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Charlotte Harbor NEP Special Issue of *Florida Scientist*
"The State of Our Watersheds and Estuaries"

(Continued on the Back Cover)

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Florida Scientist

QUARTERLY JOURNAL OF THE FLORIDA ACADEMY OF SCIENCES

JAMES AUSTIN, *Editor* DONNA L. GOSNELL, *Co-Editor*

Volume 76 Spring, 2013 Number 2

DEDICATION—This issue is dedicated to the thousands of individual partners of the Charlotte Harbor National Estuary Program who share their knowledge, efforts and enthusiasm to collectively protect and restore the exceptional natural resources found throughout our watershed.

ACKNOWLEDGMENTS—It is with sincere gratitude that the Charlotte Harbor National Estuary Program (CHNEP) and Lead Guest Editor acknowledge the outstanding efforts of the contributors to this publication. It is though the expertise, diligence, patience and flexibility of the authors and guest editors that this issue has been made possible.

This issue is based on the scientific information presented at the 2011 Charlotte Harbor Watershed Summit "State of Our Watersheds and Estuaries" held in Punta Gorda, Florida March 30 and 31, 2011 at the Charlotte Harbor Event and Conference Center. The 2011 Watershed Summit included 56 presentations and posters which focused on recent technical findings throughout the CHNEP watershed relating to wetlands and submerged vegetation, invertebrates and shellfish, fisheries, water quality and quantity, and restoration activities. The contributions of each presenter further our knowledge and understanding of intricacies and complexities of the natural systems throughout southwest Florida. The 18 manuscripts included here were prepared by Summit authors interested in conveying their results through a peer reviewed journal so that the knowledge may be shared with and guide other scientists and managers throughout the region.

The CHNEP also thanks the many sponsors of the 2011 Watershed Summit, whose essential support contributed to the success of the Summit and in bringing researchers and citizens together for this invaluable triennial conference. 2011 Summit sponsors included: Caloosahatchee River Citizens Association, CF Industries, Florida Native Plant Society Mangrove Chapter, Janicki Environmental, Incorporated, Jelks Family Foundation, Mosaic, Mote Marine Laboratory, Peace River Manasota Regional Water Supply Authority, Myakka Conservancy, Scheda Ecological Associates, Sierra Club Calusa Chapter, Southwest Florida Watershed Council, and the Southwest Water Management District.

Technical reviews of the manuscripts in this issue were conducted by many scientists who contributed their time and expertise toward making each manuscript included here shine. We genuinely thank these reviewers for their conscientious and thorough comments:

Nathan Bailey (Florida Department of Environmental Protection), Margaret Banyan (Florida Gulf Coast University), Richard Bartleson (Sanibel-Captiva Conservation Foundation), Mike Bauer (City of Naples), Lisa Beever (Charlotte Harbor National Estuary Program), Karen Bickford (Lee County Natural Resources), Steve Bortone (Gulf of Mexico Fishery Management Council), Jaime Boswell, Mike Britt (City of Winter Haven), ZinJian Chen (Southwest Florida Water Management District), Jon Clough (Warren Pinnacle), L. Kellie Dixon (Mote Marine Laboratory), Mike Duever, Ernest Estevez (Mote Marine Laboratory), L. Donald Duke (Florida Gulf Coast University), Sue Fite (Lee County Environmental Laboratory), Tom Fraser, Lizanne Garcia (Southwest Florida Water Management District), Ludovic Donaghy (Florida Gulf Coast University), Whitney Gray (Florida Fish and Wildlife Conservation Commission), Boyd Gunsalus (South Florida Water Management District), Keith Hackett (Janicki Environmental, Incorporated), Emily Hall (Mote Marine Laboratory), Kelli Hammer-Levy (Pinellas County), Marion Hedgepeth (South Florida Water Management District), Clark Hull (Southwest Florida Water Management District), Roger Johansson, Kris Kaufman (Southwest Florida Water Management District), Keith Laakkonen (Fort Myers Beach), David Karlen (Environmental Protection Commission of Hillsborough County), Ernesto Lasso de la Vega (Lee County Hyacinth Control District), Jay Leverone (Sarasota Bay Estuary Program), Shawn Liston (Audubon Society), Graham Lewis (Northwest Florida Water Management District), Eric Milbrandt (Sanibel-Captiva Conservation Foundation), Ralph Montgomery (Atkins), Barron Moody (Florida Fish and Wildlife Conservation Commission), Ernst Peebles (University of South Florida), Ann Redmond (Brown and Caldwell), Jennifer Rehage (Florida International University), Ed Sherwood (Tampa Bay Estuary Program), Michelle Sims (CF Industries), Eric Stolen (Dynamac Corporation), Serge Thomas (Florida Gulf Coast University), Martin Wanielista (University of Central Florida), Bob Weisberg (University of South Florida), and Dorothea Zysko (The Ecology Group).

We greatly appreciate the assistance of Richard L. Turner, James Austin and David Karlen from the Florida Academy of Sciences and Kelly Calohan, Trisha Klosterman and Jeff Monson from Allen Press for their expertise and patience with the multitude of technical details involved with the publication of this issue.

The CHNEP is a partnership of citizens, elected officials, resource managers and commercial and recreational resource users working to improve the water quality and ecological integrity of the greater Charlotte Harbor Watershed. A cooperative decision-making process is used within the Program to address diverse resource management concerns in the 4,700 square mile study area. Many of these partners also financially support the Program, which allows the Program to support projects throughout the watershed, including this journal issue. Partners that have financially supported the CHNEP include: Cape Coral, Charlotte County, DeSoto County, Florida Department of Environmental Protection, Fort Myers, Fort Myers Beach, Hardee County, Lee County, Manatee County, North Port, Peace River/Manasota Regional Water Supply Authority, Punta Gorda, Polk County, Sanibel, Sarasota County, South Florida Water Management District, Southwest Florida Regional Planning Council, Southwest Florida Water Management District, U.S. Environmental Protection Agency, and Venice. The contributions and support of our partners are essential to the continued success of the CHNEP. Thank you.

2011 WATERSHED SUMMIT: THE STATE OF OUR WATERSHED AND ESTUARIES

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Abstract: *The Charlotte Harbor National Estuary Program (CHNEP) updated its Comprehensive Conservation and Management Plan (CCMP) in March 2008. Two months later, CHNEP updated its Environmental Indicators Technical Report. The U.S. Environmental Protection Agency requires each of the 28 NEPs in the country to produce a "State of the Bay" report or report card from their environmental indicators rooted in their CCMP. This paper describes the process CHNEP used to develop and complete its report guided by a Citizens Advisory Committee. Using survey techniques, CHNEP selected environmental indicators that best represented the state of the Charlotte Harbor estuaries and watershed. Survey respondents included members of each of the four committees (representing citizens, scientists, resource managers and elected/appointed officials) which make up CHNEP's "Management Conference". Over the next three years, the Management Conference issued contracts to fill data gaps, decided on the report title and format, reviewed deliverables from the contracts, compiled and analyzed available data, collected additional data through volunteer citizen efforts and considered representations of the data analyses. The triennial watershed summit featured much of this work and served as a kick-off for the final review of the draft report. The published report can be requested or downloaded at www.chnep.org.*

Key Words: Indicators, report card, status, trends, measures

National Estuary Programs (NEPs) were created under section 320 of the Clean Water Act. There are 28 in the nation and four in Florida. The Charlotte Harbor National Estuary Program (CHNEP) was designated in 1995 and is among the most recent designated. The Clean Water Act requires each NEP to convene a "Management Conference." CHNEP's includes four committees, including one each for citizens, scientists/technicians, resource managers and elected/appointed officials. The Policy Committee of elected officials and top agency heads is the decision-making body of CHNEP. County commissioners, city council members, water management governing board members, state officials and federal officials sit on the Policy Committee. They receive recommendations from each of the other three committees.

Other requirements of the NEP include adopting a *Comprehensive Conservation and Management Plan* (CCMP), adopting environmental indicators and publishing either a "state of the bay" report or report card. In 2008, CHNEP updated its CCMP (originally adopted in 2000) and its environmental indicators (originally adopted in 2003).

