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Middle Mississippi River Islands: Historical Distribution, Restoration Planning, and Biological Importance

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MIDDLE MISSISSIPPI RIVER ISLANDS: HISTORICAL DISTRIBUTION, RESTORATION PLANNING, AND BIOLOGICAL IMPORTANCE

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Abstract

Loss of habitat diversity is a common problem in many large rivers around the world. Due in part to impoundment, channelization, and levee construction, the Middle Mississippi River's (MMR's) hydrologic regime and sedimentation patterns have been radically altered. These activities contribute to stabilizing, narrowing, and deepening of the navigation channel, and to the loss of shallow backwaters, islands, secondary channels, and reduced biodiversity. The U.S. Fish and Wildlife Service included reasonable and prudent alternatives (RPAs) in its Biological Opinion for the Operation and Maintenance of the Nine-Foot Channel Navigation Project in the MMR. These RPAs mandated that aquatic habitat restoration such as dike notching and island building be implemented to facilitate development of a diversified aquatic habitat to benefit fish assemblages. Hence, my dissertation consisting of three primary goals was conducted. The first goal was to examine changes in the number and relative location of islands from 1797 to 2003 as an indicator of change in habitat diversity. I found that while the total number of islands increased, the number of centrally located large islands significantly decreased, and the number of small islands along the river border increased. The second goal was to explore planning aspects of building islands to increase breeding habitat for the federally endangered interior population of the least tern (*Sterna antillarum*). I used existing river bathymetry, extensive period-of-record hydrological data, and a model representing stage-discharge relationships (UNET) to determine island elevations required to allow continuous exposure for 50, 75, and 100 days during the 15 May to 31 August least tern breeding season. The UNET model resulted in a probability table of overtopping events/successful exposure for the given elevation and time period along the river. These results should promote the development of suitable habitat throughout the MMR, thereby allowing interior least terns to expand beyond their present nesting locations. The third goal was to investigate the ecological and biological significance to fish assemblages of created islands in a dike field, in relation to dike fields without islands. While island creation through dike notching appears to be a practical technique to increase local habitat diversity, and therefore fish diversity, the impacts to fish assemblages had not been previously investigated. I collected 44,501 fishes representing 71 species and 19 families. The families comprising significant percentages included Cyprinidae, Sciaenidae, Ictaluridae, and Clupeidae. Species richness was greater at islands (67) than at reference sites (55). For habitat types, species richness was lowest at tips (48), but similar among inside (53), outside (56), and reference (55) locations. Catch per unit effort did not differ among sites or habitat types, but fish assemblages differed significantly among islands and reference sites for total standardized count and for adult standardized count. Fish communities differed significantly among each of the habitat types, with the exception of outside and reference habitat, for total standardized count; and among the habitat types, with the exception of tip and reference sites, for adult standardized count. By incorporating island restoration planning activities for the MMR with these results, I can conclude that the assertion that created islands increase local habitat diversity through creating shallow backwater-like habitats is substantiated, because islands support fish assemblages which are distinct from those found in conventional dike fields.

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Dedication

For Mom, who instilled in me the motivation to “always do your best”.

For Cole, Justin Michael, Sophia,

and the little ones yet to come,

I encourage you to “always do YOUR best”!

For my family and friends –

My successful completion of this entire Ph.D.

program from coursework, to qualifying exams,

to field and lab work, writing, and defenses

would literally not have been possible without you.

I love you all so very much.

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CHAPTER ONE

THE MIDDLE MISSISSIPPI RIVER, MISSOURI, USA: AN EXAMINATION OF ABIOTIC ALTERATIONS, NAVIGATION OPERATION AND MAINTENANCE, AND ISLAND REHABILITATION ON FISH ASSEMBLAGES

Introduction

Study Problem

Loss of habitat diversity is a common problem in many large rivers around the world (Ward and Stanford 1989, Johnson and Jennings 1998, Pedroli et al. 2002). Homogenization of river habitat often results from human attempts to stabilize rivers for navigation purposes or other functions, and results in decreased aquatic biodiversity. In the face of ongoing threats and environmental degradation, the preservation of biodiversity requires the rehabilitation of altered rivers. Rehabilitation is defined as the partial recovery of ecosystem structure or function within the context of its present-day human use (FISRWG 1998, Dudgeon 2005).

Like most large rivers (Welcomme 1985, Dynesius and Nilsson 1994), the Middle Mississippi River (MMR), stretching 314 km from the mouth of the Missouri River near St. Louis, Missouri, to the mouth of the Ohio River near Cairo, Illinois, has been extensively altered by humans (Stevens et al. 1975, Norris 1997). The first modification to the MMR for navigation began in 1824, with clearing and snagging to remove hazards for wooden-hull vessels. In the 1830's, the first channel stabilization works were built. In 1881, a comprehensive plan was authorized to maintain an 8-foot channel through bankline revetments and permeable dikes. Congress authorized the existing Nine-Foot Channel Navigation Project in 1927 for the purpose of securing a 9-foot-deep by 300-foot-wide channel between St. Louis, Missouri, and Cairo, Illinois, as well as approving the construction of 23 locks and dams upstream of St. Louis, Missouri (USACE 1999a). As a result of upstream impoundment, navigation system development, and levee construction for urban and agriculture flood damage reduction, the river's hydrologic regime and sedimentation patterns have been radically altered. This has contributed to the stabilization, narrowing, and deepening of the navigation channel, and to the loss of shallow backwaters, islands, and secondary channels (USACE 2000).

The Corps of Engineers (Corps) is responsible for Operation and Maintenance (O&M) of the Nine-Foot Channel Navigation Project, as authorized and funded by Congress. Operation and Maintenance of the navigation system involves impoundment, water level regulation, dredging and disposal, clearing and snagging, construction and maintenance of river training structures and revetment, barge traffic, fleeting areas, port facilities, recreation, cabin leases, and General Plan Lands management. In performing these duties, the Corps is also committed to complying with the Endangered Species Act of 1973 (ESA; 16 USC 1531-1543) to ensure that the project is not likely to jeopardize the continued existence of any federally listed threatened or endangered species, or result in the destruction or adverse modification of critical habitat. On May 15, 2000, the U.S. Fish and Wildlife Service provided a Biological Opinion (BO)

(USFWS 2000) of federally listed endangered and threatened species which may be impacted by the project as projected 50 years into the future. The Service concluded that the continued O&M of the Nine-Foot Channel Navigation Project jeopardized the continued existence of the federally endangered pallid sturgeon (*Scaphirhynchus albus*). Reasonable and prudent alternatives (RPA's; actions to avoid the likelihood of jeopardizing the continued existence of listed species) provided by the Service, called for (in part), the implementation of a long-term aquatic habitat rehabilitation program to restore habitat quantity, quality, and diversity so that the benefits of the dynamic natural river processes are restored; as well as the implementation of short-term aquatic habitat rehabilitation measures (e.g., pilot projects) (USFWS 2000). Such pilot projects include dike notching and island building to facilitate development of a diversified aquatic habitat to benefit fish assemblages. Small scale demonstration projects have shown possibilities for a quick recovery of suitable habitats with characteristic pioneer species after disturbance or creation by river dynamics (Pedroli et al. 2002).

River Ecology Background

Large River Concepts – Large rivers are among some of the world's most severely degraded ecosystems. Despite the importance of large rivers, our understanding of how they function and how anthropogenic activities influence river processes is limited. Large rivers have been studied relatively little compared with small streams and lakes, in part because they are more difficult to sample, but also because there is no clear theoretical basis for how large river ecosystems operate (Johnson et al. 1995). Current views of the structure and function of large river ecosystems are based primarily on three influential riverine models: 1) The river-continuum concept (RCC) (Vannote et al. 1980) was developed from observations on stable, unperturbed streams in north-temperate, forested watersheds. The concept states that forested river systems have a longitudinal structure that results from a gradient of physical forces that change predictably along the length of the river. These physical forces produce a continuum of morphological and hydrological features from the headwaters to the mouth (Vannote et al. 1980, Johnson et al. 1995). 2) The serial discontinuity concept (Ward and Stanford 1983) integrated the effects of large dams and reservoirs on the RCC. Dams cause a discontinuity in the longitudinal continuum of physical and biological features, which should shift the predictions of the RCC either up or down the stream-order axis depending on the dam's location within the river network, the number of dams in a series, the method of dam operation (i.e., surface versus deep release and continuous versus regulated flow), and the particular river characteristic of interest (Palmer and O'Keefe 1990, Johnson et al. 1995). 3) The flood-pulse concept was introduced partly in response to the shortcomings of the RCC as a model for large rivers (Water and Rivers Commission 2000). The concept states that the most important hydrologic feature of large rivers is the annual flood pulse; and introduces a lateral dimension to the dynamics of river systems by extending the focus beyond the main channel (Junk et al. 1989, Bayley 1995, Johnson et al. 1995). Within the flood-pulse concept, the river system includes the main channel, off-channel water bodies, and periodically flooded areas (Junk et al. 1989, Johnson et al. 1995). Floodplains are highly productive and usually contain extensive riparian forests and a variety of aquatic habitats, such as backwaters, marshes, and lakes (Amoros 1991, Johnson et al. 1995). The concept states that during a flood, aquatic organisms migrate out of the channel and onto the floodplain to use the newly available habitats and resources. Subsequently as flood waters recede, nutrients and organic matter from the floodplain are carried back into the main channel, side channels, and backwaters, along with newly produced organisms such as young fish, invertebrates, and waterfowl. Additionally, a fresh supply of sediment is deposited on the floodplain (Junk et al. 1989, Johnson et al. 1995).

Natural (Unregulated) Large River Systems – Natural floodplain rivers are disturbance-dominated ecosystems characterized by high levels of habitat diversity (Welcomme 1979, Salo et al. 1986, Copp 1989, Junk et al. 1989, Sheehan and Rasmussen 1993, Ward and Stanford 1995b, Decamps 1996, Petts and Amoros 1996, Ward et al. 1999, Ward et al. 2001). The fluvial actions of flooding and channel migration create a shifting mosaic of habitat patches and successional stages across the riverine landscape. Such patches and/or stages may include a variety of lentic and lotic habitat types such as runs, riffles, gravel bars, sandbars, islands, side channels, backwaters, oxbow lakes, sloughs, main channel waters, and floodplains (Junk et al. 1989, Sheehan and Rasmussen 1993, Ward and Stanford 1995b, Stanford et al. 1996, Ward et al. 1999). In effect, river ecosystems have three important spatial dimensions that are temporally dynamic: the longitudinal (upstream-downstream) dimension, the lateral (side to side) dimension, and the vertical (groundwater interaction) dimension, all of which are dynamically molded through time (the fourth dimension) (Sheehan and Rasmussen 1993, Stanford et al. 1996).

The biota of floodplain rivers are adapted to exploit the spatiotemporal heterogeneity of the system caused by hydrological disturbance (i.e., flood and droughts) (Welcomme 1979, Salo et al. 1986, Copp 1989, Junk et al. 1989, Sparks et al. 1990, Bayley 1991, Bayley 1995, Theiling 1995, Ward and Stanford 1995b, Decamps 1996, Petts and Amoros 1996, Ward et al. 1999). Resources needed by a particular life history stage of an organism have a discrete or patchy distribution within this heterogeneous landscape (Stanford et al. 1996). Each species or life history stage is most abundant where the resources they require are most abundant and/or most efficiently maintained. A species will be present wherever they have enough resources to sustain growth and reproduction and thereby sustain the presence of the species in the river at that location (Hall et al. 1992, Stanford et al. 1996). Because biotic dynamics derive from natural variation in the environmental setting, equilibrium conditions rarely exist for very long because environmental changes are constantly reconfiguring resource availability (Calow and Petts 1992, Ward and Stanford 1983, McAuliffe 1983, McAuliffe 1984, Reice 1994, Stanford et al. 1996). Ecological capacity, therefore, varies from place to place, and higher levels of biological richness are most likely to occur in ecosystems with a history of spatial and temporal environmental heterogeneity (Connell 1978, Ward and Stanford 1983, Salo et al.

1986, Poff and Ward 1990, Stanford et al. 1996, Ward 1998). Periodic constraints on species-specific productivity increase opportunities for other species to use resources, implying that levels of ecosystem biodiversity are generally related to the intensity, frequency, and duration of disturbance events (Huston 1979, Resh et al. 1988, Pimm 1991, Huston 1994, Reice 1994, Stanford et al. 1996). In contrast, the biomass of a few species, while also constrained by inherent ecosystem capacity, may be high under sustained conditions of environmental stability due to slow turnover rates. For example, a few species are often extremely abundant and persistent where disturbance events are relatively benign (Valett and Stanford 1987, Wooton 1987, Reice 1994, Shannon et al. 1994, Stanford et al. 1996). Though ecological systems require temporal variation in abiotic processes, the variation has to be consistent within a specific time scale in most cases. For example, many northern rivers experience a highly predictable variation in discharge and water level. Flooding, and especially spring flooding, is a key factor which sets the system (Nilsson and Brittain 1996). Biotic interactions (i.e., competition, predation, parasitism), while they obviously continually occur within food-webs in all habitats, may become progressively more important and apparent as the time between abiotic disturbances increases, and hence are most pronounced where abiotic forces are comparably non-variable (Ward and Stanford 1983, McAuliffe 1983, McAuliffe 1984, Reice 1994, Stanford et al. 1996). Additionally, at smaller scales, biological interactions may control local species abundance and resource use (Power et al. 1988, Schlosser 1991). In large rivers, a gradient of relative importance of physical and biological control may extend from the main channel, where physical forces might dominate, to backwaters and floodplain lakes, where biological interactions might control community structure (Statzner 1987, Pecharsky et al. 1990, Johnson et al. 1995).

In summary, natural disturbance regimes and environmental gradients, acting in concert, result in a positive feedback between connectivity and spatiotemporal heterogeneity that leads to the broadscale patterns and processes responsible for high levels of biodiversity (Salo et al. 1986, Amoros and Roux 1988, Ward and Stanford 1995b, Ward et al. 2001, USFWS 2004). Different types of riverine habitat, indeed different successional stages within them, contribute to biodiversity as the biota exploit the spatiotemporal heterogeneity (Ward 1998). Indeed, it is the mosaic structure and dynamic nature of a river system that maintains its functional integrity (Ward and Stanford 1995b, Ward et al. 2001).

Regulated Large River Systems – The majority of the world's rivers are regulated (Dynesius and Nilsson 1994), and these developments have considerably changed landscape structure and processes, and led to an impoverishment of natural diversity (Ward and Stanford 1995a, Nilsson and Brittain 1996). Regulated rivers usually have a more homogeneous habitat than unregulated rivers (Thorp 1992). Anthropogenic impacts such as flow regulation, channel straightening, dredging, bank stabilization, and levee construction eliminate upstream-downstream linkages and isolate river channels from their riparian/floodplain systems by disrupting natural disturbance regimes, truncating environmental gradients, and severing interactive pathways. These alterations decrease spatiotemporal heterogeneity, and interfere with successional trajectories, habitat diversification, migratory pathways, and other processes, which ultimately reduces biodiversity (Sheehan and Rasmussen 1993).

Lock and dam systems convert rivers from a free-flowing condition to a series of impoundments. River regulation reduces annual flow amplitude, increases baseflow variation and changes temperature, mass transport, and other important biophysical patterns and features (Stanford et al. 1996). Additionally, channel degradation (deepening) occurs downstream from dams because water velocities can be greater due to upstream impoundment and constriction of the channel. Streambed degradation can occur for hundreds of kilometers downstream of dams. Furthermore, deepening of the channel causes adjacent off-channel habitats (side channels and backwaters) to become isolated or dewatered (Sheehan and Rasmussen 1993). Persistent shallow or slack water habitats are especially important for the survival of early life history stages of fishes that cannot survive in the strong currents of the channel thalweg.

Rock dikes, running perpendicular to the shore, have long been used to guide the river and maintain the navigation channel. They are used to concentrate flow in the main channel during periods of low discharge, and to reduce the need for dredging. Dikes are often grouped together to form a “dike field” in order to address a particular navigation problem. These areas often become depositional zones that fill from the bank outward toward the channel

(Neimi and Strauser 1991, Shields 1995, Theiling 1995). Notching dikes may help to alleviate this problem.

Extensive levee systems which disconnect the river from the floodplain ecosystem have also been constructed. Levees block horizontal interactions such as transfers of organic matter from the floodplain to the river (Peck and Smart 1986), as well as isolating backwater wetland ecosystems from the main river channel. This precludes access by riverine fishes, which typically make seasonal movements to backwaters and floodplains (Theiling 1995, USACE 1999a). As a result, levee projects affect the production of forage food organisms for native fishes, and may have isolated some species from important spawning, nursery, foraging areas and/or seasonal refugia (Sheehan and Rasmussen 1993, Theiling 1995, USACE 1999a).

The general conclusion is that regulation creates a discontinuum of environmental conditions and severs the connectivity of channel, groundwater, floodplain, and upland components of the catchment ecosystem; habitats for riverine biota become spatially homogenous, limited to the permanently wetted portion of the channel thalweg that is dominated by conditions dictated by operations of upstream storage reservoirs (Baxter 1977, Ward and Stanford 1979, Lillehammer and Saltveit 1984, Ward and Stanford 1987, Petts 1989, Calow and Petts 1992, Stanford et al. 1996). Rivers with more diverse habitats, in both the channel and floodplain, are likely to be more productive (Schmier and Zalewski 1992, Johnson et al. 1995). Any constraints on the natural flow pattern, such as damming or irrigation withdrawals, or on hydraulic processes within the channel, such as dikes or channelization, are likely to reduce habitat diversity. An annual flood pulse greatly enhances habitat diversity by allowing aquatic organisms access to the floodplain, but floodplain modifications, such as levees, draining of lowlands, and urbanization, can reduce diversity (Johnson et al. 1995).

Practical Aspects to Consider

River History – Loss of river function can occur because hydrologic and geomorphic processes no longer create and maintain the habitat and natural disturbance regimes necessary for ecosystem integrity (Wohl 2005). Determining the degree to which a river has been altered from its reference condition requires knowledge of historical land use and the associated effects on rivers (Wohl 2005). Rivers have a history, and rehabilitation or other management activities conducted in ignorance of this history may cause additional impairment to river ecosystems and human society (Wohl 2005). Lack of knowledge of regional land-use and river history may lead to unrealistic rehabilitation goals. Unfortunately, characterization of a reference condition can be very difficult in a region in which most river systems have been altered as a result of land-use patterns or in which land use has changed rivers for centuries. In these situations, a reference condition is likely to represent a randomly selected moment in the ongoing history of human-induced change in rivers. A reference condition can be estimated based on (1) the river characteristics of unaltered but otherwise comparable rivers, if any are available; (2) the river characteristics that can be anticipated given the climatic and geologic attributes of the area; or (3) information on the original river properties can be derived from old maps, photographs, and field data, and can serve for the definition and mapping of the corresponding habitats (Pedroli et al. 2002, Wohl 2005).

Regardless of how a reference condition is estimated, an historical knowledge of how land use transforms rivers forms a key component of rehabilitation design. This historical knowledge provides a framework for the causes, duration, spatial extent, and intensity of human-induced changes in a river (Petts 1989, Sear 1994, Kondolf and Larson 1995, Wohl 2005). Flood levees, river training structures, and controlled discharges represent irreversible changes in the abiotic environment, and must be taken into account when describing the expected processes, configurations, and ecological communities (Pedroli et al. 2002). The existing flood levees put spatial limits on rehabilitation projects. River training structures are placed or developed in a river reach to modify the hydraulic flow and sediment response of the river. The degree of control of the discharge determines the extent to which natural hydrodynamics can act as the driving forces for ecosystem development (Pedroli et al. 2002). Regrettably, rehabilitation of most rivers to their original state is impractical given prevailing regional constraints. However, some degree of rehabilitation should be possible if relevant legislation and scientific information are promptly applied (Dudgeon 2005).

Large River Rehabilitation – Loss of biodiversity in riverine communities (Frissell 1993, Welcomme 1995) underscores the need for rehabilitation of regulated rivers (Stanford et al. 1996). Often, the term “river regulation” implies that flooding is strongly reduced. Hence, reinstatement of flooding becomes a primary target if environmental quality is to be enhanced (Nilsson and Brittain 1996). Many authors suggest that the rehabilitation of large rivers should begin with recognition of the river continuum, evaluation of the loss of ecosystem capacity to sustain biodiversity, and rehabilitation of ecosystem processes that promote abiotic and biotic diversification and maintain genetic diversity (i.e., Gore 1985, Sparks et al. 1990, Gore and Shields 1995, Power et al. 1995, Sparks 1995a, Sparks 1995b, Stanford et al. 1996, Roux et al. 2002). Stanford et al. (1996) and others suggest that the main goal should be to reduce the range of human disturbances so that interconnected riverine habitats can support diverse and productive food-webs, including species of special social and economic interest, and that management should strive to restore environmental heterogeneity by letting the river do the work (Gore 1985, Sparks et al. 1990, Gore and Shields 1995, Power et al. 1995, Sparks 1995a, Sparks 1995b, Stanford et al. 1996, Roux et al. 2002). It is often proposed that owing to the importance of flow to habitat creation, and temperature and food-web energetics, highly significant rehabilitation is possible simply by re-regulation to allow more natural seasonality of flow and temperature (Stanford et al. 1996, Nilsson and Brittain 1996). Peak flows are needed to scour and rearrange substratum and reconnect floodplain habitats with the channel; while spatial and temporal temperature variability promotes re-establishment of native biodiversity (Stanford et al. 1996, Nilsson and Brittain 1996).

Thus, many river ecologists consider restoration of ecological processes necessary to maintain natural disturbance regimes, migratory corridors, habitat diversity, landscape connectivity, and evolutionary templates (Roux et al. 2002). Roux et al. (2002) suggest the restoration of four essential freshwater processes, including: 1) The magnitude, frequency, duration, timing, and rate of change in water flow. These components interact to maintain the dynamics of in-channel, off-channel, and riparian habitats, and determine the distribution of freshwater and riparian species. Natural hydrological disturbances, such as droughts and floods, are particularly important for maintaining the geomorphological integrity of freshwater ecosystems (Roux et al. 2002); 2) Nutrient cycling, or the process whereby elements such as nitrogen, phosphorus, and carbon move through an ecosystem. Reduction or augmentation of nutrients can modify the trophic status of freshwater ecosystems, which influences primary and secondary productivity and, consequently, overall ecological integrity. Riparian habitats are particularly important because these areas serve as allochthonous sources of organic matter, filter sediment, and nutrient input from terrestrial ecosystems (Roux et al. 2002); 3) Connectivity, over both space and time, is required for the movement of species between habitats. Maintenance of instream and floodplain habitats by restoration of peak flows and revitalization of shallow and slack water habitats by stabilization of baseflows will increase ecological connectivity along all three spatial dimensions (Gore 1985, Gore and Shields 1995). Longitudinal and lateral connectivity allows organisms to move up and down the watershed or into off-channel habitats, for example, to complete their life cycles. Movement of species is especially important during times of habitat change or climatic disruptions, for instance, movement into refuge areas to survive during winter, floods, droughts, or the dry season (Roux et al. 2002); 4) Succession and evolution. River systems are longitudinal systems, meaning that most fish and some invertebrate species are restricted to a particular river system and as such are isolated from other populations in adjacent rivers. The evolutionary processes acting on species occupying perennial rivers would therefore be very different from those acting on species found in intermittent rivers (Roux et al. 2002).

