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A Multifactor Ecosystem Assessment of Wetlands Created Using a Novel Dredged Material Placement Technique in the Atchafalaya River, Louisiana

An Engineering With Nature Demonstration Project

Jacob F. Berkowitz, Sung-Chan Kim, Nathan R. Beane, Darrell
E. Evans, Elizabeth Summers, Burton Suedel, Maik Flanagan,
and Jeff Corbino

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Abstract

A multifactor ecosystem assessment of dredged material supported wetlands was conducted within the Atchafalaya River, Louisiana. The assessment included evaluations of (1) geomorphic evolution, (2) ecosystem classification and distribution, (3) floral communities, (4) avian communities, (5) aquatic invertebrates, (6) soils and biogeochemical activity, and (7) hydrodynamic and sediment transport processes. Results indicate that the innovative use of dredged materials in a riverine environment supports wetland formation and expansion while providing habitat, hydrologic, and biogeochemical functions. The strategic placement of dredged materials in locations that mimic natural process promoted additional ecological benefits, especially regarding wading bird and infaunal habitat, thus adhering to Engineering With Nature (EWN) processes. The multifactor approach improved the wetland assessment, allowing for a comprehensive ecosystem-level analysis of a diverse array of ecosystem components and functions.

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Preface

This study was conducted for the Engineer Research and Development Center's Engineering With Nature initiative and was funded by the U.S. Army Corps of Engineers Dredging Operations and Environmental Research (DOER) Program. The Program Manager was Dr. Todd Bridges, CEERD-EMD.

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COL Bryan S. Green was Commander of ERDC, and Dr. David W. Pittman was the Director.

Acronyms and Abbreviations

ACNI	<i>Acer nigrum</i>
ACRU	<i>Acer rubrum</i>
ANOVA	Analysis of Variance
AOI	Area of Interest
BUMP	Beneficial Use of Dredged Material Program
CELA	<i>Celtis laevigata</i>
CH3D	Curvilinear Hydrodynamics in 3 Dimensions
CHL	Coastal and Hydraulics Laboratory
dbh	Diameter at Breast Height
DM	Dredged Material
DOER	Dredging Operations and Environmental Research
EAB	Emergent Aquatic Bed
EL	Environmental Laboratory
ENC	Electronic Navigation Chart
ERDC	Engineer Research and Development Center
EWN	Engineering With Nature
FRPE	<i>Fraxinus pennsylvanica</i>
GDAL	Geospatial Data Abstraction Library
GIS	Geographic Information System
GIWW	Gulf Intracoastal Water Way
GTRAN	General Transport
HEC	Hydrologic Engineering Center
KCl	Potassium Chloride

LOI	Loss on Ignition
MBN	Microbial Biomass Nitrogen
MOCE	<i>Morella cerifera</i>
MVN	New Orleans District
N	Nitrogen
NH ₄ -N	Ammonium
NO ₃ ⁻	Nitrate
NO ₃ -N	Nitrate Nitrogen
NOAA	National Oceanic and Atmospheric Administration
PLOC	<i>Platanus occidentalis</i>
PODE	<i>Populus deltoides</i>
QUNI	<i>Quercus nigra</i>
RMA	Resource Management Associates
SANI	<i>Salix nigra</i>
SRP	Soluble Reactive Phosphorous
TADI	<i>Taxodium distichum</i>
TC	Total Carbon
TDN	Total Dissolved Nitrogen
TN	Total Nitrogen
TP	Total Phosphorous
TRCE	<i>Triadica cebifera</i>
USACE	U.S. Army Corps of Engineers
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
WY	Water Year

Unit Conversion Factors

Multiply	By	To Obtain
bars	100	kilopascals
feet	0.3048	meters
hectares	1.0 E+04	square meters
miles (nautical)	1,852	meters
miles (U.S. statute)	1,609.347	meters

Executive Summary

A multifactor ecosystem assessment of dredged material supported wetlands was conducted within the Atchafalaya River, Louisiana. Results indicate that the innovative use of dredged materials in a riverine environment supports wetland formation and expansion while providing habitat, hydrologic, and biogeochemical functions. The strategic placement of dredged materials in locations that mimic natural processes promotes additional ecological benefits, especially regarding wading bird and infaunal habitat, thus adhering to Engineering With Nature (EWN) principles. The multifactor approach improved the wetland assessment, allowing for a comprehensive ecosystem-level analysis of a diverse array of ecosystem components and functions. Sample locations included Island E, a traditional dredged material supported wetland island; Horseshoe Bend Island, a dredged material supported wetland island created using EWN processes; and Middle Island, a mature, naturally formed wetland island.

A multiple-ecosystem assessment was undertaken at Horseshoe Bend Island, including evaluation of (1) geomorphic evolution, (2) ecosystem classification and distribution, (3) floral communities, (4) avian communities, (5) aquatic invertebrates, (6) soils and biogeochemical activity, and (7) hydrodynamic and sediment transport processes. Results were compared to observations made at the other two sampling locations. Horseshoe Bend Island and Island E contained four distinct wetland classes while Middle Island contained only two. Horseshoe Bend Island exhibited the highest plant species richness (82 species; 85% native vegetation). Surveys of bird populations on Horseshoe Bend Island yielded a high number of breeding wading birds in dense, mixed-species nesting colonies, including 79% juveniles, indicating that the island supports a productive nesting colony.

Both dredged material supported islands contained significantly higher amounts of fine particles and organic matter than Middle Island, resulting in significantly higher infaunal abundance. Further, both dredged material supported islands provide nutrient cycling functions, including estimated nitrogen removal rates of 1577–1645 kg NO₃-N yr⁻¹, indicating the ability to improve water quality. Results of a Curvilinear Hydrodynamics in 3 Dimensions (CH3D) model proved strongly related to hydrological forcing. When linked with a General Transport (GTRAN) model, sediment transport rates indicated possible deposition toward the northern shore

and erosion from the southern shore of Horseshoe Bend Island. Sedimentation rates remained elevated during the wet season with an estimated maximum volume transport rate of $9.66 \text{ m}^3/\text{m}/\text{s}$ and $0.71 \text{ m}^3/\text{m}/\text{s}$ as suspended and bedload sediments, respectively.

1 Introduction

1.1 Background

The following section outlines Engineering With Nature (EWN) principles and describes the applicability of dredged material to support wetland ecosystems. The beneficial use of dredged material is consistent with EWN, a U.S. Army Corps of Engineers (USACE) initiative enabling more sustainable delivery of economic, social, and environmental benefits associated with water resources infrastructure (Bridges et al. 2014; Gerhardt-Smith and Banks 2014; EWN 2016). EWN concepts addressed through the current project include (1) the use of science and engineering to produce operational efficiencies supporting sustainable delivery of project benefits and (2) the use of natural processes to maximum benefit, thereby reducing demands on limited resources, minimizing the environmental footprint of projects, and enhancing the quality of project benefits.

The geographical extent of wetlands continues to decrease at national and regional scales, including major wetland losses across southern Louisiana (Barras et al. 2003; USACE 2004). Louisiana contains approximately 40% of the wetlands in the contiguous United States and continues to exhibit rapid wetland losses representing nearly 80% of the total wetland losses in the lower 48 states (Edwards and Proffitt 2003; Bourne 2000; Turner 1997; Day et al. 2007). Additionally, estimated forested wetland losses across the lower Mississippi Alluvial Valley exceed 74% with only 2.8 of an original 10 million ha remaining today (Gardiner and Oliver 2005; The Nature Conservancy 1992; King et al. 2006). Wetland losses have been accompanied by a decrease in wetland functions (Johnston 1994; Bedford and Preston 1988).

Wetlands in southern Louisiana provide a number of ecosystem functions, including (1) the maintenance of floral and faunal habitat, (2) biogeochemical functions, and (3) hydrologic functions (Boesch et al. 1994; Zedler 2000; Llewellyn et al. 1996). Wetlands in the region provide important habitat for migratory waterfowl and forest songbirds (King and Keeland 1999; Twedt and Somershoe 2009). Wetland habitats also enhance invertebrate communities that sustain recreational and commercial fisheries, including shrimp (Engle 2011; Boesch and Turner 1984; Chesney et al. 2000). Biogeochemical functions in Atchafalaya River wetlands include

nutrient cycling, carbon accumulation, and denitrification (Novitski et al. 1996; Bedford and Preston 1988). Specifically, Stern et al. (1991) and others demonstrated that wetlands in the region removed nitrate, soluble reactive phosphorus, and total suspended solids from the water column. The ability of wetlands to reduce nutrient loads in river waters has the capacity to improve water quality and decrease hypoxia in the Gulf of Mexico (Mitsch et al. 2001; Lindau et al. 2008; Rabalais et al. 1994). Additionally, wetlands across the region demonstrate high levels of carbon sequestration, impacting the greenhouse gas balance (Rabenhorst 1995; DeLaune and Pezeshki 2003). Southern Louisiana wetlands also provide a variety of hydrologic functions. For example, Wamsley et al. (2010) and others have investigated the ability of wetlands to reduce storm surge. Also, Costanza et al. (2008) identified significant relationships between wetland area and hurricane damage within the region.

In Louisiana, wetland distribution and functional losses have been linked to a lack of sediment inputs among other causes (e.g., development, agriculture and forestry, and flood control) (Day et al. 2007; Templet and Meyer-Arendt 1998). USACE conducts dredging activities to maintain navigation channels in the lower Atchafalaya River. Annual dredging projects within the lower Atchafalaya River produce over 9.9 million m³ of material at a cost of \$30 million dollars (Corbino 2014). These dredging activities remove sediment from navigation channels, which is then available for beneficial uses, including wetland creation and expansion (Boustany 2010).

1.2 Objective

The project objective was to demonstrate how dredged material can be used beneficially to nourish a naturally forming river island. Biology, ecology, geography, and hydrodynamics were examined to catalog the island's maturation for determining the effectiveness of this individual project in terms of restoring, creating, enhancing, and protecting the coastal Louisiana landscape, consistent with EWN principles.

1.3 Approach

The current study applies historical geospatial imaging and classification, hydrodynamic analysis, and a multifactor ecosystem assessment of dredged material supported wetlands located within the Atchafalaya River,

Louisiana. Assessment factors include examination of (1) historical geo-spatial imaging and landmass classification, (2) ecosystem classification and geomorphology, (3) floral communities, (4) avian communities, (5) aquatic invertebrates, (6) soils and biogeochemical activity, and (7) hydro-dynamic modeling of the study area. Initial field work at Horseshoe Bend Island (performed in August 2013; Berkowitz et al. 2014) provided a basis to conduct the current study. Following a description of the study area, separate sections address research methods and results regarding each factor. An analysis and discussion of findings is also provided along with opportunities for future research.

2 Study Area

The study area is located at Horseshoe Bend, a segment of the Atchafalaya River and Bayous Chene, Boeuf, and Black Federal navigation channel located in the lower Atchafalaya River approximately 32 km south of Morgan City, LA (Figure 1). Although dredging of the navigation channel received original authorization in 1910, active maintenance dredging of Horseshoe Bend was not required until 1990 because natural channel depths exceeded authorized dimensions. Beginning in 1990 and proceeding thru 2013, 17 maintenance dredging events within this segment occurred at a return interval of 1 to 3 years with each event resulting in the removal of between 0.38 and 2.0 million m³ of shoal material (Figure 2).

Figure 1. Study location within the lower Atchafalaya River, St. Mary Parish, LA (indicated by the *rectangle*).

Figure 2. Dredged material placement history within the study reach between 1990 and 1999 (University of New Orleans 2002). Note the presence of Island E (*rectangle*) near the top of the image; Horseshoe Bend Island (*circle*) was not present in 1999.

During the 1990s, placement of shoal material dredged from Horseshoe Bend occurred at eight wetland development sites located along the riverbank adjacent to the channel (Figure 2). These efforts resulted in the creation of 23 ha Island E during dredging events in 1995 and 1996. Capacity of these placement sites was nearly exhausted by 1999. Thus, to meet the anticipated disposal requirements for future channel maintenance, the USACE New Orleans District (MVN) evaluated three placement alternatives: (1) converting the wetland development sites into upland disposal areas; (2) open-water placement of dredged material via a long-distance pipeline into the open waters of the Atchafalaya Bay; and (3) mounding material at mid-river open water

placement sites within a 142 ha area immediately adjacent to the navigation channel and upriver of a small, naturally forming submerged shoal. Environmental and cost concerns precluded the selection of alternatives 1 and 2. The third alternative was selected on a trial basis to investigate the impacts of mid-river placement on shoaling trends downriver of the site.

Beginning in 2002, strategic placement of the sediment dredged from Horseshoe Bend from the adjacent navigation channel occurred at the mid-river open water placement area. Placement of between 0.38 to 1.4 million m³ of sediment was conducted every 1 to 3 years, which influenced and contributed to the development of an approximately 35 ha mid-river island (Figures 3 and 4). The created island is referred to herein as Horseshoe Bend Island. To compare wetland conditions and functions across a range of locations, this study also selected a traditionally created dredged material island (i.e., Island E) less than 2.4 km north of Horseshoe Bend Island and a naturally formed wetland island (Middle Island) approximately 32 km north of the Horseshoe Bend. Middle Island was naturally formed during the mid-1970s and has persisted as a 142 ha island since that time.

Figure 3. Imagery displaying island location prior to dredged material (DM) placement and subsequent formation (1992 and 1998 images), establishment, and growth since strategic dredged material placement began in 2002. (Imagery provided by USACE MVN.)

Figure 4. Horseshoe Bend Island, a 35 ha island supported by strategic dredged material placement. Note the small dredged material mound (*circled*) placed immediately upriver (north) of the island. Image dated 28 October 2012.

3 Horseshoe Bend Island Image Analysis and Geomorphic Evolution

3.1 Introduction

Under the USACE Beneficial Use of Dredged Material Program (BUMP), the MVN collects aerial photography for select dredge disposal areas on an annual basis. The imagery is used to monitor the environmental effects of dredge disposal activities. This imagery additionally allows for the quantification of the island size and geomorphology, supporting ecosystem classification efforts as described in Section 4. Using this information, the MVN analyzed the geomorphic development of Horseshoe Bend Island from 2007 to 2014.

3.2 Methods

Since 2007, the imagery has been collected in a digital format and loaded as a mosaic into the Geospatial Data Abstraction Library (GDAL), allowing easy access for visualization and analysis. Resolution of the imagery is generally 0.61 m, which is sufficient to distinguish land, water, and vegetation. The GDAL is a C library frequently used for raster analysis. GDAL provides an abstraction layer for input-output operations on numerous raster formats, allowing a user to write a program to work with many different raster sources without a substantial code rewrite. ArcGIS software was used to create the maps and perform analysis of the above-water extent of Horseshoe Bend Island. ArcGIS includes a middleware product that allows a database such as Oracle to store very large raster datasets, such as the BUMP series. Imagemagick is a C library with command-line applications to manipulate images; it was also used during the image analysis process.

The analysis of Horseshoe Bend Island examined available imagery to differentiate between landscape features. Using ArcGIS software, an area-of-interest (AOI) rectangle was created that covered the analysis area. Band 3 from each of the available BUMP imagery datasets was clipped to the AOI. A script was developed to automate the process. Even though there can be substantial differences in pixel values of the same land-cover types from year to year in the BUMP imagery, the objective was to develop a quick, consistent workflow that would be understandable and provide the project team with an estimate of the island's location and general growth trends.

A region-growing algorithm was implemented as the core component of the analysis work flow. The program was written in C++ using the GDAL library for input-output operations on the imagery. During initialization, a series of real-world coordinates from known water locations were mapped to pixel coordinates and added to a queue. During each iteration of the algorithm, the head pixel was removed from the queue; and the values of each of its eight neighboring pixels were subtracted from the head pixel's value and compared to a threshold value. Neighboring pixels with a difference lower than the threshold signified that the pixel was similar to the head pixel. These similar pixels were appended to the queue, and the algorithm proceeded through the next iteration. A check was added to ensure that a pixel was compared only once in this manner. The algorithm was completed when the queue was emptied.