In preparation for the development of a "state of the bay" report or report card, Management Conference members were surveyed regarding the most important of its 54 indicators. Members from all four committees participated. Using survey results, the Management Conference selected 12 indicators that would be the core of the "state of the bay" report. The 12 indicators were organized according to nine basic questions which addressed status, trends and a measure of quality. Because four of the 12 indicators had no data or analysis available for Charlotte Harbor, CHNEP initiated several projects to fill the data and analysis gaps for fiscal year 2009. This work included:

- Fish and Wildlife Conservation Commission (FWC) Fisheries Independent Monitoring (FIM) for Lemon Bay,
- Pre-development vegetation maps for Charlotte and Manatee Counties,
- Water clarity tracking, and
- Pollutant loads.

The last of the projects were completed in late 2010. In the meantime, available data were compiled and additional data was collected by volunteers.

A VISION—Members of the Citizens Advisory Committee (CAC) provided the vision for the report. Throughout its development, the report purpose, structure and elements were responsive to CAC members' thoughtful input.

CAC members wanted the report to be understandable by their own mother and by their County commissioners. For example, one member suggested that most people would think nutrients are good and that the CHNEP should be careful about potential misunderstanding. The potential misunderstandings began with the title of the report. CHNEP is responsible for more estuaries than Charlotte Harbor, including Dona and Roberts Bays, Lemon Bay, Pine Island Sound, San Carlos Bay and Estero Bay. The study area also includes the watershed covering seven counties, not just Charlotte County. After deliberation, the CAC settled on "Charlotte Harbor Seven-County Watershed Report." They suggested featuring a map on the cover to provide more information regarding our geographic area of interest.

CAC members wanted the watershed report to be based on rigorous scientific methods. They knew detailed evaluation and acceptance by the scientific community was essential. The report needed to live up to a high scientific standard.

CAC members reviewed similar reports issued by other NEPs. The 2002 report by Puget Sound Water Quality Action Team provided general inspiration. The table of contents was organized with each section represented by a question. The text addressed status and trends with supporting images of charts, maps, photographs and additional interpretation. The document was relatively short at 16-pages. However, CAC members preferred a booklet format on high quality glossy paper to be more of a "keep-sake" than a newsprint publication.

CAC members also wanted to incorporate a "citizens' toolkit." This would provide opportunities for readers to learn what they personally could do to improve the environment.

THE QUESTIONS—The selected indicators were formulated into questions. These questions make up the table of contents:

1. Do our waters support diverse and healthy fish communities?
2. Are the fish and shellfish safe to eat?
3. Is fish and wildlife habitat increasing or decreasing (seagrasses, mangroves and freshwater wetlands)?
4. What is the condition of our shoreline?
5. What lands are managed for the environment?
6. Who restores nature?
7. Are our river flows natural (Caloosahatchee, Peace and Myakka Rivers)?
8. Is our water clean (bacteria, nutrients and water clarity)?
9. What is the source of water pollution (nitrogen, phosphorus and suspended solids)?

THE ANSWERS—CAC members wanted direct answers to the questions with supporting text and graphics. The data came primarily from existing sources but some required new data collection efforts. Most of the analysis and production of the graphics was accomplished with CHNEP in-house staff resources, predominately by the author. CAC members also wanted to provide the reader with information about how they personally could improve the environment related to the above questions. Recommendations are located as call-out boxes and titled "What can you do to help?"

The following sections describe the data sources, analysis and findings contained in the report.

Healthy and diverse fish communities—The Florida Fish and Wildlife Commission's Fish and Wildlife Research Institute (FWRI) collects random-sample Fisheries Independent Monitoring (FIM) data in the Charlotte Harbor study area. Data were available for most of the study area, except for Lemon Bay. In 2009 and 2010, FWRI collected Lemon Bay data to augment existing data sets. With the assistance of FWRI, CHNEP prepared abundance charts for both trawl (deployed in deeper waters) and bay seine (deployed in shallower water) samples by year and by bay segment. In addition, CHNEP developed Shannon-Weiner diversity index assessments for each sample. Average diversity charts (by both trawl and bay seines samples by year and by bay segment) were prepared. In addition, CHNEP used ArcGIS spatial analyst to present a map of fish diversity based on the trawl samples. The analyses showed no change within the period of record of 1996 through 2010, except for a loss of diversity in the shallow water samples (CHNEP, 2011).

Fish and shellfish safety—Fish consumption safety information was obtained from the Florida Department of Health (FDOH). Fish consumption advisories in the Charlotte Harbor study area are related solely to methyl mercury in fish tissue. Maps showing Florida Department of Environmental Protection (FDEP) impairments for mercury in the Charlotte Harbor area and a chart with the "do not eat" fish advisories were presented. Between 2005 and 2010, a growing number of water bodies were designated as impaired for mercury because of consumption advisories and mercury found in fish tissue. Many areas such as Charlotte Harbor have had increasing mercury concentrations because atmospheric deposition has increased (EPA, 2011). Another contributing factor to increasing impairments is increased sampling which may assist in confirming an impairment that previously existed.

Shellfish harvest area closure data were provided by the Department of Agriculture and Consumer Services (FDACS). CHNEP prepared a map showing approved shellfish areas and a chart created from the FDACS data, showing improvement in the number of open harvest days during the last few years. Lower than average rainfall contributed to more open harvest days (FDACS, 2011).

Further analysis and description of fish and shellfish safety can be found in the Watershed Report (CHNEP, 2011).

Fish and wildlife habitat—The CCMP has stated objectives for eight habitat types. Three of these habitat types were selected for inclusion in the Watershed Report, including seagrasses, mangroves and freshwater wetlands. Both the Southwest Florida Water Management District (SWFWMD) and South Florida Water Management District (SFWMD) have routinely mapped the areal extent of seagrasses, mangroves and freshwater wetlands using the Florida Land Use, Cover and Forms Classification System (FLUCCS). In 2006, CHNEP contracted for the development a 1950s era historic benthic coastal resources map for aerial photo interpretation. This information provided a context for modern seagrass changes and helped to establish targets. Mangrove and freshwater wetland historic changes were analyzed by comparing pre-development vegetation maps to water management district 2004–2005 FLUCCS maps. Both water management districts had developed estimates of pre-development habitat areas using soils. In 2007, HDR, Inc completed its contract with SWFWMD to prepare more precise maps for the Peace River Basin using the 1840s-era General Land Office surveys (GLOS). In 2008, CHNEP contracted HDR, Inc. to replicate this process for Charlotte County and Manatee County to augment the Peace River Basin maps. Between these resources, CHNEP documented remaining extents of seagrass (95%), mangrove (90%) and freshwater wetland (57%) of historic (1950s in the case of seagrass and 1840s in the case of mangroves and freshwater wetlands) estimates (CHNEP, 2011).

Shoreline condition—In 2007, CHNEP initiated shoreline conditions mapping. The strategy included aerial photo interpretation coupled with citizen volunteer shoreline surveys. The contracted data provided linear values of shoreline type (e.g. seawall, mangroves, exotic vegetation) and quality (e.g. hurricane damage). The volunteer assessments documented mangrove heights and trimming. CHNEP conducted the volunteer shoreline survey again in 2010. These CHNEP datasets were augmented with a 1995 statewide shoreline study. CHNEP has found 81.5% of its shoreline in a natural condition and non-native plants dominate 3% of shorelines. Manmade shorelines such as bulk heads and rip-rap revetments make up the remaining shoreline. In 2010, 52% of urban lots had mangroves, up 4% from 2007. Of lots with mangroves, 39% trimmed them, up 7% from 2007. Of trimmed mangroves, 38% were less than 6 feet in height, a violation of state standards, down 8% from 2007.

Managed lands—Florida has a marvelous history of pursuing environmental land acquisition through various programs such as Florida Forever and Save Our Rivers (Farr and Brock, 2006). In addition, all urban county partners within the CHNEP study area have local environmental land acquisition programs, including Polk, Sarasota, Charlotte and Lee Counties. Over 186,200 hectares (460,000 acres) are in conservation management. Of this area, over 85,000 hectares (210,000 acres) have been acquired since 1998. Almost 14% of our watershed land is in conservation. CHNEP worked with the Florida Natural Areas Inventory (FNAI) to ensure accuracy of the FNAI Managed Lands geographic information systems (GIS) files.

Restored lands—When the author began compiling restorations projects to present in a restored lands map, she assumed that the focus would be on restoration of managed conservation lands. As the work progressed, it became apparent that a significant part of the story was nutrient source reduction and water conservation on private land.