Cooperation in a Multiple-Use System – Multiple-use conflicts on large rivers will continue to increase as river uses increase. Large rivers are important for transportation, water supply, waste assimilation, generation of electricity, fish production, and recreation (Sparks et al. 1990). As a result, large rivers in the United States have been irreversibly altered from natural conditions, and they are already artificially managed for other primary uses (Sheehan and Rasmussen 1993). This requires innovation and artificiality in managing biotic communities as well. Large-river managers deal with multiple-use conflicts because fisheries resources have been considered secondary to other uses (Sheehan and Rasmussen 1993). River managers should therefore consider the maintenance of a healthy ecosystem as their primary goal (Sheehan and Rasmussen 1993). Supplemental fish stocking, introductions, and habitat rehabilitation and enhancement projects may be required. From a practical standpoint, impoundment and many other modifications for navigation and flood control can be considered as *fait accompli*. It is unrealistic that there will be any large-scale dismantling of the inland waterway system in the near term. Therefore, Sheehan and Rasmussen (1993) suggest that biologists should consider using innovative approaches to managing large rivers under existing conditions. As a general rule, fish management objectives in large rivers should be established according to the following criteria: 1) threatened and endangered species and remaining unique habitats should be preserved, and reintroductions of extirpated species should be considered; 2) fish communities should be managed to maximize species diversity, making certain that existing trophic niches are occupied; 3) resources such as commercial and sport fisheries should be managed for optimal sustained yield; and 4) habitat diversity should be maintained minimally at its current level through sound conservation practices, or, optimistically, increased through construction projects directed at rehabilitation, enhancement, or creation of fish habitat and near natural flow regimes (Sheehan and Rasmussen 1993). In fact, the U.S. Fish and Wildlife Service suggests the use of short-term aquatic habitat rehabilitation measures (i.e., pilot projects), including side channel restoration, island building, and dike notching to increase habitat diversity in the MMR (USFWS 2000).

On a strictly theoretical level, restoration of natural hydrographic features and the associated ecological processes seems paramount. The strong inference is for management to protect uncolonized floodplains by re-establishing periodic overbank flooding, allowing the river to rebuild habitats (Gore 1985, Gore and Shields 1995). In actuality, however, restoration of overbank flows may be problematic in many rivers where humans have colonized the floodplains. In these cases, levees have often been extensively built to restrain flood flows. Levees are problematic because the objective is to reconnect channels and floodplains (Stanford et al. 1996). Thus, re-regulation to produce overbank flows may not be practical (Stanford et al. 1996).

When put into practice, however, water level manipulation of any kind in developed areas rarely meets with favor from riparian property owners (Sheehan and Rasmussen 1993). As a consequence of floodplain habitation, flow augmentation and water level manipulation on a large scale is rarely, if ever, practical for large-river fisheries (Sheehan and Rasmussen 1993).

Bioengineering… a Partial Solution?

Middle Mississippi River - The natural meandering processes of the MMR have been altered through construction of wing dikes, bankline revetments, levees, closing structures, bendway weirs, and channelization. Furthermore, as of 2000, approximately 80% of the floodplain of the MMR had been isolated from the main channel due to levee construction. Consequently, the MMR ecosystem is a dynamic system that responds to abiotic controls which are now largely defined by humans (Starrett 1972, Bhowmik and Adams 1986, Chen and Simons 1986, Grubaugh and Anderson 1988, Lubinski et al. 1991, Theiling 1995). River regulation in the MMR has disrupted the natural disturbance regimes that maintain a diversity of successional stages and high levels of connectivity across riverine landscapes, resulting in a loss of aquatic habitat heterogeneity and biodiversity (Thorp 1992, Ward et al. 1999, USFWS 2004, Hurley et al. 2004). With the construction of the Nine-Foot Channel Navigation Project, the riverbank top width has been reduced, side channels, islands, and ephemeral sand bars have been lost, the physical process of channel meandering has been arrested, and sediment transport and availability for habitat development have been significantly impaired (USFWS 2004). Because the MMR is highly channelized and has few secondary or abandoned channels, sandbars, or large islands, the Pallid Sturgeon Recovery Plan identified this region of the river as a recovery-priority area (Dryer and Sandoval 1993). Enhancement and restoration of downstream island tip, main channel border, and between wing-dike habitats would represent an increase in habitat diversity, which could benefit many species including the endangered pallid sturgeon (Hurley et al. 2004) and the endangered interior least tern (*Sterna antillarum)* (USACE 1999b).

The trend is slowly turning towards developing river rehabilitation concepts. However, such rehabilitation concepts for large river systems are currently in an early stage of development. Most rehabilitation ecologists agree that the concepts should be based on theoretical concepts of river ecology; ecosystem oriented; conceived at the catchment-scale; and should primarily foster the hydrological and geomorphological functions of the river (Regier et al. 1989, Gore and Shields 1995, Henry and Amoros 1995, Sparks 1995a, Sparks 1995b, Stanford et al. 1996, Schiemer at al. 1999, Ward et al. 1999).

A consequence of stabilizing a dynamic alluvial river system is that in general, new islands and side channel areas are no longer being created naturally. Therefore, humans are required to take over many of the rejuvenating functions once performed by natural scouring and erosion (Sparks et al. 1990).

Physical habitat rehabilitation in large floodplain rivers, including island creation, can be addressed on a local scale (Sparks et al. 1990). Bioengineering may be required to divert some flow from the main channel to create and maintain islands and side channels instead of a uniform habitat (Sparks et al. 1990). Because abiotic factors largely control biotic elements of rivers, protecting and conserving habitat heterogeneity should result in the protection and conservation of river organisms (Stanford 1998, Roux et al. 2002). Because island dynamics integrate several ecological processes, the presence, distribution, and turnover of islands may provide landscape-level indicators for assessing the ecological status of river corridors, as well as serving as a rehabilitation goal (Ward et al. 2001). Thus, in chapter 2, I examined the recent history of the river, specifically the distribution of islands in the MMR from the pre-navigation era through current times. Although river rehabilitation at the catchment scale is ideal, the potential for rehabilitation, as well as the constraints, differ depending on the degree of anthropomorphic change and the many functions the river may currently serve (Schiemer et al. 1999). In many cases, it is often more practical to begin basic rehabilitation efforts at local or regional scales, rather than at the catchment-level (USFWS 2000). Consequently, in Chapter 3, I modeled local island creation in the MMR. Finally, because fish are excellent indicators of terrestrial and aquatic environmental impacts caused by human activity (Bennett et al. 1974, Goulding et al. 1996) I used them as a study species. Island creation through dike notching appears to be a practical technique to increase local habitat diversity, and therefore fish diversity; however, the impacts to fish assemblages have not been investigated. Thus, in Chapter 4, I investigated the purported ecological and biological significance to fish assemblages of created islands in a dike field, in relation to dike fields without islands.

These goals were achieved by implementing several objectives as follows: Chapter 2- 1) enumerating and determining the relative location and size of islands in the MMR between 1798 and 2003; 2) determining locations in the MMR where islands are lacking based on historical distribution patterns; Chapter 3 - 3) identifying a site (or sites) in the MMR where habitat diversity is lacking and island (or island complex) creation may benefit local fish and bird populations; 4) modeling the size, relief, and inundation characteristics of an island (or island complex) proposed for construction in the MMR; Chapter 4 - 5) determining assemblage composition, diversity, and richness in dike fields with and without created islands; and 6) testing the significance of associations between habitat characteristics (water depth, current velocity, substrate type, habitat type) and biological and behavioral traits of fish species (feeding habits, habitat niche categories, size classes). It is crucial that a study that examines the relationship between islands and their influence on fish assemblages be conducted now, as many such island-building activities are currently under consideration for the MMR, and the potential impacts and purported benefits are undocumented.

Project Significance

The extensive ecological degradation and loss of biological diversity resulting from river exploitation is generating widespread concern for conservation and rehabilitation among scientists and the lay public alike (Poff et al. 1997, USFWS 2000, USFWS 2004). The importance of understanding the role that islands play in the ecological function of the MMR is great. The Corps of Engineers, state and federal resource agencies, under the auspices of the Upper Mississippi River Environmental Management Program, recently developed a habitat needs assessment (HNA) for the Upper Mississippi River (USACE 2000). It cites goals that included the creation or rehabilitation of 25,000 acres of backwater and secondary channel habitat with associated islands. Additionally, the USFWS called for the implementation of a long-term aquatic habitat rehabilitation program in the MMR to restore habitat quantity, quality, and diversity so that the benefits of the dynamic natural river processes are restored (USFWS 2000). They also required the implementation of projects such as dike notching and island building to facilitate development of a diversified aquatic habitat to benefit fish assemblages. Thus, the impetus for this work is to: 1) document the historical distribution and alteration of MMR islands since the navigation era began, 2) determine the most practical location and design criteria for island creation in the MMR to help restore local habitat diversity, and 3) provide the required quantitative data necessary to substantiate the assertions that created islands increase local habitat diversity by creating shallow backwater-like habitat, which is limited in the MMR, and support a fish assemblage which is distinct from that found in conventional dike fields.

Habitat rehabilitation can be a cost-effective method for enhancing river ecosystems and fish communities (Sheehan and Rasmussen 1993). However, most habitat rehabilitation projects on large rivers are complex and expensive.

Therefore, it is imperative that information of this nature be obtained before the numerous habitat rehabilitation and mitigation projects for the MMR are developed.

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CHAPTER TWO

THE DYNAMIC NATURE OF ISLAND GEOMORPHOLOGY IN THE MIDDLE MISSISSIPPI RIVER BETWEEN 1798 AND 2003

Introduction

Natural floodplain rivers are disturbance-dominated ecosystems characterized by high levels of habitat diversity; and the biota are adapted to exploit the spatiotemporal heterogeneity of the system (Sparks et al. 1990, Ward and Stanford 1995, Ward et al. 1999). Islands which divide the river into a main channel and secondary channels were once characteristic habitat features of many large floodplain rivers within forest biomes (Gurnell and Petts 2002). Today, most large floodplain rivers within heavily developed regions are almost vacant of vegetated riverine islands (Ward et al. 2000, 2001, Gurnell and Petts 2002).

Island formation is a direct result of high-energy dynamic processes that redistribute large amounts of sediments (Tockner et al. 2003). However, the hydrologic regime and sedimentation patterns of many large rivers, including the Middle Mississippi River (MMR) have been altered to accommodate multiple anthropogenic uses. Navigation system development has resulted in upstream impoundment, channelization, dredging, construction of river-training structures, closing structures, and bankline revetment. Levees have been constructed to reduce flood damage in urban and agricultural areas. These alterations have brought about the stabilization, narrowing, and deepening of the navigation channel, and have contributed to habitat homogenization through the loss of shallow backwaters, islands, and secondary channels (Simons et al. 1974, USACE 2000, Gurnell and Petts 2002) and, consequently, reduced biodiversity. Islands provide multiple benefits to both aquatic and terrestrial organisms, including increased habitat diversity due to the formation of side channels, backwaters, areas of reduced current velocity, shallow water habitat, and in some cases, scour holes (Schueller 1989, Johnson and Jennings 1998).

Changes in the characteristics and relative number of riverine islands provide information of recent geomorphic history and processes, and may serve as an ecosystem-level indicator of the condition of river passageways (Ward et al. 2000, 2001). In many North American and European river corridors, for instance, vegetated riverine islands have become endangered landscape elements (Ward et al. 2000, Gurnell and Petts 2002). Galat and Frazier (1994) report an 89% reduction in number of islands in the lower Missouri River ecosystem as a result of river channelization. The Illinois River lost approximately 40% of its riverine islands between 1903 and 2007 (USACE 2007), and (Simons et al. 1974) states that in the absence of further man-made changes in the hydrology or geomorphology of the MMR, all natural side channels and associated islands, except the larger ones, may disappear from the river scene. Furthermore, early maps provide evidence that wooded islands were common on many rivers across Europe prior to the modern era of river regulation (Gurnell and Petts 2002). For example, the Austrian segment of the Danube had approximately 2000 islands in historical times, but only six remain (Ward et al. 2002). In contrast, islands remain a strong feature of riverine landscapes in areas away from agricultural and urban development, and along large floodplain rivers too powerful for early regulation, such as the French upper Rhône (Amoros et al. 1987, Gurnell and Petts 2002).

As a result of riverine habitat loss in the MMR, state and federal agencies, as well as numerous authors, have indicated the need for increased habitat diversity. The use of aquatic habitat rehabilitation measures including side channel restoration and island building were suggested in order to accomplish this goal (Johnson and Jennings 1998, U.S. Fish and Wildlife Service 1990, 1993, McGuiness 2000, USACE 2000, Barko and Herzog 2003).

Prior to undertaking any rehabilitation effort, it is important to define the natural river baseline situation, and to determine the degree to which a river has been altered from its reference condition (Pedroli et al. 2002). This baseline information can then be examined in the context of known or understood river functions within the riverine landscape, under the consideration of constraints put on the system by society, to define a clear rehabilitation target for the future ecological state of the river (Pedroli et al. 2002). Accordingly, I considered it prudent to examine the earliest reliable maps and charts to document the condition of the MMR before intensive navigation alterations began in the mid-1800's.

Although sediment erosion and deposition are dynamic processes in rivers, and consequently islands both agrade and degrade, a considerable decrease in the number of islands, or a change in the characteristics of islands may indicate a significant loss of habitat diversity. The objectives of this study were to 1) examine historical maps and charts to determine if there have been systematic patterns of change in the numbers and relative location of MMR islands over time, and 2) describe any trends in changes of island characteristics.

Methods and Materials

Data Acquisition – I examined the MMR, which extends from the confluence of the Missouri River to the confluence of the Ohio River (Figure 2.1). I divided the MMR into four reaches, each approximately 50 river miles (RM) (approximately 80.5 km) in length. English units were used in this study because they are the U.S. Army Corps of Engineers' standard unit of measure for all contemporary and historic survey documents. The river miles were estimated using U.S. Army Corps of Engineers hydrographic survey maps of the Mississippi River (USACE 2005). The four reaches were demarcated based upon landmarks which were either historically identifiable or physically fixed in nature as follows: mouth of the Missouri River (start; approximately RM 200), mouth of Yoachim (Joachim) Creek (approximately RM 150), mouth of Rock Creek (approximately RM 100), limestone outcrop south of Cape Girardeau (approximately RM 50), and the Ohio River (approximately RM 0). However, because of the dynamic nature of the river, actual river miles have changed over time. I defined an island as a vegetated terrestrial area completely encircled by aquatic habitat during bank-full flows, and counted the number of islands present in each reach. I classified islands by relative location as either "center" or "border" by measuring the width of the river from bank to bank and dividing the value by three (right and left borders and the center). Islands with greater than

50% of their area in either of the outer two regions were classified as border islands, while islands situated mainly in the center region were classified as center islands. I examined eight topographical sources (charts, maps, narratives, and aerial photographs) prepared between 1797 and 2003 (Table 2.1).

Description of Data Sources – The following eight sources were used for data acquisition:

(1) *de Finiels 1797-1798* – Information from 1797-1798 was gleaned from a map of the Mississippi Valley prepared by Nicolas de Finiels, a French military engineer, who provided the most detailed 18th century map of the middle reaches of the Mississippi River. Although the map is not planimetrically accurate, it is an impressionistically accurate rendering of the late eighteenth-century Mississippi Valley landscape and is generally considered to be an excellent source of physical information for our area of investigation (Illinois State Museum 2001).

(2) *The Navigator 1814* – “The Navigator” was used to approximate the number of islands in the MMR in 1814. “The Navigator” was a guide to those who navigated or traded on the rivers described therein. It contained directions and maps for navigating the Mississippi River (and others), as well as a concise description of the islands, towns, villages, harbors, and settlements (Cramer 1814). Islands were assigned a number, navigation instructions, and often a brief description as well. Maps of the Mississippi River contained in “The Navigator” were taken partly from survey and partly from private charts, “taken as accurately as the nature of that river will permit”.

(3) *Young et al. 1821* – This reconnaissance of the Mississippi and Ohio rivers was made during the months of October, November, and December 1821, by Captain H. Young, Captain W.T. Poussin, and Lieutenant S. Tuttle. According to the map summary, the soundings on the bars were made during the months of October and November, and it is believed that they are accurate for the lowest stage of waters in common years. The map is a basic assessment of the river, and contains information on islands, sand bars, exposed rocks, snags, the navigation channel location, as well as houses and plantations. The map does not include the river upstream of St. Louis, Missouri.

(4) *Mississippi River Commission (MRC) 1876-1881*– The information for the period between 1876 and 1881 was obtained from “Detail Map of the Upper Mississippi River from Mouth of the Ohio River to Minneapolis, Minn”, published by the Mississippi River Commission. Chart numbers 101-116 entitled “Detail Charts of the Mississippi River from Cairo, Illinois, to Hannibal, Missouri” were examined. These maps contained topographical and hydrographical data.

(5) *Board Sheets 1908* – Information for 1908 was obtained from the “The Mississippi River from St. Louis, Missouri to Cairo, Illinois in 17 Charts, Made Under the Direction of the Board on Examination and Survey of Mississippi River Created by River and Harbor Act of March 2, 1907.” The map contains elevations, soundings, dikes and hurdles, revetments or bank protections, gage records, and channel distances with origin at Eads Bridge, St. Louis, Missouri. The map does not include the river upstream of Chouteau Island, located near Granite City, Illinois.

(6-8) *USACE Aerials* – U.S. Army Corps of Engineers aerial photographs were used to count the number of islands present in 1928, 1965/1966, and 2003. These photographs provide full coverage of the Middle Mississippi River.

Data Analysis – The data used in the analyses are shown in Tables 2-6. Where data collection spanned several years (1797-1798, 1876-1881 and 1965-1966), the final year was used to represent those data points in the analyses. The 1814 data did not separate islands by type, so only the total number of islands was available for analysis. For the 200-150 mile reach, data for 1821 and 1908 were known to be incomplete; hence, these years were excluded from the analyses for that reach.

To test for changes in the number of islands over time, I used simple regression models of the form:

$$n_i = \beta_0 + \beta_1 t_i + \beta_2 t_i^2 + \varepsilon_i$$

where n_i is the number of islands at time t_i, β_0, β_1 and β_2 are fitted parameters and ε_i is a random error or residual, assumed to have a normal distribution with a mean of zero and a variance estimated by the residual mean-square. Models were fitted by standard least-squares methods, finding the combination of β parameters that minimizes the residual mean-square (variance of the ε terms). The aim was to find the most parsimonious model that provided an adequate description of the data. I first fitted a linear model, omitting the term for t^2. A quadratic model, including the t^2 term, was then fitted. The fit of each model was assessed using the standard *F* test. If the null hypothesis (β_2=0) was rejected at the 0.05 level, I used the quadratic model; otherwise the linear model was used. There was one case (center islands for the 150-100 mile reach, using all years) where both the linear and quadratic models failed to achieve significance but the *t*-tests showed that both β_1 and β_2 for the quadratic model were significantly different from zero. I therefore repeated the fit, omitting the intercept term (β_0). The resulting quadratic model had a significant fit and was therefore accepted.

Models were fitted for each category of island count (total number of islands, border islands, center islands) in each river reach, using both the full set of available years (1798 - 2003) and a reduced set of years (1881-2003). The latter omitted the first three earliest pre-navigation data sources (de Finiels 1797/1798, The Navigator 1814, Young et al. 1821). Early sources are often disregarded by various researchers when considering baseline island conditions in the MMR (i.e., Brauer et al. 2005) because they do not contain as much detail as later maps and photographs.

Predicted values for each accepted model were calculated and used to prepare plots showing both the observations and the fitted regression line or curve. Regression analyses were performed using procedure FIT MODEL in JMP IN version 5.1.2 (SAS Institute Inc. 1989-2004) and regression plots were prepared using Sigma Plot version 11.0 (Systat Software Inc. 2008).

Results

Change in Total Number of Islands – A summary of the regression results is provided in Table 2.2. Of the 12 combinations of island count (total, center, border) by river reach (200-150, 150-100, 100-50, and 50-0) examined, nine showed significant trends in island numbers over time (six linear and three quadratic) for the full range of years and three showed significant trends (three linear) using data from only 1881-2003. For every reach, at least one category of island count showed a significant trend for the full range of years. Three of the four reaches showed a significant trend for the reduced range of years.

There were only two cases for which no model was accepted for either time period: center islands in the 50-0 mile reach, and border islands in the 200-150 mile reach. Fitted parameters (β_0, β_1 and β_2) for each selected model are in the last three columns of Table 2.2.

Change in Island Numbers within Reaches –

Missouri River (RM 200) to Joachim (Yoachim) Creek (RM 150) – The total number of islands located in this reach showed no systematic trend when either the full range of years (1798-2003, p=0.88) or the limited range of years (1881-2003, p=0.09) is considered. Center islands showed a statistically significant linear decrease in number of islands for both time frames (1798-2003, p=0.02; 1881-2003, p=0.05); while border islands showed no systematic trend for either the full range of years (1798-2003, p=0.93) or the limited range of years (1881-2003, p=0.28) (Figures 2.2-2.3).

Joachim Creek (RM 150) to Rock Creek (RM 100) – The total number of islands located in this reach showed a linear increase when the full range of years (1798-2003, p=0.02) is considered. However, when only the limited number of years are examined, no statistical trend is detected (1881-2003, *p*=0.75). Center islands show a quadratic response for all years, with number of islands increasing until approximately 1902, then decreasing (*p*=0.01). However, no trend is detected when only the limited years are examined (1881-2003; p = 0.06). Border islands show a linear increase for all years (*p*=0.02). However, no trend is detected when only the limited years are examined (1881-2003; p = 0.26) (Figures 2.2-2.3).

Rock Creek (RM 100) to Cape Girardeau (RM 50) – In this reach, the total number of islands displays a quadratic trend over the 1798-2003 period, increasing to a maximum in about 1948 and then declining somewhat by 2003 (*p*=0.02). No consistent trend could be detected for total number of islands using the 1881-2003 data only (*p*= 0.25). Center islands also show a quadratic trend over the full range of years, increasing until around 1880, and then decreasing in number through 2003 (*p*=0.01). Using only the 1881-2003 data, center islands show a linear decrease over time (*p*=0.02). Border islands in this reach show an increasing linear trend using the full range of years (1798-2003; *p*=0.01), but no trend using the 1881-2003 data (*p*=0.08) (Figures 2.2-2.3).

Cape Girardeau (RM 50) to Ohio River (RM 0) – In this reach, the total number of islands has shown a linear increase from six to 10 over time according to the full range of years (1798-2003; p=0.03). However, the limited range of years showed no trend (1881-2003; p=0.10). When islands were broken down by position, center islands showed no significant trend for either range of years (1798-2003; p=0.75) (1881-2003; p=0.65). Border islands showed a linear increase over time for both year ranges (1798-2003; p=0.01) (1881-2003; p=0.05) (Figures 2.2-2.3).

Discussion

From the establishment of St. Louis in 1764 until the beginning of the nineteenth century, the Mississippi River at St. Louis was deep and narrow (Strauser 1978). However, since the arrival of the first steamboat in St. Louis in 1817, the MMR has undergone immense change (Norris 1997). By the late 1800s, the intensive and extensive clearing and gathering of wood, primarily (but not exclusively) for steamboat fuel, had denuded both banks of the MMR of most stands of mature trees (Norris 1997), making the riverbanks highly unstable. The first river training structures constructed in the MMR were a system of permeable wood and pile dikes engineered by Lt. Robert E. Lee in 1838 to move the navigation channel from the Illinois bank to the Missouri bank, where the harbor was located. In the process, they also incorporated Bloody Island, a large sandbar in St. Louis Harbor, into the Illinois shore line (USACE 1939, Dobney 1978). In 1866, Congress authorized the establishment of the 4-foot navigation channel through dredging, snagging, clearing overhanging trees, and the removal of sunken vessels. Construction of wing dikes and closing dams, which narrowed the main-stem of the river, was incorporated with the authorization of the 4.5-foot channel in 1878, and the 6-foot channel in 1907. In 1930, Congress authorized the 9-foot channel project, approving the construction of 23 locks and dams upstream of St. Louis, Missouri. The majority of the locks and dams were constructed between 1940 and 1964. The construction of stone dikes began in the 1960's, with most of the dikes built after 1963 having been constructed of stone (Brauer et al. 2005).