To improve the results of the algorithm, a number of preprocessing and post-processing steps were required. Because pixels representing water are typically surrounded by other pixels representing water and the distinction between values that are not water are inconsequential for this analysis, a filter was applied to remove noise from the imagery. Input imagery was processed in ArcGIS through a series of Majority Filters, which replace a pixel value with the mode of the eight surrounding neighbors to bring water values closer to each other in value and to reduce the required threshold for the region growing algorithm. The output raster was converted into polygons using the ArcGIS geoprocessing library based on contiguous land and water regions. An area was calculated for each region. The sizes of the output polygons classified as land were used to determine whether the polygon was actual land or noise. Parameters were determined for the 2012 imagery and were applied to the prior years to generate the land-water geomorphic analysis.

3.3 Results and conclusions

Using a combination of automated analysis and manual editing, developers were able to distinguish general geomorphic features for the years 2007 to 2014 based on available BUMP imagery (Figure 5). Additionally, yearly results were compared to the source BUMP imagery collected in 2012. The trend points to a general net increase in total land acreage over time, representing the geomorphic evolution of the Horseshoe Bend Island, with the island increasing from 26 to 35 ha from 2007 to 2014 (Figure 6).

Figure 5. Land-water geomorphology for Horseshoe Bend Island for years 2007 through 2012, with 2012 representing the base condition. The areas in orange indicate the algorithm-estimated aerial land cover detected during each of the years examined.

Figure 6. BUMP images of Horseshoe Bend Island (*a*) 2008, (*b*) 2009, (*c*) 2010, (*d*) 2011, (*e*) 2012, (*f*) 2013, and (*g*) 2014.

It is possible for algorithms to identify a wooded area from an aerial photo but not the level of maturity of the trees within the wooded area, which has an impact on ecosystem classification and mapping. Similarly, it remains difficult for algorithms to determine the difference between open water and non-wooded land when the water is covered by floating or other aquatic vegetation species. Despite these limitations, polygonal output formats facilitate future additions and revisions because a human operator can override specific outputs of the algorithm. Additionally, geomorphic analysis can be applied to the time period before 2008; although the BUMP imagery remains in hard copy format, it can be used in the future to establish more of the early history of the island's landscape geomorphology.

4 Ecosystem Classification and Distribution

4.1 Wetland classification

Classifying wetlands into ecosystem types has been widely applied to ecosystem assessments and leads to decreased variability when making comparisons across both large and small spatial scales (Brinson 1993; Brinson et al. 1994; Smith et al. 2013). To classify the wetlands within the study area, visits were conducted during August 2013 (Berkowitz et al. 2014), July 2014, and August 2014. Based on field observations; GIS (geographic information system) coordinate data collected onsite; and interpretation of aerial photographs, GIS files, and other off-site sources, each wetland community type was delineated. The *Classification of Wetlands and Deep-water Habitats of the United States* (Cowardin et al. 1979) formed the basis of the wetland classification scheme used. The results of the ecosystem classification effort reveal four distinct ecological communities occurring within the study area (Table 1; Figures 7–9). A brief generic description of each community is presented below, consistent with the classification system developed by Cowardin et al. (1979). Specific elements (e.g., vegetation, fauna, soils, etc.) of the multifactor wetland assessment are provided in subsequent sections.

Table 1. Areal extent of wetland classification types within each island studied. Note that all wetland area created at Horseshoe Bend Island and Island E resulted from dredged material placement; Middle Island formed naturally.

Wetland Class	Horseshoe Bend Island	Island E	Middle Island
Forested	10.2 (29)	6.9 (29)	127.3 (89)
Scrub Shrub	1.3 (4)	0.8 (3)	0.0 (0)
Emergent	4.0 (11)	2.0 (9)	14.4 (10)
Aquatic Bed	19.1 (55)	13.5 (58)	0.0 (0)
Total	35.0	23.1	141.7

Units are in hectares with percentages in parenthesis.

Figure 7. Ecosystem classification results at (a) Horseshoe Bend Island, (b) Island E, and (c) Middle Island. Note that palustrine forested areas are shown in green, aquatic bed features appear in blue, scrub-shrub wetlands are in yellow, and emergent wetlands are denoted in purple.

Figure 8. Average elevation of wetland classes identified at Horseshoe Bend Island and Island E. Error bars represent standard error of the mean. Wetland classes with a different letter had statistically different elevations.

Figure 9. Habitat map of the Horseshoe Bend portion of the Atchafalaya River in 2001 (University of New Orleans 2002). The *circle* indicates the location of Island E; the *square* outlines Horseshoe Bend Island and a small dredged material feeder mound to the north. Note that the authors identified forested, shrub/scrub, and marsh (i.e., emergent) wetland classes.

Palustrine broadleaf forested wetlands consist of temporarily flooded zones exhibiting established woody vegetation with an overstory approximately more than 6.1 m in height. These wetlands receive intermittent exposure to freshwater inputs (<0.05% salinity) from groundwater saturation, overbank, and backwater flooding regimes. The frequency and duration of flooding varies on both local and regional scales. Common dominants include species such as ashes (*Fraxinus pennsylvanica* and *F. nigra*), black gum (*Nyssa sylvatica*), tupelo gum (*N. aquatica*), swamp white oak (*Quercus bicolor*), overcup oak (*Q. lyrata*), and willow (*Salix* spp.). Wetlands in this subclass generally occur on mineral soils or highly

decomposed organic soils. Palustrine broadleaf forested wetlands occur within the most stable, established, and highest-elevation portions of the island, representing the most mature and structurally complex community type observed.

Palustrine broadleaf scrub-shrub wetlands consist of temporarily flooded zones exhibiting established woody vegetation less than 6.1 m in height. These wetlands receive intermittent exposure to freshwater inputs (<0.05% salinity) from groundwater saturation, overbank, and backwater flooding regimes. The frequency and duration of flooding varies on both local and regional scales. The species include true shrubs, young trees, and trees or shrubs that are small or stunted because of environmental conditions. Scrub-shrub wetlands may represent a successional stage leading to forested wetland, or they may be relatively stable communities. They represent one of the most widespread classes in the United States. Dominance types include sea-myrtle (*Baccharis halimifolia*), marsh elder (*Iva* spp.), alders (*Alnus* spp.), willows (*Salix* spp.), buttonbush (*Cephalanthus occidentalis*), red osier dogwood (*Cornus stolonifera*), honeycup (*Zenobia pulverulenta*), spirea (*Spiraea douglasii*), bog birch (*Betula pumila*), and young trees of species such as red maple (*Acer rubrum*) or black spruce (*Picea mariana*). Wetlands in this subclass generally occur on mineral soils or highly decomposed organic soils. Palustrine broadleaf scrub-shrub wetlands occur within less stable, geomorphically younger, and moderate-elevation portions of the island, representing the second-most mature and structurally complex ecosystem type observed.

Palustrine persistent emergent wetlands consist of seasonally flooded herbaceous areas characterized by erect, rooted, herbaceous hydrophytes, excluding mosses and lichens. This vegetation is present for most of the growing season in most years. These wetlands usually remain dominated by perennial plants. Palustrine persistent emergent wetlands contain a vast array of grasslike plants such as cattails (*Typha* spp.), bulrushes (*Scirpus* spp.), saw grass (*Cladium jamaicense*), and sedges (*Carex* spp.) and true grasses such as reed (*Phragmites australis*), manna grasses (*Glyceria* spp.), slough grass (*Beckmannia syzigachne*), and whitetop (*Scolochloa festuacea*). There is also a variety of broadleaved persistent emergents such as purple loosestrife (*Lythrum salicaria*), dock (*Rumex mexicanus*), waterwillow (*Decodon verticillatus*), and many species of smartweeds (*Polygonum* spp.). Palustrine persistent emergent wetlands occur within the transition zones between forest and scrub-shrub wetlands and aquatic bed

features. Emergent-dominated ecosystems represent dynamic ecosystems with homogenous vegetative structure.

Palustrine aquatic bed wetlands consist of wetlands and deepwater habitats dominated by plants that grow principally on or below the surface of the water for most of the growing season in most years. Water regimes include irregularly exposed, regularly flooded, permanently flooded, intermittently exposed, semipermanently flooded, and seasonally flooded. Aquatic beds represent a diverse group of plant communities that require surface water for optimum growth and reproduction. They are best developed in relatively permanent water or under conditions of repeated flooding. The plants are either attached to the substrate or float freely in the water above the bottom. Rooted vascular aquatic plants occur at all depths within the photic zone. They often occur in sheltered areas where there is little water movement. Typical inland plants include pondweeds, horned pondweed (*Zannichellia palustris*), ditch grasses (*Ruppia* spp.), wild celery (*Apium graveolens*), and waterweed (*Elodea* spp.). Some of the rooted vascular species are characterized by floating leaves. Typical dominants include water lilies (*Nymphaea* spp.; *Nuphar* spp.), floating-leaf pondweed (*Potamogeton natans*), and water shield (*Brasenia schreberi*). Plants such as yellow water lily (*Nuphar luteum*) and water smartweed (*Polygonum amphibium*), which may stand erect above the water surface or substrate, may be considered either emergent or rooted vascular aquatic plants, depending on the life form adopted at a particular site. Beds of floating vascular plants occur mainly in the Palustrine system. The plants float freely either in the water or on its surface. Dominant plants that float on the surface include duckweeds (*Lemna* spp.; *Spirodela* spp.), water lettuce (*Pistia stratiotes*), and water hyacinth (*Eichhornia crassipes*). Palustrine aquatic bed wetlands occur at the lowest island elevations.

4.2 Ecosystem distribution

As depicted in Figures 7–9 and Table 1, Horseshoe Bend Island and Island E each contain all four of the wetland classes listed above (forested, scrub-shrub, emergent, and aquatic bed). Conversely, Middle Island consists of only forested and emergent areas. As a result, Middle Island exhibits lower habitat diversity despite being substantially larger than the other two study areas. Notably, the strategic placement of dredged materials resulted in the creation of four distinct wetland classes.

The wetland class distributions result from a number of factors, including elevation gradients with aquatic bed and emergent habitats developing in low-lying, more frequently flooded areas while forested and scrub-shrub communities become established in higher topographic locations. Examining elevation data from Horseshoe Bend Island and Island E, wetland class elevations showed statistically significant differences ($df = 29$; $F = 24.62$; $P < 0.001$) with aquatic bed habitats occurring below the river surface, emergent wetlands occurring in slightly elevated positions, followed by forested and shrub scrub wetlands (Figure 8).

For context, this study examined the results of two other studies conducted at larger spatial scales. The University of New Orleans (2002) made wetland determinations across an approximate 70 km² area (Figure 9). That study reported similar wetland classification results as presented herein (Figure 9; Table 2; Table 3), indicating the presence of forested and scrub-shrub wetland types and marsh and intertidal wetlands (i.e., emergent and aquatic bed wetlands). Stedman and Dahl (2008) and Engle (2011) presented data at a much larger spatial scale, encompassing the Gulf of Mexico from Texas to Florida. They also classified wetlands in terms of palustrine emergent, forested, and scrub (i.e., scrub-shrub) wetlands. The results of the current study compare well with the habitat distributions provided by the University of New Orleans (2002). In that study, freshwater marsh habitat (i.e., emergent aquatic bed) accounted for 67% of the natural wetland area surveyed. Our results indicate that 51% and 58% of the dredged material created habitat occurs as emergent aquatic bedforms at Horseshoe Bend Island and Island E, respectively. Similarly, the University of New Orleans (2002) study reported that forested wetlands occupied 28% of the natural landscape; Horseshoe Bend Island and Island E each contained 29% forested wetlands. Finally, the University of New Orleans (2002) reported 1.3% scrub-shrub wetlands in a natural setting; our findings indicate 0.9% scrub-shrub areas on the dredged material supported islands.

In summary, the dredged material supported islands exhibited a variety of ecological classes and distinct habitat types. The observed distribution of ecosystem classes corresponds with the findings reported in other studies. Examining landscape characteristics and geomorphology represents an important component contributing to the multifactor-assessment approach.

Table 2. Habitat classification changes (hectares) of Horseshoe Bend Island between 1985 and 2001. (Adapted from University of New Orleans [2002].)

Habitat Type	1985	2001	Change 1985–2001
Natural Fresh Marsh	1023	916	–106
Natural Upland	0	0.4	0.3
Natural Scrub-Shrub	19.6	16.7	–2.9
Natural Forested Wetland	130	383	253
Natural Bare Land	0	41.4	41.4
Total Natural Habitats	1173	1358	184
Other-Made Fresh Marsh	0	2.7	2.7
Other-Made Forested Wetland	80.9	75.8	–5.1
Other-Made Bare Land	0	0.2	0.2
Total Other-Made Habitats	80.9	78.8	–2.1
BUMP-Made Marsh	0	131.4	131
BUMP-Made Upland	0	3.3	3.3
BUMP-Made Shrub/scrub	0	6.0	6.0
BUMP-Made Forested Wetland	0	112	112
BUMP-Made Bare Land	0	16.5	16.5
BUMP-Made Beach	0	0.6	0.6
Total BUMP-Made Habitats	0	270	270
Habitat Total	1254	1707	453

Table 3. Estimated area (hectares) of estuarine and palustrine wetlands along the Gulf of Mexico. (Adapted from Stedman and Dahl [2008] and Engle [2011]).

Wetland Class	1998	2004	Change 1998–2004
Estuarine Emergent	982,969	965,127	–17,842
Estuarine Shrub	274,838	274,296	–542
Palustrine Emergent	1,124,913	1,104,812	–20,101
Palustrine Forested and Shrub	3,744,884	3,604,418	–140,466
Total	6,127,604	5,948,652	–178,952

5 Floral communities

5.1 Introduction

Vegetation plays a major role in the ecosystem functions associated with wetlands in the lower Atchafalaya River. The diversity of floral communities in the region provides habitat for a variety of terrestrial, aquatic, and avian species (Johnson et al. 1985; Castellanos and Rozas 2001). Additionally, vegetated wetlands stabilize river sediments, allowing for accretion and land-building to occur. Van Heerden and Roberts (1980), Shaffer et al. (1992), and others have focused on the floral establishment, succession, and sustainment; and Faulkner and Poach (1996) compared vegetation between natural and dredged material supported islands in the region. However, few studies examine vegetative characteristics between wetland islands established using different techniques (i.e., natural establishment, traditional dredged material placement, and non-traditional dredged material placement). As a result, the following sections provide a basis for comparison across wetland island types, a contributing element of the multifactor ecosystem assessment.

5.2 Methods

All vegetation sampling occurred in August 2013 and August 2014. Vegetation sampling included quantification of dominant species within each distinct wetland class (forested, scrub-shrub, emergent, and aquatic bed). Sample-plot selection examined representative areas within each of the ecosystem classes; a randomized sampling design remains beyond the scope of the current assessment. Several variables were measured within each wooded area sampled, including (1) the identification of dominant overstory species; (2) stem counts of overstory trees within 0.04 ha plots; (3) measurements of shrub-sapling stem density (i.e., woody stems less than 10 cm in diameter at breast height (dbh) and greater than 1.2 m tall) within two nested 0.004 ha subplots; and (4) estimates of herbaceous and woody vegetation ground cover less than or equal to 1.4 m within four representative 1 m² subplots (Figure 10). A comprehensive list of all plants observed was developed (Appendix A). Plants were identified to species or lowest taxonomic level possible with nomenclature following the U.S. Department of Agriculture (USDA) PLANTS Database (NRCS 2013). On Horseshoe Bend Island, vegetation sampling occurred using seven plots in forested, scrub-shrub, and emergent wetland areas (Figure 9). There were

13 sample plots similarly located on Island E, and 14 sample plots were established on Middle Island.

Figure 10. Vegetation sampling scheme used for characterize forested, scrub-shrub, and emergent wetland habitats.