In 2007, the Southwest Florida Regional Planning Council (SWFRPC) adopted a resolution with provided guidance to local governments recommending local ordinances which would reduce fertilizer application on urban property. Since then the four coastal counties and seven of eight coastal cities within the CHNEP area adopted such ordinances. Combined, the adopted urban fertilizer ordinances apply to over 118,200 hectares (292,000 acres) (CHNEP, 2011).

SWFWMD offers an agricultural best management practice (BMP) cost-share reimbursement program for both water quantity and water quality, entitled Facilitating Agricultural Resource Management Systems (FARMS) Program. By 2011, over 46,500 hectares (115,000 acres) of FARMS projects have been constructed or approved (CHNEP, 2011). After publication of the Watershed Report, an additional 32,400 hectares (80,000 acres) of enrollment under the DACS Agricultural BMP manuals were identified.

By adding together managed conservation land, property subject to urban fertilizer ordinances, FARMS projects and the additional Agricultural BMP enrollment, nearly 405,000 hectares (1,000,000 acres), or over 36% of CHNEP's land area, is under some form of active land or water conservation management.

River flows—The three major rivers in the CHNEP study area are the Caloosahatchee, Peace and Myakka rivers. Caloosahatchee flows are the least natural of the three rivers. Originally, the Caloosahatchee did not have a direct connection to Lake Okeechobee. In the 1880s, 1940s and 1960s, successive channelization and construction of dams and locks have changed the timing of the flow. Continuous measurement of Caloosahatchee flows in the CHNEP study area began only after construction of the Franklin Lock (S-79) and associated works in 1965. However, in 1926, river flows were measured daily in LaBelle (George B. Hills Co., 1927). No day was below 300 cfs or above 2,800 cfs for this year of normal rainfall. Doering et. al. (2002) estimated the salinity needs of habitat forming species required freshwater flows at S-79 to be between 300 cfs and 2,800 cfs. After the Franklin Lock was constructed, a significant number of days with river flows below the minimum and above the maximum river flows have been measured each year. Since 1966, flows at S-79 have averaged less than 300 cfs for at least one month of the year in 40 of 44 years (all but 1983, 1987, 2003 and 2005). In 2001, an additional measure of salinity of 10 parts per thousand for a 30-day average or a single daily average of 20 parts per thousand at the gage in Fort Myers was adopted as part of the Minimum Flows and Levels for the Caloosahatchee. Such data were available beginning in 1992 and suggest that an additional 2 years (1995 and 1998) met this additional measure of salinity. Since 1966, 20% of all months exceeded unhealthy average flows of 2,800 cfs (CHNEP, 2011).

Since the mid-1970s, the upper Peace River has not met minimum flows for a significant number of days in most years. That trend has increased in the middle Peace River at Arcadia. The Myakka River receives too much flow from dry season irrigation. Diversion of Cow Pen Slough flows from the Myakka River basin balances some of this increased flow.

Clean water—A wide variety of water quality parameters are assessed by Florida Department of Environmental Protection (FDEP) to identify impaired water quality. In the CHNEP study area the most notable impairments are bacteria, nutrients, dissolved oxygen, metals and salts. For the Watershed Report, the CAC decided to focus on bacteria and nutrient impairments. The Management Conference, including the CAC, also elected to develop water clarity measures for which FDEP does not determine impairments. The two sources of data for bacteria and nutrients are the FDEP impairment assessments and CHNEP's water quality status and trends assessments. In general, bacteria and nutrient problems are numerous and growing worse.

Bacteria impairments included beach advisories, bacteria in shellfish and fecal coliform. In 2008, 18% of the CHNEP study area was designated as impaired for at least one bacteria parameter. By 2010, bacteria impairments rose to 22% of the CHNEP study area. However, more long term stations showed statistically significant improving trends (16%) than those that degraded (10%). The other 74% of stations had no trend for their period of record of at least 6 years and more typically 15 years. Most of the improving stations tended to be in estuaries, probably due to adjacent cities replacing septic tanks with central sewer.

Nutrient impairments were identified by chlorophyll a and Trophic State Index (TSI) values in excess of State standards. In 2008, 11% of the CHNEP study area was designated as impaired for at least one nutrient parameter. By 2010, nutrient impairments rose to 18% of the CHNEP study area. As with bacteria, more chlorophyll a and TSI nutrient stations showed long term improving trends (24%) than degrading trends (14%). The remaining 62% had no trend.

In Charlotte Harbor's estuaries, water clarity is a function of colored dissolved organic matter (CDOM), turbidity and chlorophyll (phytoplankton). Phytoplankton (microscopic algae) and CDOM are critical for estuarine food webs. Unfortunately, unnaturally high levels of phytoplankton can bloom from excess nutrients and unnaturally high levels of CDOM can be present from increased freshwater flow resulting from drainage projects. Since 2002, the Coastal Charlotte Harbor Monitoring Network (of various partner agencies) has provided stratified random sample data to assess ambient water quality conditions. Sarasota County began its monitoring program earlier for Lemon Bay and the Myakka. Under this program, water clarity is measured by light attenuation (Kd), a measure of how much light is lost through the water column. The Watershed Report explains the interaction of light with the water column, chlorophyll a, CDOM and turbidity. These components reduce the red and blue part of the light spectrum that seagrass need for survival.

Seagrass segments which have as much or more seagrasses than measured in the 1950s include Tidal Myakka River, Charlotte Harbor, Pine Island Sound and San Carlos Bay. These have been identified for protection. The CHNEP established restoration targets for the other segments which include Lemon Bay, Tidal Peace River, Matlacha Pass, Tidal Caloosahatchee and Estero Bay.

Through each of the ten segments periods of record, four segments have shown improvement in Kd: Lemon Bay, Tidal Myakka, Tidal Peace and Charlotte Harbor. Three of these segments went from a degraded condition to an improving condition (against the same measures for each year). All four segments are within the jurisdiction of the Southwest Florida Water Management District which placed a priority on improving water quality through partnerships with local governments and others. The approach appears to be working well.

The only segment that appears to be getting worse is Estero Bay. The Estero Bay watershed has had much development during the recent decades. Data from the Charlotte Harbor Estuaries Volunteer Water Quality Monitoring Network suggests that turbidity is the root of Estero Bay's water clarity problems (Ott et al., 2006). In addition to construction and other activities adding more sediment to waterways, boating in this shallow, lagoonal bay stirs up the sediments, causing continual problems with turbidity.

Water pollution (pollutant loads)—This factor complements a section presented earlier which focused on concentrations of pollutants in natural systems. The Management Conference chose pollutant loads as an additional environmental indicator to feature. More total nitrogen and phosphorus is delivered to water bodies during times of high rain and creek flow. The total amount is called a "load." For estuaries, the load can be as much of a problem as the concentration. Two kinds of substances were evaluated: excess nutrients and suspended sediments.

Prior to this report, the latest pollutant load information for the CHNEP study area was created in 1999 using data for the period of 1975 through 1990. CHNEP initiated a project in 2008 to update the pollutant load information. This occurrence had fortunate results. In 2010, CHNEP embarked on developing recommended estuarine numeric nutrient criteria for FDEP and EPA to consider as adopted water quality standards. The pollutant load data were critical for the development of these recommended criteria. The updated pollutant load data period of record was from 1995 through 2007.

Pollution loads varied each year because the amount of water flowing to creeks, rivers and estuaries varies with rainfall. The amount of water flowing to a water body is called a "hydrologic load." From 1995 through 2007, the greatest hydrologic load occurred in 1995 (more than 12 million cubic meters) and the lowest was in 2007 (under 2 million cubic meters). The average hydrologic load was more than seven million cubic meters.

Pollutants are carried with water flow. Total nitrogen (TN) loads varied from more than 2 million kilograms (2,000 tons) in 2007 to more than 17 million kilograms (19,000 tons) in 2005, with an annual average of more than 9 million kilograms (10,000 tons). Total phosphorus (TP) varied from less than 500 thousand kilograms (500 tons) in 2007 to more than 3 million kilograms (4,000 tons) in 2005, with an average of more than 2 million kilograms (2,000 tons). Total suspended sediments (TSS) varied from more than 9 million kilograms (10,000 tons) in 2007 to more than 125 million kilograms (135,000 tons) in 2005, with an average of more than 56 million (62,000 tons) (Janicki Environmental, Inc., 2010). The reason 2005 had such high pollution loading relates to a two year period of intense rain and numerous hurricanes, as well as releases of impounded water in advance of each impending hurricane to protect structures.