During periods when the river widened and became shallower, the Mississippi River channel became more braided, resulting in an increase in the total number of islands (Figure 2.2a) (Brauer et al. 2005). River widening was due to bank instability and collapse (Strauser 1978, Norris 1997). In all of the MMR reaches, except between RM 200-150, the number of islands increased significantly (1798-2003 data). However, while islands in RM 150-100 and RM 50-0 increased, islands in RM 100-50 increased only through 1948, then decreased. The failure of RM 200-150 to show a trend in the number of total islands is not unexpected, as two data sources were eliminated from the analyses due to a lack of data above St. Louis, Missouri. Furthermore, the RM 200-150 reach is highly developed and includes the present-day St. Louis Harbor region.

Center islands in all reaches significantly decreased in number by 1902, with the exception of RM 50-0 (1798-2003 data) (Figure 2.2b). Brauer et al. (2005) asserted that the closure of many of the side channels in the MMR led to a decrease in number of larger islands (often center islands). This is supported by the 1798-2003 data (Figure 2.3).

Conversely, an increase in the number of border islands could also result from the placement of closing structures across center island side channels, because this often merges these islands to the bankline or creates smaller islands situated closer to the bankline. Middle Mississippi River RM 200-150 did not show a trend in border islands, likely due to the reduced data available for analyses (1978-2003 data). However, RM 150-100, RM 100-50, and RM 50-0 all demonstrated an increase in the number of border islands as expected if closing structures were in-part responsible for the observed trends (1978-2003 data; RM 50-0, 1798-2003 data also).

This review of historic documents highlighted how appreciably the character of the river and many of its islands has changed since before the navigation era began on the MMR. Before navigation modifications took place, islands separated from the main channel by substantial side channels were frequently encountered. This resulted in split current flows, and created diverse aquatic habitat conditions. Islands were also commonly surrounded by extensive sand bars. Brauer et al. (2005) noted that prior to the construction of river training structures in 1881, large islands were scattered within the channel and were surrounded by wide side channels. After the construction of river training structures, the islands were much smaller and were located away from the river center and surrounded by much narrower side channels (Brauer 2005). Barko and Herzog (2003) document that modern side channels appear to have been drastically narrowed and experience reduced or restricted flow, often as a result of closing structures. Figure 2.4 illustrates the changes in MMR geomorphology at the Grand Isle complex, which later became known as Power’s Island, located near RM 39-35(R). The island complex remained relatively stable through 1881. However, by 1908, at least two closing structures had been constructed, as indicated by arrows, in order to reduce the width of the river as a means of securing a minimum navigation depth in the main channel. The 1928 through 2003 aerial photos show the loss of the islands and side channels as terrestrial intrusion occurred. “Powers Island” no longer meets the definition of an “island” in this study.

The considerable loss of off-channel areas in the MMR has also been described in Theiling (1995), and Theiling et al. (2000). Theiling et al. (2000) examined six secondary channels in the open river reach, using historical aerial photography for four time periods from the early 1950s to 1994, and compared acreage changes in three ‘analytical landscape units’: secondary channel, island, and main channel. The results of this analysis indicated that, though varying in degree and rate, all of the areas have shown a loss in aquatic area and a gain in terrestrial area; in some cases large areas of aquatic habitat have been lost. At low river stages, all secondary channel characteristics were lost. The extreme flooding in 1993 acted to restore lost aquatic habitat in some cases, but not to a great extent.

Natural hydraulic processes which result in sediment erosion and deposition create a dynamic environment. It was the mosaic structure and dynamic nature of the Mississippi River system that maintained its functional integrity (i.e., Ward and Stanford 1995b, Ward et al. 2001). For biological organisms, this likely meant that critical habitat required by various life stages was almost certainly available and accessible within a reasonable distance. As the river became increasingly stabilized, opportunities for habitat diversification, including natural large-scale island formation and associated off-channel habitat creation, diminished drastically.

Unquestionably, natural variability in the number of MMR islands exists over time. However, the loss of island dynamics as well as associated side channel habitat, in conjunction with the loss of access to backwater floodplain wetlands, has likely altered the community structure and composition of fish, birds, amphibians, and reptiles in the MMR ecosystem. Many species of frogs, turtles, snakes, and salamanders thrive in the aquatic and terrestrial habitats of the Mississippi River system. The reptiles and amphibians generally prefer the slow-moving channels, backwaters, isolated pools, or moist terrestrial island habitats (USACE 2004). Many species of fish, such as sunfish, killifish, and mosquitofish also require or prefer slow-moving channels, shallow waters, or backwaters (Barko and Herzog 2003).

Regression analysis successfully detected trends in the numbers of islands over time within four reaches of the Middle Mississippi River. Based on these results, I concluded that significant changes in the numbers and characteristics of islands in the Middle Mississippi River have occurred due to anthropogenic river alterations over the entire survey period. These results would not have been properly detected by simply examining 1881-2003 data alone. Early pre-navigation data were essential in revealing the impact that navigation operation and maintenance has had on the number and characteristics of Middle Mississippi River islands.

Because island dynamics integrate several ecological processes, the presence, distribution and turnover of islands may provide landscape-level indicators for assessing the condition of river corridors, as well as serving as a restoration goal. In natural or near natural river corridors, protecting the ecological processes that create and wear away islands would be a far-reaching conservation goal. In impacted river systems, returning islands to the river corridor should be a long-term rehabilitation goal.

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Figure 2.1. Location of the Middle Mississippi River, which extends from the confluence of the Missouri River to the confluence of the Ohio River.

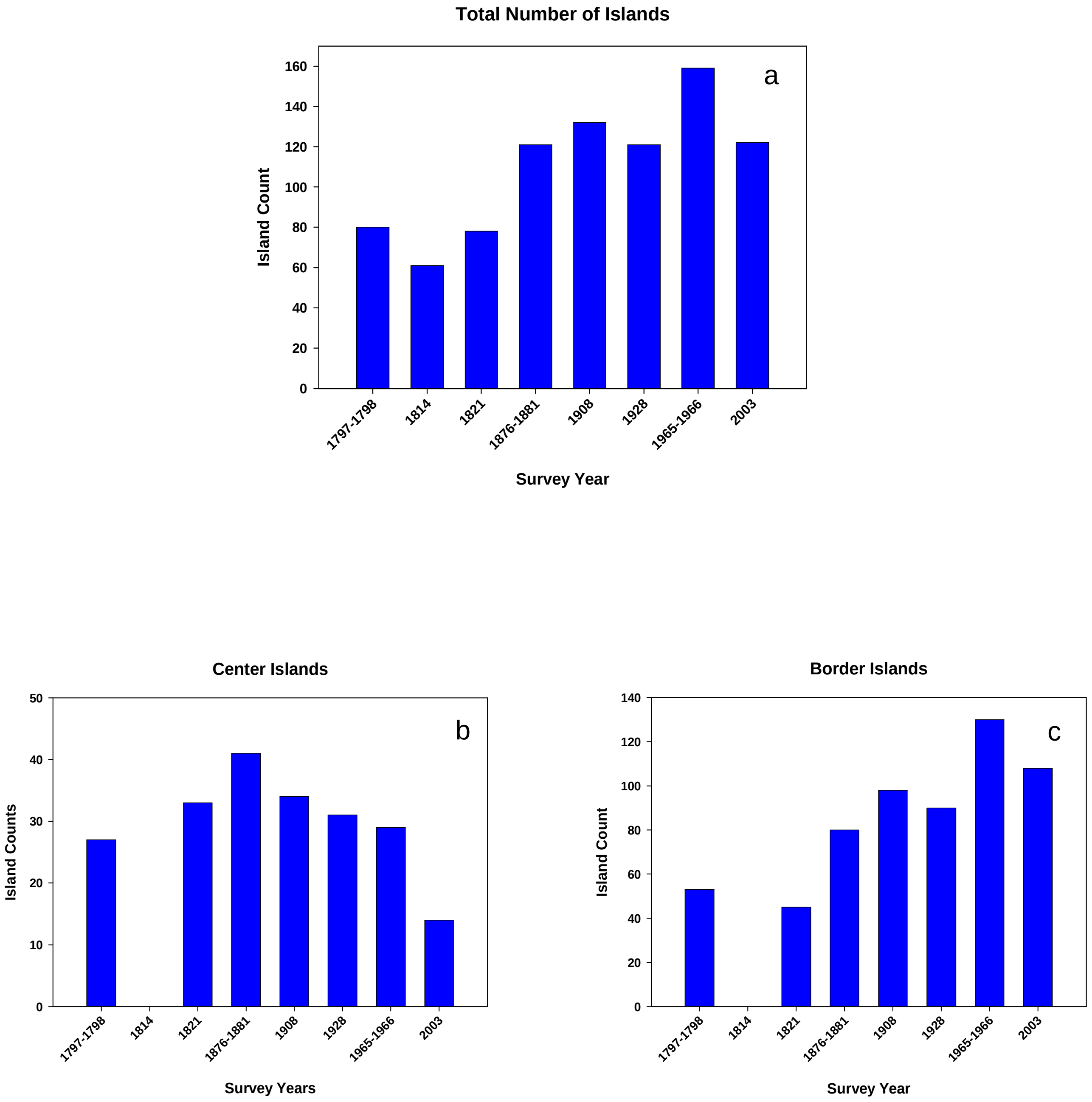


Figure 2.2. Number of total (a), center (b) and border (c) Middle Mississippi River islands present between 1797 and 2003.

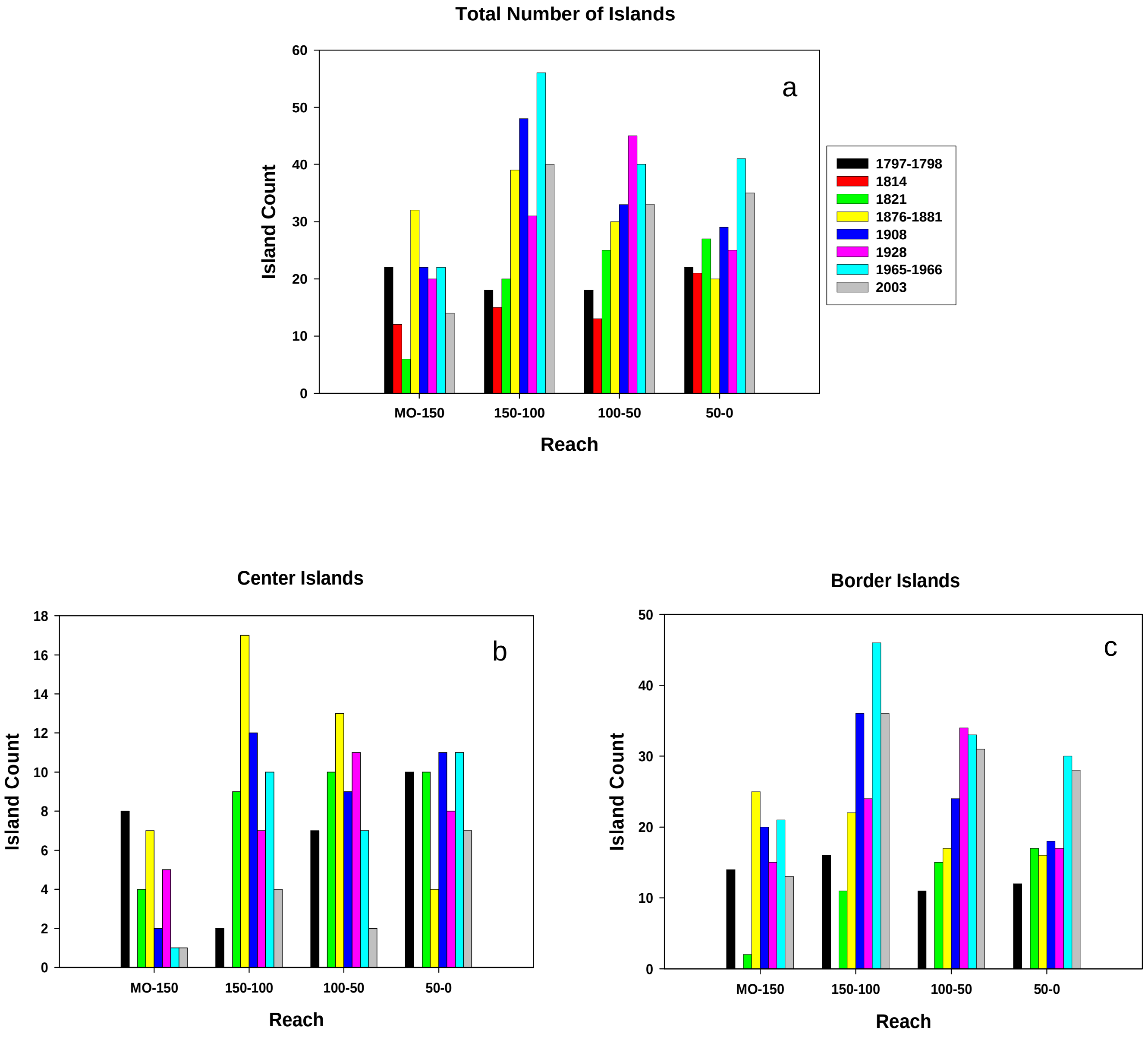
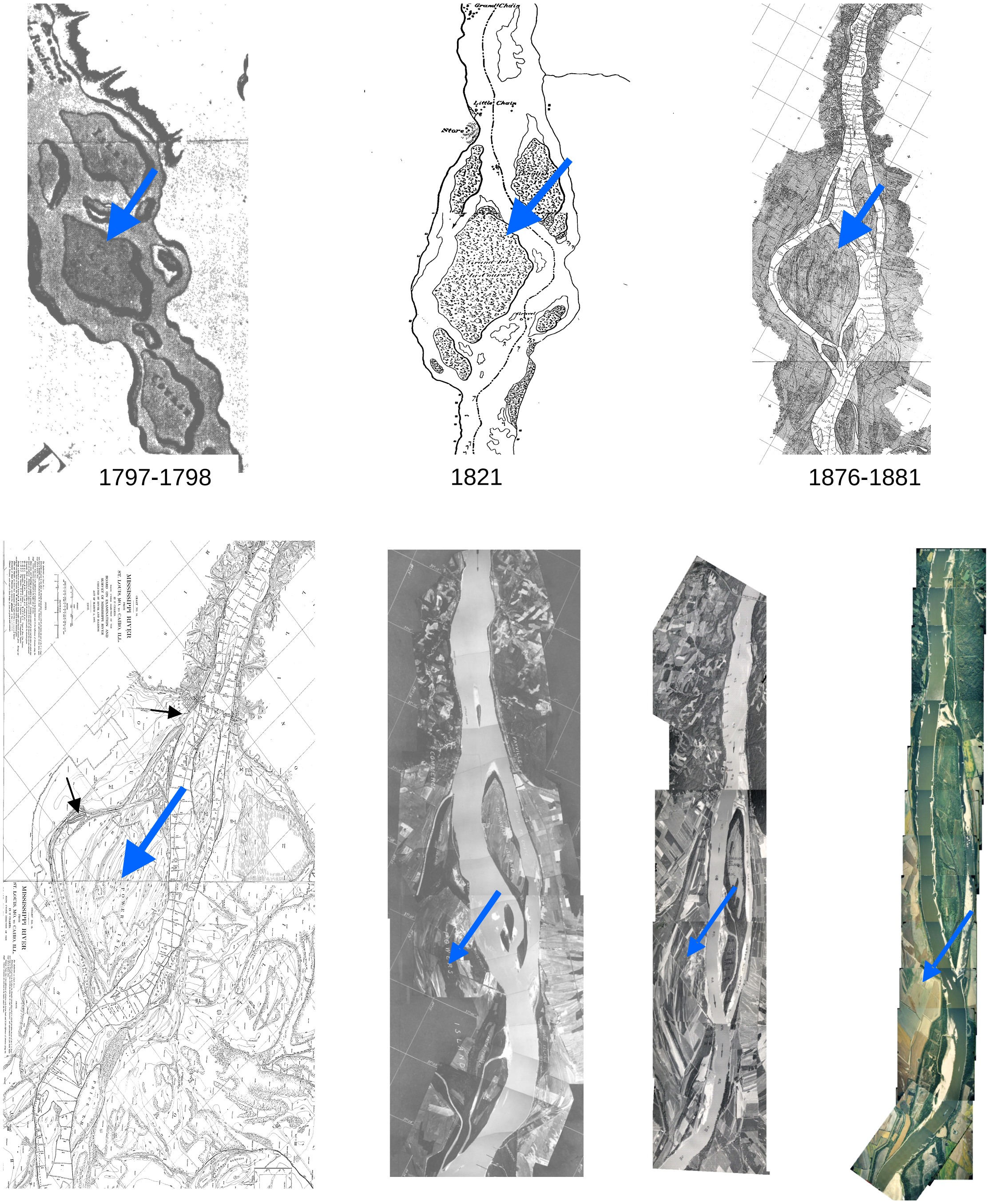


Figure 2.3. Number of total (a), center (b) and border (c) Middle Mississippi River islands present by reach between 1797 and 2003.



1797-1798

1821

1876-1881

1908

1928

1965-1966

2003

Figure 2.4. An example of change in the geomorphology of the Middle Mississippi River between 1798 and 2003**.** The focal area is Powers Island, located at approximately RM 39-35 (R). Powers Island is identified in each illustration by a blue arrow. By 1928, it no longer meets the definition of an island as defined in this study.

Table 2.1. Data sources used to compute the number of islands present between the years 1797 and 2003.

Year	Originator	Data Format	Source Scale
1797-1798	de Finiels	Survey map	N/A
1814	The Navigator	Narrative	N/A
1821*	Young et al.	Survey map	Length: 1” = 1 mile; Breadth: 2” = 1 mile
1876-1881	MRC	Survey map	1:20,000
1908*	Board Sheets	Survey map	1” = 2400’
1928	USACE	Aerial photographs	1:20,000
1965-1966	USACE	Aerial photographs	1:24,000
2003	USACE	Aerial photographs	1:12,000

*The Mississippi River above St. Louis, Missouri was not included in the data.

Table 2.2. Summary of regression analyses of Mississippi River islands data. For each time period, river reach and island type, both linear and quadratic regressions were fitted, using an alpha level of 0.1. The tabulated regression statistics are r^2 (proportion of the variance in island number explained by the regression), Residual DF (the degrees of freedom for the residual mean-square), F (ratio of regression mean-square to residual mean-square) and p (probability of obtaining an F at least as large if there were no relationship between number of islands and year). If neither model attained significance ($p<0.1$), the model type is shown as "none" and regression statistics are shown for the linear model. A quadratic model was accepted only if the t-test (not shown) indicated that the additional fitted parameter β_2 for Year^2 was significantly different from zero ($p<0.1$), otherwise the linear model was used. Fitted parameters β_0, β_1 and β_2 are shown for each selected model.

Time Period	**Reach (river miles)**	**Islands**	**Model Type**	r^2	**Residual DF**	F	p	**Intercept** β_0	**Year** β_1	**Year^2** β_2
1798 – 2003	200-150	Total	none	0.006	4	0.02	0.8837			
		Border	none	0.0034	3	0.01	0.9262			
		Main	linear	0.8667	3	19.50	0.0215	8.90	-0.0384	
	150-100	Total	linear	0.6492	6	11.10	0.0158	19.23	0.1567	
		Border	linear	0.7055	6	11.98	0.0180	12.74	0.1417	
		Main	quadratic	0.8433	5	13.46	0.0097		0.22941	-0.00110
	100-50	Total	quadratic	0.8113	5	10.75	0.0155	14.49	0.3185	-0.00106
		Border	linear	0.8195	5	22.70	0.0050	11.78	0.1148	
		Main	quadratic	0.9003	4	18.07	0.0099	7.46	0.1046	-0.00064
	50-0	Total	linear	0.5707	6	7.98	0.0302	20.72	0.0738	
		Border	linear	0.7259	5	13.24	0.0149	11.84	0.0766	
		Main	none	0.0225	5	0.12	0.7483			

1881-2003	200-150	Total	linear	0.8322	2	9.92	0.0878	41.14	-0.1307	
		Border	none	0.5208	2	2.17	0.2783			
		Main	linear	0.9037	2	18.77	0.0494	11.50	-0.0546	
	150-100	Total	none	0.0399	3	0.12	0.7472			
		Border	none	0.3904	3	1.92	0.2598			
		Main	linear	0.7396	3	8.52	0.0616	22.32	-0.0885	
	100-50	Total	none	0.7455	2	2.93	0.2545			
		Border	quadratic	0.9211	2	11.67	0.0789	-36.62	0.8542	-0.00257
		Main	linear	0.8798	3	21.96	0.0184	19.85	-0.0823	
	50-0	Total	linear	0.6539	3	5.67	0.0975	10.70	0.1386	
		Border	linear	0.7738	3	10.26	0.0492	4.88	0.1216	
		Main	none	0.0773	3	0.25	0.6506			

CHAPTER THREE

INTERIOR LEAST TERN BREEDING HABITAT IN THE MIDDLE MISSISSIPPI RIVER IN MISSOURI, USA: AN ESTIMATION OF INUNDATION RISKS ASSOCIATED WITH CONSTRUCTED BREEDING HABITAT

Introduction

The interior population of the least tern (*Sterna antillarum*) was added to the Federal list of endangered and threatened wildlife and plants in 1985 (USFWS 1985). The species faces a number of threats including habitat loss, predation, human disturbance, and inundation of nesting habitat (USFWS 1990). The interior least tern is migratory and breeds primarily on barren sandbars, sand islands, and beaches in several midwestern river systems. The nest is a shallow and inconspicuous depression in an open area (USFWS 1990). Least terns select elevated areas away from the water's edge with little to no vegetation; although, nests are often built near driftwood which is then used as shelter from the wind, blowing sand, and sun (Smith and Renken 1993). Least terns nest in colonies, and nests can be a few meters to hundreds of meters apart (USFWS 1990). The interior least tern is piscivorous, feeding in shallow waters close to their nesting sites (USFWS 1990). The terns generally prey on river fishes ≤ 5 cm in size (Dugger 1997).