Vegetation sampling also occurred within aquatic bed areas. Variables measured at each sample point included an ocular estimation of percent cover of herbaceous and woody vegetation and estimates of percent cover of any free-floating or visibly submerged aquatic species (USACE 2010). Additionally, all plants observed within floating bed environments adjacent to the island were identified or collected for further identification in the lab. All specimens were identified to species, when possible, to provide a comprehensive baseline survey of all observed herbaceous and woody flora. Seven aquatic bed sample points were located on Horseshoe Bend Island, and five aquatic bed sample points were located on Island E.

Aquatic bed wetlands were not present at Middle Island at the time of sampling.

5.3 Results

5.3.1 Horseshoe Bend Island

Forested zones occupied approximately 12 ha of the Horseshoe Bend Island (Figures 7 and 11; Table 4). The majority of the forested areas displayed younger stand ages representing 10.5 ha while the older portions of the forest represented approximately 1.3 ha. Both of these forest wetland habits remained disproportionately dominated by black willow (*Salix nigra*) with eastern baccharis (*Baccharis halimifolia*) and common elderberry (*Sambucus nigra canadensis*) frequently observed as codominants. Other infrequently observed codominants included annual marsh elder (*Iva annua*) and red mulberry (*Morus rubra*). The average diameter of woody stems within well-established portions of the island was 12.5 cm dbh, ranging from 5 to 20 cm dbh. Total tree density per hectare ranged from 1500 to 17,325 woody stems, resulting in varying amounts of ground cover across all forested sites (Table 4). The average tree height was 6 m across all plots measured with a maximum height of approximately 10 m.

Figure 11. A forested wetland sample location on Horseshoe Bend Island.

Table 4. Vegetation summary for wooded areas of Horseshoe Bend Island. Data include the number of trees within each sample plot, dominant tree species, tree density, shrub-sapling density, ground vegetation within each sample plot, and average percent ground cover.

Site	Tree Stem Count	Dominant Species	Tree Density (stems/ha)	Sapling Stem Density (stems/ha)	Ground Vegetation Cover (%)				Ground Cover (%)
					Plot				
					1	2	3	4	
1	287	SANI*	7175	125	35	20	30	40	31.3
2	225	SANI	5625	0	25	30	10	15	20
3	693	SANI	17325	0	30	65	60	40	48.8
4	472	SANI	11800	0	0	0	10	10	5
5	370	SANI	9250	0	0	0	25	40	16.3
6	60	SANI	1500	750	0	5	0	10	3.8
7	298	SANI	7450	0	35	5	10	5	13.8

* *Salix nigra*

The herbaceous layer was diverse both within and among the younger- and older-aged stands, with clear distinctions of understory conditions by age class not readily observed. However, across both stand types, consistent patterns of understory cover varied from densely aggregated clumps of vegetation less than 1 m in height (e.g., cocoyam [*Colocasia esculenta*]) to extensive ground cover of invasive torpedo grass (*Panicum repens*), smartweed (*Polygonum* spp.), or invasive alligatorweed (*Alternanthera phloxeroides*). On the highest elevational areas, a pronounced abundance of dogfennel (*Eupatorium capillifolium*) was observed, with a distinct absence of coco-yam. Stiff dogwood (*Cornus foemina*) was found only in the higher elevation areas located on the southeast portion of the island. Additionally, black willow, common elderberry sprouts, and peppervine (*Ampelopsis arborea*) were frequently encountered. Less commonly observed, but scattered throughout both stand types, were buttonbush (*Cephalanthus occidentalis*), climbing hempvine (*Mikania scandens*), whorled marsh pennywort (*Hydrocotyle verticillata*), smallspike false nettle (*Boehmeria cylindrica*), flatsedge (*Cyperus* spp.), goldenrod (*Solidago* spp.), hairy pod cowpea (*Vigna luteola*), broadleaf cattail (*Typha latifolia*), invasive Chinese tallowtree (*Triadica sebifera*), rosemallow (*Hibiscus lasiocarpus*), and broadleaf arrowhead (*Sagittaria latifolia*). The unvegetated areas of the ground throughout the island exhibited small-sized downed dead wood deposited during flooding events. Appendix A provides a comprehensive plant species list.

On Horseshoe Bend Island, dominant aquatic bed wetland species included common water hyacinth (*Eichhornia crassipes* [Mart.] Solms), alligatorweed (*Alternanthera philoxeroides* [Mart.] Griseb.), water spangles (*Salvinia minima* Baker), common duckweed (*Lemna minor* L.), and primrose-willow (*Ludwigia* spp.) (Figure 12; Table 5). Several invasive species were observed during the aquatic bed wetland survey (Appendix A).

Figure 12. An aquatic bed wetland sample location on Horseshoe Bend Island.

Table 5. Percent cover estimates for floating and submerged vegetation on Horseshoe Bend Island.

Species	Common name	Average cover (%)
<i>Eichhornia crassipes</i>	common water hyacinth	37.5
<i>Alternanthera philoxeroides</i>	alligatorweed	34.8
<i>Salvinia minima</i>	water spangles	31.7
<i>Lemna minor</i>	common duckweed	12.5
<i>Ludwigia</i> spp.	primrose-willow	10.0
<i>Hydrilla verticillata</i>	waterthyme	6.5
<i>Nelumbo lutea</i>	American lotus	4.6
<i>Leersia oryzoides</i>	rice cutgrass	1.0
<i>Colocasia esculenta</i>	coco-yam	0.2

5.3.2 Island E

Ten forested wetland and three scrub-shrub wetland sampling locations were sampled on Island E (Figure 13; Table 6). Tree stem density ranged from 100 to 1050 stems per hectare, with the average diameter at breast height ranging from 12–30 cm. Sapling density ranged from 200 to 18375 stems per hectare with a mean of 7000 stems per hectare. Average ground cover of herbaceous vegetation ranged from 50% to 99% with a mean coverage of 88%.

The scrub-shrub areas were dominated by willows and wax-myrtle (*Morrellia cerifera* [L.] Small) with an increased abundance of buttonbush and alligatorweed. Otherwise, the forested and scrub-shrub areas displayed similar vegetation communities. Black willow disproportionately dominated sample plots, with wax-myrtle as a codominant in approximately half of the sample locations. Other woody species commonly observed included baldcypress (*Taxodium distichum* [L.] Rich.), annual marsh elder, and non-native Chinese tallowtree. In one sample location, red maple (*Acer rubrum* L.) saplings dominated with black willow interspersed; however, red maple was not frequently observed across the entire island. Honeylocust (*Gleditsia aquatica* L.) and water locust (*Gleditsia aquatica* Marsh.) were infrequently encountered. In the understory, buttonbush and coco-yam were frequently observed. Coco-yam represented the dominant herb in many areas of the island, particularly in open areas and along the islands perimeter. Vines were also common within sample locations, with morning-glory (*Ipomea* spp.), cowpea (*Vigna unguiculata* [L.] Walp.), and peppervine being most prevalent. Knotweed (*Polygonum* spp. L.), alligatorweed, lizardtail (*Saururus cernuus* L.), and whorled marsh pennywort (*Hydrocotyle verticillata* Thunb.) were the most commonly observed low-growing herbs. Appendix A provides a full list of species observed.

Within the aquatic bed environment surrounding Island E, the dominant aquatic plants, with average percent cover greater than 25%, were common water hyacinth, American lotus (*Nelumbo lutea* Willd.), and water spangles (Table 7).

Figure 13. A sample location on Island E. A forested wetland area can be seen in the foreground with an emergent wetland area in the background of the image.

Table 6. Vegetation summary for forested and scrub-shrub wetlands on Island E. Data include the number of trees within each sample plot, dominant tree species, tree density, shrub-sapling density, ground vegetation within each sample plot, and average percent ground cover.

Site	Tree Stem Count	Dominant Species	Tree Density (stems/ha)	Sapling Stem Density (stems/ha)	Ground Vegetation Cover (%)				Ground Cover (%)
					Plot				
					1	2	3	4	
1	287	SANI*	7175	125	35	20	30	40	31.3
2	24	SANI	600	7750	15	85	80	30	53
3	42	SANI	1050	3250	100	100	100	95	99
4	16	SANI	400	3000	85	100	80	90	89
5	17	ACRU ⁺ /SANI	425	12375	90	95	80	90	89
6 [‡]	4	MOCE [‡]	100	18375	85	80	60	65	73
7 [‡]	9	SANI	225	5250	95	90	95	95	94
8	19	SANI	475	4500	100	90	95	80	91
9	34	SANI	850	3375	90	100	90	85	91
10	24	SANI	600	2125	100	95	90	80	91
11	14	MOCE	350	14875	85	90	90	80	86
12	21	SANI	525	2000	100	95	95	85	94
13 [‡]	12	SANI/MOCE	300	8625	100	100	95	90	96

* *Salix nigra*

† *Acer rubrum*

‡ *Morella cerifera*

Table 7. Percent cover estimates for floating and submerged vegetation on Island E.

Species	Common name	Average cover (%)
<i>Eichhornia crassipes</i>	common water hyacinth	63.3
<i>Nelumbo lutea</i>	American lotus	36.0
<i>Salvinia minima</i>	water spangles	29.0
<i>Lemna minor</i>	common duckweed	16.7
<i>Ludwigia</i> spp.	primrose-willow	10.0
<i>Leersia oryzoides</i>	rice cutgrass	10.0
<i>Alternanthera philoxeroides</i>	alligatorweed	5.0
<i>Amaranthus</i> spp.	amaranth	5.0

5.3.3 Middle Island

Sampling occurred at 14 forested wetland locations on Middle Island (Figure 14; Table 8). Overstory dominant tree species were black willow, American sycamore (*Platanus occidentalis* L.), boxelder (*Acer negundo* L.), baldcypress, Chinese tallowtree, water oak (*Quercus nigra* L.), sugarberry (*Celtis laevigata* Willd.), and eastern cottonwood (*Populus deltoides* W. Bart. ex Marshall), respectively. Tree density ranged from 150 to 800 stems per hectare; average tree diameter ranged from 18 to 38 cm. Sapling density ranged from 1000 to 6125 stems per hectare with a mean of 2500 stems per hectare. Stem density for dominant trees averaged 400 stems per hectare and sapling density averaging 2500 stems per hectare.

Shrub species, including wax myrtle, swamp privet (*Forestiera acuminata* [Michx.] Poir.), and possumhaw (*Ilex decidua* Walter), were observed although not dominant on the island. Dominant vines on Middle Island included American buckwheat vine (*Brunnichia ovata* [Walter] Shinnars) and eastern poison ivy (*Toxicodendron radicans* [L.] Kuntze), with Japanese climbing fern (*Lygodium japonicum* [Thunb.] Sw.), trumpet creeper (*Campsis radicans* [L.] Seem. Ex Bureau), pepper vine, and climbing hempvine. Average ground cover was 60% with a range of 10%–100%. No emergent aquatic vegetation was observed surrounding Middle Island. Appendix A provides a full list of species observed on Middle Island.

Figure 14. Emergent wetland surrounded by a forested wetland area at a Middle Island sampling location.

Table 8. Vegetation summary for wooded areas of Middle Island. Data include the number of trees within each sample plot, dominant tree species, shrub-sapling density, ground vegetation within each sample plot, and average percent ground cover.

Site	Tree Stems (count)	Dominant Species	Tree Density (stems/ha)	Sapling Density (stems/ha)	Ground Vegetation Cover (%)				Ground Cover (%)
					Plot				
					1	2	3	4	
1	12	SANI/TADI	300	1750	85	20	25	45	44
2	11	SANI/FRPE	275	1875	40	25	60	70	49
3	16	SANI/PLOC	400	2500	30	10	80	40	40
4	10	PLOC/SANI	250	1500	100	100	100	100	100
5	30	SANI/TRCE	750	1750	70	70	70	60	68
6	19	TADI/QUNI	475	3625	30	40	40	10	30
7	6	PLOC/ACNI	150	3125	40	100	90	100	83
8	7	ACNI/PLOC	175	625	30	80	40	90	60
9	10	SANI/FRPE	250	1000	100	100	100	100	100
10	10	PLOC/TRCE	250	1125	100	70	100	50	80
11	10	SANI/PLOC	250	2000	70	100	100	100	93
12	17	SANI/PLOC	425	5500	30	90	100	100	80
13	32	PLOC/CELA	800	2625	20	30	20	20	23
14	27	PLOC/PODE	675	6125	5	10	5	20	10

ACNI = *Acer nigrum*

PLOC = *Platanus occidentalis*

SANI = *Salix nigra*

CELA = *Celtis laevigata*

PODE = *Populus deltoides*

TADI = *Taxodium distichum*

FRPE = *Fraxinus pennsylvanica*

QUNI = *Quercus nigra*

TRCE = *Triadica cebifera*

5.4 Discussion

The Horseshoe Bend Island displayed a diverse array of plant species supporting the larger Atchafalaya River wetland ecosystem, with 81 plant and 23 animal species observed (see Berkowitz et al. 2014). Lower species-richness values were observed at Island E and Middle Island. For contextual purposes, Faulkner and Poach (1996) assessed the functional capacity of created and natural wetlands within the Atchafalaya Delta and determined the functional capacity within each wetland type. Faulkner and Poach conducted a vegetation survey within created and natural wetlands, identifying a total of 53 plant species compared to more than 80 observed on Horseshoe Bend Island. Approximately 60% of the plant species reported by Faulkner and Poach listed occurred on Horseshoe Bend Island.

These findings suggest that the plant species richness and diversity observed at Horseshoe Bend Island remains comparable, if not surpasses, the values reported for other ecosystem studies conducted in the region. Additionally, Faulkner and Poach (1996) reported that created wetlands require 5 to 10 years prior to approaching the characteristics observed within natural wetland ecosystems. Thus, the successional development of Horseshoe Bend Island compares favorably with natural wetland ecosystems in coastal Louisiana based on the deposition periods of dredged materials and the resulting island geomorphic features and accretion rates.

6 Avian communities

6.1 Introduction

The creation of dredged material islands has the potential to result in ecosystems that fulfill some of the ecosystem functions observed on natural islands (Faulkner and Poach 1996; University of New Orleans 2002). If engineered to mimic natural processes, over time, created dredge islands may begin to display ecosystem characteristics similar to natural islands although some studies suggest that long-term restoration outcomes remain unclear (Streever 2000; Shafer and Streever 2000). However, this dredged material island demonstration project is advancing the knowledge needed to support the efficacy of this novel dredging and dredged material placement technique.

Biological assessments using community composition metrics of a particular taxonomic group can be used to evaluate ecosystem condition in created habitats (USEPA 2002). Biological community characteristics can yield a great deal of information about a particular habitat because community composition can reflect the cumulative effect of multiple stressors. Additionally, repeated monitoring of community metrics can be used to track the success of created habitats over time. For created dredged material supported islands and other habitats, bird communities serve as useful indicators of habitat condition because they have complex communities that span a variety of ecological niches (Rabalais 1990). Additionally, many bird species have exhibited population declines and thus have been designated as species of concern (Partners in Flight 2006). Within the study area, Mallach and Leberg (1999) compared nesting densities and substrate characteristics within natural and dredged material supported islands. Many studies focus on wading-bird populations when assessing habitat quality because wading species selectively choose nesting colony locations based on specific habitat criteria (Piazza and Wright 2004; Frederick and Ogden 1997; Strong et al. 1997).

This section examines avian habitat on a non-traditionally engineered dredged material island and compares community metrics with those observed on a traditional dredge island (Island E) and a natural island (Middle Island). Through this comparison, the effectiveness of dredge island creation techniques for creating functional avian habitats can be evaluated as part of a multifactor assessment.

6.2 Methods

Bird surveys occurred on 1–2 July 2014. Transect surveys involved an observer traversing a straight line perpendicular to the long-axis orientation of the island and recording the number, species, and age class of all birds observed within 5 m of the transect center. Transects were located on the portion of the islands that contained forested and scrub-shrub habitats that potentially provided nesting habitat for wading birds or foraging habitat for passerine species. The extent and location of forest habitat (Table 1) on each island also dictated the number, location, and total length of transects evaluated. Transects were located approximately 20 m apart to maintain independence among transects.