In 1999, the CHNEP assessed pollutant loads from 1975 through 1990. We compared this 15-year period ending in 1990 to the 17-year period ending in 2007. It appears that there may have been a reduction in pollution loading of total nitrogen by 48%, of total phosphorus by 36% and of total suspended sediments by 47%. There are alternative explanations for the decrease in pollution loading in the face of urban growth. Fewer urban development and mining regulations, as well as agricultural best management practices, were in place. Developments permitted before 1975 were allowed to build under the old standards. As better standards were put into place, pollutant loading may have decreased in general from the 1975–1990 period to the 1995–2007 period. Another explanation is that pollution loading was overestimated for the earlier period to account for greater uncertainties due to less available water quality data (CHNEP, 2011).

Emerging issues—CHNEP has sponsored research for several emerging issues, including pollution from pharmaceuticals and personal care products (PPCP) and climate change. In 2006, 2008 and 2009, Mote Marine Laboratory conducted separate studies investigating the presence of pharmaceuticals in the CHNEP study area. In 2006, ecoestrogens were measured by Mote Marine Laboratory in water samples from Charlotte Harbor, Peace River, Myakka River and the Caloosahatchee. Ecoestrogens are environmental chemicals capable of altering estrogen-regulated processes in aquatic organisms (Gelsleichter et al., 2009.) Of the three rivers, detection of ecoestrogens occurred most frequently in the Caloosahatchee. Ecoestrogens were found near other developed areas also. In 2008, ecoestrogens were measured in the Caloosahatchee and in the tissue of a fish species [hogchokers (*Trinectes maculatus*)] collected in the Caloosahatchee. Ecoestrogens do not appear to pose significant health threats to wildlife populations residing in the Caloosahatchee, based on low ecoestrogen concentrations and the apparent lack of ecoestrogen effects in hogchokers found in the Caloosahatchee. In 2009, Mote Marine Laboratory tested for the presence of steroid, impotence treatment, lipid-lowering drug and six antidepressant chemicals in water samples, wastewater samples and the blood plasma of bull sharks found in the Caloosahatchee and, as a control, the Myakka River. Presence of these chemicals was often at undetectable or near detectable levels.

Beginning in 2008, CHNEP and the Southwest Florida Regional Planning Council (SWFRPC) conducted several studies under EPA's Climate Ready Estuaries (CRE) program. The studies included a comprehensive climate change vulnerability assessment (Beever et al., 2009a), a publicly accessible vulnerability assessment document (CHNEP 2010) and a climate change adaptation plan for a small city (Beever et al., 2009b). Reviewing local data, CHNEP documented that average air temperatures have increased, the number of days in the year over 90° F have increased, rainfall delivered in the rainy season has increased, rainfall delivered in the dry season has decreased and sea level has risen about 8 inches, during the past 100 years. Since 1965, sea level has risen at the Fort Myers gage by one inch per decade. In addition, salt marshes and seagrass beds have migrated landward by approximately 100 yards since 1950.

Summary and Conclusions—CHNEP's 2011 *Charlotte Harbor Seven County Watershed Report* provides concise and rich analyses of the factors that the Management Conference (the four committees that comprise the CHNEP) found important. The Citizens Advisory Committee (CAC) guided extensive staff analysis and preparation of the document. The CAC's thoughtful guidance resulted in a document that is meaningful to a majority of people. General responses from the public confirm that the casual reader can easily leaf through the document and can drill down for more detailed information where they choose. In the end, the Watershed Report is a physical manifestation of the science-based and consensus-driven partnership of citizens, scientists, resource managers, top agency heads and elected officials that make up CHNEP.

This paper describes data sources and analysis processes for nine components of the Watershed Report. These sources and methods may be useful to agencies that write reports of this kind intended for both the public and member/oversight agencies. This paper also served as the introductory presentation for the 2011 Watershed Summit: The State of our Watershed and Estuaries. All presentations which followed fit into and later augmented the Watershed Report.

Concurrently with the publication of the Watershed Report, CHNEP launched its online Water Atlas (www.chnep.wateratlas.usf.edu). The Water Atlas provides citizens, scientists and decision-makers with constantly updated data and analysis of water-related issues.

ACKNOWLEDGMENTS—This work is the result of hundreds of scientists' and citizens' efforts. The hard work and dedication of people monitoring, analyzing and communicating natural systems data of the Charlotte Harbor study area is extraordinary. This wealth of information helps to guide good decision-making to protect our valuable natural resources.

Ernesto Lasso de la Vega chaired the CAC Environmental Progress Subcommittee. Deb Highsmith, Pete Quasius, Kayton Nedza, Warren Bush, Lou Kovach and Forest Reynolds served on the subcommittee and provided valuable suggestions throughout the process.

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RESULTS OF THE FLORIDA DEPARTMENT OF ENVIRONMENTAL PROTECTION, CHARLOTTE HARBOR AQUATIC PRESERVES' SEAGRASS MONITORING PROGRAM FROM 1999–2009

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Abstract: *Seagrass monitoring is conducted annually throughout the Charlotte Harbor estuarine complex by the Florida Department of Environmental Protection, Charlotte Harbor Aquatic Preserves office. This program provides baseline, status and trends data of seagrass parameters for assessing estuarine health. Results from 1999–2009 show the three most common seagrass species throughout the Charlotte Harbor area are Halodule wrightii, Thalassia testudinum and Syringodium filiforme. Seagrass appears relatively stable across the study area with minor declines associated with considerable wet years and hurricane events. Since 2004, the total abundance of all seagrass species, as well as the density of H. wrightii, increased significantly. In 2009, H. wrightii and S. filiforme had the highest mean shoot count in Gasparilla Sound and Lemon Bay. The tidal Peace and Myakka river systems have the lowest occurrence, abundance and densities of seagrass. The maximum depth of seagrass growth has increased since 1999 with San Carlos Bay having the deepest growing seagrass. San Carlos Bay experienced declines in seagrass abundance during high flow events from the Caloosahatchee River. Continued monitoring will be important to track changes, understand potential causes of trends, and to aid in estuarine management so the aquatic preserves can be maintained in an essentially natural condition.*

Key Words: Seagrass, Monitoring, Charlotte Harbor Aquatic Preserves, *Halodule wrightii*, *Thalassia testudinum*, *Syringodium filiforme*

The Florida Department of Environmental Protection (FDEP) Charlotte Harbor Aquatic Preserves (CHAP) office, through the Office of Coastal and Aquatic Managed Areas (CAMA), has been monitoring seagrasses since 1999 at fifty fixed transects throughout the Lemon Bay, Gasparilla Sound-Charlotte Harbor, Cape Haze, Pine Island Sound and Matlacha Pass Aquatic Preserves (Fig. 1). Aquatic preserves are exceptional sovereign submerged lands set aside by the Florida Legislature to be preserved in an essentially natural condition for future generations to enjoy (Chapter 18-20.001(2) F.A.C.). To properly manage these aquatic preserves, CHAP staff monitor water quality and seagrass conditions to obtain baseline conditions, assess status and trends, and identify areas of concern. Long term quantitative monitoring of seagrass beds at repeatable intervals along fixed transects provides valuable information to resource managers such as seagrass species distribution, density, abundance, and the deep edge of the meadows. The data from this program has been provided to other agencies for statewide seagrass