The natural hydrologic fluctuations of many rivers used by least terns have been greatly altered (USFWS 1990). The greatest hazard to nesting terns appears to be natural flooding during the breeding season (Sidle et al. 1992, Smith and Renken 1993, Dugger et al. 2002). In a study of reproductive success along a 340 km stretch of the Mississippi River within Missouri, Smith and

Renken (1993) found that flooding had destroyed 41% of 92 existing nests in 1986 and 40% of 436 nests in 1989. They concluded that least terns nesting in the Mississippi River valley have a greater chance of losing nests to flooding than to predation or human disturbance. Similar flood related nesting losses have been observed on the Missouri River (Sidel et al. 1992). On the Missouri River, most riverine nesting by least terns occurs in river reaches immediately below reservoirs (Sidel et al. 1992). Untimely discharges from Missouri River dams, for example, can continue to kill eggs and chicks (USACE 1991). Dugger et al. (2002) investigated the relationship between river hydrology and interior least tern reproductive success on the lower Mississippi River (LMR) from April to July 1986 – 1993. They found a strong negative correlation between tern fledging success and July water levels, suggesting that high summer water levels decrease tern reproduction on the LMR (Dugger et al. 2002). They encountered the highest reproductive success during years with the lowest July water levels, and suggest that low water late in the nesting season may increase the availability of fish prey or least tern foraging efficiency by concentrating fish. Additionally, the benefits of low summer water levels may also be associated with an increase in the total amount of suitable sandbar nesting habitat for terns (Dugger et al. 2002). Tibbs and Galat (1998) examined the relationship between spatial and temporal availability of small fishes suitable as forage for least terns to the annual least tern reproductive period in the LMR within Missouri. They concluded that river stage dictates when the floodplain is available to fishes for spawning, which in turn influences the timing and abundance of most small fishes. Availability of sandbars, which influences when least terns begin nesting and require forage for their young, is also controlled by river stage. Thus, Tibbs and Galat (1998) indicate that the coupling of forage availability and least tern reproduction is strongly regulated by river stage. A proper understanding of natural flow fluctuations, including the timing and frequency of peak flows, is required to properly manage rivers to create or maintain least tern habitat, and avoid flow regimes that cause frequent mortality (Sidel et al. 1992). Sustained high flows during the spring through summer prevent birds from nesting because the habitat is always inundated. High flows that recede early in the nesting season will not affect many nests. Likewise, high flows late in the season will not affect many nests; and chicks could survive if some portion of the sandbar remains exposed (Sidel et al. 1992). Although high flows can result in substantial mortality, periodic scouring flows are required to maintain exposed sandbar habitat (Sidel et al. 1992).

Recent counts for the interior population exceeded the population recovery goal of 7000 (USFWS 1990) due to the large number of least terns on the lower Mississippi River (Jones 2009). Least terns in most other areas remain below recovery objectives (Kirsch and Sidle 1999). Smith and Renken (1993) suggested that adding more potential nesting colony sites above the mouth of the Ohio River would increase the likelihood of improving fledging success in the Mississippi River Valley. This would be especially true if untimely floods, originating in the Ohio River Valley, inundated nesting habitat in the lower Mississippi River, as in 1989. Smith and Stucky (1988) suggested that dredged material could be used to enhance existing colony sites and to create new island habitat.

In order to enhance or develop least tern breeding habitat, an understanding of water elevations at the enhancement site is required for the breeding season. Least terns require a minimum of 50 consecutive days of sandbar exposure to complete courtship, lay eggs, incubate a clutch (21 days), and raise young to fledging (approximately 21 days) (Smith and Renken 1991, 1993). Although 50 days is the minimum number of days required to fledge a brood, Smith and Renken (1991) found that least terns were more likely to use sites that were continuously exposed for at least 100 days during the period 15 May - 31 August. This study was conducted to determine the inundation risk associated with constructing least tern breeding habitat at 5-mile increments along the Middle Mississippi River.

Methods and Materials

Geometric Model – The UNET model was developed for the reach of the Middle Mississippi River from St. Louis, Missouri, (RM 179.6) to the mouth of the Ohio River near Cairo, Illinois (RM 0.0) (Figure 3.1). The upstream boundary was a flow hydrograph at St. Louis, and the downstream boundary was a stage hydrograph from the Cairo gage on the Ohio River. Three major tributary inflows entering the Mississippi River were also modeled, including the Meramec River from the Eureka gage downstream; the Kaskaskia River from the Venedy Station gage downstream; and the Big Muddy River from the Murphyboro gage downstream (Figure 3.1).

Hydrology – Hydraulic modeling was performed using UNET, a one-dimensional unsteady flow hydraulic model (Barkau 1995). The 70 year period of record from 1940 to 2009 was simulated. From 1940 to 1979, the flow input used was the synthetic, regulated period of record from the Mississippi River Flood Flow Frequency Study (USACE 2004b). From 1979 to 2009, the observed flow record was used. The simulation assumed all reservoirs were in place. The U.S. Geological Service (USGS) mean daily flow was converted into a sequence of daily histograms. The model routed the rectangular hydrograph. The output hydrographs were reconstituted as mean daily flow in and out.

Drainage area accounting between St. Louis and Thebes, Illinois, was used (Table 3.1). Flow from the 5,860 square miles of ungaged drainage was added into the simulation. Ungaged inflow between Thebes and Cairo was ignored. Ungaged inflow from 1940 through 2009 was computed using UNET's ungaged inflow optimization procedure. The procedure computed the ungaged inflow that exactly reproduced the flow at the Chester, Illinois, and Thebes USGS gages. Water loss to storage and possibly groundwater resulted in negative inflow. Any negative flow was contributed to two assumed aquifers between St. Louis and Chester, and between Chester and Thebes. The aquifers discharge back into the river but at a slower rate, as modeled by Darcy's Equation:

$$Q = \frac{-\kappa A}{\mu} \frac{(P_b - P_a)}{L}$$

where Q = total discharge, κ= permeability of the medium, A = cross-sectional area to flow, $P_b - P_a$ = pressure drop, μ = dynamic viscosity, and L = the length the pressure drop is taking place over.

Calibration – Two sets of gages, primary and secondary, were used. Primary gages include USGS gages and U.S. Army Corps of Engineers (USACE) stage gages; the latter are located at USACE structures. The difference in accuracy between the gages is due to the number of visits by streamgagers. During a visit, the Data Collection Platform (DCP) is recalibrated to match the manual, outside reading. The USGS gages along the Mississippi are visited fortnightly when the river flow is measured. The USACE gages at the structures are used for regulating the structures. These gages are constantly watched and any discrepancy is immediately corrected. Secondary stage gages are operated by the USACE at intermediate points between the primary gages along the Mississippi River. These gages are visited only when maintenance is required; the interval may be months.

The calibration period extended from 1 January 2007 through 31 December 2008. Since the river is constantly changing, calibration is done to the most recent period for which the stage data at the secondary gages are available (primary gage records are always available). For 2009, most of the data at secondary gages were either missing or erroneous.

The Mississippi River has two calibration regimes: A winter, cold water, regime and a summer, warm water, regime. During the winter, the water is more viscous and erosive; the dunes (bed forms) are smaller and shorter. During the summer the water is less viscous; the dunes are larger and longer. The transition between winter and summer occurs between late April and mid-May. In the fall, the transition between summer and winter regimes, slowly occurs between October and early December. On average, the transitions occur between 1 May and 15 May and between 15 October and 15 December. Stages are about 1 to 2 feet (30.5 – 61.0 cm) higher during the summer, which translates into a difference in Manning's coefficient of roughness from 0.028 in the summer (the base period) to 0.026 in the winter.

Elevation – This portion of the analysis determined the annual cumulative elevation probability curves for 50, 75, and 100 day durations between 15 April and 31 August. The river was simulated for regulated conditions from 1940 through 2009, a 70 year period. Period of record hydrographs were output at five mile intervals from St. Louis, Missouri, to Cairo, Illinois (37 locations). For each year, the lowest maximum water surface elevation for a duration of “n” days was determined. Inundation/exposure probability tables for each location were produced.

Results

The results are provided in the form of probability tables. These tables present the probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year. Tables are provided for every five (5) river miles between St. Louis, Missouri, and Cairo, Illinois (Tables 3.2-3.38).

Discussion

A reasonable initial approach in meeting recovery plan goals for least terns (USFWS 1990), is to provide additional suitable colony sites (Smith and Renken 1991) which are protected from predation, human disturbance, and inundation during the breeding season. The results of the UNET model provide valuable information in designing islands to an elevation which should allow them be continuously exposed for at least 100 days during the 15 May - 31 August breeding season.

In the Mississippi River, much of the historic sand island and sandbar habitat has been eliminated, and the breeding range in Missouri is now restricted to an area south of Cape Girardeau (Smith 1985). Currently, reoccurring nesting in the MMR is known only at Marquette Island (RM 50.5), Bumgard Island (RM 30), and Brown's Bar (RM 24.5-23.5) (USACE 2004a, Jones 2009). Because suitable breeding habitat is currently limited in the MMR, least terns are forced to nest on fewer riverine sites, making them more vulnerable than if they were dispersed among more locations (Smith and Renken 1991).

Loss of islands and associated secondary channels results, in part, from the construction of closing structures and unnotched wing dikes, followed by terrestrial intrusion (Shields 1995, USACE 2004a). In the MMR, wing dikes are prominent features of the riverine environment. They are used to concentrate flow in the main channel in order to reduce the need for dredging. Like dikes, side channel closing structures were traditionally used to improve navigation by diverting flow into the main channel (USACE undated publication). Both unnotched dikes and closing structures tend to lead to sediment build-up, resulting in the conversion of aquatic habitat to terrestrial habitat (Shields 1995). Notching wing dikes appears to be a viable method for maintaining terrestrial and aquatic habitats in channelized large rivers, and conserving small fish biodiversity important to survival of the interior least tern (Tibbs and Galat 1998). Dike notching causes hydraulic scour to create holes downstream of the notches, with the scoured material being deposited further downstream. Under the proper conditions, sandbars with an isolating side channel will develop. Notching a closure structure tends to keep the side channels from being filled with sedimentation (USACE undated publication), and maintains the isolation of the sandbar or vegetated island.

Smith and Renken (1991) suggest the creation or restoration of sandbars in the Mississippi River through construction of chevron dikes. Chevrons are dike structures designed as a blunt nosed, arch shape (Davinroy et al. 1996). They are constructed parallel to flow and, like regular dikes, utilize the energy of the river to redistribute flow and sediment. They are usually placed in close proximity to the river bank to allow flow separation and create both channel deepening, side channel development, and isolated sandbar formation. Placement of dredge material to the appropriate height downstream of chevrons would create suitable least tern nesting habitat.

As sandbars accrete to the shore or become vegetated, nesting habitat will be lost and the vulnerability of the population to nest predation and human disturbance will increase (Smith and Renken 1993). Thus, periodic high flows are necessary to scour vegetation from sandbars creating barren islands of sand and gravel, ideal nesting habitat for least terns (Currier et al. 1985, USFWS 1990, Sidel et al. 1992). Sidel et al. (1992) found that on the Platte River in Nebraska, high water of a scouring magnitude can be expected to occur about once every nine years. Although increased nesting mortality may occur, high water events provide opportunities to create and renew least tern nesting habitat.

River level fluctuations also influence the degree of predation a colony site experiences (Szell and Woodrey 2003). Low river levels increase the potential for mammalian predators to gain access to the colony sites via dikes or accreted shoreline (Szell and Woodrey 2003). Such a concern is important, given many sandbars are found within diked reaches of the river (Szell and Woodrey 2003). Nest predation by coyotes, raccoons, domestic dogs, American crows (Smith and Renken 1991), hawks (Latka et al 1993), and barred owls (Szell and Woodrey 2003) takes a heavy toll on least tern colonies. All-terrain vehicles (ATVs) also present a hazard to nesting terns (Smith and Renken 1993). ATVs are likely to gain access to isolated habitat when river levels are low and islands accreted to the shore. Harassment by humans recreating on islands has resulted in the death of juvenile and adult least terns (Smith and Renken 1993).

For the population of interior least terns in the Mississippi River valley to remain at or above its current size, several management actions should be encouraged (Smith and Renken 1993). Most important, nesting habitat of appropriate elevation needs to be available. Smith and Renken (1991) believe the sites used by least terns are the tallest sandbars and sand islands because they are the first sites to be exposed above the water after spring floods. It appears that sand islands and sandbars continuously exposed for at least 100 days are important to terns not only because they are the first sites to be exposed in the spring, but they are also available to nesting terns in most years. There is a reproductive advantage for least terns to nest as soon as suitable sites are available (Smith and Renken 1991, Schwalbach et al 1993). Early nesters are usually older, more successful breeding adults (Massey and Atwood 1981, Schwalbach et al 1993). Smith and Renken (1991) noted that least terns nesting early in the season experience greater daily nest survival rates (in 1987, 0.99 vs. 0.94, early vs. late nesters, respectively, $Z = 6.1$, $P < 0.001$) and produce more young (in 1987,2.2 vs. 0.5 chicks/pair, early vs. late nesters, respectively) than late nesters. Gauthier (1989) also observed that early nesters experience greater nesting success. In many species of birds, young hatched early in the season often experience greater survival rates than later hatched young (Arcese and Smith 1985, Dow and Fredga 1984, Perrins 1970) and are more likely to be recruited into the breeding population than late-hatched young (Cooke et al. 1984, Gauthier 1989).

Additionally, dikes should be modified to ensure river flows between the sandbar and shore (Renken and Smith 1993). This would reduce travel corridors for mammalian predators and provide slack-water areas rich in fish prey (Tibbs and Galat 1998).

Control of vegetative encroachment on higher sandbar and island sites is also essential. Besides river scouring, Latka et al. (1993) describe various methods of vegetation removal, including use of herbicides, mechanical, burning, removal by hand, and island build up by bulldozers or deposition of appropriate dredge material. It is important that the substrate not be too silty in order to avoid egg stick syndrome. A deposition with over 20% shells could interfere with nest construction (Kotliar 1984). Furthermore, on beaches devoid of vegetation, the provision of driftwood can provide important shelter from a storm (Haddon and Knight 1983), as well as protection from the sun and predators. Adding debris may increase the attractiveness of a sandbar to breeding terns.

Ultimately, potential least tern colony sites should be posted as seasonal refuges. Signs educating the public about the importance of sandbars and minimizing disturbance during the nesting season should be placed at public boat ramps (Szell and Woodrey 2003), as well as on sand islands. As public use of sandbar habitat along the Mississippi River increases, education (Szell and Woodrey 2003), including use of informative websites, will play an increasingly important role in minimizing human disturbance on nesting least terns. In Missouri, the combination of posting seasonal refuges and a public information campaign has reduced the amount of human disturbance to nesting terns (Smith and Renken 1993).

The least tern, like most other long-lived colonial water birds, is adapted to patchy, dynamic environments characteristic of large alluvial streams such as the Mississippi River (USACE 1999). The overall goal of this study is to promote the development of suitable sand islands throughout the Middle Mississippi River, thereby allowing interior least terns to expand beyond their present nesting locations. Scattered nesting sites would reduce the overall threat from inundation, predation, and human disturbance. With reduced risks, overall nesting success, and population numbers, should improve.

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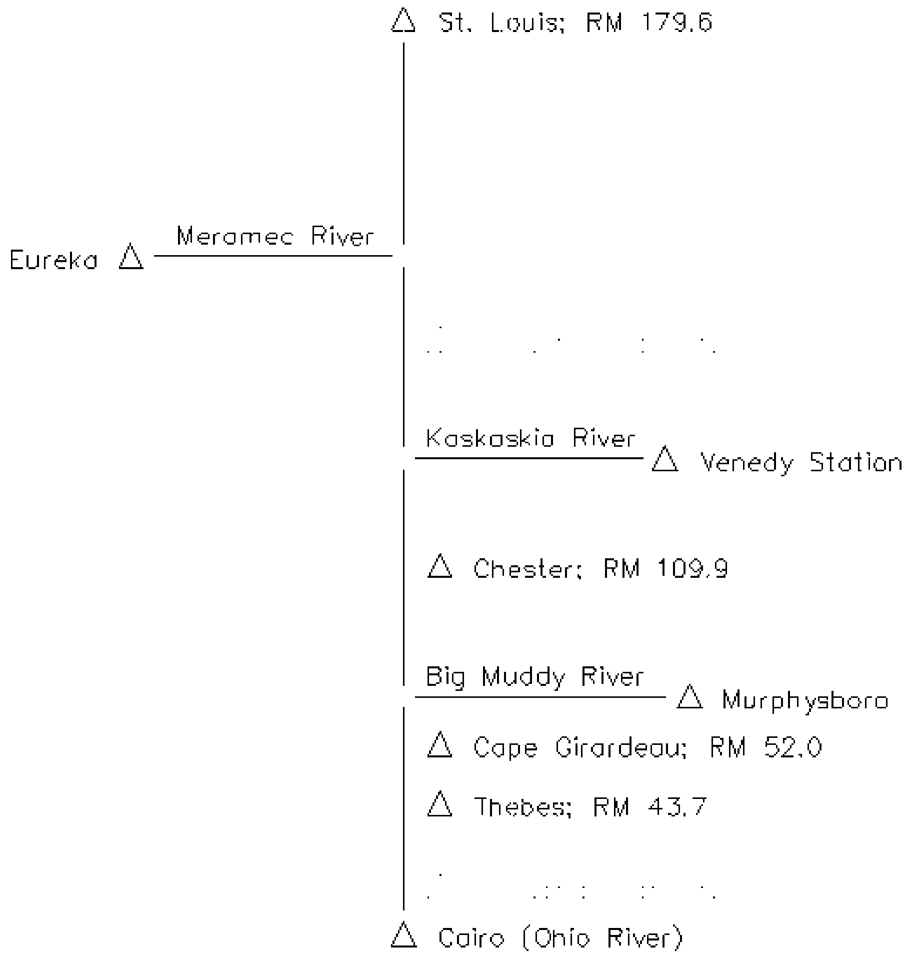

Figure 3.1. The Middle Mississippi River UNET model, extending from St. Louis, Missouri, to Cairo, Illinois.

Table 3.1. Drainage area accounting from St. Louis, Missouri, to Thebes, Illinois.

River	Station	Drainage Area	Ungaged Area
Mississippi River	St. Louis, MO	697,000	
Meramec River	Eureka, MO	3,788	
Kaskaskia River	Venedy Station, IL	4,393	
Mississippi River	Chester, IL	708,600	3,419
Big Muddy River	Murphysboro, IL	2,159	
Mississippi River	Thebes, IL	713,200	2,441

Table 3.2. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 179.6.

St. Louis RM 179.6 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	4.6	7.0	12.1
0.90	7.6	12.8	13.8
0.85	9.4	13.4	16.6
0.80	11.0	15.5	17.4
0.75	12.1	16.6	19.0
0.70	13.1	17.4	19.8
0.65	14.0	18.2	21.4
0.60	14.2	19.5	21.6
0.55	14.7	20.3	22.8
0.50	15.6	21.4	23.6
0.45	16.1	22.2	24.3
0.40	16.9	23.1	26.3
0.35	18.1	25.3	28.2
0.30	18.6	26.4	29.3
0.25	19.2	28.3	30.0
0.20	21.3	29.2	30.9
0.15	24.1	29.6	34.2
0.10	26.1	30.3	34.9
0.05	27.8	33.4	38.0

Table 3.3. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 175.

RM 175 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	382.4	384.8	389.8
0.90	385.3	390.4	391.4
0.85	387.0	391.1	394.1
0.80	388.7	393.1	394.9
0.75	389.7	394.2	396.5
0.70	390.7	394.9	397.2
0.65	391.6	395.7	398.8
0.60	391.8	397.0	399.1
0.55	392.3	397.9	400.2
0.50	393.1	398.9	401.1
0.45	393.6	399.6	401.6
0.40	394.4	400.4	403.6
0.35	395.6	402.6	405.5
0.30	396.2	403.6	406.5
0.25	396.6	405.5	407.3
0.20	398.7	406.4	408.2
0.15	401.4	406.9	411.3
0.10	403.4	407.6	412.2
0.05	405.1	410.6	415.2

Table 3.4. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 170.

RM 170 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	379.1	381.7	386.8
0.90	382.3	387.3	388.3
0.85	383.8	388.0	390.9
0.80	385.6	390.1	391.8
0.75	386.5	391.1	393.4
0.70	387.6	391.8	394.1
0.65	388.6	392.5	395.7
0.60	388.8	393.9	396.0
0.55	389.2	395.0	397.0
0.50	389.9	395.8	398.0
0.45	390.5	396.5	398.5
0.40	391.3	397.3	400.6
0.35	392.5	399.5	402.6
0.30	393.2	400.5	403.5
0.25	393.5	402.4	404.3
0.20	395.6	403.4	405.3
0.15	398.3	404.1	408.2
0.10	400.4	404.6	409.7
0.05	402.1	407.6	412.3

Table 3.5. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 165.

RM 165 Cumulative Probability	Minimum Elevation		
	50 Days	**75 Days**	**100 Days**
0.95	375.9	378.6	383.7
0.90	379.3	384.1	385.3
0.85	380.8	385.0	387.9
0.80	382.4	387.3	388.9
0.75	383.3	388.2	390.4
0.70	384.5	388.9	391.3
0.65	385.5	389.7	392.9
0.60	385.8	391.1	393.4
0.55	386.3	392.0	394.5
0.50	386.9	393.0	395.2
0.45	387.4	393.7	395.9
0.40	388.3	394.5	397.9
0.35	389.6	396.9	400.0
0.30	390.3	397.8	400.8
0.25	390.8	399.7	401.7
0.20	392.8	400.8	402.7
0.15	395.6	401.5	405.4
0.10	397.7	401.9	407.4
0.05	399.5	405.0	409.7

Table 3.6. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 160.

RM 160 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	373.2	376.1	381.3
0.90	376.9	381.7	383.0
0.85	378.4	382.7	385.6
0.80	380.0	385.1	386.6
0.75	380.9	386.0	388.2
0.70	382.1	386.6	389.1
0.65	383.3	387.5	390.6
0.60	383.5	389.0	391.2
0.55	384.0	389.8	392.3
0.50	384.6	390.8	393.1
0.45	385.2	391.6	393.7
0.40	386.2	392.2	395.7
0.35	387.4	394.8	397.8
0.30	388.0	395.5	398.6
0.25	388.7	397.4	399.4
0.20	390.6	398.5	400.5
0.15	393.3	399.3	403.1
0.10	395.5	399.6	405.4
0.05	397.3	402.7	407.5

Table 3.7. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 155.

RM 155 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	370.9	373.9	379.2
0.90	374.6	379.5	380.8
0.85	376.2	380.5	383.3
0.80	377.8	382.9	384.4
0.75	378.7	383.8	385.9
0.70	379.9	384.3	386.8
0.65	381.1	385.2	388.3
0.60	381.3	386.7	388.9
0.55	381.8	387.4	389.9
0.50	382.4	388.5	390.7
0.45	382.9	389.1	391.3
0.40	383.9	389.8	393.2
0.35	385.2	392.4	395.3
0.30	385.7	393.1	396.1
0.25	386.5	394.8	396.9
0.20	388.3	396.0	398.0
0.15	390.9	396.7	400.4
0.10	393.0	397.0	402.8
0.05	394.9	400.1	404.8

Table 3.8. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 150.

RM 150 Cumulative Probability	Minimum Elevation		
	50 Days	**75 Days**	**100 Days**
0.95	369.4	372.3	377.5
0.90	372.9	377.8	379.1
0.85	374.6	378.8	381.4
0.80	376.1	381.1	382.5
0.75	377.1	382.0	383.9
0.70	378.2	382.5	384.9
0.65	379.4	383.4	386.3
0.60	379.5	384.8	386.9
0.55	380.1	385.5	387.9
0.50	380.7	386.5	388.7
0.45	381.1	387.2	389.2
0.40	382.2	387.7	391.2
0.35	383.3	390.4	393.1
0.30	383.8	391.1	393.9
0.25	384.6	392.7	394.7
0.20	386.3	393.8	395.8
0.15	388.8	394.5	398.1
0.10	390.9	394.7	400.5
0.05	392.9	397.9	402.5

Table 3.9. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 145.