A total of eight transect surveys on Horseshoe Bend Island resulted in a total transect length surveyed of 760 m. Bird surveys of Island E included two transects totalling 244 m surveyed. A single 324 m transect was surveyed on Middle Island. Bird community metrics included species richness and individual bird density, allowing for comparisons of avian habitat use between islands. Species-richness estimates represent the total number of bird species observed on transects at each sample location. Bird density was calculated as the total number of birds observed per meter of transect surveyed. Statistical analysis used Analysis of Variance (ANOVA) to compare bird richness and density between Horseshoe Bend Island and Island E; significance was evaluated at $\alpha = 0.05$ (SPSS IBM, Inc. Version 20).

6.3 Results

6.3.1 Horseshoe Bend Island

Bird surveys of Horseshoe Bend Island yielded high numbers of breeding wading birds in dense, mixed-species nesting colonies (Figures 15–19). Total species richness was ten species (Table 9), including nine species of wading birds and one passerine species (Table 10). The most abundant species observed within the nesting colony was great egret (*Ardea alba*), followed by glossy ibis (*Plegadis falcinellus*). Of the 193 individuals recorded on surveys at Horseshoe Bend Island, 79% were juveniles, suggesting that the island supports a highly productive nesting colony.

Table 9. Comparison of bird community metrics across the three islands studied.

Location	Species Richness	Density (birds/m transect length)	% Juveniles
Horseshoe Bend Island	10	0.27	78.76
Island E	7	0.05	0
Middle Island	13	0.09	0

Table 10. Bird Species observed at Horseshoe Bend Island, Island E, and Middle Island.

Species	Scientific Name	Horseshoe Bend Island	Island E	Middle Island
<i>Agelaius phoeniceus</i>	red-winged blackbird	X		
<i>Ajaia ajaja</i>	roseate spoonbill	X		
<i>Archilochus colubris</i>	ruby-throated hummingbird	X		X
<i>Ardea alba</i>	great egret	X		
<i>Baeolophus bicolor</i>	tufted titmouse	X		
<i>Bubulcus ibis</i>	cattle egret	X		
<i>Buteo lineatus</i>	red-shouldered hawk	X		
<i>Cardinalis</i>	northern cardinal	X		
<i>Egretta caerulea</i>	little blue heron	X	X	
<i>Egretta rufescens</i>	reddish egret	X	X	
<i>Egretta thula</i>	snowy egret		X	
<i>Egretta tricolor</i>	tricolored heron		X	X
<i>Eudocimus albus</i>	white ibis		X	X
<i>Melanerpes carolinus</i>	red-bellied woodpecker		X	X
<i>Porphyryula martinica</i>	purple gallinule		X	
<i>Strix varia</i>	barred owl			X
<i>Polioptila caerulea</i>	blue-gray gnatcatcher			X
<i>Thryothorus ludovicianus</i>	Carolina wren			X
<i>Melanerpes carolinus</i>	red-bellied woodpecker			X
<i>Buteo lineatus</i>	red-shouldered hawk			X
<i>Archilochus colubris</i>	ruby-throated hummingbird			X
<i>Baeolophus bicolor</i>	tufted titmouse			X
<i>Vireo griseus</i>	white-eyed vireo			X
<i>Meleagris gallapavo</i>	wild turkey			X

Figure 15. Wading-bird nesting colony on Horseshoe Bend Island.

Figure 16. Juvenile tricolored heron observed during a transect survey on Horseshoe Bend Island.

Figure 17. Juvenile snowy egret observed during a transect survey on Horseshoe Bend Island.

Figure 18. Juvenile glossy ibis observed during a transect survey on Horseshoe Bend Island.

Figure 19. Forested wetland area within the wading-bird rookery on Horseshoe Bend Island. Note the presence of standing water below the nesting colony.

6.3.2 Island E

Total species richness on Island E was seven species (Table 9). Total bird density was low (0.05 birds/m transect length). Only one individual wading bird was documented on Island E (Table 10), an adult white ibis (*Eudocimus albus*). Island E exhibited no sign of a wading-bird nesting colony.

6.3.3 Middle Island

A total richness of 13 species, the highest of the three islands surveyed, was documented at Middle Island (Table 9). The majority of birds documented were passerines (i.e., songbirds) (Table 10). A single great egret (*Ardea alba*) was the only wading bird documented, suggesting that Middle Island does not support a wading-bird nesting colony. Total bird density (0.09 birds/m transect length) was substantially lower than at Horseshoe Bend Island but somewhat higher than at Island E.

6.4 Discussion

Results of statistical comparisons between Horseshoe Bend Island and Island E indicate that species richness of wading birds was significantly greater at Horseshoe Bend Island than at Island E ($F(1,6) = 16.84$, $P = 0.01$). No juvenile birds were observed at Island E, resulting in a significantly higher estimate of percent juveniles at Horseshoe Bend Island ($F(1,6) = 85.42$, $P < 0.001$). However, the two islands displayed similar species richness ($F(1,6) = 0.75$, $P = 0.42$); and overall bird density was not significantly different between the two islands ($F(1,6) = 1.06$, $P = 0.34$).

Based on the July 2014 bird survey data, Horseshoe Bend Island provides an important breeding habitat for wading birds, an ecosystem function not observed at the other two islands surveyed. Previous studies have also documented the use of dredged material islands by at least 26 species of colonial sea and wading birds (Schreiber and Schreiber 1978). The observed difference between bird community characteristics at the two dredged material supported islands suggests that Horseshoe Bend Island displays more favorable conditions for breeding wading birds, possibly as a result of the novel technique used for forming the island. Traditional dredge island construction at Island E resulted in an area of emergent and scrub-shrub wetlands near the center of the island, creating a lack of core forested habitat (Figure 5). In contrast, Horseshoe Bend Island contains proportionally more core forested habitat, which is related to nest site selection for herons (Gibbs and Kinkel 1997). McCrimmon (1978) found that within mixed-species nesting colonies, distance to forest edge influenced nest-site selection with some species more likely to be found closer to the edge than others.

Previous studies indicate that site selection for wading-bird nesting colonies is related to vegetation structure (Burger 1978; McCrimmon 1978). Of particular importance is the presence of rigid woody plants capable of offering nest stability (Beaver et al. 1980). Horseshoe Bend Island contains a high density of woody vegetation, providing structural support for heavy nests and nestlings. The vegetation density at Horseshoe Bend Island also results in high percent canopy closure (Table 4), which offers cover and mitigates temperature fluctuations in a region where average summer temperatures often exceed 90°F (32°C).

Hydrologic condition is another factor that may influence the availability of nesting and foraging habitat for wading birds (Strong et al. 1997). The

two dredge islands examined show different hydrologic regimes, which may help to explain the difference in bird community characteristics between locations. Notably, the nesting colony area on Horseshoe Bend Island remains inundated with standing water during much of the nesting season (Figure 19). One can speculate that the presence of standing water beneath nesting trees during breeding season may facilitate nestling survival by discouraging predators, such as raccoons, rats, and snakes, a trend that has been documented for passerine species that nest in wetlands (Picman et al. 1993; Cain et al. 2003; Hoover 2006).

In addition to habitat characteristics that may influence the presence or absence of nesting wading birds, it should be noted that social stimulation is an important determinant in nest site selection, with the presence of other wading birds often being a key determinant in site selection (Beaver et al., 1980; Custer and Osborn 1978; Krebs 1974). For this reason, habitat use by colonial nesters can sometimes be unpredictable, and habitats with seemingly ideal conditions may not be selected.

Statistical comparisons regarding Middle Island, the naturally formed island, were not performed because only a single transect was conducted. A descriptive comparison between islands shows that Middle Island supported greater species richness than Horseshoe Bend Island. However, nesting wading birds were not observed on Middle Island, and overall bird density was comparatively higher at Horseshoe Bend Island as a result of its dense nesting colony. Despite the high numbers of wading birds documented on Horseshoe Bend Island, very few passerine species were observed. Tree size on Horseshoe Bend Island remained low in comparison with Middle Island, and many species of neotropical migrants using forested habitats prefer later successional forests. Should ongoing monitoring occur at Horseshoe Bend Island, it is possible that observations of passerine species would increase as forest structure continues to develop.

In summary, the strategic placement of dredged material at Horseshoe Bend Island resulted in the development of an exceptionally rich and productive colony of wading birds and thus represents a potential model for the future creation of dredge islands. Horseshoe Bend Island displayed higher wading-bird richness than a traditional dredged material island and a natural island in the region. The examination of avian communities contributes to a multifactor-assessment approach by examining vertebrate species near the top of the food chain.

7 Aquatic Invertebrates

7.1 Introduction

Infaunal communities provide the basis of many aquatic food webs, supporting both demersal-feeding fish and a wide variety of wading bird species (Wells et al. 1973; Mathews et al. 1992). Examining infauna allows for comparisons between habitat types in early to mid-successional stages, as infauna have proven an effective measure of wetland restoration trajectory (Evans and Short 2005). Ewel et al. (2001) further points out that benthic infaunal communities quickly follow the establishment of restored wetlands, providing a responsive measure of a restoration success trajectory. For example, Ray (2000) examined infauna in natural and dredged material supported wetlands in the northeastern U.S., reporting similar species richness, abundance, composition, and diversity within 10 years of wetland creation. As a result, infaunal communities provide a valuable addition to a multifactor-assessment approach.

Many studies demonstrate that restored wetlands infaunal densities increased rapidly following restoration, including dredged material placement (Craft 2000). However, community metrics often fail to reach the levels observed in natural wetlands (Streever 2000; Craft et al. 2003). The following section examines sediment characteristics and infaunal community metrics at the three islands investigated. These results were used as an element of the multifactor approach assessing the strategic placement of dredged material for supporting wetlands.

7.2 Methods

Infaunal community sampling was conducted in two aquatic habitat types on Horseshoe Bend Island and Island E, including (1) along the wetland edge and (2) within emergent aquatic bed locations (Figure 20, left). Middle Island lacked emergent aquatic bed features, thus no aquatic bed samples were collected at that site. Invertebrate sampling occurred at three to five stations per sampling location, and each station included three replicate samples. All core samples were collected at water depths between 1 and 1.5 m (Figure 20, right).

Figure 20. *Left*, the infaunal community sampling locations at Horseshoe Bend Island. Note that triplicate core samples were collected at the wetland edge and at five emergent aquatic bed (EAB) locations. *Right*, infaunal sampling used a 7.5 cm coring device.

Infaunal sampling followed procedures outlined in Ray (2000) in which a 7.5 cm diameter coring tube was inserted to a sediment depth of 10 cm. Samples were rinsed over a 0.5 mm mesh screen, fixed in 4% formalin, and transported to the laboratory where they were transferred to 70% ethanol and stained with rose bengal solution to facilitate sorting of specimens. After staining, the samples were examined under 3× magnification and the specimens separated from the remaining sediment and detritus and stored in 70% ethanol. When all infaunal samples had been processed, 10% of the samples were randomly selected and reexamined to ensure that all the invertebrate organisms were collected during initial processing efforts. Specimens were then identified to the lowest practicable taxonomic level and counted.

Particle size was determined using laser diffraction as described in Loizeau et al. (1994) and Konert and Vandenberghe (1997). Grain size analysis was conducted using the GRADISTAT software package following Blott and Pye (2001). Organic matter content determination followed the loss on ignition (LOI) method; dried ground samples underwent combustion at

550°C in a muffle furnace for 4 hr (Sparks 1996). Community metrics calculated for comparison between sampling locations were taxonomic richness (i.e., total number of invertebrate taxa in each sample), abundance (total number of organisms), percent dominance (i.e., the percentage of each sample that is made up of the most abundant taxonomic group), and percent Chironomids (percentage of all organisms in each sample that were in the order Chironomidae). Comparisons of community metrics between islands combined data from both emergent aquatic bed and wetland edge sampling locations. Comparisons of sediment characteristics and community metrics included ANOVA when data were normally distributed; the non-parametric Mann-Whitney U and Kruskal-Wallis H tests were applied for comparisons involving non-normally distributed data. Normality testing used the Shapiro-Wilk test ($\alpha = 0.05$; SPSS statistics v.17.0).

7.3 Results

7.3.1 Sediment characteristics

Analysis of sediment particle size distribution demonstrates that all of the wetland sites examined were composed of a mixture of fine sands, silts, and clays with no very coarse sand or gravels present (Table 11). Silts remained the most abundant size fraction at Island E emergent aquatic bed locations and at Middle Island while the sand fraction dominated all other sampling locations. Horseshoe Bend Island wetland edge samples contained significantly higher sand content ($89.8 \pm 2.7\%$; $p < 0.001$) than Island E ($70.2 \pm 3.0\%$). Submerged aquatic bed habitats showed similar results with Horseshoe Bend Island EAB containing significantly higher sand content ($60.1 \pm 7.4\%$; $p < 0.001$) than Island E EAB ($10.2 \pm 7.9\%$).

Table 11. Average sediment particle size distribution for Horseshoe Bend river wetland edge (HB) and emergent aquatic bed (HB EAB), Island E wetland edge (IE) and emergent aquatic bed (IE EAB), and Middle Island (MI) sample locations.

Particles	HB	HB EAB	IE	IE EAB	MI
% Sand	89.8 ± 2.7	60.1 ± 7.4	70.2 ± 3.0	10.2 ± 7.9	39.8 ± 6.2
% Silt	7.1 ± 6.3	34.4 ± 6.7	28.8 ± 2.4	73.0 ± 4.7	51.5 ± 7.1
% Clay	3.1 ± 1.9	5.5 ± 0.7	1.0 ± 0.6	16.8 ± 3.2	8.69 ± 4.2

Figure 21 presents the sediment organic matter content from each wetland. Craft (2000) and others report significant relationships between organic matter content and infaunal community metrics. Statistical analysis

of organic matter content did not detect significant differences when comparing across wetlands (e.g., Horseshoe Bend Island vs. Island E). However, significant differences were detected between the wetland edge and emergent aquatic bed sampling locations (Mann-Whitney $U = 26.5$; $p = 0.025$) with emergent aquatic bed locations containing significantly higher organic matter concentrations than wetland edge locations.

Figure 21. Sediment organic matter content at Horseshoe Bend wetland edge (HB) and emergent aquatic bed (HB EAB), Island E river shoreline (IE) and emergent aquatic bed (IE EAB), and Middle Island sample locations. Note that the emergent aquatic bed sediments (*) contained significantly more organic matter than river shoreline locations.

7.3.2 Infaunal community metrics

Taxonomic richness at both dredged material supported islands remained higher than at Middle Island. The highest richness values occurred in the Horseshoe Bend Island emergent aquatic bed sampling locations (average 13.8 species per sample), and the lowest richness values were observed in the Middle Island samples (1.3 species; Table 12). When samples from emergent aquatic bed and wetland edge locations were combined for each island, richness was significantly higher at Horseshoe Bend than at Middle Island ($P = 0.01$; Table 13). Island E also displayed significantly higher richness than Middle Island ($P = 0.03$). Species richness at Horseshoe Bend and Island E were not significantly different.

Table 12. Average infauna community metrics at Horseshoe Bend Island (HB), Island E (IE), and Middle Island (MI). Sampling locations included wetland edge (Edge) and emergent aquatic bed (EAB).

Wetland	Sample location	Richness	% Dominance	% Chironomids	Abundance
IE	Edge	6	79.2	3.2	77
IE	EAB	11.3	66.3	10.1	190
HB	Edge	4.7	72.4	12	43
HB	EAB	13.8	70.5	7.1	263
MI	Edge	1.3	99.8	0.2	104

Table 13. Statistical comparisons of infauna community metrics between sampling locations. Wetland site interaction included all three islands: Horseshoe Bend Island (HB), Island E (IE), and Middle Island (MI). Pair-wise comparisons were conducted between each wetland combination (e.g., HB vs. IE) and between the two sampling locations (Edge and EAB). Statistically significant results appear in bold ($\alpha = 0.05$).