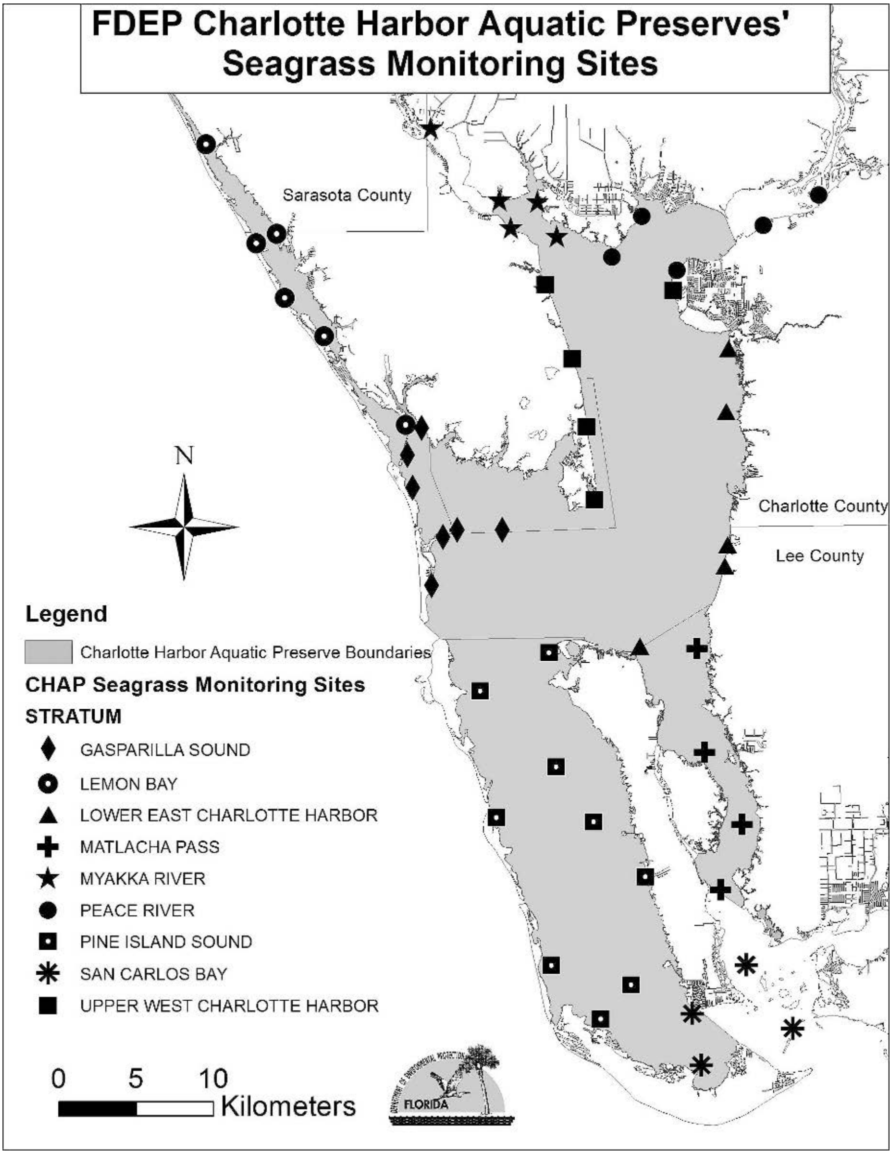

FIG. 1. Florida Department of Environmental Protection (FDEP), Charlotte Harbor Aquatic Preserve's (CHAP) seagrass monitoring sites by region.

reports, establishing water clarity targets based on the seagrass deep edge depth data, and has provided information for regulatory review of activities proposed in the aquatic preserves.

Seagrass meadows are considered to be one of the most productive ecosystems (Larkum et al., 2010), playing an integral role in the estuarine environment by improving water quality, stabilizing sediment, removing

TABLE 1. Braun Blanquet abundance categories for seagrass coverage in a square meter and corresponding code (as seen in graphs).

Code	Braun-Blanquet Abundance %
0	no cover
1	<5
2	5–25
3	26–50
4	51–75
5	76–100

suspended materials from the water column, aiding in nutrient cycling and providing shelter and food for many juvenile estuarine and marine species (Hemminga et al., 2000). Several seagrass characteristics can help determine the ecosystem health and quality of an estuary, including the presence or absence of seagrass, abundance, density, species type, epiphytic growth (plant and animal), blade lengths and the water depth at which they are found. Each year these key parameters are monitored along fixed transects that are representative of a defined waterbody and watershed within the CHAP.

The CHAP will continue to monitor this important submerged resource to track yearly changes in various seagrass characteristics and create summary reports. The FDEP CHAP Seagrass Report (Brown, 2011) is a graphical summary of seagrass species, occurrence, abundance and density between regions and by year from 1999–2009. This paper includes a statistical analysis of the FDEP CHAP's 2011 report, highlighting significant trends and discussion of the results.

MATERIALS AND METHODS—*Field methods*—The CHAP seagrass monitoring program occurs annually throughout late summer and fall (August to November) in order to capture seagrass at its prime abundance. The program includes fifty sites covering an extensive geographic area; from the southern end of Sarasota County in Lemon Bay throughout Charlotte and Lee Counties including five aquatic preserves, San Carlos Bay and the Peace and Myakka River systems. The transect locations (FIG. 1) are influenced by various watersheds and have been grouped accordingly. Each transect starts perpendicular to shore at the beginning of the seagrass bed and ends waterward at the deep edge (the last shoot) of the seagrass bed. Every year, the transects are monitored at fixed repeatable intervals or stations, which are physically marked with PVC stakes and geo-referenced using a sub-meter accuracy Trimble© GPS unit. Transects range from 10m to over 600m long and stations are typically set every 50m, except for transects <50m in length, where stations are set every 10m. At each station, ten parameters are measured within a one square meter quadrat (divided into 100 cm^2 squares) including depth of the water, sediment type, total abundance (using Braun-Blanquet (BB) abundance ranges, Fourqurean et al., 1999; TABLE 1), species type and species abundance, shoot density per species, five blade lengths per species, epiphyte type and epiphyte density, and relative location of the station (i.e. beginning, middle or end of bed). Shoot density is measured for each species at each station using a pre-determined pattern relative to the BB abundance. A BB abundance of 5 would require the shoots counted in only 5 squares (every other square along a diagonal pattern within the quadrat), while every shoot would be counted in a quadrat with a BB of 1. The shoot count measurements are then calculated appropriately in the database as the density of the entire quadrat (i.e. the shoot count number is multiplied by a factor, relative to the BB abundance, to represent a total of 100 squares). Epiphyte densities are recorded as either clean (1), light (2), moderate (3) or heavy (4). All parameters have been measured since 1999, except for total abundance which began in 2004 and density/shoot counts which began in 2005. Detailed monitoring procedures are outlined in the FDEP CHAP protocols (Stearns, 2007).

Analyses methods—The field monitoring data is entered into an Access© database each year following the field survey. Water depths (cm) measured in the field are converted to mean water depths in Excel© using the beginning and ending tide stage for the transect, the closest NOAA benchmark tidal datum and the time recorded at each station. Status and trends by year, hydrological region and species were developed using SPSS© statistical software.

For analyses of BB abundance, the number code associated with the abundance category was used. A BB of 0, or no cover, was included in the analysis of total abundance but not for species abundance, as this relates only to the abundance of a particular species by region and year. The 50 transects were grouped and summarized according to nine hydrological regions: Peace River, Myakka River, Upper West Charlotte Harbor (UWCH), Lower East Charlotte Harbor (LECH), Lemon Bay, Gasparilla Sound, Pine Island Sound, Matlacha Pass and San Carlos Bay (Fig. 1). Individual transect analyses were not examined, as transects were grouped by hydrological region to search for trends. Only the repeatable stations (monitored nine out of the eleven years) along each transect were analyzed for consistency. Beginning and end of bed data were not used for the determination of abundance and density, as the seagrass beds typically vary from year to year in extent. However, deep edge data were used to determine maximum seagrass growth depths.

Assumptions of normality were tested using: Johnson's SU transformation for skew; Anscombe & Glynn's transformation for kurtosis; and Jarque & Bera LM test. Outliers were identified using Mahalanobis D^2. The assumption of homoscedasticity was tested using Levene's test of homogeneity of variance. Assumptions of linearity were examined using plots of observed versus predicted values and residuals versus predicted values. In addition, assumptions of independence were assessed using autocorrelation function (ACF) and the Durbin-Watson d test. Where the assumptions were violated, data were either transformed or robust nonparametric regressions (Theil-Kendall regression) were used for detection of trends. However, if assumptions were met or the data was transformed, linear regression was used. These statistical methods for trend analyses are those employed by Leary, 2011.

Analyses of flow versus abundance for San Carlos Bay were conducted using the mean annual discharge from the S-79 Franklin Locks and Dam against the average annual abundances of all species combined, *H. wrightii*, *S. filiforme*, and *T. testudinum*. Average abundances were regressed against flow, and included both Pearson's correlation and ANOVA tables. Likewise, paired t-tests were run on flow versus average abundances. In addition, ANOVA with the Brown-Forsythe F test (a modified ANOVA which is robust against heteroscedastic data) and Dunnett's T^3 *post hoc* (a robust *post hoc* test when data are heteroscedastic) comparisons were run for abundances against the years (used as a proxy for mean flow since there was only one value per year) to determine which flow-years were significantly different. Pearson correlation were run correlating Matlacha Pass average annual seagrass (all species combined) and mean annual rainfall in Ft. Myers. A significance value of $p<0.05$ was used to determine if the trends were significant or not.