RM 145 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	366.4	369.4	374.7
0.90	369.9	374.9	376.3
0.85	371.8	376.0	378.7
0.80	373.3	378.3	379.8
0.75	374.2	379.4	381.1
0.70	375.3	379.7	382.2
0.65	376.5	380.8	383.7
0.60	376.7	382.1	384.1
0.55	377.4	382.8	385.2
0.50	377.9	383.8	386.0
0.45	378.4	384.5	386.5
0.40	379.5	385.0	388.5
0.35	380.6	387.9	390.5
0.30	381.0	388.5	391.2
0.25	381.9	390.0	392.0
0.20	383.7	391.2	393.3
0.15	386.1	391.9	395.4
0.10	388.3	392.1	397.8
0.05	390.3	395.3	399.9

Table 3.10. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 140.

RM 140 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	364.6	367.6	372.6
0.90	367.9	372.8	374.2
0.85	369.8	373.9	376.4
0.80	371.3	376.1	377.5
0.75	372.1	377.2	378.8
0.70	373.2	377.4	379.8
0.65	374.4	378.6	381.3
0.60	374.5	379.8	381.8
0.55	375.2	380.5	382.8
0.50	375.7	381.5	383.6
0.45	376.3	382.2	384.2
0.40	377.3	382.6	386.1
0.35	378.4	385.5	388.0
0.30	378.7	386.2	388.7
0.25	379.7	387.5	389.6
0.20	381.4	388.7	390.9
0.15	383.7	389.3	392.8
0.10	385.9	389.6	395.2
0.05	388.0	392.8	397.4

Table 3.11. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 135.

RM 135 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	362.3	365.3	370.2
0.90	365.6	370.4	371.8
0.85	367.6	371.6	374.0
0.80	368.9	373.6	375.0
0.75	369.7	374.7	376.2
0.70	370.8	374.9	377.3
0.65	371.9	376.2	378.8
0.60	372.1	377.3	379.2
0.55	372.8	378.0	380.3
0.50	373.3	379.0	381.1
0.45	373.9	379.6	381.8
0.40	374.9	380.0	383.6
0.35	375.9	383.1	385.4
0.30	376.2	383.6	386.1
0.25	377.2	384.9	387.2
0.20	378.9	386.1	388.4
0.15	381.1	386.7	390.3
0.10	383.3	387.1	392.8
0.05	385.5	390.3	394.9

Table 3.12. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 130.

RM 130 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	359.9	362.8	367.6
0.90	363.1	367.7	369.3
0.85	364.9	369.0	371.4
0.80	366.4	371.0	372.5
0.75	367.2	372.3	373.8
0.70	368.2	372.5	374.9
0.65	369.3	373.7	376.4
0.60	369.5	374.9	376.8
0.55	370.3	375.6	378.0
0.50	370.7	376.7	378.8
0.45	371.4	377.3	379.5
0.40	372.4	377.6	381.2
0.35	373.4	380.8	383.2
0.30	373.7	381.3	383.9
0.25	374.8	382.6	385.1
0.20	376.5	383.9	386.2
0.15	378.7	384.5	388.1
0.10	381.0	384.9	390.8
0.05	383.3	388.2	392.8

Table 3.13. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 125.

RM 125 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	356.8	359.7	364.6
0.90	360.1	364.6	366.3
0.85	361.9	366.1	368.4
0.80	363.3	367.9	369.4
0.75	364.2	369.2	370.7
0.70	365.1	369.5	371.7
0.65	366.2	370.5	373.2
0.60	366.5	371.7	373.6
0.55	367.3	372.5	374.9
0.50	367.7	373.5	375.6
0.45	368.3	374.1	376.3
0.40	369.4	374.4	378.0
0.35	370.3	377.6	379.9
0.30	370.6	378.1	380.6
0.25	371.7	379.4	381.9
0.20	373.4	380.7	382.9
0.15	375.4	381.2	384.8
0.10	377.8	381.7	387.5
0.05	380.2	384.9	389.5

Table 3.14. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 120.

RM 120 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	353.0	356.1	361.1
0.90	356.5	361.2	363.0
0.85	358.3	362.7	364.9
0.80	359.9	364.3	366.1
0.75	360.8	365.7	367.4
0.70	361.6	366.3	368.4
0.65	362.7	367.1	369.9
0.60	363.0	368.4	370.4
0.55	363.8	369.2	371.6
0.50	364.2	370.3	372.3
0.45	364.8	370.9	373.1
0.40	366.1	371.1	374.8
0.35	366.9	374.3	376.8
0.30	367.2	375.0	377.6
0.25	368.4	376.1	378.8
0.20	370.1	377.7	379.9
0.15	372.0	378.0	381.8
0.10	374.5	378.6	384.6
0.05	377.1	381.9	386.6

Table 3.15. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 115.

RM 115 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	350.8	353.8	358.6
0.90	354.2	358.7	360.4
0.85	355.9	360.2	362.2
0.80	357.4	361.6	363.3
0.75	358.3	362.9	364.6
0.70	359.1	363.6	365.6
0.65	360.1	364.3	367.0
0.60	360.5	365.6	367.5
0.55	361.3	366.4	368.7
0.50	361.6	367.5	369.5
0.45	362.2	368.0	370.1
0.40	363.4	368.3	371.8
0.35	364.1	371.3	373.7
0.30	364.5	372.0	374.6
0.25	365.6	373.0	375.7
0.20	367.2	374.6	376.9
0.15	369.0	374.9	378.8
0.10	371.5	375.6	381.6
0.05	374.1	378.9	383.6

Table 3.16. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 110.

RM 110 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	347.9	350.9	355.7
0.90	351.3	355.8	357.4
0.85	353.0	357.2	359.3
0.80	354.6	358.6	360.3
0.75	355.4	359.8	361.6
0.70	356.1	360.6	362.7
0.65	357.1	361.3	363.9
0.60	357.6	362.5	364.5
0.55	358.3	363.3	365.7
0.50	358.7	364.5	366.5
0.45	359.3	364.9	367.1
0.40	360.4	365.3	368.8
0.35	361.0	368.2	370.7
0.30	361.5	369.1	371.6
0.25	362.7	370.0	372.7
0.20	364.2	371.6	374.2
0.15	366.0	372.0	376.0
0.10	368.5	372.8	379.0
0.05	371.1	376.0	380.9

Table 3.17. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 105.

RM 105 Cumulative Probability	Minimum Elevation		
	50 Days	**75 Days**	**100 Days**
0.95	346.0	349.0	353.7
0.90	349.5	353.8	355.3
0.85	351.1	355.2	357.1
0.80	352.7	356.5	358.2
0.75	353.5	357.6	359.4
0.70	354.1	358.4	360.4
0.65	355.1	359.1	361.6
0.60	355.5	360.2	362.2
0.55	356.2	361.0	363.3
0.50	356.6	362.2	364.2
0.45	357.2	362.5	364.8
0.40	358.2	363.1	366.4
0.35	358.8	365.8	368.3
0.30	359.3	366.8	369.2
0.25	360.4	367.6	370.2
0.20	362.0	369.2	371.8
0.15	363.6	369.7	373.4
0.10	366.0	370.2	376.3
0.05	368.6	373.4	378.1

Table 3.18. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 100.

RM 100 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	342.8	345.9	350.6
0.90	346.4	350.6	352.2
0.85	347.8	352.0	354.0
0.80	349.6	353.3	355.1
0.75	350.3	354.4	356.4
0.70	351.0	355.4	357.5
0.65	351.9	356.1	358.6
0.60	352.4	357.2	359.2
0.55	353.1	358.1	360.3
0.50	353.4	359.3	361.4
0.45	354.1	359.6	361.9
0.40	355.2	360.3	363.6
0.35	355.8	362.9	365.6
0.30	356.3	364.1	366.5
0.25	357.4	364.9	367.6
0.20	359.2	366.5	369.3
0.15	360.7	367.1	370.8
0.10	363.2	367.5	373.8
0.05	365.9	370.8	375.5

Table 3.19. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 95.

RM 95 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	338.3	341.8	346.9
0.90	342.4	347.0	348.5
0.85	343.9	348.4	350.6
0.80	345.9	349.9	351.8
0.75	346.6	351.0	353.1
0.70	347.3	352.1	354.3
0.65	348.4	352.8	355.3
0.60	348.9	353.9	356.0
0.55	349.7	354.9	357.1
0.50	349.9	356.0	358.3
0.45	350.8	356.5	358.8
0.40	352.0	357.1	360.7
0.35	352.5	359.9	362.8
0.30	353.1	361.2	363.6
0.25	354.1	362.1	364.7
0.20	356.1	363.6	366.6
0.15	357.6	364.2	368.1
0.10	360.2	364.8	371.2
0.05	362.9	368.2	372.9

Table 3.20. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 90.

RM 90 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	336.5	339.9	344.8
0.90	340.5	344.9	346.4
0.85	342.0	346.3	348.5
0.80	343.8	347.8	349.6
0.75	344.6	348.8	350.9
0.70	345.2	349.9	352.1
0.65	346.3	350.6	353.0
0.60	346.9	351.6	353.6
0.55	347.5	352.6	354.7
0.50	347.7	353.6	355.9
0.45	348.7	354.1	356.5
0.40	349.7	354.7	358.2
0.35	350.3	357.3	360.5
0.30	350.9	358.7	361.1
0.25	351.8	359.6	362.2
0.20	353.8	361.1	364.1
0.15	355.2	361.7	365.6
0.10	357.7	362.3	368.6
0.05	360.4	365.7	370.3

Table 3.21. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 85.

RM 85 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	334.4	337.9	342.7
0.90	338.5	342.7	344.2
0.85	339.9	344.1	346.3
0.80	341.6	345.6	347.5
0.75	342.4	346.7	348.7
0.70	343.1	347.7	349.8
0.65	344.1	348.4	350.6
0.60	344.8	349.3	351.3
0.55	345.3	350.3	352.3
0.50	345.6	351.2	353.5
0.45	346.6	351.7	354.2
0.40	347.4	352.3	355.8
0.35	348.1	354.9	358.1
0.30	348.7	356.2	358.6
0.25	349.6	357.2	359.6
0.20	351.6	358.6	361.5
0.15	352.8	359.1	363.0
0.10	355.4	359.7	365.8
0.05	357.9	363.0	367.5

Table 3.22. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 80.

RM 80 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	332.2	335.7	340.3
0.90	336.3	340.2	341.7
0.85	337.6	341.7	343.9
0.80	339.1	343.1	345.0
0.75	340.0	344.2	346.1
0.70	340.6	345.2	347.2
0.65	341.7	345.8	347.9
0.60	342.4	346.7	348.6
0.55	342.9	347.7	349.6
0.50	343.1	348.4	350.7
0.45	344.1	349.0	351.5
0.40	344.9	349.7	353.0
0.35	345.6	352.0	355.4
0.30	346.1	353.4	355.6
0.25	347.0	354.5	356.6
0.20	349.0	355.6	358.5
0.15	350.0	356.1	359.8
0.10	352.7	356.7	362.4
0.05	355.0	359.9	364.0

Table 3.23. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 75.

RM 75 Cumulative Probability	Minimum Elevation		
	50 Days	**75 Days**	**100 Days**
0.95	329.4	332.9	337.6
0.90	333.5	337.5	339.2
0.85	334.8	339.1	341.3
0.80	336.3	340.6	342.5
0.75	337.2	341.6	343.7
0.70	338.0	342.8	344.7
0.65	339.1	343.3	345.4
0.60	339.9	344.2	346.1
0.55	340.4	345.2	347.1
0.50	340.5	345.9	348.2
0.45	341.6	346.5	349.0
0.40	342.5	347.2	350.6
0.35	343.1	349.5	352.9
0.30	343.7	350.8	353.0
0.25	344.6	352.0	354.0
0.20	346.5	353.0	356.0
0.15	347.5	353.6	357.3
0.10	350.2	354.2	359.7
0.05	352.4	357.4	361.2

Table 3.24. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 70.

RM 70 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	326.8	330.2	334.8
0.90	330.8	334.7	336.4
0.85	332.1	336.2	338.5
0.80	333.6	337.7	339.7
0.75	334.4	338.8	340.8
0.70	335.2	340.0	341.9
0.65	336.3	340.5	342.6
0.60	337.1	341.4	343.2
0.55	337.6	342.3	344.2
0.50	337.7	342.9	345.4
0.45	338.8	343.7	346.2
0.40	339.7	344.3	347.6
0.35	340.3	346.5	349.8
0.30	340.9	347.9	350.1
0.25	341.8	349.1	351.0
0.20	343.6	350.0	353.0
0.15	344.6	350.5	354.3
0.10	347.4	351.2	356.6
0.05	349.3	354.3	358.1

Table 3.25. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 65.

RM 65 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	323.7	327.2	331.7
0.90	327.7	331.6	333.3
0.85	328.9	333.1	335.2
0.80	330.5	334.6	336.4
0.75	331.2	335.5	337.5
0.70	332.1	336.7	338.5
0.65	333.1	337.3	339.3
0.60	333.9	338.1	339.8
0.55	334.4	339.0	340.9
0.50	334.6	339.5	342.0
0.45	335.6	340.3	342.9
0.40	336.4	340.9	344.3
0.35	337.0	343.1	346.5
0.30	337.6	344.6	346.9
0.25	338.5	345.8	347.6
0.20	340.2	346.6	349.6
0.15	341.2	347.1	351.0
0.10	344.1	347.9	353.0
0.05	345.8	350.9	354.5

Table 3.26. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 60.

RM 60 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	321.9	325.2	329.4
0.90	325.6	329.3	331.0
0.85	326.7	330.7	332.7
0.80	328.3	332.3	333.9
0.75	328.9	333.1	334.9
0.70	329.7	334.1	335.9
0.65	330.7	334.7	336.7
0.60	331.5	335.6	337.3
0.55	332.1	336.4	338.2
0.50	332.3	336.9	339.3
0.45	333.2	337.7	340.3
0.40	333.9	338.3	341.6
0.35	334.4	340.4	343.8
0.30	335.0	341.9	344.3
0.25	336.0	343.2	344.9
0.20	337.5	343.8	347.0
0.15	338.5	344.4	348.3
0.10	341.4	345.3	350.1
0.05	343.1	348.2	351.8

Table 3.27. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 55.

RM 55 Cumulative Probability	Minimum Elevation		
	50 Days	**75 Days**	**100 Days**
0.95	319.6	323.0	326.8
0.90	323.2	326.7	328.4
0.85	324.3	328.1	330.1
0.80	325.7	329.8	331.3
0.75	326.3	330.5	332.2
0.70	327.2	331.5	333.4
0.65	328.1	332.1	334.1
0.60	328.9	332.9	334.8
0.55	329.4	333.7	335.6
0.50	329.7	334.2	336.6
0.45	330.6	335.0	337.7
0.40	331.3	335.6	339.0
0.35	331.9	337.7	341.1
0.30	332.4	339.3	341.8
0.25	333.3	340.7	342.4
0.20	334.8	341.2	344.5
0.15	335.8	341.8	346.0
0.10	338.7	342.8	347.6
0.05	340.5	345.7	349.3

Table 3.28. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 50.

RM 50 Cumulative Probability	Minimum Elevation		
	50 Days	75 Days	100 Days
0.95	316.5	320.1	323.7
0.90	320.1	323.5	325.3
0.85	321.1	325.0	326.9
0.80	322.6	326.7	328.3
0.75	323.1	327.3	329.0
0.70	324.0	328.4	330.4
0.65	325.0	329.1	331.0
0.60	325.8	329.8	331.7
0.55	326.3	330.6	332.5
0.50	326.7	331.1	333.4
0.45	327.5	331.8	334.8
0.40	328.4	332.5	336.0
0.35	328.7	334.6	338.0
0.30	329.3	336.2	338.7
0.25	330.3	337.8	339.4
0.20	331.6	338.2	341.3
0.15	332.7	338.7	342.8
0.10	335.6	339.7	344.2
0.05	337.5	342.5	345.9

Table 3.29. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 45.

RM 45 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	312.4	316.4	319.7
0.90	315.7	319.4	321.4
0.85	316.9	321.1	322.9
0.80	318.5	322.7	324.5
0.75	319.1	323.3	325.4
0.70	320.0	324.5	326.6
0.65	321.0	325.3	327.4
0.60	321.9	326.2	328.3
0.55	322.3	326.8	329.2
0.50	323.0	327.4	329.8
0.45	323.6	328.1	331.4
0.40	324.5	328.9	332.4
0.35	325.3	331.0	334.1
0.30	325.5	332.6	334.9
0.25	326.6	334.0	335.7
0.20	328.0	334.4	337.3
0.15	329.1	334.8	338.7
0.10	331.8	335.9	340.0
0.05	333.9	338.4	341.8

Table 3.30. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 40.

RM 40 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	309.7	314.1	316.9
0.90	312.8	316.6	318.7
0.85	314.1	318.4	320.2
0.80	315.7	320.0	321.9
0.75	316.3	320.8	322.9
0.70	317.3	321.9	324.2
0.65	318.3	322.7	324.9
0.60	319.3	323.5	326.0
0.55	319.6	324.2	326.9
0.50	320.4	324.8	327.4
0.45	320.9	325.4	328.9
0.40	321.7	326.3	330.0
0.35	322.6	328.4	331.5
0.30	323.1	330.1	332.4
0.25	324.3	331.3	333.0
0.20	325.4	331.9	334.5
0.15	326.5	332.2	335.9
0.10	329.2	333.2	337.0
0.05	331.3	335.6	338.7

Table 3.31. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 35.

RM 35 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	306.7	311.5	313.8
0.90	309.5	313.4	315.6
0.85	310.8	315.3	317.0
0.80	312.4	316.7	318.9
0.75	313.1	317.5	320.0
0.70	314.0	318.8	321.1
0.65	315.3	319.8	322.0
0.60	316.1	320.4	323.4
0.55	316.4	321.2	324.1
0.50	317.3	322.0	324.7
0.45	317.7	322.2	325.9
0.40	318.7	323.3	327.2
0.35	319.3	325.4	328.6
0.30	319.9	327.2	329.7
0.25	321.4	328.2	330.2
0.20	322.3	329.0	331.9
0.15	323.4	329.4	332.9
0.10	326.2	330.2	333.9
0.05	328.3	332.7	335.4

Table 3.32. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 30.

RM 30 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	303.7	308.2	311.2
0.90	306.1	310.2	312.5
0.85	307.6	312.2	314.0
0.80	309.0	313.5	316.0
0.75	309.8	314.4	317.1
0.70	310.7	315.8	318.1
0.65	312.3	317.0	319.1
0.60	312.9	317.4	320.7
0.55	313.4	318.2	321.6
0.50	314.3	319.0	322.3
0.45	314.8	319.6	323.4
0.40	315.7	320.4	324.6
0.35	316.3	322.5	325.9
0.30	316.8	324.4	327.2
0.25	318.4	325.4	327.9
0.20	319.3	326.4	329.7
0.15	320.6	326.9	330.4
0.10	323.3	327.8	331.5
0.05	325.6	330.3	332.8

Table 3.33. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 25.

RM 25 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	301.6	305.6	308.6
0.90	303.6	307.5	310.4
0.85	305.0	309.8	311.8
0.80	306.3	311.3	313.3
0.75	307.2	311.7	314.5
0.70	308.1	313.4	315.3
0.65	309.8	314.2	316.6
0.60	310.2	314.7	318.0
0.55	310.7	315.6	318.8
0.50	311.8	316.1	319.9
0.45	312.1	316.7	321.4
0.40	313.0	318.4	322.0
0.35	313.8	319.7	323.1
0.30	314.0	321.5	325.0
0.25	315.7	322.4	325.3
0.20	316.6	323.6	327.1
0.15	317.9	324.2	327.8
0.10	320.4	325.3	328.6
0.05	322.8	327.4	330.2

Table 3.34. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 20.

RM 20 Cumulative Probability	Minimum Elevation		
	50 Days	75 Days	100 Days
0.95	298.5	301.1	304.3
0.90	299.2	303.0	306.1
0.85	300.7	305.5	308.6
0.80	301.7	306.7	309.6
0.75	302.8	308.6	310.4
0.70	303.8	309.5	311.0
0.65	305.0	310.4	312.4
0.60	305.8	311.1	314.4
0.55	306.5	311.3	315.0
0.50	307.5	311.9	316.8
0.45	307.9	313.2	318.3
0.40	308.7	315.5	319.1
0.35	309.3	316.7	320.4
0.30	310.1	317.7	321.8
0.25	311.7	318.9	322.6
0.20	312.8	320.2	323.9
0.15	314.1	321.2	324.4
0.10	316.7	322.5	325.5
0.05	319.4	324.3	327.9

Table 3.35. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 15.

RM 15 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	296.2	298.7	302.4
0.90	297.3	300.9	303.6
0.85	298.6	303.4	306.1
0.80	299.4	304.0	307.4
0.75	300.6	306.4	308.6
0.70	301.5	307.2	309.2
0.65	302.5	308.0	310.0
0.60	303.6	308.7	312.0
0.55	304.8	309.1	312.8
0.50	305.1	309.9	315.0
0.45	305.8	311.2	315.9
0.40	306.4	313.0	316.9
0.35	306.9	314.5	318.4
0.30	308.0	315.5	319.4
0.25	309.3	316.1	320.5
0.20	310.5	317.5	321.4
0.15	311.7	318.8	321.9
0.10	314.1	320.2	323.0
0.05	316.6	321.9	326.3

Table 3.36. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 10.

RM 10 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	291.2	293.9	298.4
0.90	293.1	296.5	299.2
0.85	294.5	298.3	301.2
0.80	295.3	299.7	302.8
0.75	296.4	301.9	304.4
0.70	296.8	302.6	305.3
0.65	297.9	302.9	306.9
0.60	298.9	303.9	308.3
0.55	299.8	305.2	309.5
0.50	300.7	306.7	310.8
0.45	300.9	307.9	311.9
0.40	301.7	308.5	313.5
0.35	302.2	309.7	315.1
0.30	304.4	310.9	316.1
0.25	304.8	311.9	317.2
0.20	305.7	313.5	318.1
0.15	307.1	314.5	318.8
0.10	309.7	315.6	320.2
0.05	311.6	317.9	325.0

Table 3.37. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 5.

RM 5 Cumulative Probability	**Minimum Elevation**		
	50 Days	**75 Days**	**100 Days**
0.95	288.7	291.8	296.0
0.90	290.9	294.0	297.6
0.85	292.2	295.8	298.1
0.80	293.1	297.7	301.0
0.75	294.4	299.0	301.9
0.70	295.2	299.8	303.6
0.65	296.1	300.8	304.2
0.60	296.6	301.4	306.7
0.55	297.5	302.6	307.0
0.50	298.4	304.2	308.3
0.45	298.7	305.7	309.5
0.40	299.6	306.2	311.7
0.35	300.0	307.0	313.7
0.30	301.9	308.2	314.3
0.25	302.6	309.9	314.7
0.20	303.6	310.6	316.9
0.15	304.8	312.8	317.7
0.10	306.8	313.8	319.1
0.05	308.7	316.6	324.4

Table 3.38. Probability of inundation for the elevation indicated over 50, 75, and 100 continuous days during the period 15 May - 31 August in any given year at RM 0.