Data source	Community Metric	Statistical Test	df	n	H / F	p
Interaction between all wetland sites						
All	Richness	Kruskal-Wallis	2	17	7.44	0.02
	% Dominance	Kruskal-Wallis	2	17	7.02	0.03
	% Chironomid	Kruskal-Wallis	2	17	6.94	0.03
	Abundance	Kruskal-Wallis	2	17	0.11	0.95
Pair-wise comparisons between wetland sites						
HB vs. MI	Richness	Kruskal-Wallis	1	11	6.06	0.01
	% Dominance	Kruskal-Wallis	1	11	6.03	0.01
	% Chironomid	Kruskal-Wallis	1	11	6.03	0.01
	Abundance	ANOVA	1	11	0.64	0.44
IE vs. MI	Richness	Kruskal-Wallis	1	9	4.94	0.03
	% Dominance	Kruskal-Wallis	1	9	5.45	0.02
	% Chironomid	Kruskal-Wallis	1	9	3.38	0.07
	Abundance	Kruskal-Wallis	1	9	0.07	0.8
HB vs. IE	Richness	ANOVA	1	14	0.3	0.59
	% Dominance	Kruskal-Wallis	1	14	0.02	0.9
	% Chironomid	ANOVA	1	14	0.45	0.51
	Abundance	Kruskal-Wallis	1	14	0.02	0.9
Pair-wise comparisons between sampling locations						
Edge vs. EAB	Richness	ANOVA	1	17	20.27	<0.001
	% Dominance	ANOVA	1	17	3.15	0.1
	% Chironomid	ANOVA	1	17	0.99	0.33
	Abundance	Kruskal-Wallis	1	17	5.8	0.02

Compared to Middle Island, both created dredge islands showed lower percent dominance, a community metric that is associated with disturbance due to the disproportionately high occurrence of a single taxa. Percent dominance was comparable between the two dredge islands, Horseshoe Bend and Island E. Middle Island displayed significantly higher percent dominance than either Horseshoe Bend Island or Island E ($P \leq 0.02$). Of the three samples collected at Middle Island, two contained only Oligochaete species and no other taxa, and the third sample contained 99.5% Oligochaetes. Horseshoe Bend and Island E showed a high percent dominance of Oligochaetes and Polychaetes (Annelid worms) but contained various other organisms. At Horseshoe Bend Island, six of the eight samples collected were dominated by Oligochaetes with the remaining two samples dominated by Polychaetes. Five of the six samples from Island E were dominated by Oligochaetes, and the remaining sample was dominated by Polychaetes.

Samples from Horseshoe Bend Island had the highest percent Chironomids, a taxa associated with tolerance to disturbance, while Middle Island contained very few Chironomids. Combined emergent aquatic bed and wetland edge samples at Horseshoe Bend Island showed significantly higher percent Chironomids than at Middle Island ($P = 0.02$). At Island E, percent Chironomids was not significantly different from either Horseshoe Bend Island or Middle Island.

Abundance was highest in the Horseshoe Bend Island emergent aquatic bed location and lowest in the Horseshoe Bend wetland edge location. There were no significant differences in abundance between dredged material supported islands. Samples from emergent aquatic bed locations showed significantly higher abundance than wetland edge samples, with two to six times the number of organisms observed in aquatic bed samples ($H = 0.02$). Taxa richness also was significantly higher in aquatic bed locations than in wetland edge locations ($P < 0.001$). Percent dominance and percent Chironomids were not significantly different between sampling locations.

7.4 Discussion

Sediment grain size distributions followed expected patterns observed in created riverine wetland locations. Faulkner and Poach (1996) examined created and naturally formed wetlands in the lower Atchafalaya River, re-

porting that created wetlands exhibited coarser sediment grain size distributions in comparison to naturally formed wetlands. Our results conform to this trend. During the creation of Horseshoe Bend Island and Island E, river bedload sediments were dredged from within the river channel and deposited to an aerial height. As indicated above, Horseshoe Bend Island contained significantly higher sand contents than Island E ($p < 0.001$).

The higher sand content at Horseshoe Bend Island can be attributed to construction technique, due to its mid-river location and the natural reworking of sediments following dredged material placement, allowing for the settlement of coarser particles while the fine soil materials were carried further downstream. Conversely, Island E was created using a technique in which sediment was deposited into a low velocity area along the river edge, promoting settling of fine sediment fractions during a single dredging event. The movement of fine materials downstream has potential implications for dredging economics and engineering. For example, if less material is deposited within a dredged material placement area, additional capacity is maintained for future maintenance dredging activities (USACE 1983). Emergent aquatic bed environments occur within the interior, protected portions of the dredged material supported islands, allowing for incoming suspended sediments to settle and accumulate. These emergent aquatic bed habitats contained significantly higher infaunal richness and abundance than other sampling locations.

Craft (2000) examined the density of Oligochaetes in created wetlands, reporting densities up to 3800 organisms/m². The current study sites yielded between 2777 and 19104 Oligochaetes/m². These values fall within the reported range of 380–99000 organisms/m² occurring in a natural wetland setting (Craft 2000). Both dredged material supported islands contained rich communities of infaunal organisms. However, the created islands, along with Middle Island, support a high percentage of disturbance-tolerant organisms, as observed in the percent dominance and percent Chironomid results. Data suggest that the two dredged material supported islands function similarly for providing infaunal habitat. The lack of a significant difference in abundance between dredged material supported islands suggests that differences in taxa richness are not due to a larger number of organisms being sampled on one island compared with another. Both islands displayed relatively high taxa richness and low percent dominance in comparison to the natural island sampled, indicating that the created islands are capable of supporting infaunal habitats comparable

to the natural island examined. The inclusion of infaunal community metrics aids in the application of a multifactor-assessment approach.

8 Soils and Biogeochemical Activity

8.1 Introduction

Soils play an important role in providing wetland functions, including nutrient cycling, carbon sequestration, and the removal of harmful elements and compounds (Smith and Klimas 2002). For example, Lindau and Hossner (1981) and Craft et al. (1999) demonstrated the ability of wetland soils to accumulate carbon following restoration and vegetation establishment. Mitsch et al. (2005) and Mitsch and Day (2006) discuss the ability of wetlands to remove excess nitrogen from surface waters, decreasing eutrophication and Gulf of Mexico hypoxic events. Those studies report nitrate-nitrogen removal capacities on the order of $46 \text{ g-N m}^{-2} \text{ year}^{-1}$ in Louisiana wetlands. Faulkner and Poach (1996) investigated the phosphorus dynamics of natural and dredged material supported wetlands in the Atchafalaya River, indicating that natural phosphorous conditions will develop in created wetlands within 10–20 years of establishment.

Additionally, wetland soils provide the physical substrate required to support wetland habitat and hydrology functions (Vepraskas et al. 2001; Reddy and DeLaune 2008). For example, Berkowitz and White (2013) reported that more-developed soil profiles showed increased hydrologic function in restored Mississippi Valley wetlands. Craft (2000) demonstrated increases in invertebrate infaunal habitat associated with a restored soil chronosequence in North Carolina wetlands. Many studies link restored wetlands with avian and higher-organism functionality (Rabalais 1990; Mallach and Leberg 1999).

Although many studies examine the ability of wetlands to perform various ecosystem functions, few studies compare soil and biogeochemical parameters across dredged material placement techniques. The following section examines the soils within each of the three islands studied and compares soil parameters and activities between Horseshoe Bend Island and Island E, both formed by dredged material. This approach provides a platform for examining the impact of the dredged material placement technique on soil parameters and biogeochemical functions.

8.2 Methods

8.2.1 Soils

Basic wetland soil descriptions were made at sample locations within forested and scrub-shrub wetland communities on each of three islands examined (USACE 2010; NRCS 2010). Soil descriptions included the determination of soil horizon, Munsell color, redoximorphic features, and field textural analysis. Soil descriptions focused on near-surface 50 cm horizons. Additionally, soil sample collections used 98 cm³ soil cores to determine volumetric water content and bulk density. Surface-soil horizon samples were analyzed for total moisture content (105°C), total nitrogen, and total carbon concentration via combustion on an Elementar Vario Macro Carbon Nitrogen analyzer (900°C) (Kahn 1998; Klute 1986).

8.2.2 Soil Biogeochemistry

Four soil cores were taken from representative areas within each habitat type during August 2014. Push cores were collected using 7 cm diameter acrylic tubes to collect approximately the top 30 cm of soil. The cores were placed in a core rack, covered, and stored at 4°C. Homogenized soil subsamples from the top 0–5 cm of each core were used to investigate soil conditions within the zone of highest microbial activity (Gardner and White 2010). Analyses included particle size, soil organic matter content, bulk density, total carbon (TC), total nitrogen (TN), total phosphorus (TP), extractable ammonium (NH₄-N), and soluble reactive phosphorus (SRP), and microbial biomass nitrogen (MBN).

Determining soil organic matter content followed the LOI method; dried ground samples underwent combustion at 550°C in a muffle furnace for 4 hours (Sparks 1996). Dry-weight bulk density was calculated from the total wet weight of the soil sample, corrected for moisture content, and divided by volume of the soil section. The TC and TN were measured on dried, ground soil samples by using an Elemental Combustion System (Costech Analytical Technologies, Inc., Valencia, CA). The TP was quantified using the ashing-digestion method as described by Andersen (1976). Extractable NH₄-N and SRP were determined by a 2 M KCl (potassium chloride) extraction followed by filtering through a 0.45 µm Supor-450 membrane filter, acidified to a pH < 2, and stored at 4°C until analysis using a SEAL AQ2 Automated Discrete Analyzer (SEAL Analytical, West Sussex, England) (USEPA 1993). Detection limits for NH₄-N and SRP were

0.012 and 0.005 mg L⁻¹, respectively. The MBN used the chloroform fumigation-extraction method (Vance et al. 1987) in which the total dissolved nitrogen (TDN), determined on a Shimadzu Scientific Instrument TOC-VCSN (Columbia, MD), in non-fumigate samples is subtracted from TDN in fumigate samples.

Nutrient flux and potential nitrate reduction rates were determined using intact soil cores. The existing water column was siphoned out of each core and replaced with a 2 mg L⁻¹ nitrate nitrogen (NO₃-N) solution by using filtered Atchafalaya River water to create a water column of 15 cm (0.57 L) (Theriot et al. 2013). Cores were incubated in a water bath maintained at 23.5°C under aphotic conditions. Each core was aerated using PharMed BPT tubing and Tetra® Whisper 40 UL Air Pumps to maintain uniform, aerobic conditions representative of the surface water in the river channel. Dissolved oxygen and pH were measured using a YSI 556 MPS (YSI Incorporated) on days 4 and 10 of incubation. Water samples were taken on initial reflooding and then every other day for a period of 10 days. During sampling, 6 ml of water was removed from each water column and replaced with 6 ml of deionized water. Each core was filled with deionized water to 15 cm prior to sampling to compensate for evaporative losses in the water column. Water samples were filtered through a 0.45 µm syringe filter, acidified to pH < 2 for preservation, and stored at 4°C for NO₃-N, NH₄-N, and SRP colorimetric determination as described above. Nitrate reduction rates were calculated by standardizing nitrate concentrations to core surface area over time.

The length of time each site was inundated was needed to estimate NO₃⁻ removal in 2013 for both Horseshoe Bend Island and Island E. To do this, 2013 river stage data from the Lower Atchafalaya River at the Morgan City U.S. Geological Survey (USGS) monitoring station (USGS 07381600; www.USGS.gov) was paired with individual site elevation data to determine how many days in the year each island was submerged. Days inundated was averaged over each habitat type. The estimated days of inundation were then used in conjunction with average NO₃⁻ reduction rates and corresponding habitat area to determine approximate total NO₃⁻ removed by each island during 2013. Nitrate-nitrogen removal estimates used the nitrate removal rates from core incubations combined with the habitat area and inundation period; this calculation assumed that denitrification occurred throughout periods of inundation.

Statistical relationships were examined between habitat types (e.g., palustrine emergent vs. aquatic bed) and study areas (e.g., Island E vs. Horseshoe Bend Island) using ANOVA ($\alpha = 0.05$) with Tukey's post hoc analysis (SPSS v.17.0). Model assumptions were tested using Levene's test for equality of variance and the Shapiro-Wilk test for normality ($\alpha = 0.05$). Data were transformed, as necessary, to meet model assumptions. Study area and habitat type were considered independent variables. Dependent variables included bulk density, soil organic matter, TC, TN, TP, extractable $\text{NH}_4\text{-N}$, SRP, nitrate flux rates, MBN, site elevation, and days inundated in 2013.

8.3 Results

8.3.1 Soil descriptions

The soils examined throughout the study area consisted of textures ranging from loamy sands to very fine sands. In general, surface soil layers contained thin, dark, organic-rich horizons (10YR 3/1 or 10YR 3/2) underlain by depleted materials (e.g., 10YR 5/2) in subsurface horizons. Soil horizons also exhibited redoximorphic iron and manganese concentrations in the form of rust-colored (e.g., 10YR 5/4) pore linings and masses within the soil matrix.

The presence of dark soil layers near the surface indicates the accumulation of carbon and organic materials due to frequent inundation and saturation and associated retardation of microbial decomposition rates. Additionally, the presence of depleted materials and redoximorphic features at depth indicates that biogeochemical reduction is occurring, an important function of hydric soils. All soils examined met one or more field indicators of hydric soils (NRCS 2010). Observed indicators included F3 (depleted matrix), S5 (sandy redox), and A5 (stratified layers). Many of the soils exhibited buried surface horizons, signatures of recent sediment inputs, and indicators of active soil-forming processes associated with over-bank flooding and material deposition. Table 14 provides complete soil descriptions.

Table 14. Soil descriptions indicating layer depth (cm), matrix color, redoximorphic features, and soil texture at each sample plot.

Horseshoe Bend1	Horseshoe Bend 2	Horseshoe Bend 3	Horseshoe Bend 4
0–30; 10YR 5/2; 20% 7.5YR 5/4 C/M/PL/LC	0–20; 10YR 5/2; 20% 7.5YR 5/4 C/M/PL/LC	0–15; 10YR 5/3; 10% 10YR 5/6 C/PL/LC	0–2; 10YR 3/1; high OC
30–45; 10YR 4/1; 7% 7.5YR 5/4 C/M/PL/S	20–35; 10YR 4/1; 2% 7.5YR 5/4 C/M/PL/LC	15–33; 10YR 5/1; 20% 7.5YR 5/6 C/PL/LC	2–30 10YR 4/4; 7% 10YR 5/6 C/M/LC
		33–43; 10YR 4/1; 10% 7.5YR 5/4 C/M/S	30–40; 10YR 5/3; 7% 10YR4/6 CM; 15% 10YR 5/2 D/M/LC
Horseshoe Bend 5	Horseshoe Bend 6	Horseshoe Bend 7	Island E 1
0–1; 10YR 5/2; 20% 10YR5/6 C/M/PL/LC	0–1; 10YR 3/2; high OC	0–1; 10YR 2/1; high OC	0–7; 10YR 5/2; 10% 7.5YR 5/6 C/PL/LC
1–12; 10YR 4/2; 5% 10YR 5/6 C/M/PL/Bw/LC	1–10; 10YR 5/2; 10% 10YR 5/6 C/PL/LC	1–15; 10YR 4/2; 20% 10YR 5/4 C/PL/M/LC	7–15; 10YR 5/2; 35% 7.5YR 5/6 C/PL/LC
12–43; 10YR 5/1; 10% 10YR 4/6 C/M/PL/LC	10–16; 10YR 3/1; 25% 10YR 5/6 C/M/PL/Bw/LC	15–30; 10YR 5/1; 5% 10YR 5/6 C/M/LC	15–25; 10YR 4/1; LC
	16–40; 2.5Y 5/1; 15% 10YR 4/6 C/PL/S	30–40; 10YR 5/3; 10% 10YR 5/6 C/M/LC	25–38; 10YR 5/1; S
Island E 2	Island E 3	Middle Island 1	Middle Island 2
0–7; 10YR 5/2; high OC	0–1; 10YR 2/1; OC/Peat	0–1; 10YR 2/1; L/C	0–0.5; 10YR 2/1; LC
7–20; 10YR 4/1; LC	1–12; 10YR 4/2; 30% 10YR 5/6 C/PL/LC	1–10; 10YR 4/2; 10% C 10YR 4/6 PL/LC	0.5–15; 10YR 4/1; 20% 10YR 4/6 C/PL/LC
20–33; 10YR 4/1; S	12–18; 10YR 4/1; 2% 10YR 4/4 C/PL/LC	10–30; 10YR 5/1; 15% 10YR 4/4 C/PL/LC	15–30 10YR 3/3; 10% 10YR 4/6 C/PL/LC
	18–38; 10TR 4/2; S		

C = redoximorphic concentration

D = redoximorphic depletion

M = matrix

PL = pore lining

S = sandy

LC = loamy-clayey

Bw = buried soil horizon

OC = high organic matter content.