Results—As a whole, the seagrass parameters measured were stable throughout the region from 1999–2009. There were some decreases in abundance and density in 2004 and 2005, the two years characterized by higher than average rainfall and hurricanes. However, since that period, seagrasses have rebounded with some of the highest recorded abundances and densities in the CHAP monitoring program, and were found at the deepest depths in 2009. Variations in species abundance, occurrence and densities by year and hydrological region were observed over the study period within the

TABLE 2. Percentage of occurrence of seagrass species (including no cover) by year within the CHAP. (H. Species refers to the genus *Halophila.*).

Year	No Cover	H. wrightii	T. testudinum	S. filiforme	R. maritima	H. Sp.
1999	10	46.5	31.5	9.2	1.9	0.8
2000	11.9	47.8	30.4	9.3	0.7	0
2001	16.2	40.5	32	9.5	1.4	0.4
2002	15.5	44.5	31.7	8.3	0	0
2003	19.9	41.3	29.9	8.9	0	0
2004	19.9	41.6	30.1	8.4	0	0
2005	24.3	41	26.5	8.2	0	0
2006	20.3	44.5	27.2	7.9	0	0
2007	15.8	47.4	26.8	9.3	0	0.7
2008	16	47	25.4	8.7	2.8	0
2009	12.5	51.2	27.5	8.8	0	0
Mean	16.6	44.8	29.0	8.8	0.6	0.2

estuary primarily due to the influence of the hydrologic regions' watershed and annual variations in climatic conditions.

Seagrass species occurrence—The three most frequently occurring seagrass species throughout the Charlotte Harbor area are *Halodule wrightii*, *Thalassia testudinum*, and *Syringodium filiforme* occurring approximately 45%, 29% and 9% of the time respectively (TABLE 2). Seagrass absence (i.e. no cover) was observed approximately 17% of the time along consistently sampled transect stations. *Ruppia maritima, Halophila engelmannii* and *Halophila decipiens* are also found in the study area but with no major occurrence or abundance, and were not used for these analyses or for finding the deep edge of bed. *H. wrightii* occurs in all estuary regions, while *T. testudinum* and *S. filiforme* are found in most regions with the exception of the Peace and Myakka Rivers. *S. filiforme* is absent in Matlacha Pass as well. For all regions, Leary (2011) found *H. wrightii* frequency, based on density not occurrence, was significantly increasing over the years ($p=0.030$).

Total abundance—Since the initiation of monitoring total abundance of all seagrass species combined (2004), total abundance has increased significantly ($p<0.001$) from an average BB of 1 to 2 (FIG. 2). Gasparilla Sound has the most abundant seagrass (BB of 2) over the 2004–2009 time period, while Myakka and Peace Rivers have the lowest average total abundances (BB of 0.5 and 0.4 respectively). Six of the nine regions showed significant increasing coverage trends from 2004–2009 (Peace River $p=0.004$, Myakka River $p=0.025$, UWCH $p<0.001$, LECH $p<0.001$, Gasparilla Sound $p=0.002$ and San Carlos Bay $p<0.001$; Leary, 2011). The three remaining stratum show increasing, but non- significant trends in total BB abundance.

Species abundance—Throughout the study area from 1999–2009, *H. wrightii* was the only species to have a significant increasing trend in

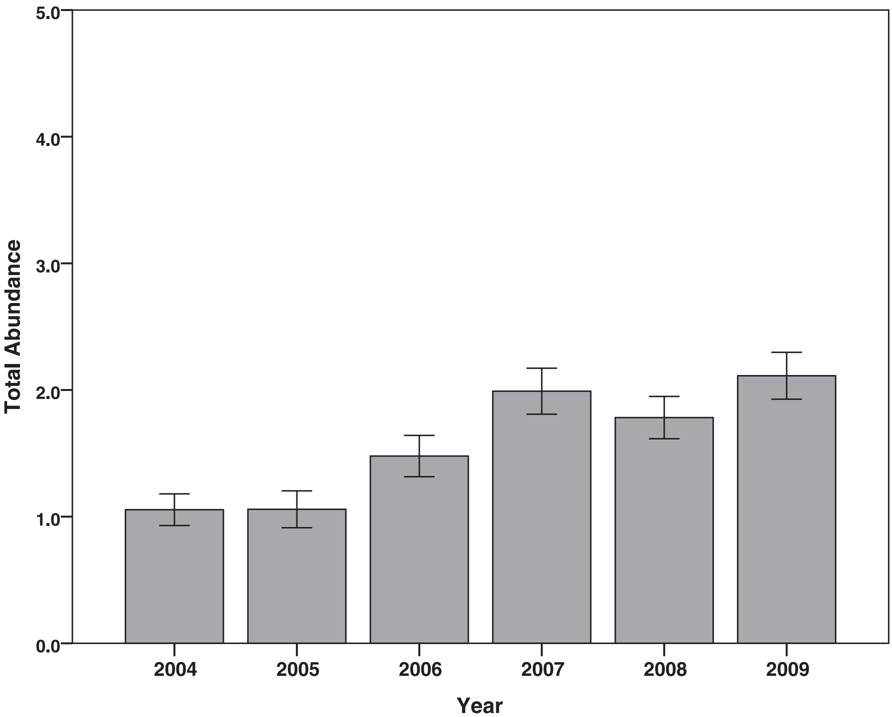

FIG. 2. Mean annual Braun-Blanquet (BB) total quadrat abundance (+/− SE) for the three major seagrass species, including no cover, over the period of record (2004–2009) for the CHAP study area.

abundance ($p<0.001$). By region, *H. wrightii* abundance increased significantly in San Carlos Bay ($p<0.001$), Myakka River ($p=0.019$), UWCH ($p< 0.001$), LECH ($p<0.001$) and Matlacha Pass ($p<0.001$). Lemon Bay and San Carlos Bay had significant declines in *T. testudinum* abundances ($p=0.025$ and $p<0.001$ respectively), and *S. filiforme* decreased significantly in Lemon Bay ($p=0.025$). With all years combined, Gasparilla Sound had the highest average abundances of *H. wrightii* (BB of 2) and *S. filiforme* (BB of 3), while the LECH region had the highest average abundance of *T. testudinum.* The Peace and Myakka Rivers had the lowest average *H. wrightii* abundance (BB of 1); while the lowest average abundances of *T. testudinum* and *S. filiforme* (BB of 2) occur in San Carlos Bay.

Region wide, all three seagrass species displayed a decline in species abundance during the years 2002–2005. Species abundance then increased with the abundances from 2007–2009 similar to 1999–2001 coverages. San Carlos Bay and Matlacha Pass were the only two regions with decreases in seagrass coverage in 2005 for all species present (FIG. 3). San Carlos Bay mean annual seagrass abundance (all species combined) significantly declined in 2005 ($p<0.001$), which was significantly different from all other years (p ranges from