Cairo, Illinois, RM 0 Cumulative Probability	Minimum Elevation		
	50 Days	**75 Days**	**100 Days**
0.95	286.7	290.2	293.7
0.90	289.1	291.4	294.4
0.85	290.1	293.9	296.7
0.80	291.2	294.9	298.0
0.75	292.8	296.2	299.6
0.70	293.5	296.9	301.3
0.65	294.1	297.9	302.7
0.60	294.3	299.1	304.5
0.55	295.1	300.3	305.5
0.50	296.2	302.7	306.0
0.45	296.6	303.2	308.2
0.40	297.2	303.5	309.7
0.35	298.0	304.3	311.1
0.30	298.9	306.5	313.0
0.25	300.5	308.2	313.9
0.20	302.0	308.8	315.4
0.15	302.6	310.2	316.6
0.10	303.4	312.9	317.6
0.05	306.5	315.2	323.5

CHAPTER FOUR

THE BIOLOGICAL IMPORTANCE OF MIDDLE MISSISSIPPI RIVER ISLANDS ON FISH ASSEMBLAGES, MISSOURI, USA

Introduction

Loss of habitat diversity is a major problem in the Middle Mississippi River (MMR) (Simons et al. 1975, Theiling 1999), as it is in many large rivers around the world (Ward and Stanford 1989, Johnson and Jennings 1998, Pedroli et al. 2002). Alterations to vital river ecosystem functions bring about decreases in habitat diversity, and as a consequence, result in a loss of biological diversity (Junk et al. 1989, Thorp 1992, Ward and Stanford 1995, Ward et al. 1999, Theiling et al. 2000).

Islands are natural features in large rivers. In its natural state, an alluvial river often divides itself into two or more channels by the processes of either erosion or deposition (Simons et al. 1974). Side channels, which are obliterated by deposition, are replaced by new side channels caused by floods and/or river migrations. In the MMR, the river is no longer free to migrate and produce new side channels and islands (Simons et al. 1974) due to navigation related features such as flow regulation (upriver dams), channel straightening, bank stabilization, wing dikes, revetments, and closing structures. Additionally, agricultural levees disconnect the main river channel from the floodplain, wetlands, and associated backwaters (Simons et al. 1974, Theiling 199). In the MMR, such anthropogenic activities have contributed to the stabilization, narrowing, and deepening of the navigation channel, and to the loss of shallow backwaters, islands, and secondary channels (Simons et al. 1974, Theiling 1999, Barko and Herzog 2003).

The loss of MMR side channels and their corresponding islands is well documented. Side channels have frequently been closed off and others have sedimented in (Simons et al. 1974, Theiling 1999). In 1797 there were 55 major side channels (Collot 1826), 35 in 1860 (Simons et al. 1974), 27 in 1968 (Simons et al. 1974), and only 28 currently. Many that remain are degraded and much smaller than in the past (Theiling et al. 2000) and function more as backwater habitat since they are disconnected from the main channel during large portions of the year (Barko and Herzog 2003). In the absence of further human-induced changes in hydrology or geomorphology of the MMR, some of the remaining side channels may disappear (Theiling 1999).

Fishes benefit from islands in several aspects. Sheltered areas with reduced current velocity may benefit fish by decreasing their energy expenditure and increasing their growth rate through hover feeding (Bachman 1984, Todd and Rabeni 1989, Putman et al. 1995, Barko et al. 2004). The shade from island trees may alter light penetration and water temperatures, providing fish with a variety of microhabitats. Scour holes, which may be created upstream of islands in dike fields, may function like deep water holes associated with snags in un-channelized river systems (Barko et al. 2004). Deep scour holes are often used by fish to hide from terrestrial predators, such as mammals and birds (Matthews et al. 1986), and are also used as overwintering habitat (Hesse and Newcomb 1982, Logsdon 1993). Islands also provide fish with shelter and refuge from predators by supplying cover in the form of woody debris (Lehtinen et al. 1997), undercut banks, vegetation, or large rocks along the shoreline (Johnson and Jennings 1998). Aquatic macroinvertebrates, which grow on the submerged woody debris (Thorp 1992), as well as insects which often fall into the water from riparian plants, are both considered a major source of high-quality fish food in rivers (Benke et al. 1984, Zalewski et al. 2003). The shallow back waters or side channel habitats created by islands provide refuge from the swift currents and harsh environment of the thalweg (Environmental Sci. and Eng. 1982, Fremling et al. 1989, Barko and Herzog 2003). Such backwaters are particularly beneficial as spawning, rearing, food production, feeding, and seasonal refuge areas for several species of fish (i.e., centrarchids) (Chipps et al. 1997). Islands also increase habitat diversity by providing conditions suitable for a variety of forest, shrub, grassland, and wetland communities. Large sandbars, which often form prior to vegetated islands, are critical nesting habitat for the federally endangered interior least tern (*Sterna antillarum*).

The U.S. Fish and Wildlife Service suggested the use of aquatic habitat rehabilitation measures (i.e., pilot projects), including side channel restoration, island building, and dike notching to increase habitat diversity in the MMR (USFWS 2004). However, few studies have assessed relationships among islands, side channels, and fish assemblages in riverine systems. The ecology and use of side channels by fishes in the MMR is poorly understood (Barko and Herzog. 2003). Thus, I tested the hypothesis that dike fields with created islands support a fish assemblage which is distinct from that found in conventional dike fields. I also tested the hypothesis that physical habitat characteristics (water depth, current velocity, habitat type, etc.) would influence fish assemblage composition and structure. Habitat restoration is necessary to re-establish habitat quantity, quality, and diversity so that the benefits of the dynamic natural river processes are restored. While island creation through dike modification appears to be a practical technique to increase local habitat diversity, and therefore fish assemblage diversity, the potential impacts and purported benefits have not been investigated and are undocumented.

Methods and Materials

Study Site – The Middle Mississippi River, approximately 322 km (200 miles) in length, lies between the mouth of the Missouri River at St. Louis, Missouri, and the mouth of the Ohio River at Cairo, Illinois (Figure 4.1). The Mile 100 dike field is located near Chester, Illinois, between river mile (RM) 100.1 and 98.9 on the right descending bank (RDB). The study area consists of six notched dikes and five islands (Figure 4.2). The dikes were built in the early 1970's for the expressed purpose of sediment management and channel improvement. Notches were designed in the dikes at the time of construction with the intent of creating a scour pattern that would eventually form a secondary channel and associated islands. The notches were designed to pass flows approximately 50% of the time. Scour holes developed immediately downstream of the notches, with scoured material being deposited further downstream. These depositional areas eventually increased in size and elevation until vegetation became established and finally converted into terrestrial habitat with distinct island boundaries. Vegetation became established on the depositional areas 10-20 years after dike construction. Denser vegetation assisted in greater sediment deposition, which raised the islands to even greater elevations. The formation of backwater areas between the islands and the leveed floodplain followed.

The islands range in size from approximately 2.0 acres to 11.0 acres (0.8 to 4.5 ha). The highest average point on the islands is just over +30 feet low water reference plane (LWRP), with vegetation establishment as low as +19 feet LWRP. The wetted perimeter of the islands ranges from approximately 460 to 1290 feet (140 to 393 m). This area is the only reach in the MMR where small islands were purposely engineered using a set of notched dikes.

In addition to the five islands, five areas between wing dikes which do not contain islands served as "controls" or reference sites to determine if observed differences between fish assemblages were due to the habitat modifications related to the presence of islands. An example of a reference site is shown in Figure 4.2. The reference sites are located on the RDB between RM dikes 107.4-107.2, 107.2-106.8, 105.0-104.7, 104.7-104.4, and 100.6-100.4. Each study site was sampled eight times over a 24-month, four season period from October 2004 through August 2006. Approximately seven to ten days of intensive sampling effort were expended in each season.

Habitat Characteristics – Habitat characteristics between dikes were recorded along transects during each seasonal sampling period (Figure 4.3). Average velocity was measured just below the surface using a SonTek/YSI FlowTracker handheld acoustic Doppler velocity meter (SonTek/YSI Inc., San

Diego, CA). The device takes velocity readings at a single point if the velocity is within the range of 0.001 m/s (0.1 cm/s, 0.003 ft/s) to 5 m/s (500 cm/s, 16 ft/s). Average water depth along each transect was measured to the nearest 0.03 m (0.1 ft) using boat-mounted sonar (Lorance LCX-19C). Water temperature, dissolved oxygen, and conductivity were measured *in situ* for each sample site using a Hydrolab water quality probe. A pH meter (Hanna PHep HI 98128) was used to measure pH, and a Secchi disk was used to measure visibility at each sample site. No aquatic vegetation occurs at the study sites.

Fish Assemblage – To examine fish use of dike field habitat, the area between dikes was sampled by running transects using Missouri otter trawls and daytime electrofishing (Gutreuter et al. 1995). Missouri otter trawls were 4.9 m (16 ft.) wide, 0.9 m (3 ft.) high, 7.6 m (25 ft.) long, with 19.05 mm (0.75 in.) inner bar mesh and 4.76 mm (0.19 in.) outer chafing mesh (Herzog et al. 2005). The opening of the trawl net was maintained by outward forces generated by water pressure and bottom friction against 76.2- by 38.1-cm (30 by 16 in.) plywood boards (trawl doors) as it was towed (Herzog et al. 2005). Trawls were towed just faster than the current. Electrofishing was conducted using pulsed direct current (DC) set at a 120 Hz pulse frequency and a 25% duty cycle (Smith-Root, Inc. Model GPP Electrofisher). Each transect was timed. All electrofishing was conducted parallel to the shore. At island sites, two trawling and two electrofishing transects were run on the main channel side of the island (outside) as well as behind each island (inside; water depth permitting). One trawling and one electrofishing transect were run along the upstream tip (tip) of each island.

At reference sites, two trawling and two electrofishing transects were run between dikes (Gutreuter et al. 1995). Mini fyke nets were used to sample shallow, low-velocity areas at each habitat type (Hubert 1996). Small Wisconsin-type fyke nets consisted of a 4.5 m (15 ft.) lead, two rectangular steel frames, and two circular hoops. The netting was 3.2 mm (1/8 in.) ace type nylon mesh, coated with green latex net dip. The two rectangular frames were 1.2 m (4 ft.) wide and 0.6 m (2 ft.) high (Herzog et al. 2005). Mini fyke nets were set perpendicular to the shoreline for approximately 24 hours. One overnight set was considered one unit of effort. Two nets were set behind each island, two at each upstream island tip, two on the outside of each island, and two along the shore of each reference site (Figure 4.3). A multiple gear approach is usually warranted in sampling fish communities in large rivers because of biases associated with various types of gear and because of strong interactions between the environment and sampling efficiency (Sheehan and Rasmussen 1993, Hayes 1996). Fish, which were identifiable in the field, were identified to species level, measured, and released near the collection site. All other fishes were anesthetized and euthanized in a concentration ≥ 250 mg/L of tricaine methane sulfonate (TMS, MS 222) by leaving them in the solution for at least 10 minutes following cessation of opercular movement. Following euthanasia, fishes were fixed in 10% formalin, and returned to the laboratory for preservation in 70% ethanol, identification and measurement (AVMA 2001).

Treatment of Study Animals – The use and care of fishes in this study adhered to humane guidelines. A dissertation proposal entitled "Middle

Mississippi River Islands: Historical Distribution, Biological Importance and Restoration Planning" (05-03-01) was approved by the University of Missouri-St. Louis' Animal Care and Use Committee. Approval by the UMSL ACUC assures compliance with the Public Health Service Policy on Humane Care and Use of vertebrate research animals.

Data Analyses –

Ordination – In order to visualize variation in fish communities among sites and habitats, ordination by non-metric multidimensional scaling (NMDS) was used (Kruskal 1964). The term "sampling unit" was used to refer to a fish assemblage sampled from a particular habitat within a given site on a particular date. NMDS is effective for summarizing community data when the aim is to extract the major dimensions of community variation that are correlated with underlying ecological factors (Minchin 1987). It represents each sampling unit as a point in a coordinate system, such that the distances between all pairs of sampling unit points are, as far as possible, in rank order agreement with their degree of difference in community composition. To express community differences, the Bray-Curtis dissimilarity index (Bray and Curtis 1957) which has been shown to be one of the best indices for summarizing trends in community data, was used (Faith *et al.* 1987). To reduce the influence of occasional high counts for some species, count data were transformed to square roots, and were then standardized by species maximum (i.e., transformed counts within each species were divided by the maximum transformed count attained by that species over all sampling units), as recommended by Faith *et al.* (1987) and

Sandercock (1997). This standardization allows a stronger consensus among species and makes the dissimilarities more informative regarding ecological variation among sampling units.

NMDS does not have an algebraic solution. It is necessary to find the optimal ordination by a successive improvement algorithm, during which the positions of sampling units within the ordination are gradually adjusted, in order to improve the rank-order fit between ordination distances and community dissimilarities. A statistic known as “stress” measures the badness-of-fit of a rank-order regression of distance on dissimilarity and the optimization process seeks to minimize stress. Ordinations were performed with the number of dimensions ranging from one through six. To avoid problems of entrapment at local minima, 50 different random starting configurations were used in each case. The scree plot (line graph of minimum stress versus number of dimensions) was examined to identify the number of dimensions beyond which further reductions in stress were relatively minor (Kruskal and Wish 1978).

Vector Fitting – Vector fitting (Dargie 1984; Kantvilas and Minchin 1989) was used to examine correlations between explanatory variables (e.g., physicochemical parameters, fish species) and the NMDS ordination. For each explanatory variable, this method determines the direction of a vector through the ordination, such that scores of sampling units along the vector are maximally correlated with values of the variable. Statistical significance of the correlation is tested by randomly permuting the values of the explanatory variable among sampling units (Faith and Norris 1989). Ten thousand (10,000) random permutations were used for each test.

ANOSIM – Differences in fish community between levels of site type (island or reference), habitat (inside, tip, outside, reference), were tested using analysis of similarities or ANOSIM (Clarke 1993), a multivariate, non-parametric test based on the ranks of dissimilarities. The test statistic, *R*, measures the extent to which “between group” dissimilarity values are greater in rank than “within group” values. An *R* value of +1 indicates that all the between dissimilarities are ranked higher than all of the within dissimilarities and the groups are as different as they can possibly be. A value of *R* close to zero suggests that the groups are not different. The statistical significance of *R* was tested by randomly permuting group membership 10,000 times. Separate ordinations, vector fitting and ANOSIM analyses were done for each of two data matrices:

1. Transformed, standardized count data for all individuals (adult fish and young-of-the-year).

2. Transformed, standardized count data for adult individuals only.

Analyses were not conducted separately for YOY fishes because this size class is known to be highly dynamic depending on yearly environmental spawning and rearing conditions.

Indicator Species Analysis – When ANOSIM indicated a difference in community composition, indicator species analysis (ISA) (Dufrene and Legendre

1997) was used to identify the species that best differentiated the groups of sampling units. Indicator species analysis is based on the concepts of fidelity (the degree to which a species is confined to a particular group) and constancy (the proportion of sampling units in a group in which the species occurs). The ideal indicator species for a group should be both faithful to that group (does not occur in other groups) and constant within that group (all sampling units in the group contain the species).

The fidelity of species *j* to group *k* is calculated as:

$$\mathbf{F}_{kj} = \frac{\bar{x}_{kj}}{\sum_{l=1}^{g} \bar{x}_{lj}}$$

where $\bar{x}_{kj}$ is the mean abundance of species *j* in group *k*. The denominator is the sum of the mean abundances of species *j* over all *g* groups. These fidelity values range from 1.0 when species *j* is confined to group *k* to 0.0 when the species is absent from group *j*. The constancy of species *j* in group *k* is computed as:

$$\mathbf{C}_{kj} = \frac{n_{kj}}{n_{k.}}$$

where n_{kj} is the number of sampling units in group *k* in which species *j* occurs and $n_{k.}$ is the number of sampling units in group *k*. Constancy values are proportions which range from 0.0 (species does not occur in group *k*) to 1.0

(species occurs in every sampling unit in group *k*). Fidelity and constancy are combined into a single Indicator Value as follows:

$$IV_{kj} = 100F_{kj}C_{kj}$$

In order to attain a high *IV*, a species must be both faithful and constant. The statistical significance of $IV_{\max}$, the highest *IV* attained by a species over all *g* groups, is tested by a random permutation of group membership among sampling units. In each test, I used 10,000 random permutations.

Catch Per Unit Effort (CPUE) – Total catch per unit effort (CPUE) was examined using repeated-measures analysis of variance (ANOVA), in which sampling date was treated as a repeated factor with eight levels. Multivariate tests were used for within-habitat (among-sampling date) effects and the habitat X sampling date interaction. The four tests applied were Wilks' Lambda, Pillai's Trace, Hotelling-Lawley Trace, and Roy's Maximum Root (Morrison 1976). Among-habitat effects were tested with conventional F ratios computed using type III sums-of-squares. Least-squares means and their standard errors, corrected for other terms in the model, were computed for significant effects and interactions. Multiple-comparisons, with probabilities adjusted using the Tukey-Kramer approach (Kramer 1956), were used to determine which pairs of means were significantly different, using an experiment-wide alpha level of 0.05.

Software – NMDS ordinations, Vector Fitting and ANOSIM tests were performed using DECODA version 3 (Minchin 1998). Indicator Species Analysis was performed using PCORD version 4 (McCune and Mefford 1999). Repeated-measures Analyses of Variance was performed using procedure GLM in SAS version 9.1 (SAS Institute Inc. 2004). Graphs were prepared using Sigma Plot version 11.0 (Systat Software Inc. 2008).

Results

Seven hundred seventy-eight samples were taken over a 2-year period. These were comprised of 138 inside, 160 tip, and 240 outside for island samples, and 240 for reference sites. A total of 44,501 fishes in identifiable condition representing 71 species and 19 families were collected. The families comprising the highest percentage of fishes collected (raw numbers) included Cyprinidae (minnows), Sciaenidae (drums), Ictaluridae (catfishes), and Clupeidae (herring) (Table 4.1).

Assemblage Structure and Organization at Islands and Reference Sites – At island sites, I collected a total of 33,368 fishes representing 67 species and 18 families. Of these, 14,650 were adult fishes representing 48 species and 16 families, and 18,718 were young-of-the-year (YOY) fishes representing 55 species and 15 families. At reference sites, I collected a total of 11,133 fishes representing 55 species and 15 families. Of these, 4,963 were adult fishes representing 42 species and 14 families, and 6,170 were YOY fishes representing 37 species and 12 families (Table 4.2).

Assemblage Structure and Organization at Island Habitat Types and Reference Sites – Islands were further divided into habitat types in order to reveal potential differences in fish assemblages in the different areas. Habitat types at the islands included inside, tip, and outside. Theses areas were compared with each other as well as with the reference sites.

At inside habitat, I collected a total of 8,194 fishes representing 53 species and 15 families. Of these, 5,127 were adult fishes representing 33 species and 11 families, and 3,067 were YOY fishes representing 44 species and 13 families (Table 4.2).

At tip habitat, I collected a total of 7,923 fishes representing 48 species and 15 families. Of these, 2,845 were adult fishes representing 39 species and 14 families, and 5,078 were YOY fishes representing 35 species and 12 families (Table 4.2).

At outside habitat, I collected a total of 17,251 fishes representing 56 species and 14 families. Of these, 6,678 were adult fishes representing 40 species and 13 families, and 10,573 were YOY fishes representing 41 species and 12 families (Table 4.2).

CPUE at Islands and Reference Sites – Fish catch per unit effort (CPUE) for total (P = 0.8673) and adult (P = 0.9563) counts did not differ significantly between islands and reference sites.

CPUE at Island Habitat Types and Reference Sites – CPUE did not differ significantly between habitat type and reference sites for total count (P =0.1341) or for adult count (P = 0.1480).

Assemblage Composition at Islands and Reference Sites – ANOSIM results indicated that fish assemblages differed significantly between islands and references sites for total standardized count (R= 0.1784, p= 0.0000) (Figure 4.4), and for adult standardized count (R= 0.2150, p= 0.0000) (Figure 4.5). Twenty species significantly differentiated between islands and reference sites based on total standardized count. All species were indicative of islands, while none implied an affinity for reference sites. The species with the highest indicator values were silverband shiner (*Notropis shumardi*), bullhead minnow (*Pimephales vigilax*), and river carpsucker (*Carpiodes carpio*) (Table 4.3).

When the same analysis was conducted using only adult fishes, bullhead minnow, orangespotted sunfish (*Lepomis humilis*), and silverband shiner had the highest indicator values of the 16 species with significant scores. Again, all 16 species were indicators of islands (Table 4.4).

Assemblage Composition at Island Habitat Types and Reference Sites – ANOSIM results indicate that habitat types differ significantly in terms of fish assemblage differences for standardized count data for all individuals (adult fish and young-of-the-year) (R= 0.1508, p= 0.0000). The fish communities differed significantly among each of the habitat types (p< 0.000), with the exception of outside and reference habitat (p= 0.3319) (Figure 4.6). Thirteen species significantly differentiated island habitat from reference areas. Six species were most closely associated with inside habitat, one with tip, two with outside habitat, and four with reference (Table 4.5).

Further, ANOSIM results suggest that habitats and reference sites differ significantly in terms of fish assemblages for standardized adult count data in terms of habitat type (R= 0.1594, p= 0.0000) with the exception of tip and reference site (p= 0.2401) (Figure 4.7). Eleven species were found to significantly explain these differences. Four species were most closely associated with inside habitat, three with outside habitat, two with tip, and two with reference (Table 4.6).

Relationship Between Physicochemical Parameters and Fish Assemblages – Ordination by NMDS using the Bray-Curtis dissimilarity index (Bray and Curtis 1957) indicated that average depth, conductivity, pH, velocity, water temperature, and Secchi visibility were significantly related to differences in both total count and adult fish community assemblages between islands and reference sites and between habitat type and reference sites (Figures 4.4-4.7). The single exception was the non-significance of Secchi visibility in differentiating between habitat types and reference sites for total standardized counts.

Discussion

Little information is available on the role or importance of island side channels to riverine fish assemblages (Barko and Herzog 2003). Access to side channel and floodplain habitat is being lost to fish species because of sedimentation and levee construction (Simons et al. 1975, Grubaugh and Anderson 1988, Theiling 1999. USACE 2001, Barko and Herzog 2003). In the MMR, nearly the entire floodplain is disconnected from the main river by levees (Theiling 1995) most of the time, and side channels are being lost (Theiling et al. 2000).

Throughout the year, island habitat undergoes physical changes in correlation with river stage and season. During low water conditions, usually in the summer and winter, the inside (side channel) habitat may become isolated from the main river channel. This transforms a fluvial (flowing) habitat into a lentic (standing) habitat, and may also result in differences in water quality characteristics between the pooled area and the main river. During high water, all habitats are fluvial with variations in velocity and depth, but with otherwise similar water quality characteristics. The MMR typically has two high water seasons, spring and fall, with spring usually having the higher peak flow. In general, reference sites would experience conditions similar to the main river channel throughout the year, with the exception of lower velocity and depth.

One premise of this study was that the physically complex habitat created by islands in dike fields would support a fish assemblage that differed from dike fields without islands. This research strongly supported that hypothesis. Although islands and reference sites did not differ significantly in CPUE, islands had a greater total, adult, and YOY species richness than reference sites (Table 4.2). Furthermore, species composition differed between islands and reference sites as revealed by indicator species analysis. Results for both total and adult count identified numerous sunfish (Centrarchidae) and bass (Moronidae), including orangespotted sunfish, bluegill (*Lepomis macrochirus*), white crappie (*Pomoxis annularis*), green sunfish (*Lepomis cyanellus*), and white bass (*Morone crysops*) as being associated with island habitat. These species generally inhabit pools or backwaters and are tolerant of moderate to low turbidity and/or current (Smith 1979, Pflieger 1997). No fishes were indicative of reference sites. Indicator species analysis was not conducted on YOY fishes because this size class is known to be highly dynamic depending on yearly environmental spawning and rearing conditions. Chipps et al. (1997) evaluated trends in fish abundance associated with in-water disposal of dredged material in Lower Granite Reservoir, Idaho-Washington. They sampled fish assemblages before and after construction of a 0.37-ha disposal island to assess local changes in fish community structure. Results of the island indicator species support the findings of Chipps et al. (1997) in which several centrarchid and moronid species were absent at a sampling site before construction of the island, but were present after construction. Chipps et al. (1997) attribute this to the creation of shallow water habitat associated with islands. They conclude that islands constructed from dredged material can reduce local water depth and provide rearing areas for several resident fish species.