8.3.2 Soil nutrients and biogeochemical cycling

As described above, soils on Horseshoe Bend Island displayed pedogenesis and the development of morphological features associated with developing wetland areas. Nutrient data demonstrates that Horseshoe Bend Island exhibits increasing soil TC and TN concentrations on the older, more mature portions of the island (Figure 22). These findings demonstrate that the island is accumulating soil nutrients as vegetative communities mature and carbon sequestration occurs. In the current study, soil C concentrations reached 1.8% following 14 years of island development. Craft (2000) reported similar results with restored wetlands near coastal North Carolina, accumulating 2.0% C within 15 years (converted from soil organic carbon as described in Pribyl [2010]). Nitrogen showed similar results with observed increases of 0.1% nitrogen occurring within 14 years of island establishment. Craft (2000) reported average concentrations of 0.1 % over a 15-year-old restored wetland.

Figure 22. Soil carbon and nitrogen concentrations showed increases with stand age on Horseshoe Bend Island.

When comparing the two treatments, a number of soil parameters related to soil development and nutrient concentrations were significantly different between Horseshoe Bend Island and Island E (Table 15; Figure 23). The Horseshoe Bend Island was established in 1992, 8 years prior to the initial dredged material placement at Horseshoe Bend Island. As a result, portions of Horseshoe Bend Island were established 8–18 years following the construction of Horseshoe Bend Island. The differences in wetland age coupled with the construction practices help to account for the observed differences in soil properties. For example, bulk density was significantly

higher at Horseshoe Bend Island (mean 0.67 g cm^{-3}) than at Island E (mean 0.35 g cm^{-3} ; $p = 0.003$). Craft (2000) reported similar results with bulk density values of 0.7 g/cm^3 in 15-year-old created wetlands, decreasing to 0.45 g/cm^3 after 25 years.

Table 15. Soil physical, nutrient, and biogeochemical properties and estimated nitrate removal calculation ($\pm$ standard error) at Horseshoe Bend Island (HB) and Island E (IE) habitats.

Parameter	Wetland	Aquatic	Habitat type		Scrub-Shrub
			Emergent	Forested	
Bulk Density (g cm^{-3})	HB	0.6 ± 0.15	0.56 ± 0.18	0.65 ± 0.05	0.85 ± 0.07
	IE	0.37 ± 0.06	0.27 ± 0.03	0.23 ± 0.05	0.52 ± 0.08
Organic Matter (%)	HB	10.71 ± 4.17	10.08 ± 2.88	9.34 ± 1.95	3.95 ± 0.63
	IE	10.23 ± 1.14	26.39 ± 3.2	35.6 ± 7.55	16.04 ± 6.68
TC (g kg^{-1})	HB	32.8 ± 15.67	34.16 ± 10.8	31.0 ± 9.28	12.66 ± 2.06
	IE	23.18 ± 0.62	111.17 ± 9.7	153 ± 31.4	61.64 ± 31.7
TN (g kg^{-1})	HB	3.09 ± 1.58	2.93 ± 0.89	2.5 ± 0.72	1.44 ± 0.52
	IE	2.56 ± 0.28	8.55 ± 0.53	12.29 ± 2.5	4.51 ± 2.1
TP (mg kg^{-1})	HB	1007.8 ± 409	741.8 ± 198	585.33 ± 89	1650 ± 1113
	IE	961.69 ± 73	707.8 ± 154	828.12 ± 97	463.02 ± 106
Extractable $\text{NH}_4\text{-N}$ (mg kg^{-1})	HB	56.41 ± 28.29	42.27 ± 17.2	26.09 ± 7.2	7.14 ± 2.14
	IE	83.14 ± 13.19	48.49 ± 18.2	36.61 ± 2.7	13.39 ± 8.12
Extractable SRP (mg kg^{-1})	HB	0	0	0.36 ± 0.36	1.56 ± 0.98
	IE	0	0.04 ± 0.04	0.06 ± 0.06	0.01 ± 0.01
MBN (mg kg^{-1})	HB	39.33 ± 13.71	19.93 ± 3.7	62.0 ± 28.0	29.33 ± 11.24
	IE	18.25 ± 6.93	37.67 ± 15.9	51.2 ± 19.1	47.73 ± 7.76
$\text{NO}_3\text{-N}$ flux ($\text{mg m}^{-2} \text{ day}^{-1}$)	HB	-19.3 ± 4.2	-35.7 ± 3.4	-22.5 ± 6.7	-23.4 ± 4.4
	IE	-27.8 ± 9.3	-31.0 ± 4.7	-14.2 ± 4.7	-23.4 ± 4.9

Wetland	Habitat Type	Area (m^2)	Average $\text{NO}_3\text{-N}$ flux ($\text{mg m}^{-2} \text{ d}^{-1}$)	Inundation (d yr^{-1})	$\text{NO}_3\text{-N}$ removed ($\text{kg NO}_3\text{-N yr}^{-1}$)	Total $\text{NO}_3\text{-N}$ removed ($\text{kg NO}_3\text{-N yr}^{-1}$)
HB	Aquatic	191092	-19.3 ± 4.2	360.0 ± 2.8	1325.8	1645.5
	Emergent	40266	-35.7 ± 3.4	97.8 ± 3.7	140.5	
	Forested	100160	-22.5 ± 6.8	69.8 ± 8.6	157.5	
	Shrub/Scrub	14488	-23.0 ± 4.4	63.8 ± 4.2	21.7	
IE	Aquatic	134760	-27.7 ± 9.3	351.0 ± 3.5	1312.7	1577.2
	Emergent	19546	-31.0 ± 4.7	89.3 ± 10.6	54.1	
	Forested	68716	-41.2 ± 5.4	70.0 ± 4.5	198.2	
	Shrub/Scrub	8458	-32.4 ± 5.0	61.5 ± 11.4	12.2	

Figure 23. Comparison of (a) soil carbon (C), (b) soil nitrogen (N), (c) soil organic matter, and (d) bulk density at Horseshoe Bend Island and Island E. Columns with different letters were statistically different; error bars represent standard error.

Soils at Horseshoe Bend Island contained significantly lower concentrations of soil nutrients than at Island E with regards to soil organic matter ($p < 0.001$), TC ($p < 0.001$), and TN ($p = 0.001$). No significant differences were identified in TP concentrations, extractable $\text{NH}_4\text{-N}$, or extractable SRP. The low extractable SRP at several sample locations (below detection) documents the phosphorous limitation of the system, similar to low phosphorous concentrations seen in the Mississippi River water (Roy et al. 2013).

The disparity in soil nutrient properties between the study sites cannot be easily attributed to either construction technique or site age (Craft et al. 1999). However, several general observations can be made to further discuss the implications of novel dredged material placement techniques. The

Island E area is the product of dredged materials distributed via a hydraulic-pipeline delivery mechanism (Palermo and Francingues 1992; Palermo et al. 2009), which is very successful in building land though it leaves depressions, or “doughnuts,” near the center of deposition where the pipeline aperture is positioned during sediment delivery. This technique leads to the deposition of fine materials. Conversely, the formation of Horseshoe Bend Island resulted from the reworking of dredged material placed up-river, resulting in creation of a linear feature oriented perpendicular to the direction of flow, similar to other mid-river wetlands in the region (Berkowitz et al. 2014). The relationship between soil texture and nutrient content is well documented (Jenny 1980; Parton et al. 1987), with finer fractions associated with higher organic matter and nutrient content. Therefore, the lower nutrient values observed at Horseshoe Bend Island may be related to the coarse nature of sediments associated with the construction technique that mimics natural processes, in addition to difference in wetland age.

The microbial biomass represents a small but very active nutrient pool in the soil environment and, as a result, regulates the amounts of bioavailable nutrients in the system. Therefore, MBN serves as a proxy measure of microbial pool size in soils that can be linked to biogeochemical cycling (White and Reddy 2001). MBN averaged $37.6 \pm 14.1 \text{ mg kg}^{-1}$ at Horseshoe Bend Island and $38.7 \pm 12.4 \text{ mg kg}^{-1}$ at Island E. No significant differences in microbial pool sizes were detected between wetlands. The observed values compare well with the range of 20–35 mg kg^{-1} reported by VanZomeran et al. (2013) and agree with Groffman et al. (1996) who report microbial nutrient pools on the order of 1%–3% of the total nutrient concentrations. Conversely, these findings remain significantly lower than values reported by Gardner and White (2010) who reported MBN concentrations of 340 mg N kg^{-1} in a wetland receiving flood waters from the Mississippi River. Notably, the soils examined by Gardner and White (2010) occurred within a well-established wetland exhibiting thick organic soils with more than 75% higher soil TN concentrations and significantly lower bulk density than those from the current study, accounting for the observed differences.

Extractable $\text{NH}_4\text{-N}$ represents the readily bioavailable nitrogen pool in the soil system, which can drive ecosystem primary productivity; Island E exhibited significantly more extractable $\text{NH}_4\text{-N}$ ($40.0 \pm 7.6 \text{ mg kg}^{-1}$) than

Horseshoe Bend Island ($25.8 \pm 5.5 \text{ mg kg}^{-1}$; $p = 0.025$). This result suggests that there is ample bioavailable nitrogen in the soil to support the wetland vegetation.

Average $\text{NO}_3\text{-N}$ flux at Horseshoe Bend Island was $-25.2 \pm 2.94 \text{ mg NO}_3\text{-N m}^{-2} \text{ d}^{-1}$ while at Island E the average $\text{NO}_3\text{-N}$ flux rate was $-30.8 \pm 2.19 \text{ mg NO}_3\text{-N m}^{-2} \text{ d}^{-1}$. There were no significant differences detected in $\text{NO}_3\text{-N}$ flux rates among wetlands or habitats, which suggest both wetlands have sufficient carbon availability and reducing conditions to process nitrate. These rates are comparable with the low ranges of nitrate reduction rates determined from Lake Pontchartrain sandy sediments at $-24.3 \text{ mg NO}_3\text{-N m}^{-2} \text{ d}^{-1}$ (Roy and White 2012) and lower than maximum rates of $-40.3 \text{ mg NO}_3\text{-N m}^{-2} \text{ d}^{-1}$ found in an highly organic, freshwater wetland soil (Yu et al. 2006).

The only difference in potential removal of nitrate between sites is linked to the difference in elevation of each habitat type, with the lowest elevation flooded for a longer portion of the year with greater potential for removal of nitrate annually. To investigate this, potential $\text{NO}_3\text{-N}$ removal during 2013 was calculated for each wetland as a function of $\text{NO}_3\text{-N}$ flux, days inundated in 2013, and total wetland area (Table 15). The Horseshoe Bend Island wetland removed an estimated 1645 kg $\text{NO}_3\text{-N}$ while Island E removed an estimated 1577 kg $\text{NO}_3\text{-N y}^{-1}$ for 2013. There was no significant difference between potential nitrate reduction capacity for the two wetlands. Aquatic bed habitats displayed significantly higher potential $\text{NO}_3\text{-N}$ removal rates than other wetland types ($p < 0.001$) because of their lower elevations. Submerged aquatic bed habitats are inundated year-round, promoting anaerobic conditions; contain significantly higher organic matter contents than other habitats ($p = 0.034$); and have continuous inputs of nitrogen in river water, providing a hot spot for nutrient reduction via denitrification, assimilation, and other mechanisms.

8.4 Discussion

The soils observed within the created wetland islands display common characteristics associated with riverine wetlands exposed to periodic inundation, overbank flooding, and sediment deposition. Soil profiles indicate an abundance of depleted materials interspersed with higher chroma sandy minerals from recent flooding events. Notably, buried soil horizons and wavy boundary transitions demonstrate the frequency of deposition events as observed through the presence of stratified layers (NRCS 2010).

Further, the development of dark, organic rich surface horizons indicates a decrease in microbial decomposition rates due to periodic saturation or inundation. The profusion of redoximorphic features within subsurface layers demonstrates that soil biogeochemical reduction regularly occurs within island soils. The onset of reduced, anaerobic conditions promotes wetland biogeochemical functions, including carbon sequestration, nutrient cycling, removal and sequestration of elements and compounds, and denitrification (Reddy and DeLaune 2008; Smith and Klimas 2002).

As expected, the soil characteristics of Horseshoe Bend Island continue to develop, with older portions of the island displaying increased nutrient concentrations. Additionally, Horseshoe Bend Island contains less soil carbon, nitrogen, and organic matter and higher bulk density (i.e., the soils are less developed) than Island E, which was established 7 year prior. Rates of soil microbial activity at the dredged material supported islands proved comparable to results reported in the literature for other natural and constructed wetlands in southern Louisiana. Also, the dredged material supported islands demonstrated the capacity to remove nutrients from overlying floodwaters and to contribute to water quality improvements associated with decreasing the hypoxic zone in the Gulf of Mexico.

The calculations presented above are theoretical, assuming anaerobic conditions during periods of inundation, a constant $\text{NO}_3\text{-N}$ concentration in the river water, and no fluctuations in $\text{NO}_3\text{-N}$ removal rates over time. The potential $\text{NO}_3\text{-N}$ removal in the study area represents only a minute fraction of the 20 million tons of nitrate entering the Mississippi River system per year (Goolsby et al. 1999), contributing less than 0.001% of the estimated 21,000–52,000 km^2 of created or restored wetlands required to significantly decrease $\text{NO}_3\text{-N}$ loading to the Gulf of Mexico (Mitsch et al. 2001). However, the results of the current study demonstrate that despite the differences in soil physical parameters and nutrient concentrations, the wetlands examined display equivalent levels of biogeochemical function and nutrient removal capacity.

An additional potential benefit of the Horseshoe Bend Island site is that the river water can access the marsh from all edges due to its location fully within the main river channel while the Island E sites have limited exchange with the river water on one side as they are located along the edge of the channel. Consequently, a wetland created using EWN techniques may be able to process more nitrate than an equally sized traditional

dredged material constructed wetland (e.g., Island E). Future work should seek to quantify the difference in hydrologic connectivity with the river. Results of this study suggest that the novel engineering and construction techniques associated with EWN are a viable option for creating wetlands capable of nitrate reduction. Additionally, future projects using EWN principles should focus on maximizing creation of submerged aquatic bed habitat to take advantage of the great nutrient reduction potential of this habitat type.

9 Hydrodynamics and Sediment Transport Modeling

9.1 Introduction

The strategic placement of dredged materials has been shown to provide habitat, storm surge reduction, improved navigation, and other benefits (Yozzo et al. 2004; Day et al. 2007). However, few studies have examined the impact of open-water dredged material placement, especially in a riverine setting. To address this data gap, the following section describes results from two modeling efforts examining the hydrodynamics and sediment transport characteristics occurring at Horseshoe Bend Island.

The Lower Atchafalaya River is connected to Gulf Intracoastal Water Way (GIWW), Bayou Boeuf, Bayou Penchant, Avoca Island Cutoff, and Shell Pass (Figure 24), thus representing an important navigable waterway linking the Gulf of Mexico and inland communities and waterways. The range of hydrological cycles and tidal conditions of Atchafalaya Bay provides a typical river-lake-estuary-bay system hydrodynamics response (Wang 1984). The system is thus controlled by hydrology, meteorology, and astronomical tides with varying time scales. Upstream, river hydrodynamics near Morgan City remain dominated by hydrological influence and tidal flows that modulate the strong river inflows, especially during the spring and summer wet season. Downstream areas are dominated by micro-tidal circulation patterns influenced by weather patterns.