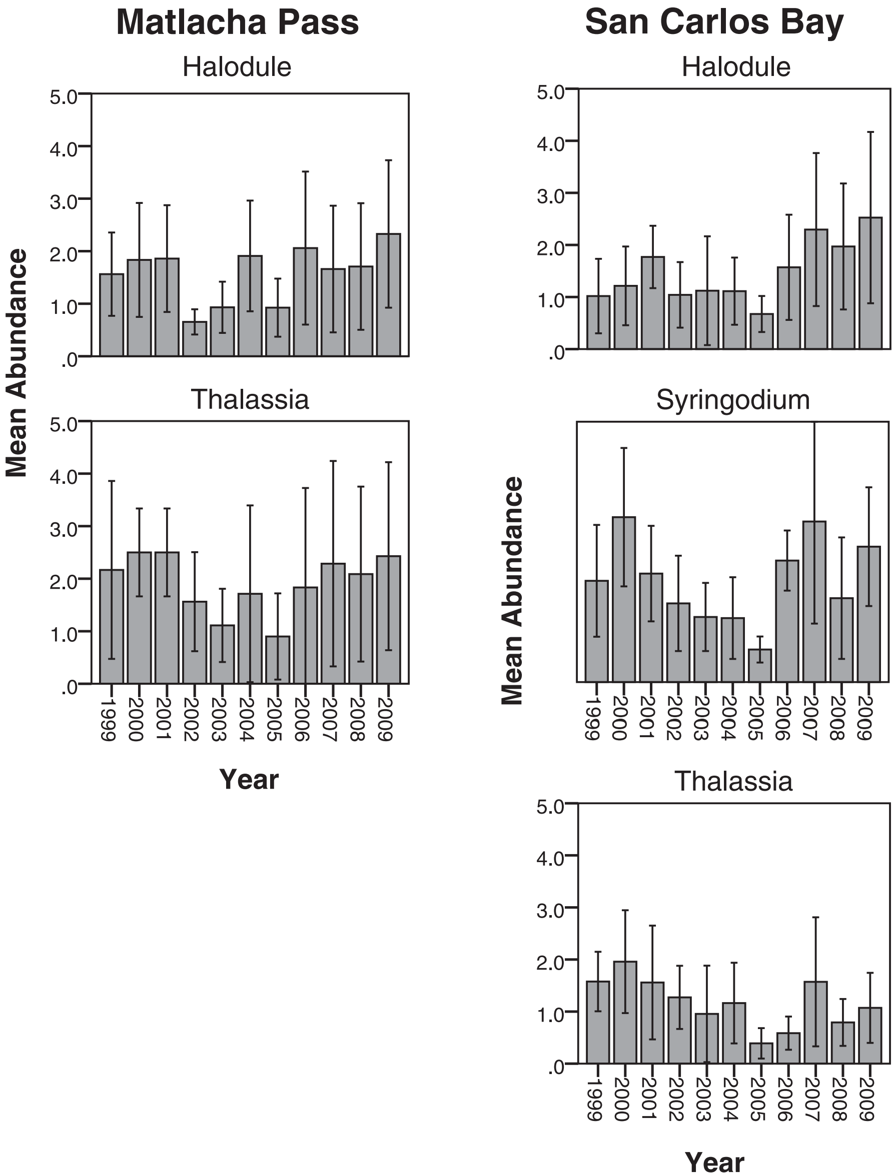

FIG. 3. Mean BB abundance (+/− SD) by species and year for Matlacha Pass and San Carlos Bay over the study period (1999–2009).

0.020 for 2003 and 2006 to <0.001 for all other years) and was negatively influenced by the Caloosahatchee River flow (p=0.001). Matlacha Pass seagrass abundance was negatively correlated to annual rainfall (−0.774, p=0.005), not flow.

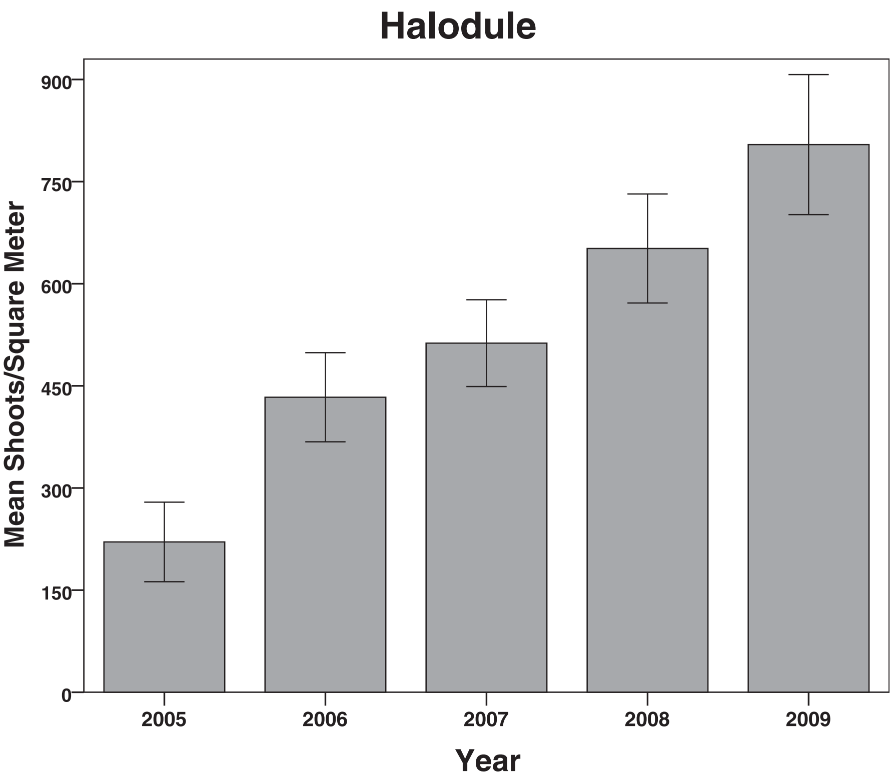

FIG. 4. *Halodule wrightii* mean annual shoots per square meter (+/− SE) over the CHAP study area from 2005–2009. (Note: shoot counts were not conducted before 2005).

Shoot density—Recording shoot counts, or density, per species within each quadrat began in 2005. The highest densities of *H. wrightii, T. testudinum* and *S. filiforme* occurred in 2009. Across the study area, mean *H. wrightii* shoot counts significantly increased from 2005 to 2009 ($p<0.001$); from 221 to 841 shoots/m^2 (FIG. 4). Myakka and Peace Rivers, Gasparilla Sound and San Carlos Bay all had significantly increasing trends in densities for *H. wrightii* ($p=0.007$, $p=0.013$, $p=0.002$ and $p<0.001$ respectively) and for all seagrass species combined ($p=0.017$, $p=0.014$, $p=0.002$ and $p<0.001$ respectively). UWCH and LECH had significant increasing trends for all species combined ($p=0.023$ and $p=0.004$ respectively), while Pine Island Sound had an increasing trend in *H. wrightii* density ($p=0.028$).

Over the five years that density was measured, Lemon Bay and Gasparilla Sound regions had the highest mean *H. wrightii* densities at 907 and 856 shoots /m^2 respectively. The average density of *S. filiforme* was 562 shoots/m^2, throughout all the regions, with the highest occurring in 2009 (631 shoots/m^2). Gasparilla Sound had the highest average density of *S. filiforme* (744 shoots/ m^2) with all years combined. *T. testudinum* densities were highest in 2009

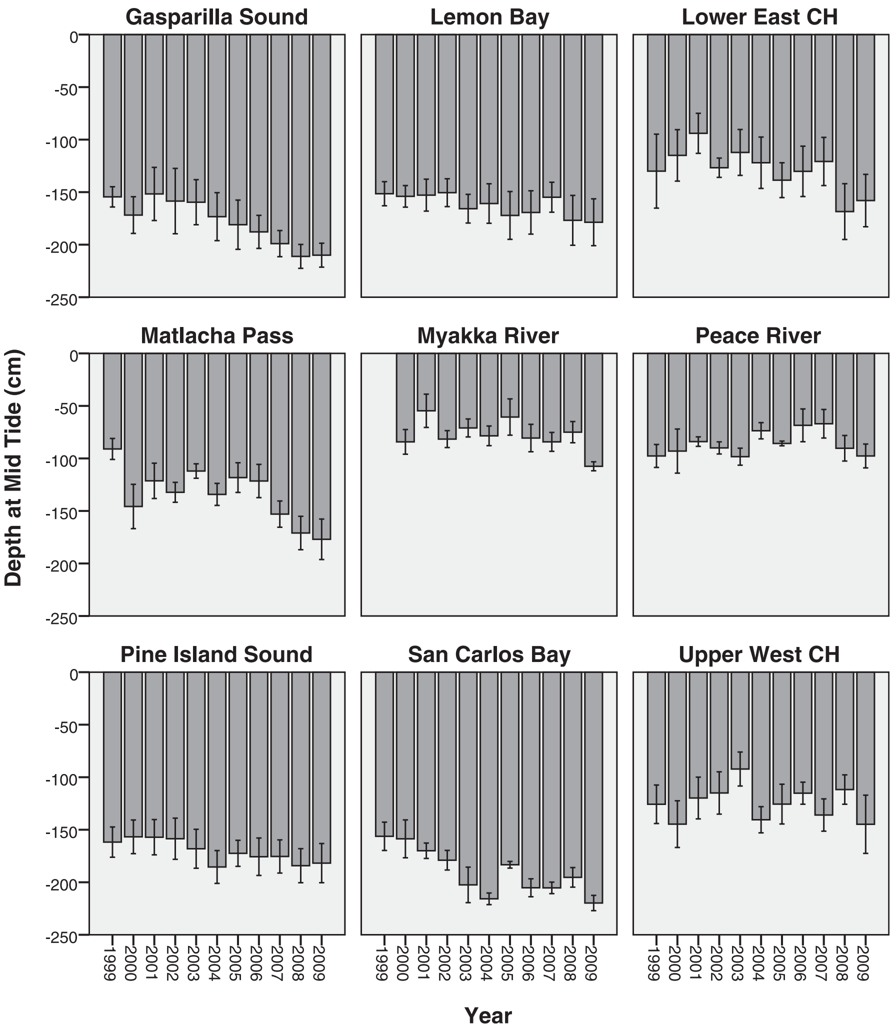

FIG. 5. Deep edge seagrass mean depth (+/− SE) by region over the study period.

throughout all the regions, with LECH having the greatest densities (281 shoots/m^2) and San Carlos Bay having the least (43 shoots/m^2).