Indicator species analysis also revealed numerous additional species associated with islands, including fluvial specialists such as mooneye (*Hiodon tergisus*), channel shiner (*Notropis wickliffi*), speckled chub (*Macrhybopsis aestivalis*), silverband shiner, river shiner (*Notropis blennius*), inland silverside (*Menidia beryllina*), and sauger (*Stizostedion canadense*), as well as generalists or lentic specialists such as threadfin shad (*Dorosoma petenense*), red shiner (*Cyprinella lutrensis*), bullhead minnow, bluntnose minnow (*Pimephales notatus*), bighead carp (*Hypophthalmichthys nobilis*), silver carp (*Hypophthalmichthys molitrix*), bigmouth buffalo (*Ictiobus cyprinellus*), smallmouth buffalo (*Ictiobus bubalus*), river carpsucker, and mosquitofish (*Gambusia affinis*).

Barko and Herzog (2003) examined six side channels of varying connectivity located in the MMR. They found that side channels which were disconnected from the main river at one end were primarily lentic. The adult assemblage associated with these areas included red shiner, orangespotted sunfish, and green sunfish. Some YOY of fishes correlated with the same chutes included silverband shiner and white crappie. They classified these species as tolerant of moderate to low turbidity and/or current (Smith 1979, Pflieger 1997, Barko and Herzog 2003). The side channels which were intermediate in connectivity exhibited both lentic and lotic characteristics. Adult and YOY assemblages were dominated by pool-dwellers and schooling species, both of which seek cover and prefer little to no current (Smith 1979, Pflieger 1997). Adults of species correlated with these chutes included smallmouth buffalo, black buffalo (*Ictiobus niger*), bigmouth buffalo, black crappie (*Pomoxis nigromaculatus*), and white crappie. The YOY species most correlated with these chutes was the emerald shiner (*Notropis atherinoides*) (Barko and Herzog 2003). The open side channels were connected to the main river at both ends and were lotic in character. Adults of some species associated with the open side channels included channel catfish (*Ictalurus punctatus*), channel shiner, emerald shiner, and sauger, while YOY of species included common carp (*Cyprinus carpio*), sauger, river carpsucker, and goldeye (*Hiodon alosoides*). These species are large river inhabitants that are tolerant of currents and/or turbidity (Smith 1979, Pflieger 1997, Barko and Herzog 2003). Results of species distributions revealed by this study generally substantiate those of Barko and Herzog (2003).

In order to design islands which advance habitat and biotic diversity, it is essential to ascertain which particular feature or features support maximum diversity. Investigating islands in terms of inside, tip, and outside habitat and comparing these to reference sites allowed these factors to be revealed. During high water conditions, water flows over the dikes and fishes at tip habitat experience high velocity and turbulent conditions. Some adult species and many YOY species may not be tolerant of these conditions. Additionally, YOY may avoid the area in order to reduce predation risk. Therefore, lower species richness would be expected at tip habitat. Total species richness was similar among all habitat types, with the exception of tip habitat which had fewer species (Table 4.2). Indicator species analysis characterized white bass as a tip species for total count and white bass and sauger as tip species for adult count. White bass are common in large rivers where they inhabit open water with moderate current (Mathias et al. 1996), such as that found at upstream island tips. Sauger is exclusively a fish of flowing waters (Pflieger 1997). As the river drops and flow over the dike ceases, it is realistic to expect riverine species to move from tip and inside areas to outside habitat or the open river to avoid being trapped in the isolated side channels. The exception would be YOY individuals attempting to avoid predation and adult lentic specialists searching for lower velocity areas. At low water stages when inside habitat becomes pooled, species intolerant of increased water temperatures and low dissolved oxygen levels would not be likely to survive these periods of isolation. Based on this scenario, species richness should be lowest at inside and tip habitat, and higher at outside and reference habitat. This was true for both total and adult counts (Table 4.2). Inside habitat would be expected to contain primarily YOY individuals and adult lentic specialists. Statistically, inside habitat was significantly different from all other habitat types for both total and adult count. The lowest adult species richness was found at inside habitat, while the other areas had similarly greater adult species richness (Table 4.2). Conversely, inside habitat had the greatest YOY fish species richness, followed by outside, reference, and tip (Table 4.2). Indicator species analysis of both total and adult count identified orangespotted sunfish, bluegill, mosquitofish, and white crappie as being linked with inside habitat. Once again, the data show that it was the relatively shallow, low velocity, backwater habitat which supports these species. Pflieger (1997) maintains that orangespotted sunfish, bluegill, mosquitofish and white crappie typically occur in backwaters and overflow pools of larger streams. Indicator species analysis also identified bighead carp and silver carp as indicators of inside habitat for total count. This is not unexpected, as these generalist, omnivore, non-native fishes commonly inhabit backwaters and are notoriously difficult to catch in other areas. Similar microhabitat should exist at outside and reference habitats under most flow conditions, thus they should be comprised of similar species assemblages. Outside habitat was characterized by channel catfish and speckled chub (total and adult count) and blue catfish (*Ictalurus furcatus*) (adult count); while gizzard shad (*Dorosoma cepedianum*) (total and adult catch), blue catfish (total count), goldeye (total count), sturgeon chub (*Macrhybopsis gelida*) (total count), and emerald shiner (adult count) characterized the reference sites. Since outside and reference sites were indistinguishable based on total count of fish assemblages, it is not surprising that the species associated with these habitats all share a common habitat requirement; the open channel of large rivers or streams with moderate to swift currents (Pflieger 1997).

For freshwater fishes, studies have shown that, at a local scale, abiotic factors including habitat diversity (Gorman and Karr 1978), water chemistry (Rahel 1986), flow regime, temperature and channel morphology (Horwitz 1978, Matthews 1985, Schlosser 1985), appear to influence fish assemblages in variable environments (Capone and Kushlan 1991, Oberdorff et al. 1995). Large river systems are dynamic systems; thus, the hypothesis that physicochemical characteristics would influence fish assemblage composition and structure was also investigated. In the absence of physical barriers, the distribution of fish is determined by the presence of suitable environmental conditions that support the activities of the individual (Shirvell and Dungey 1983, Schueller 1989). The results of this study indicated that average depth, conductivity, pH, Secchi visibility, velocity, and water temperature were significantly related to differences in community assemblages between islands and reference sites for both total and adult count. Dissolved oxygen concentrations were not significantly related to variation in community assemblages. Average depth, conductivity, and velocity all tended to be higher at reference sites. Average water temperature was slightly cooler and Secchi visibility was slightly greater at island sites.

Schueller (1989) investigated habitat utilization of a river island in navigation pool 7 of the Upper Mississippi River by young-of-the-year fishes. He measured microhabitat characteristics including temperature, dissolved oxygen, pH, conductivity, turbidity, current, depth, and vegetation characteristics. He found that slack-water, vegetated areas around the island had more species than the river side locations where currents were higher and there was less vegetative cover. Johnson and Jennings (1998) investigated habitat associations of small fishes around islands in the Upper Mississippi River, by seining at 62 sites around 20 islands disbursed over a 180 km area. They found that macrohabitat features (island location, shape, or maximum depth around the island) of islands were less important than mesohabitat features (current velocity, depth, sediment type, and vegetation abundance) of sites in determining density of small fishes. The primary distinguishing factors were water depth and distance from the main channel (Johnson and Jennings 1998).

The hypothesis that physicochemical factors would be related to differences in community assemblages among habitat types was also examined. Abiotic factors significantly related to differentiating community assemblages between habitat types and reference sites included average depth, conductivity, pH, Secchi visibility (adults only), velocity, and water temperature. Inside habitat was the shallowest, followed by tip, outside, and reference. Conductivity was similar for all habitats except inside, where it was notably reduced. Dissolved oxygen levels were similar among island habitat types but greater at reference sites, likely as a result of wave action. Secchi visibility was similar for all habitat types with the exception of inside, where visibility averaged 10 cm less. This result is counter-intuitive to our observations and is likely the result of the water being too shallow to obtain accurate readings, since the Secchi disk would often reach the bottom while still in view. Average velocity was significantly reduced at inside habitat, slightly greater at tip habitat, while outside and reference habitats had comparable higher readings. Water temperature was slightly lower at inside habitat, while pH was consistent in all areas.

It is apparent that both islands and reference sites are being used as nursery habitat since over half of the total catch in each area was comprised of YOY individuals (56% and 55%, respectively). The importance of shallow water areas as nursery habitat is supported by the findings of Brown-Peterson and Eames (1990). They investigated the fish assemblage associated with 90 spoil islands along a 190 km section of Indian River Lagoon, Florida. They found that, in general, fish assemblages at islands were similar to the fish assemblages from other littoral areas within Indian River Lagoon. They suggested that seasonal recruitment of juvenile fishes indicated the importance of spoil islands as nursery habitat. Schueller (1989) also concluded that island environment can be a useful nursery habitat, and can provide an alternative to traditional backwaters where backwaters are absent or at a minimum, such as in the MMR. Our results supported the findings of Schueller (1989), implying that off channel, lentic, and seasonally flooded habitats can provide an alternative to traditional backwaters, as well as supply valuable nursery habitat.

In many community ecology studies, rare species are discarded from the analyses. However, it is these species that often provide valuable insight. While no federally or state threatened or endangered species were collected during this study, several Missouri Species of Conservation Concern were encountered. Western sand darter (*Ammocrypta clara*), along with mooneye, Mississippi silvery minnow (*Hybognathus nuchalis*), silver chub (*Macrhybopsis storeriana*), ghost shiner (*Notropis buchanani*), and pugnose minnow (*Opsopoedus emiliae*) all generally inhabit quiet pools and backwaters (Pflieger 1997). Relative to large rivers, the presence of these species in the MMR suggests that the Mile 100 Island complex has created a habitat that functions similarly to backwaters, a much needed habitat in this section of the river (Barko and Herzog 2003). River darter (*Percina shumardi*), plains minnow (*Hybognathus placitus*), and sturgeon chub generally occupy areas with moderate to swift current. Surprisingly good numbers of sturgeon chubs, silverband shiners, and river shiners were also identified in the samples. In recent times, decreases in numbers of the latter two species in the MMR have been observed (Hrabik, pers. comm.). Apparently, the Mile 100 Islands are providing a diversity of habitat types that support some rare and unusual fish for the MMR. Changes to the island complex, such as more or swifter water behind the islands, would compromise this productive habitat.

This study improves our knowledge of large river fish communities and their use of island and dike field habitat in the Middle Mississippi River. Where little or no floodplain or backwater habitat exists, an alternative is needed, and the island environment may be that alternative. Creation of islands in large rivers may help to restore some of the local habitat diversity lost as a result of navigation and agricultural alterations, and should promote an increase in the diversity of local riverine communities. Long-term positive benefits to native fishes are anticipated as a result of island building, since island habitats provide bathymetric and hydraulic diversity. This environment most likely cannot replace classic floodplain or backwater habitat, but could, to some degree, mitigate their continuing loss under proper conditions. The seasonal channels and pools provided by island side channels would supply refugia, spawning and juvenile rearing habitat, and forage food production for numerous fluvial and lentic species in the MMR.

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Figure 4.1. Location of the Middle Mississippi River, extending from its confluence with the Missouri River north of St. Louis, Missouri to its confluence with the Ohio River near Cairo, Illinois.

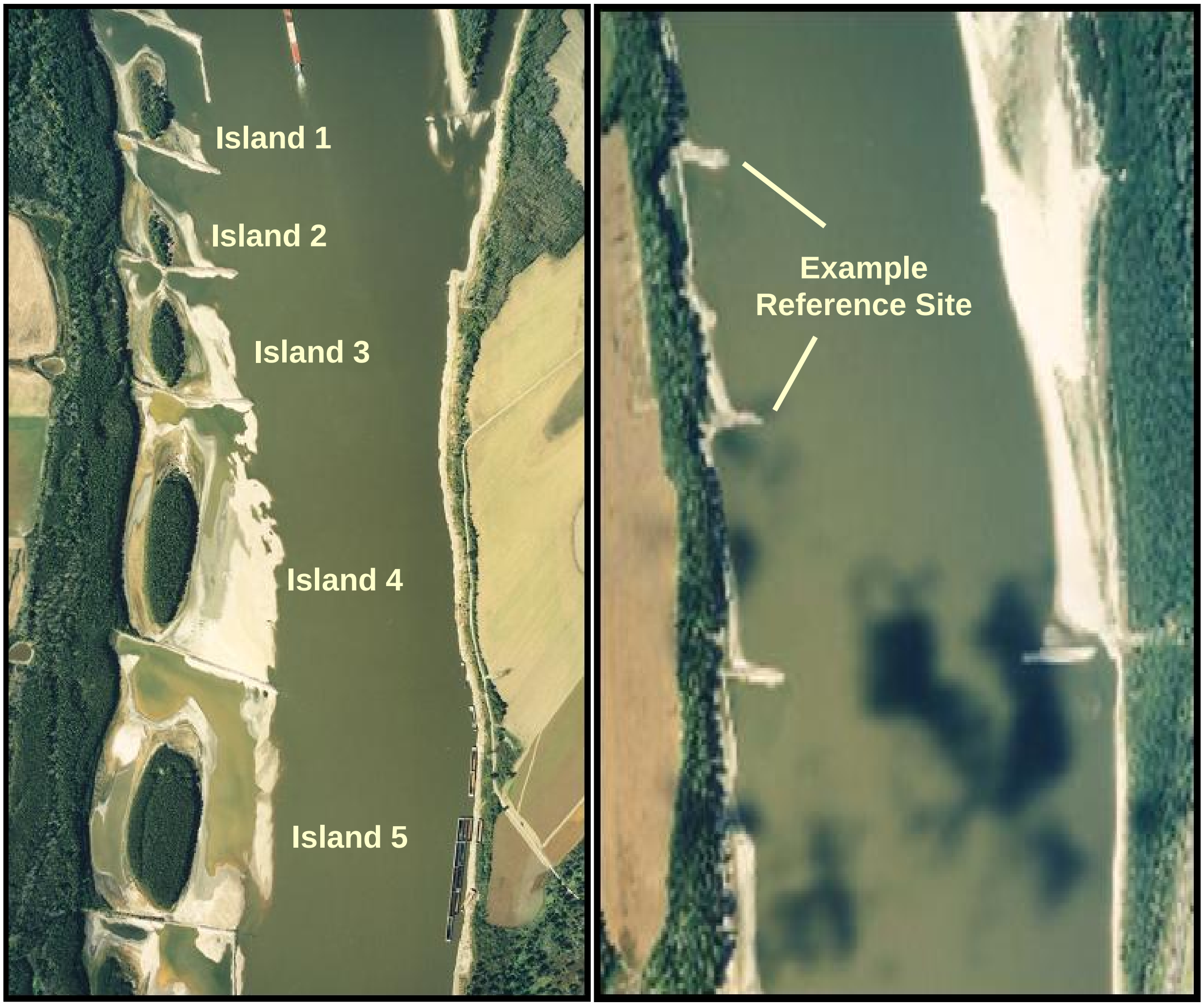


Figure 4-2. Island sites (left) and an example of a reference site (right). Five island and five non-island (reference) dike field sites in the Middle Mississippi River located between RM 107.4 and 98.9 were examined between October 2004 and August 2006. The Mile 100 Islands are located between RM 100.6 and 98.9. The reference sites are located between dikes 107.4-107.2, 107.2-106.8, 105.0-104.7, 104.7-104.4, and 100.6-100.4.

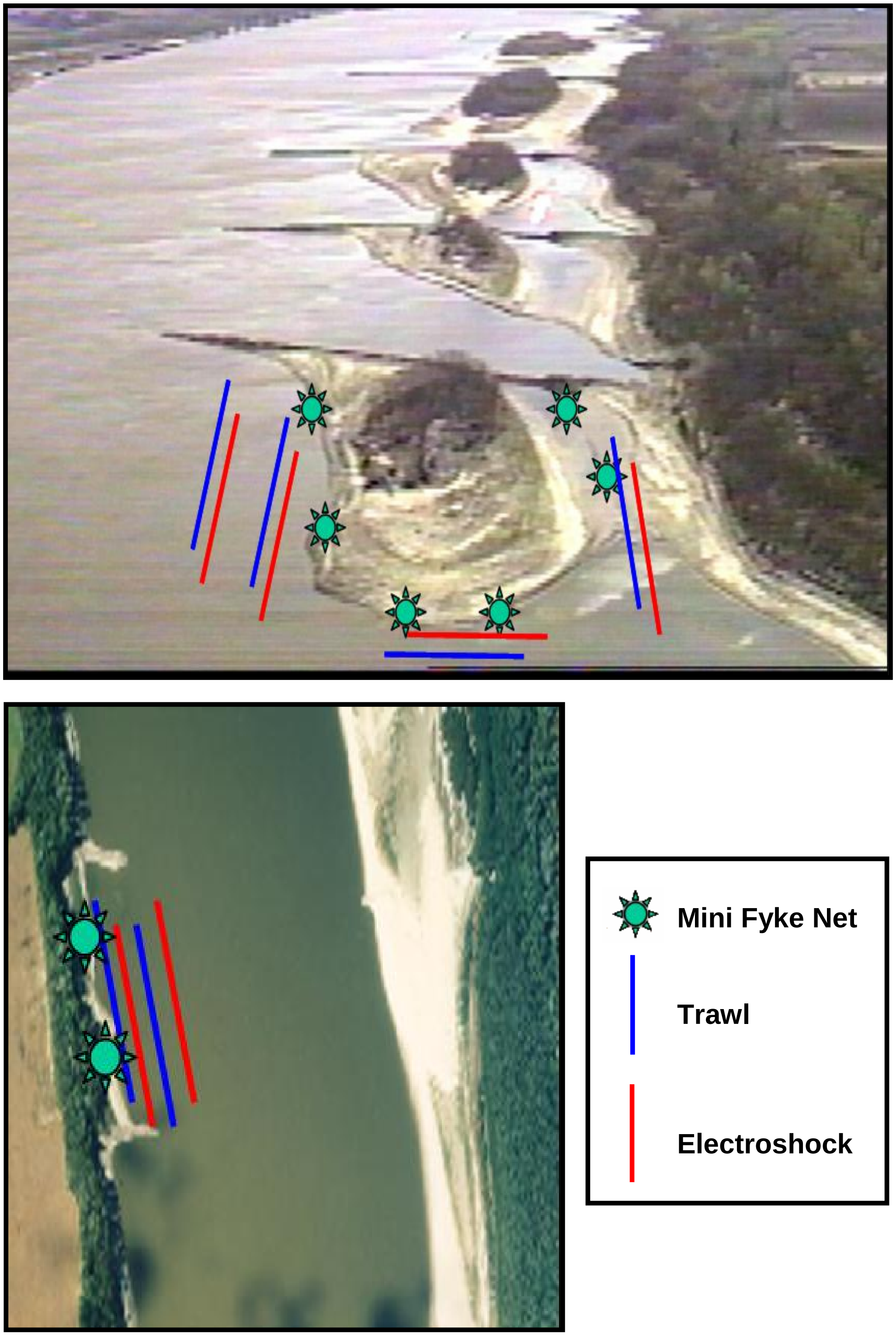


Figure 4-3. Approximate locations of mini fyke net placement, trawl transects, and electrofishing transects**.**

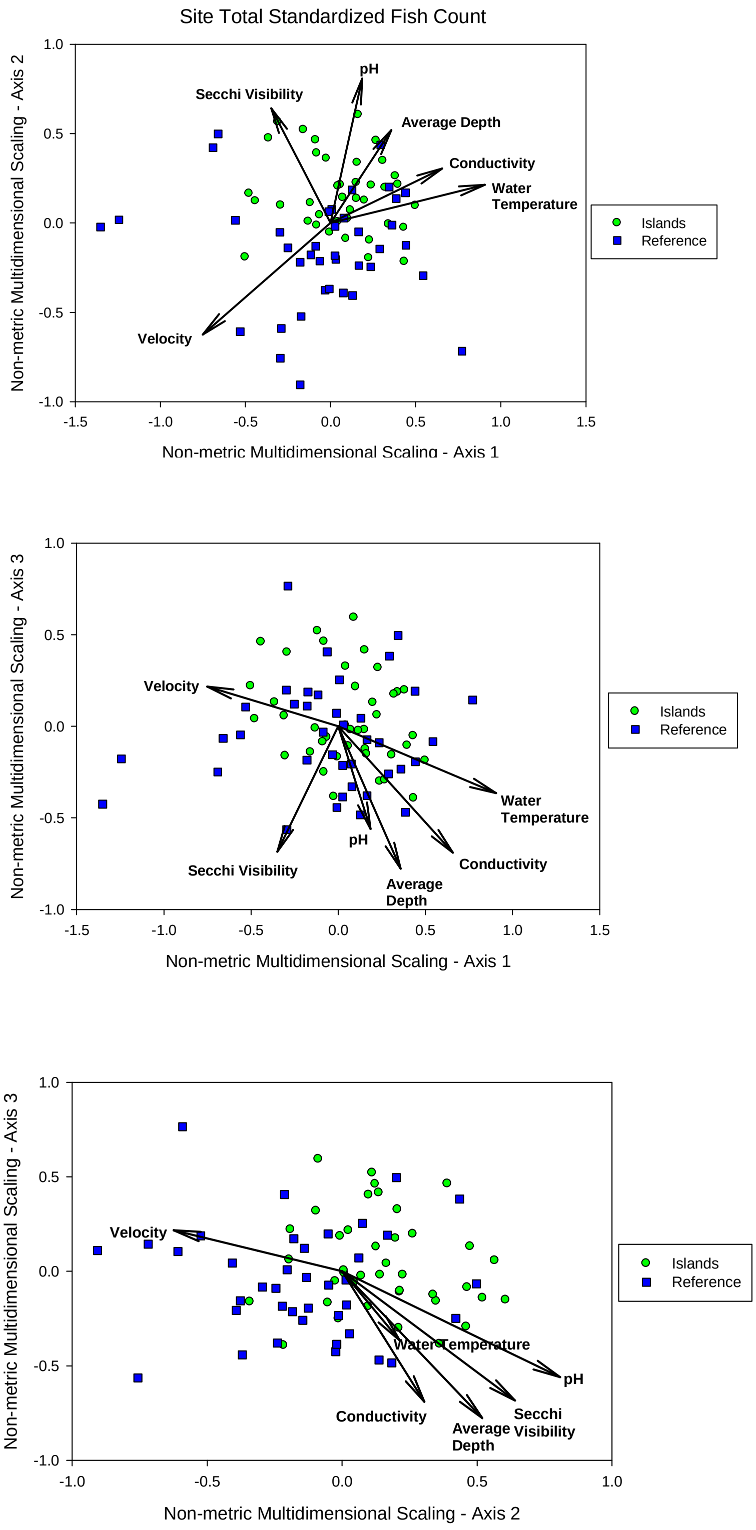


Figure 4.4. Total standardized fish count NMDS graphs illustrating the significant difference between islands and reference sites**.**