Previous studies suggest that relocation of the Federal navigation channel to the east of Horseshoe Bend Island would reduce maintenance dredging efforts. Thus, two modeling studies—through a one-dimensional model, Hydrologic Engineering Center (HEC)-6T, and a two-dimensional model, Resource Management Associates (RMA)-2, were applied to the lower Atchafalaya River. Study results provide insight into the impact of strategic dredged material placement and the formation of Horseshoe bend Island on current hydrodynamics and sediment processes within the river. Additionally, modeling approaches allow for predictions regarding the future trajectory of Horseshoe Bend Island development and the impact of hydrodynamics and sediment transport on navigation.

Figure 24. Horseshoe Bend Island and the lower Atchafalaya River basin in coastal Louisiana. North is at the top of the figure.

The one-dimensional sediment transport model, HEC-6T, was used to estimate the maintenance dredging requirement for 52 km (32.5 miles) of the Lower Atchafalaya River between Morgan City at River Mile –119 and Eugene Island at River Mile –151.5 over a 50-year period. The most critical dredging reaches are Horseshoe Bend between River Miles –134.4 and –137.9 and the Bay Channel between River Miles –151.5 and –139.8. The study also included (1) 22.9 km of Bayou Chene from the Avoca Island Cutoff to the Gulf Intracoastal Waterway, (2) 2.4 km of Bayou Boeuf between Amelia and Bayou Chene, and (3) 5.5 km of Bayou Black between Bayou Chene and Bay Wallace. The two upstream boundary control points were at Morgan City on the Lower Atchafalaya River (Control Point 6) and at Amelia on Bayou Boeuf (Control Point 3). The discharge hydrograph was simulated in the numerical model by a series of steady-state events. The duration of these events was set at 1 day in the model. Mean daily flow data for water years 1998–2002 and January–June 1975, from the USGS website (USGS 2016), were used to develop a hydrograph at the two upstream boundary points. Downstream water-surface elevations at Eugene Island and Bay Wallace were used as boundary conditions in the numeri-

cal model. The downstream water surface elevation was set based on existing mean annual values. Starting water-surface elevations were increased over time to simulate estimated delta growth.

As part of the Dredged Material Management Plan in force, the USACE New Orleans District studied the relocation of the authorized Federal channel through Horseshoe Bend. The proposed relocation of the channel to the east at Crewboat Cut can reduce maintenance dredging by over 0.76 million m³/year. A two-dimensional model, RMA-2, was developed to simulate three Atchafalaya River flow conditions. The changes in velocities were analyzed to determine potential impacts of relocating the channel. Sediment transport was also simulated to study erosion and deposition. The second model uses the model's grid and bathymetry data for defining the boundary conditions.

To assess the formation of the Horseshoe Bend Island in the Atchafalaya River, the U.S. Army Engineer Research and Development Center (ERDC) developed a hydrodynamic model to gain an understanding about how the island footprint is growing and to serve as input into models determining the transport of sediments from dredged material seed mounds to the island. The model analyzed the hydrodynamics around the island and also determined sediment transport characteristics. The hydrodynamic model will help identify the processes associated with hydrodynamics and sediment transport of the island's creation and growth.

9.2 Curvilinear Hydrodynamics in Three Dimensions (CH3D) model

The CH3D model (Chapman et al. 1996) is based on the assumptions of hydrostatic pressure distribution, Boussinesque approximation, and the eddy-viscosity concept. It is composed of mass and momentum conservation equations. In the momentum equation, the velocity variations in time and space are dependent on Coriolis force, horizontal pressure gradients, surface and bottom stresses, and eddy viscosities. For the closure of the equations, the k- ϵ turbulence model is solved. In the horizontal plane, the coordinates are transformed to a boundary-fitted curvilinear grid system. Vertically, sigma-stretched coordinates are used. For this simulation, calendar year 2012 was selected based on data availability. The first 9-month period includes wet spring and dry summer conditions and associated meteorological events.

The solutions are controlled by bathymetry and are subject to boundary conditions especially for barotropic response. The following sections describe the model grid and boundary conditions that were used.

9.2.1 Computational grid

From the National Oceanic and Atmospheric Administration (NOAA) Electronic Navigation Chart (ENC), 5800 sounding data points were extracted for use in the present study (Figure 25). The data were obtained over a 70-year period between 1935 and 2006 and may not reflect the most up-to-date bathymetry data. The existing RMA-2 model grid also contains bathymetry data (Figure 26). Note that NOAA sounding data units are in meters whereas the RMA-2 bathymetry data are in feet. Overall, the depths between NOAA ENC and RMA data are in agreement. Figure 27 shows the portion of RMA-2 grid overlaid at the Horseshoe Bend Island and vicinity. The highest grid resolution is about 50 m near the Horseshoe Bend Island. In general, the channel resolution is about 50–100 m whereas low resolution of about 1800 m was applied near the open boundary grid. The total number of nodes was 12,314.

Figure 25. Sounding data from NOAA's Electronic Navigation Chart (units in meters).

Figure 26. RMA-2 grid with bathymetry data (units in feet).

Figure 27. The RMA-2 grid overlaid in the vicinity of the Horseshoe Bend Island.

The domain for the current CH3D model was set comparable to the RMA-2 model domain in plan view (Figure 28). The resolution at the focus area around Horseshoe Bend Island is roughly 50 m (Figure 29). The CH3D grid cell dimension is 104 by 209, and the total number of surface cells is 8723. The model was set to run in three-dimension. The model is using bathymetry following vertical coordinates, which will give the same number of vertical layers all over the model domain. The number of vertical layers was set to five so that the vertical resolution for the 6 m channel around Horseshoe Bend becomes about 1.2 m.

Figure 28. The RMA-2 model grid in plan view.

Figure 30. Gage locations for Lower Atchafalaya River. Triangles (*blue*) represent USGS gage locations, circles (*purple*) are for NOAA tide gage locations, and the asterisk (*green*) denotes a meteorological station.

Table 16. Tidal harmonics at the Lawma gage station, LA (NOAA 2013).

Constituents	Amplitude (ft)	Phase (degrees)	Description
K1	0.423	313.1	Lunar diurnal constituent
O1	0.377	312.4	Lunar diurnal constituent
M2	0.292	120.8	Principal lunar semidiurnal constituent
P1	0.161	309.5	Solar diurnal constituent
S2	0.112	105.9	Principal solar semidiurnal constituent
Q1	0.089	300	Larger lunar elliptic diurnal constituent

Table 17. USGS gage locations in the study area.

STID	STATION
73816501	Avoca Island Cutoff south of Morgan City, LA.
73814675	Bayou Boeuf at Railroad Bridge at Amelia, LA
7381654	Atchafalaya Bay at Eugene Island, LA
73816503	Bayou Penchant south of Morgan City, LA
7381600	Lower Atchafalaya River at Morgan City, LA
73816202	GIWW at mile 103 S of Morgan City, LA
293229091230800	CRMS0465-H01-RT

The hydrology record at the five USGS gages over the 7-year period between WY 2007 and WY 2013 shows intra- and interannual signals (Figure 31). Gages along the Atchafalaya River (7381600 and 73816503), the GIWW (73816202), and Bayou Boeuf (73814675) exhibit similar hydrological responses. The maximum gage height variation in the upstream station (7381600) is about 3.0 m whereas it is about 1.5 m about 20 km downstream (73816503). Overall, these gage data indicated that WY 2011 was a wet year and WY 2012 was a dry year.

Figure 31. Gage heights from USGS gages over the 7-year period between water years 2007 and 2013. Gages are 7381600 (*upper left panel*), 73816202 (*lower left panel*), 73814675 (*upper right panel*), and 73816503 (*lower right panel*).

To assess intra-annual variability, the 2012 water level data from USGS gage at upstream (7381600) was compared with NOAA tide data from upstream (8764044) and downstream (8764227) gages (Figure 32). Persistent hydrological input over April was observed. Meteorological conditions were observed by using wind speed and surface pressure data. A meteorological

logical event associated with passage of a low-pressure system on 4 September 2011 resulted in a high-water-level event of about 1.5 m (Figure 33, lower left panel).

Figure 32. Intra-annual variability of the hydrology record across select USGS and NOAA gages for water year 2012. Shown are gage height (ft) for USGS gage 7381600 (*upper left* panel), and for NOAA gages 8764044 (*blue*) and 8764227 (*green*), water levels (m; *lower left* panel), surface pressure (mb, *upper right* panel), and wind speed (m/s; *lower right* panel).

Figure 33. Flow boundary conditions in 2012 using data from three USGS gage stations.

The upstream flow boundary was set near Morgan City by using data from USGS gage 7381600. Other flow boundary conditions were set at GIWW and at flows from the east via Bayou Penchant and the Avoca Island Cut-off. Figure 34 shows flows at the three boundaries, indicating that the flows at Morgan City dominate the system.

Figure 34. Seven locations (*green triangles*) used for obtaining water level data along the open boundaries. The *green circle* near the entrance of Atchafalaya River marks the location of NOAA gage 8764227.

Water levels were set using seven points along the open boundaries (Figure 35). The relationships of amplitude and phase between each location and NOAA gage 8764227 were obtained from published tide tables. Figure 36 shows the water levels at NOAA gage 8764227. The gap in data indicates a 10-day period between 6 November 2012 and 16 November 2012 when data were unavailable. These data indicate six meteorological events: 18-Feb-2012 20:00:00; 21-Mar-2012 14:00:00; 04-Apr-2012 20:00:00; 19-Aug-2012 11:00:00; 29-Aug-2012 05:00:00; and 30-Aug-2012 20:00:00. July appears to be quiescent in terms of meteorological forcing. Figure 37 shows wind speed and direction observed at NOAA gage 8764227. The plot indicates strong winds associated with a frontal passage is related to a 1-day event between 19 and 29 August 2012. It also suggests

a persisting southwest wind caused surges in water levels around 21 March 2012.

Figure 35. Water levels at NOAA gage 8764227 for 2012 that identified six meteorological events in the study area.

Figure 36. Wind speed (*top panel*) and direction (*bottom panel*) at NOAA gage 8764227.

Figure 37. Predicted (*red lines*) and observed (*blue dots*) hourly water levels for the first 4 months of 2012 at NOAA gage 8764044 at Berwick, LA, and NOAA gage 8764227 at Lawma, LA.

9.2.3 Model calibration results

Water levels at two tide gages, one at NOAA gage 8764044 at Berwick, LA, and NOAA gage 8764227 at Lawma, LA, were compared, showing the predicted water levels against observed hourly water levels at these two locations (Figure 37). The predicted water levels resolve tidal signals and sub-tidal signals for meteorological events. There are no other observed data for water levels nor are currents available to validate the model. Further calibration and validation of the model will require additional field data from the study area.

9.3 General Transport Model (GTRAN)

The GTRAN model uses near-bottom velocities from the CH3D model and calculates transport rates of both suspended load and bedload transport. The calculation also depends on bed characteristics obtained from the study area. Points depicting sediment data sources are shown on the CH3D model grid in Figure 38. Surface sediment grain size data were analyzed and assigned to CH3D model grid cells (Figure 39). Figure 40 shows the median grain size interpolated over the model grid around Horseshoe Bend. Northwest of the island and along the flank of the island, coarse bottom sediments prevail (sands) whereas the northern end of the island is comprised of fine-grained sediments (silts and clays).

Figure 38. Median grain size of bottom sediments in the study area.

Figure 39. Assigned median grain size based on existing data shown in Fig. 40.

Figure 40. Assigned median grain size to CH3D model grid.

Bottom currents vary with the hydrological cycle. The wet season from January through March 2012 shows much stronger bottom currents (Figure 41; maximum about 1.44 m/s) around the island than the dry season from July through September 2012 (Figure 42; maximum about 0.47 m/s). Both suspended load and bedload transport is controlled by bottom currents and by bottom sediment grain size. Typically, the suspended sediment transport rate is an order of magnitude higher than bedload transport. The maximum volume transport rate during the wet season is about 9.66 m³/m/s as suspended load and about 0.71 m³/m/s as bedload. These rates decrease during the dry season—1.50 m³/m/s as suspended load and 0.11 m³/m/s as bedload. Figures 43 and 44 show the mean transport rates during the wet season for suspended sediment and bedload transport, respectively, and indicate possible deposition toward the northeast shore and erosion from the southeast shore of Horseshoe Bend Island.

Figure 41. Mean bottom current during the relatively wet season from January through March 2012.

Figure 42. Mean bottom currents during the relatively dry season from July through September 2012.

Figure 43. Mean suspended sediment transport rate from January through March 2012.

Figure 44. Mean bedload transport rate from January through March 2012.

9.4 Conclusions

A three-dimensional hydrodynamic model, CH3D, was applied to the Lower Atchafalaya River and adjacent Atchafalaya Bay to assess the formation of the Horseshoe Bend Island in the Atchafalaya River to gain a more complete understanding how the island footprint is being created. A one-year simulation during water year 2012 showed the hydrodynamic model behavior is strongly related to hydrological forcing. A sediment transport model, GTRAN, was linked to CH3D to estimate sediment transport rates. The results indicated dynamic response of sediment transport during the wet season extending from January to March and indicated along the eastern shore of Horseshoe Bend Island possible deposition toward the northern shore and erosion from the southern shore. To a lesser degree, the western shore of the island showed erosion along the northern shore and deposition along the southern shore. The model will require more hydrodynamic and sediment transport data from the area to calibrate and validate. Nonetheless, these model results can serve as valuable information so that this innovative and strategic use of dredged sediment can be applied in other island building projects.

10 Discussion and Conclusions

Landin (1989) identified over 100 reports examining wetland creation with dredged material, and dozens more have been published since on this topic. The majority of these studies focus on marsh creation, including information on engineering aspects of dredged material placement and vegetation establishment and comparisons of select ecosystem components between natural and created wetlands (Woodhouse 1979; Knutson 1977; USACE 1986; Webb 1982; Webb and Newling 1985). For example, Broome et al. (1986) evaluated vegetation establishment and trajectory, Kurz et al. (1998) compared fish populations in natural and created wetlands in Florida, and Cammen (1976) examined invertebrate community colonization in created wetlands. Although these studies provide valuable data, the focus on a single ecological factor (e.g., fish) limits the utility of these assessment results. Several studies examined a combination of ecosystem factors to expand the scope of the assessment approach. For example, Craft (2000) examined both invertebrate populations and soil characteristics (e.g., nutrients and bulk density) in natural and restored areas while Alphin and Posey (2000) combined vegetation and invertebrate data sets collected in created marshes.

Streever (2000) presents a review of dredged material supported wetlands, investigating how the success of wetland creation projects should be determined. In that study, the author combines results of a number of previously published studies to examine a variety of factors, including above- and below-ground biomass, invertebrate communities, crustacean communities, and soil characteristics (e.g., organic carbon content). As a result, Streever (2000) presents one of the most comprehensive ecosystem assessments regarding dredged material created wetlands; however, as Streever indicates, the variety of data collection techniques used in the published studies poses challenges to comparative data interpretation.

As a result, the current study applied a multifactor ecosystem assessment of dredged material supported wetlands. Assessment factors included (1) landscape geomorphology, (2) ecosystem classification, (3) floral communities, (4) avian communities, (5) aquatic invertebrates, (6) soils and biogeochemical activity, and (7) hydrodynamic and sediment modeling. By examining ecological components comprising primary producers, microbial communities, invertebrates that form the basis of aquatic food webs,

and higher organisms, the current study provides a comprehensive approach to the assessment of dredged material supported wetlands (Figure 45). The current study provides a framework for future studies examining the ecological, societal, and economic value of the strategic placement of dredged material applied in this manner. The multifactor-assessment approach used in the current study allows for a comprehensive analysis of ecosystem condition across of range of ecosystem components. Study results demonstrate that each of the factors examined at Horseshoe Bend Island proved comparable or exceeded the other study areas examined, including the naturally formed Middle Island and the traditionally created dredged material supported Island E. Therefore, the novel dredged material placement approach used in the creation of Horseshoe Bend Island could be applied in other scenarios to investigate the success and potential benefits of EWN projects that use natural process in such a manner to improve wetland creation and restoration outcomes.