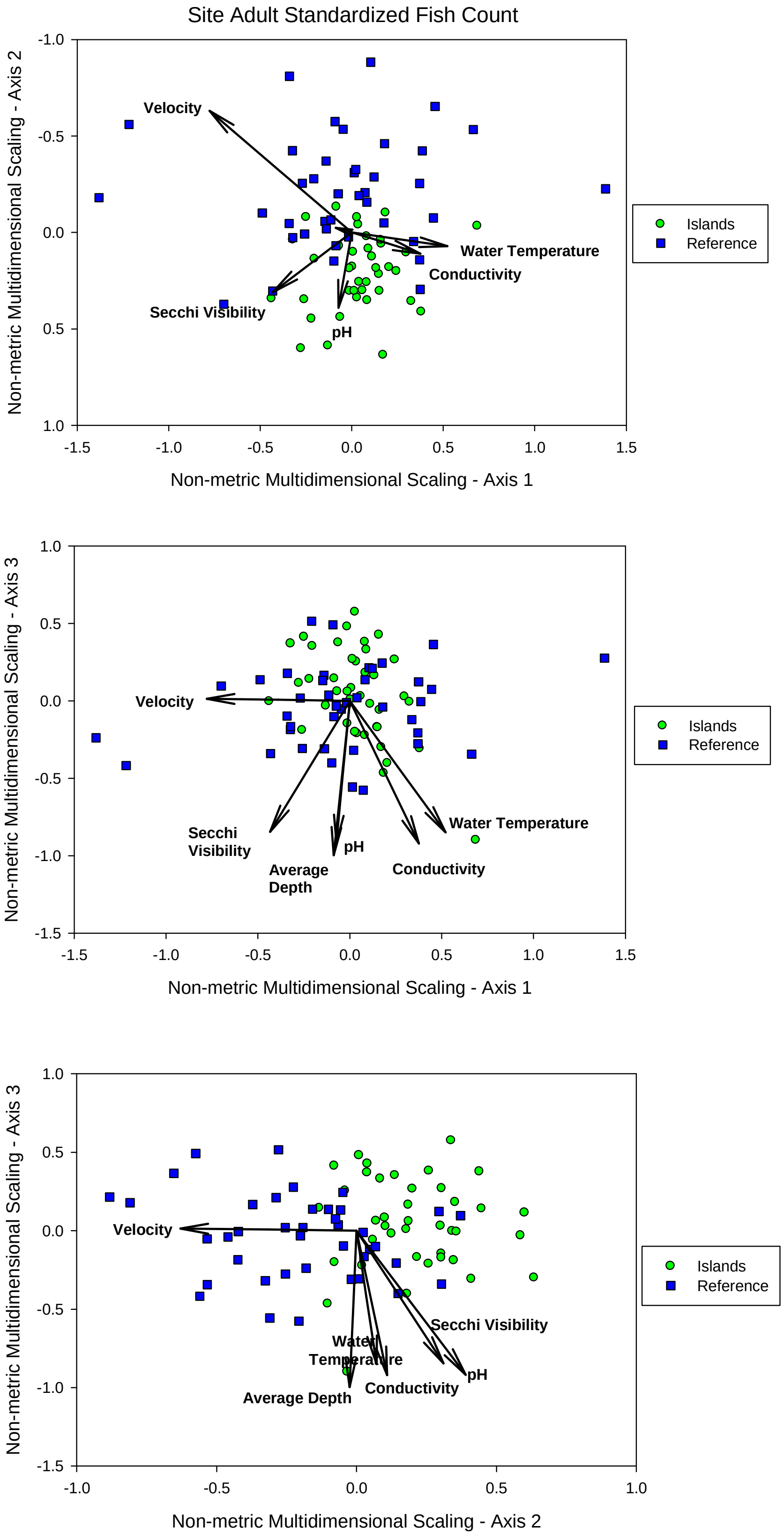


Figure 4.5 Adult standardized fish count NMDS graphs illustrating the significant difference between islands and reference sites**.**

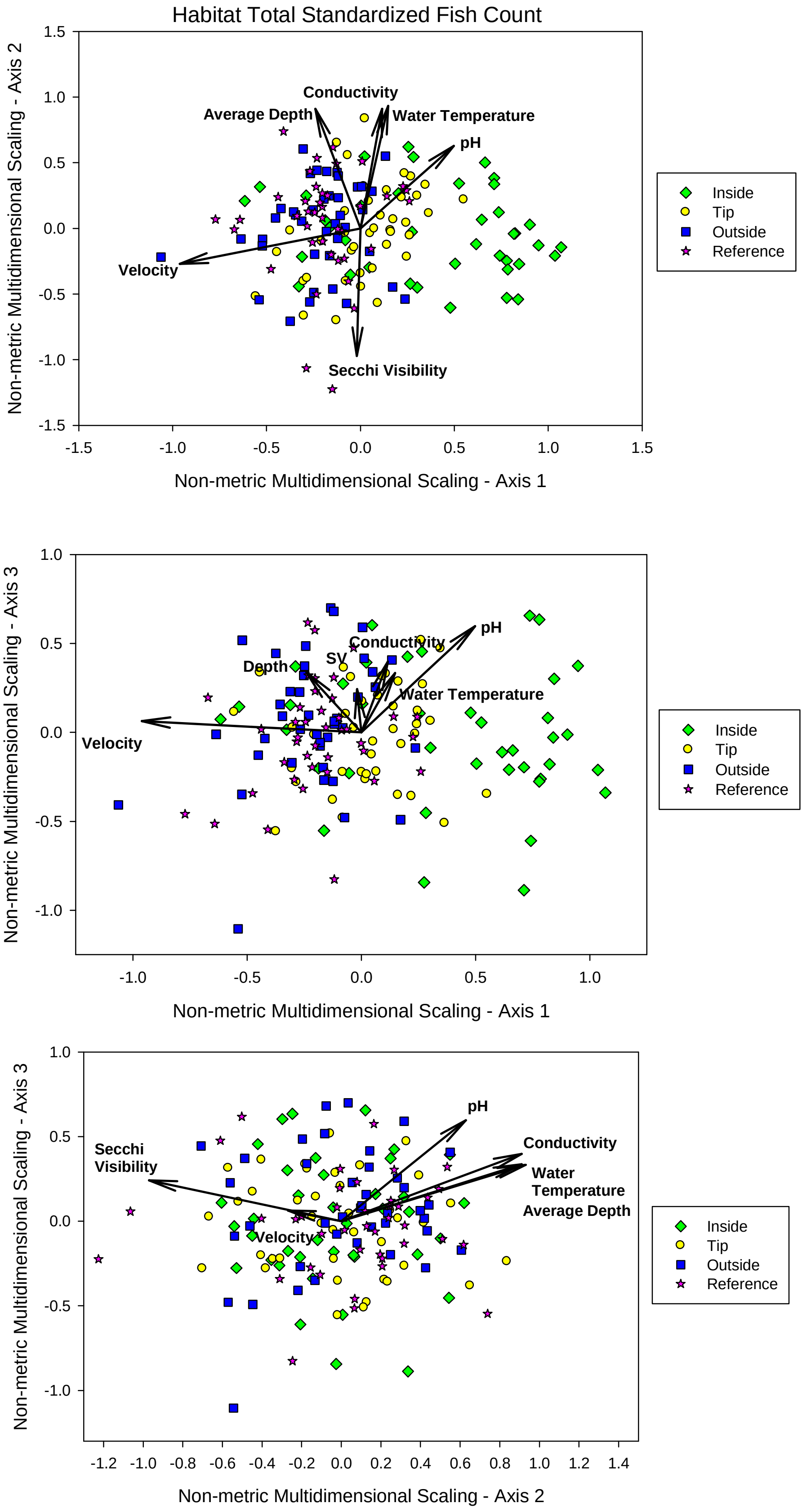
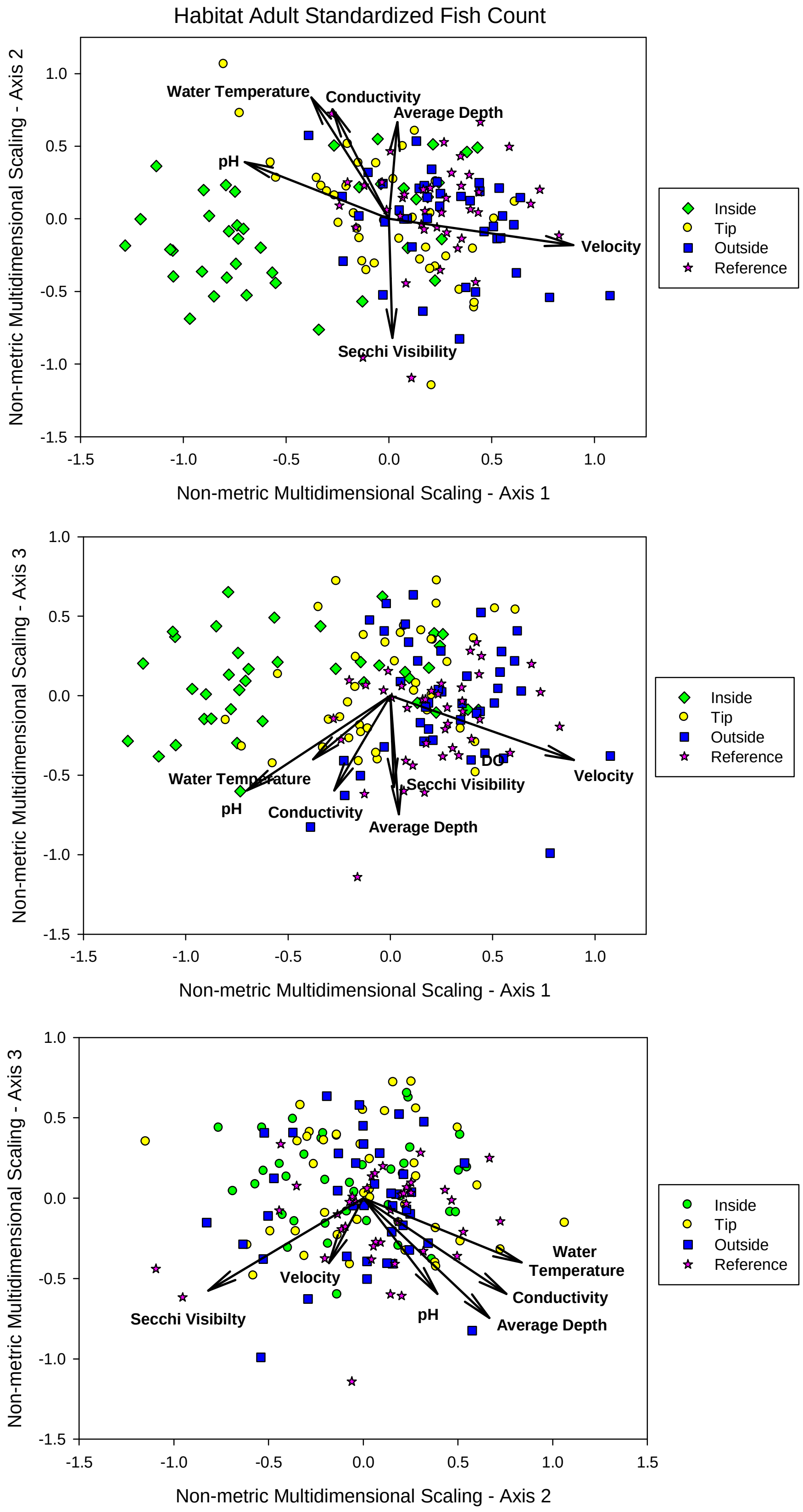


Figure 4.6. Total standardized fish count NMDS graphs illustrating the significant difference between habitats and reference sites**.**



Habitat Adult Standardized Fish Count

Figure 4.7. Adult standardized fish count NMDS graphs illustrating the significant difference between habitats and reference sites.

Table 4.1. Number of adult and YOY fish species collected at island habitat, islands, and reference sites. Bold indicates groupings comprising at least 5% of the collection at each location.

Family	Common Name	Scientific Name	Inside	Tip	Outside	Island	Reference
Petromyzontidae	Chestnut lamprey	*Ichthyomyzon castaneus*	2/0	0/0	0/0	2/0	0/0
Acipenseridae	Shovelnose sturgeon	*Scaphirhynchus platorynchus*	0/0	1/0	9/3	10/3	1/1
Polyodontidae	Paddlefish	*Polyodon spathula*	0/0	1/11	0/48	1/59	0/6
Lepisosteidae	Shortnose gar	*Lepisosteus platostomus*	51/0	52/0	79/2	181/2	88/0
	Longnose gar	*Lepisosteus osseus*	0/2	0/2	0/0	0/4	1/0
	Spotted gar	*Lepisosteus oculatus*	0/0	0/0	0/0	0/0	0/1
Amiidae	Bowfin	*Amia calva*	0/0	1/0	0/0	1/0	0/0
Clupeidae	**Gizzard shad**	***Dorosama cepedianum***	239/191	**403**/**407**	322/93	**964**/691	**443**/**321**
	Skipjack herring	*Alosa chrysochloris*	0/0	1/55	3/4	4/59	3/15
	Threadfin shad	*Dorosoma petenense*	0/1	2/5	5/0	7/6	1/0
Hiodontidae	Goldeye	*Hiodon alosoides*	0/20	0/26	5/155	5/201	6/71
	Mooneye*	*Hiodon tergisus*	0/42	0/1	0/14	0/57	0/1
Cyprinidae	**Channel shiner**	***Notropis wickliffi***	96/**863**	135/**2474**	197/**5391**	428/**8728**	225/**2979**
	Emerald shiner	***Notropis atherinoides***	**792**/**654**	**1088**/**449**	**2755**/**1586**	**4635**/**2689**	**2735**/**328**
	Red shiner	***Cyprinella lutrensis***	**865**/12	**473**/16	**1412**/49	**2750**/77	**483**/1
	Silverband shiner	***Notropis shumardi***	**957**/148	**151**/37	84/25	**1192**/210	45/3
	Speckled chub	***Macrhybopsis aestivalis***	34/3	85/92	**778**/238	**897**/333	136/106
	River shiner	*Notropis blennius*	43/13	73/76	132/54	248/143	60/10
	Common carp	*Cyprinus carpio*	148/2	24/0	60/1	232/3	86/3
	Bullhead minnow	*Pimephales vigilax*	106/6	58/24	30/11	194/41	10/7
	Sicklefin chub*	*Macrhybopsis meeki*	4/1	13/55	41/21	58/77	27/3
	Silver chub*	*Macrhybopsis storeriana*	3/5	1/30	7/16	11/51	3/9
	Silver carp	*Hypophthalmichthys molitrix*	1/53	4/1	0/2	5/56	8/0
	Bluntnose minnow	*Pimephales notatus*	0/18	1/11	0/8	1/37	0/0
	Spotfin shiner	*Cyprinella spilopterus*	10/2	5/1	3/2	18/5	3/0
	Sand shiner	*Notropis stramineus*	1/8	0/10	0/3	1/21	0/12
	Bighead carp	*Hypophthalmichthys nobilis*	0/21	0/0	0/0	0/21	0/0
	Redfin shiner	*Lythrurus umbratilis*	0/5	0/0	0/11	0/16	0/0
	Sturgeon chub*	*Macrhybopsis gelida*	0/2	0/1	0/10	0/13	2/20

	Pugnose minnow*	*Opsopoedus emiliae*	0/2	0/3	0/3	0/8	1/1
	Mississippi silvery minnow*	*Hybognathus nuchalis*	0/1	1/0	3/0	4/1	1/1
	Fathead minnow	*Pimephales promelas*	0/0	3/0	1/0	4/0	1/0
	Grass carp	*Ctenopharyngodon idella*	0/2	0/0	1/0	1/2	1/0
	Central stoneroller	*Campostoma pullum*	0/0	0/1	0/2	0/3	0/0
	Plains minnow	*Hybognathus placitus*	0/2	0/0	0/0	0/2	0/2
	Mimic shiner	*Notropis volucellus*	0/0	0/0	0/2	0/2	0/0
	Bleeding shiner	*Luxilus zonatus*	0/1	0/0	0/0	0/1	0/1
	Ghost shiner*	*Notropis buchanani*	0/1	0/0	0/0	0/1	0/0
	Golden shiner	*Notemigonus crysocleucas*	0/0	0/0	0/0	0/0	0/3
Catostomidae	**River carpsucker**	***Carpiodes carpio***	22/**609**	15/72	46/90	83/771	40/55
	Smallmouth buffalo	*Ictiobus bubalus*	5/5	2/0	12/4	19/9	6/0
	Bigmouth buffalo	*Ictiobus cyprinellus*	12/2	2/0	3/1	17/3	1/0
	Black buffalo	*Ictiobus niger*	0/0	4/0	7/0	11/0	8/0
	Blue sucker	*Cycleptus elogatus*	0/1	0/1	0/1	0/3	0/6
	Shorthead redhorse	*Moxostoma macrolepidotum*	0/0	0/1	0/0	0/1	0/2
	Quillback	*Carpiodes cyprinus*	0/0	0/0	0/1	0/1	0/1
Ictaluridae	**Channel catfish**	***Ictalurus punctatus***	130/28	100/56	**520**/**802**	**750**/886	**393**/**731**
	Blue catfish	*Ictalurus furcatus*	1/3	4/19	20/242	25/264	16/139
	Flathead catfish	*Pylodictis olivaris*	4/0	1/3	8/2	13/5	6/0
	Freckled madtom	*Noturus nocturnus*	0/1	0/0	0/10	0/11	1/2
	Tadpole madtom	*Noturus gyrinus*	0/0	0/0	2/0	20/	0/0
	Slender madtom	*Noturus exilis*	0/0	0/0	1/0	1/0	0/0
Osmeridae	Rainbow smelt	*Osmerus mordax*	0/1	0/0	0/0	0/1	0/0
Aphredoderidae	Pirate perch	*Aphredoderus sayanus*	0/1	0/0	0/0	0/1	0/0
Fundulidae	Blackstripe topminnow	*Fundulus notatus*	0/0	0/0	0/0	0/0	1/0
Poeciliidae	**Mosquitofish**	***Gambusia affinis***	**1015**/49	15/2	4/0	**1034**/51	10/0
Atherinidae	Inland silverside	*Menidia beryllina*	8/0	1/0	2/0	11/0	0/0
	Brook silverside	*Labidesthes sicculus*	1/0	2/0	3/0	6/0	7/0
Moronidae	White bass	*Morone crysops*	1/21	9/37	4/35	14/93	2/19
Centrarchidae	**Orangespotted sunfish**	***Lepomis humilis***	**402**/52	25/19	6/0	433/71	20/5
	Bluegill	*Lepomis macrochirus*	70/0	14/0	11/0	95/0	7/0
	White crappie	*Pomoxis annularis*	14/2	6/3	2/0	22/5	0/4
	Green sunfish	*Lepomis cyanellus*	7/0	3/0	1/0	11/0	1/0
	Black crappie	*Pomoxis nigromaculatus*	5/0	0/0	0/1	5/1	1/0

	Warmouth	*Lepomis gulosus*	1/0	0/0	0/0	1/0	0/0
	Spotted bass	*Micropterus punctulatus*	0/0	0/0	0/1	0/1	0/0
Percidae	Sauger	*Stizostedion canadense*	0/86	6/35	2/50	8/171	0/42
	Western sand darter*	*Ammocrypta clara*	0/1	0/0	0/3	0/4	0/2
	River darter*	*Percina shumardi*	0/0	0/0	1/0	1/0	0/0
	Mud darter	*Etheostoma asprigene*	0/0	0/0	0/0	0/0	1/0
Sciaenidae	**Freshwater drum**	***Aplodinotus grunniens***	77/124	60/**1042**	96/**1576**	233/**2742**	72/**1257**
Totals			5127/3067	2845/5078	6678/10,573	33,368/18,718	4963/6170

*Missouri Species of Conservation Concern

Table 4.2. Number of individual, species, and families collected over a two-year period at islands, habitats, and reference sites.

	Inside			Tip			Outside			Island			Reference		
	Individuals	**Species**	**Families**	**Individuals**	**Species**	**Families**	**Individuals**	**Species**	**Families**	**Individuals**	**Species**	**Families**	**Individuals**	**Species**	**Families**
Total	8194	53	15	7923	48	15	17,251	56	14	33,368	67	18	11,133	55	15
Adult	5127	33	11	2845	39	14	6678	40	13	14,650	48	16	4963	42	14
YOY	3067	44	13	5078	35	12	10,573	41	12	18,718	55	16	6170	37	12

*YOY – Young-of-the-Year

Table 4.3. Results of indicator species analysis of total standardized count for islands and reference sites.

Common Name	**Indicator Values**		**p**
	Island	**Reference**	
Bighead carp	**20**	0	0.005
Bigmouth buffalo	**22**	0	0.005
Bluegill	**48**	4	0.000
Bluntnose minnow	**18**	0	0.012
Bullhead minnow	**61**	6	0.000
Channel shiner	**53**	33	0.030
Green sunfish	**20**	0	0.013
Inland silverside	**15**	0	0.029
Mooneye	**16**	0	0.039
Mosquitofish	**52**	3	0.000
Orangespotted sunfish	**56**	3	0.000
Red shiner	**54**	34	0.013
River carpsucker	**59**	20	0.000
Silver carp	**30**	2	0.002
Silverband shiner	**62**	12	0.000
Smallmouth buffalo	**28**	2	0.006
Speckled chub	**55**	32	0.008
Threadfin shad	**15**	0	0.049
White bass	**43**	13	0.008
White crappie	**26**	2	0.013

Table 4.4. Results of indicator species analysis of adult standardized count for islands and reference sites.

Common Name	**Indicator Values**		**p**
	Island	**Reference**	
Bigmouth buffalo	**17**	0	0.0226
Bluegill	**48**	4	0.0001
Bullhead minnow	**62**	4	0.0001
Channel shiner	**48**	16	0.0063
Green sunfish	**20**	0	0.0155
Inland silverside	**15**	0	0.0242
Mosquitofish	**52**	3	0.0001
Orangespotted sunfish	**58**	1	0.0001
Red shiner	**54**	34	0.0121
River carpsucker	**47**	16	0.0063
River shiner	**44**	17	0.0144
Sauger	**18**	0	0.0122
Silverband shiner	**58**	12	0.0002
Speckled chub	**53**	31	0.0134
White bass	**27**	1	0.0028
White crappie	**30**	0	0.0002

Table 4.5. Results of indicator species analysis of total standardized count for habitat types.

Common Name	**Indicator Values**				**p**
	Inside	**Outside**	**Tip**	**Reference**	
Bighead carp	**21**	0	0	0	0.0001
Mosquitofish	**43**	0	3	2	0.0001
Orangespotted sunfish	**42**	1	7	2	0.0001
Silver carp	**21**	0	2	1	0.0004
Bluegill	**21**	4	4	3	0.0026
White crappie	**11**	0	3	2	0.0352
Channel catfish	5	**33**	16	29	0.0001
Speckled chub	3	**32**	18	20	0.0003
White bass	2	11	**19**	8	0.0287
Blue catfish	0	24	3	**27**	0.0002
Gizzard shad	9	18	22	**28**	0.0100
Goldeye	3	19	2	**21**	0.0148
Sturgeon chub	1	2	0	**10**	0.0372

Table 4.6. Results of indicator species analysis of adult standardized count for habitat types.

Species	**Indicator Value**				**p**
	Inside	**Outside**	**Tip**	**Reference**	
Orangespotted sunfish	**45**	1	6	1	0.0001
Mosquitofish	**44**	0	3	2	0.0001
Bluegill	**21**	4	4	3	0.0022
White crappie	**15**	1	2	0	0.0016
Speckled chub	2	**31**	12	21	0.0004
Channel catfish	5	**31**	12	28	0.0005
Blue catfish	0	**17**	1	10	0.0070
White bass	0	3	**11**	1	0.0240
Sauger	0	1	**9**	0	0.0218
Emerald shiner	13	20	17	**30**	0.0095
Gizzard shad	4	18	18	**27**	0.0059