Figure 45. Conceptual model of the multifactor-assessment approach applied at Horseshoe Bend Island.

Examining each ecosystem factor, results indicate that Horseshoe Bend Island provides habitat and biogeochemical functions at rates comparable or

exceeding those at a traditional dredged material supported island and a natural reference island. First, wetland classification and analysis of geomorphic features demonstrated that Horseshoe Bend Island provides a variety of habitat types supporting complex communities of vegetation, invertebrates, soil microbes, and higher organisms (i.e., avian species). The distribution of forested, scrub-shrub, emergent, and emergent aquatic bed habitat types corresponds to the natural distribution reported throughout the study area (University of New Orleans 2002). Second, Horseshoe Bend Island contains a wide variety of vegetation, including more than 85% native species, with species-richness values exceeding observations from both traditional dredged material supported and natural reference areas. Third, the EWN design used at Horseshoe Bend Island resulted in landscape and landform characteristics (e.g., distance from shore and flooding regime) that support a large, successful wading-bird rookery. Fourth, Horseshoe Bend Island supports more invertebrate abundance and diversity than natural islands in the region that lack the emergent aquatic bed landforms resulting from the strategic placement of dredged materials. Finally, the soils at the Horseshoe Bend Island display a capacity to sequester nutrients and other compounds and to perform water quality functions at levels comparable to natural wetlands in the region.

Further research is required to verify the results described herein and to expand the multifactor-assessment approach to other environmental settings. Additionally, increasing the number of observations and study sites promotes confidence in study results while addressing any potential autocorrelation associated with ecosystem assessments in which a small number of manipulated study sites are compared with natural reference areas (Hurlbert 1984). The application of the multifactor-assessment approach to additional ecosystem types and components will help assess the success of wetland creation and restoration practices using both traditional and EWN applications.

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Appendix A: List of Plant Species Observed at Horseshoe Bend Island, Middle Island, and Island E

Scientific Name	Common Name	Family	Middle Island	Island E	Horseshoe Island
<i>Acer negundo</i> L.	boxelder	Aceraceae	X		
<i>Acer rubrum</i> L.	red maple	Aceraceae	X	X	
<i>Alternanthera philoxeroides</i> (Mart.) Griseb.	alligatorweed	Amaranthaceae	X	X	X
<i>Amaranthus australis</i> (Gray) Sauer	southern amaranth	Amaranthaceae			X
<i>Amaranthus</i> spp. L.	pigweed	Amaranthaceae	X	X	X
<i>Ammannia</i> spp.	redstem	Lythraceae			X
<i>Amorpha fruticosa</i> L.	false indigo bush	Fabaceae	X	X	X
<i>Ampelopsis arborea</i> (L.) Koehne	peppervine	Vitaceae	X	X	X
<i>Ampelopsis cordata</i> Michx.	heartleaf peppervine	Vitaceae	X		X
<i>Apios americana</i> Medik.	groundnut	Fabaceae	X		
<i>Aster</i> spp.	aster	Asteraceae	X		
<i>Baccharis halimifolia</i> L.	eastern baccharis	Asteraceae		X	X
<i>Bacopa monniერი</i> (L.) Pennell	herb of grace	Scrophulariaceae	X	X	
<i>Bidens laevis</i> (L.) Britton, Sterns & Poggenb.	smooth beggartick	Asteraceae			X
<i>Boehmeria cylindrica</i> (L.) Sw.	smallspike false nettle	Urticaceae	X	X	X
<i>Borrchia</i> Adans.	seaside tansy	Asteraceae	X		
<i>Brunnichia ovata</i> (Walter) Shinnars	American buckwheat vine	Polygonaceae	X		
<i>Campsis radicans</i> (L.) Seem. Ex Bureau	trumpet creeper	Bignoniaceae	X		
<i>Carex bromoides</i> Schkuhr ex Willd.	broom-like sedge	Cyperaceae	X		
<i>Carex</i> spp.	unknown sedge	Cyperaceae		X	
<i>Carya</i> spp.	hickory	Juglandaceae	X		
<i>Celtis laevigata</i> Willd.	sugarberry	Ulmaceae	X		
<i>Cephalanthus occidentalis</i> L.	common buttonbush	Rubiaceae		X	X
<i>Ceratophyllum demersum</i> L.	coon's tail	Ceratophyllaceae		X	
<i>Cicuta maculata</i> L.	spotted water hemlock	Apiaceae		X	X
<i>Cirsium horridulum</i> Michx.	yellow thistle	Asteraceae		X	X
<i>Colocasia esculenta</i> (L.) Schott	coco-yam	Araceae	X	X	X

Scientific Name	Common Name	Family	Middle Island	Island E	Horseshoe Island
<i>Commelina virginica</i> L.	Virginia dayflower	Commelinaceae	X	X	
<i>Cornus foemina</i> Mill.	stiff dogwood	Cornaceae	X	X	
<i>Crinum americanum</i> L.	seven sisters	Liliaceae		X	X
<i>Cuscuta</i> spp. L.	dodder	Convolvulaceae	X		
<i>Cyperus esculentus</i> L.	yellow nutsedge	Cyperaceae			X
<i>Cyperus pseudovegetus</i> Steud.	marsh flatsedge	Cyperaceae			X
<i>Cyperus</i> spp.	flatsedge	Cyperaceae		X	X
<i>Dichanthelium clandestinum</i> (L.) Gould	deertongue	Poaceae	X	X	
<i>Diospyros virginiana</i> L.	persimmon	Ebenaceae	X		
<i>Echinochloa walteri</i> (Pursh) A. Heller	coast cockspur grass	Poaceae			X
<i>Eichhornia crassipes</i> (Mart.) Solms	common water hyacinth	Pontederiaceae	X	X	X
<i>Eleocharis</i> spp. R. Br.	spikerush	Cyperaceae			X
<i>Equisetum hyemale</i> L.	scouringrush horsetail	Equisetaceae	X		
<i>Eupatorium capillifolium</i> (Lam.) Small	dogfennel	Asteraceae		X	X
<i>Euthamia</i> spp. Nutt. Ex Cass.	goldentop	Asteraceae			X
<i>Forestiera acuminata</i> (Michx.) Poir.	eastern swampprivet	Oleaceae	X		
<i>Fraxinus pennsylvanica</i> Marshall	green ash	Oleaceae	X	X	
<i>Galium</i> spp.	bedstraw	Rubiaceae		X	X
<i>Gleditsia aquatica</i> Marsh.	water locust	Fabaceae		X	X
<i>Gleditsia triacanthos</i> L.	honeylocust	Fabaceae		X	
<i>Heliotropium indicum</i> L.	Indian heliotrope	Boraginaceae			X
<i>Heteranthera dubia</i> (Jacq.) MacMill.	grassleaf mudplantain	Pontederiaceae		X	
<i>Hibiscus laevis</i> All.	halberdleaf rosemallow	Malvaceae	X	X	
<i>Hibiscus lasiocarpus</i> Cav.	rosemallow	Malvaceae			X
<i>Hydrilla verticillata</i> (L. f.) Royle	waterthyme	Hydrocharitaceae		X	X
<i>Hydrocotyle ranunculoides</i> L. f.	floating marshpennywort	Apiaceae		X	
<i>Hydrocotyle verticillata</i> Thunb.	whorled marshpennywort	Apiaceae		X	X
<i>Hymenocallis liriosme</i> (Raf.) Shinnars	spring spiderlily	Liliaceae	X	X	
<i>Hypericum mutilum</i> L.	dwarf St. Johnswort	Clusiaceae			X
<i>Ilex decidua</i> Walter	possumhaw	Aquifoliaceae	X		

Scientific Name	Common Name	Family	Middle Island	Island E	Horseshoe Island
<i>Ipomea</i> spp.	morning-glory	Convolvulaceae	X	X	X
<i>Iris</i> spp. (L.)	iris	Iridaceae		X	
<i>Iris virginica</i> L.	Virginia iris	Iridaceae			X
<i>Iva annua</i> L.	annual marsh elder	Asteraceae		X	X
<i>Juncus effusus</i> L.	common rush	Juncaceae		X	
<i>Justicia lanceolata</i> (Chapm.) Small	looseflower water-willow	Acanthaceae		X	
<i>Kosteletzkya virginica</i> (L.) C. Presl ex A. Gray	Virginia saltmarsh mallow	Malvaceae			X
<i>Lactuca</i> spp.	lettuce	Asteraceae			X
<i>Leersia oryzoides</i> (L.) Sw.	rice cutgrass	Poaceae		X	X
<i>Lemna minor</i> L.	common duckweed	Lemnaceae		X	X
<i>Ludwigia alterniflora</i> L.	seedbox	Onagraceae		X	
<i>Ludwigia decurrens</i> Walter	wingleaf primrose-willow	Onagraceae	X	X	
<i>Ludwigia glandulosa</i> Walter	cylindricfruit primrose-willow	Onagraceae		X	
<i>Ludwigia peploides</i> (Kunth) P.H. Raven	floating primrose-willow	Onagraceae		X	
<i>Ludwigia</i> spp.	primrose-willow	Onagraceae		X	X
<i>Lygodium japonicum</i> (Thunb.) Sw.	Japanese climbing fern	Lygodiaceae	X		X
<i>Lythrum alatum</i> L.	winged lythrum	Lythraceae			X
<i>Lythrum lineare</i> L.	wand lythrum	Lythraceae		X	X
<i>Menispermum canadense</i> L.	common moonseed	Menispermaceae	X		
<i>Mikania scandens</i> (L.) Willd.	climbing hempvine	Asteraceae	X	X	X
<i>Morella cerifera</i> (L.) Small	way myrtle	Myricaceae	X	X	X
<i>Morus rubra</i> L.	red mulberry	Moraceae	X		X
<i>Nelumbo lutea</i> Willd.	American lotus	Nelumbonaceae		X	X
<i>Nymphaea odorata</i> Aiton ssp. <i>odorata</i>	American white waterlily	Nymphaeaceae			X
<i>Nyssa sylvatica</i> Marshall	blackgum	Cornaceae	X		
<i>Packera glabella</i> (Poir.) C. Jeffrey	butterweed	Asteraceae		X	
<i>Panicum repens</i> L.	torpedo grass	Poaceae	X	X	X
<i>Parthenocissus quinquefolia</i> (L.) Planch.	Virginia creeper	Vitaceae	X	X	
<i>Paspalum</i> L.	crowngrass	Poaceae			X
<i>Phanopyrum gymnocarpon</i> (Elliott) Nash	savannah-panicgrass	Poaceae	X		
<i>Phragmites australis</i> (Cav.) Trin. Ex Steud.	common reed	Poaceae			X

Scientific Name	Common Name	Family	Middle Island	Island E	Horseshoe Island
<i>Phyla lanceolata</i> (Michx.) Greene	lanceleaf fogfruit	Verbenaceae			X
<i>Phyla nodiflora</i> (L.) Greene	turkey tangle fogfruit	Verbenaceae	X	X	X
<i>Phytolacca americana</i> L.	American pokeweed	Phytolaccaceae		X	
<i>Pistia stratiotes</i> L.	water lettuce	Araceae		X	X
<i>Platanus occidentalis</i> L.	American sycamore	Platanaceae	X		X
<i>Pluchea rosea</i> Godfrey	rosy camphorweed	Asteraceae			X
<i>Polygonum hydropiperoides</i> Michx.	swamp smartweed	Polygonaceae		X	X
<i>Polygonum</i> L.	knotweed	Polygonaceae	X	X	
<i>Polygonum punctatum</i> Ell.	dotted smartweed	Polygonaceae			X
<i>Pontederia cordata</i> L.	pickerelweed	Pontederiaceae			X
<i>Populus deltoides</i> W. Bart. ex Marshall	eastern cottonwood	Salicaceae	X	X	
<i>Populus heterophylla</i> L.	swamp cottonwood	Salicaceae	X	X	
<i>Quercus nigra</i> L.	water oak	Fagaceae	X		
<i>Rhynchospora colorata</i> (L.) H. Pfeiffer	starrush whitetop	Cyperaceae	X		
<i>Rubus argutus</i> Link	sawtooth blackberry	Rosaceae	X	X	X
<i>Rubus</i> spp. L.	blackberry	Rosaceae	X	X	
<i>Ruppia maritima</i> L.	widgeongrass	Ruppiaceae		X	X
<i>Sagittaria latifolia</i> Willd.	broadleaf arrowhead	Alismataceae		X	X
<i>Salix nigra</i> Marshall	black willow	Salicaceae	X	X	X
<i>Salvinia minima</i> Baker	water spangles	Salviniaceae		X	X
<i>Sambucus nigra</i> L. ssp. <i>canadensis</i>	common elderberry	Caprifoliaceae	X	X	X
<i>Samolus valerandi</i> L. ssp. <i>parviflorus</i> (Raf.) Hulten	seaside brookweed	Primulaceae			X
<i>Saururus cernuus</i> L.	lizard's tail	Saururaceae	X	X	X
<i>Schoenoplectus americanus</i> (Pers.) Volkart ex Schinz & R. Keller	chairmaker's bulrush	Cyperaceae			X
<i>Schoenoplectus californicus</i> (C.A. Mey.) Palla	California bulrush	Cyperaceae			X
<i>Sesbania drummondii</i> (Rydb.) Cory	poisonbean	Fabaceae		X	X
<i>Sesbania herbacea</i> (Mill.) McVaugh	bigpod sesbania	Fabaceae			X
<i>Sicyos angulatus</i> L.	oneseed bur cucumber	Cucurbitaceae		X	X
<i>Smilax auriculata</i> Walter	earleaf greenbriar	Smilacaceae	x		
<i>Solanum americanum</i> Mill.	American black nightshade	Solanaceae			X

Scientific Name	Common Name	Family	Middle Island	Island E	Horseshoe Island
<i>Solidago altissima</i> L.	Canada goldenrod	Asteraceae			X
<i>Solidago sempervirens</i> L.	seaside goldenrod	Asteraceae			X
<i>Sphenoclea zeylanica</i> Gaertn.	chickenspike	Sphenocleaceae			X
<i>Taxodium ascendens</i> Brongn.	pondcypress	Cupressaceae	X	X	
<i>Taxodium distichum</i> (L.) Rich.	bald cypress	Cupressaceae	X	X	X
<i>Thelypteris kunthii</i> (Desv.) Morton	Kunth's maiden fern	Thelypteridaceae	X	X	X
<i>Toxicodendron radicans</i> (L.) Kuntze	eastern poisonivy	Anacardiaceae	X	X	
<i>Tradescantia</i> spp. L.	spiderwort	Commelinaceae		X	
<i>Triadica sebifera</i> (L.) Small	Chinese tallowtree	Euphorbiaceae	X	X	X
<i>Typha latifolia</i> L.	broadleaf cattail	Typhaceae		X	X
<i>Ulmus americana</i> L.	American elm	Ulmaceae	X		
<i>Ulmus rubra</i> Muhl.	slippery elm	Ulmaceae	X		
<i>Valerianella radiata</i> (L.) Dufr.	beaked cornsalad	Valerianaceae		X	
<i>Vallisneria americana</i> Michx.	American eelgrass	Hydrocharitaceae		X	
<i>Vigna luteola</i> (Jacq.) Benth.	hairypod cowpea	Fabaceae			X
<i>Vigna unguiculata</i> (L.) Walp.	cowpea	Fabaceae		X	X
<i>Wisteria</i> spp. Nutt.	wisteria	Fabaceae	X		
<i>Woodwardia virginica</i> (L.) Sm.	Virginia chainfern	Blechnaceae		X	
<i>Xanthium strumarium</i> L.	rough cocklebur	Asteraceae			X

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