Deep edge—On average, across the study area, the maximum depth of seagrass growth (deep edge) increased significantly from 1999 (−1.42 m) to 2009 (−1.69 m; $p=0.006$). San Carlos Bay has, on average, the deepest growing seagrass growing (−1.91 m), followed by Gasparilla Sound (−1.77 m), Pine Island Sound (−1.71 m) and Lemon Bay (−1.63 m; FIG. 5). From 1999 to 2009, deep edge measurements trended significantly deeper only in the San Carlos Bay ($p=0.001$), Matlacha Pass ($p=0.001$), Gasparilla Sound ($p<0.001$), and LECH ($p=0.017$) regions.

Epiphytes—From 1999–2009, epiphyte densities have increased significantly ($p<0.001$) over the study area. Regionally across all seagrass species, epiphyte density significantly increased in Pine Island Sound ($p=0.014$), Matlacha Pass ($p<0.001$) and San Carlos Bay ($p<0.001$), while there were significant decreasing trends in UWCH ($p=0.040$) and Gasparilla Sound ($p=0.017$). *T. testudinum* epiphyte densities decreased significantly in Lemon Bay ($p<0.001$), UWCH ($p<0.001$), LECH ($p=0.005$), and Gasparilla Sound ($p<0.001$) while epiphyte densities for *H. wrightii* increased in LECH ($p=0.031$), Pine Island Sound ($p<0.001$), Matlacha Pass ($p<0.001$) and San Carlos Bay ($p<0.001$) over the study period. The majority of epiphyte densities were characterized as light or moderate, with *T. testudinum* exhibiting more moderate to heavy loading than the other two species. *H. wrightii's* heaviest loading occurred in San Carlos Bay, while the heaviest loading on *T. testudinum* and *S. filiforme* occurred in UWCH and Lemon Bay, respectively.

Discussion—The overall trends in seagrass abundance and density over the study area and period correspond well with one another. The trends are influenced by several interacting variables, but a primary driver for the overall trends appears to be related to the amount of freshwater the watershed and estuary received. Freshwater influence from seasonal rainfall as well as natural and anthropogenic flow, can lead to a decline in salinity and water quality (i.e. increases in nutrients, chlorophyll a, color, turbidity, etc.) in the receiving estuary (McPherson and Miller, 1987). Color, chlorophyll and other suspended matter, such as turbidity, are primary factors causing reduced water clarity and light penetration to the seagrass beds (McPherson and Miller, 1987; Corbett et al., 2005; Greenawalt-Boswell et al., 2006). Water clarity increases with increased salinity levels, therefore reduced water clarity as a result of freshwater flow can cause adverse conditions for optimum seagrass growth from decreased light penetration (Johansson, 2000; Tomasko et al., 2001; Doering et al., 2002; Corbett et al., 2005; Greenawalt-Boswell et al., 2006). Losses of seagrass coverage in Tampa Bay, Sarasota Bay and Upper Charlotte Harbor have been linked to reduced water clarity from increased freshwater inflow and stormwater runoff (Tomasko et al., 2005). Seagrass in the CHAP region is highly influenced by freshwater flows from the Caloosahatchee, Peace and Myakka Rivers.

Species occurrence is dependent on salinity, and areas that are subject to freshwater flow and high variations in salinity, such as the Peace and Myakka Rivers, cannot support stable seagrass populations (Greenawalt-Boswell et al., 2006). The Peace and Myakka Rivers do in fact have the lowest occurrence, abundance and densities of seagrass, as well some of the lowest salinities and water clarity in the Charlotte Harbor complex (Duffey et al., 2007). Seagrass beds near the Caloosahatchee River are also influenced by changes in salinity, and the quantity and timing of freshwater flows are especially important in this region as it can be controlled through the gate and lock system upstream (McPherson and Miller, 1987; Doering et al., 2002; Corbett et al., 2005).

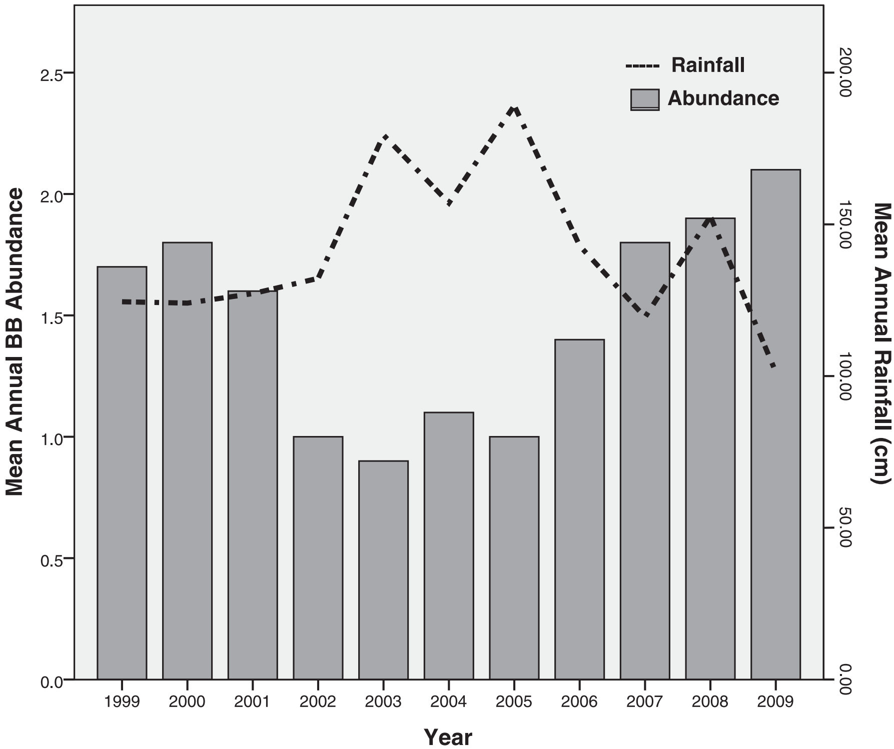

FIG. 6. Mean *H. wrightii* BB abundance for the CHAP study area over the study period in relation to mean annual rainfall in Ft. Myers, FL.

According to Florida State University's Florida State Climate Center (2010), Ft. Myers' lowest average annual rainfalls over the study period occurred in 1999, 2000, 2007 and 2009, with 2009 having the lowest average annual rainfall (101.3 cm). In response to the low rainfall, it appears that seagrasses such as *H. wrightii,* flourished, having the highest abundances and densities during the dry years (FIG. 6). Corbett (2006) also documented increases in aerial seagrass coverage during the drought conditions of 1999 to 2002. The years 2003, 2004, 2005 and 2008 mark some of the wettest years of the study period, with the numerous tropical storms and hurricanes that inundated southwest FL in 2004 (Hurricane Charley) and 2005 (Hurricane Wilma). In 2005, the region experienced 189.1 cm of rain (FSU, 2010), the highest recorded during the study period which likely contributed to a decline in species' abundance (FIG. 6) and the lowest densities in the study period. In Matlacha Pass, annual rainfall was negatively correlated to species abundance. Dawes and Avery (2010) found that *H. wrightii* coverage in Hillsborough Bay (Tampa, FL) decreased as well during the wet hurricane years of 2003–2005.

The frequency of freshwater releases through the gate and lock system on the Caloosahatchee River also increased in response to high rainfall conditions.

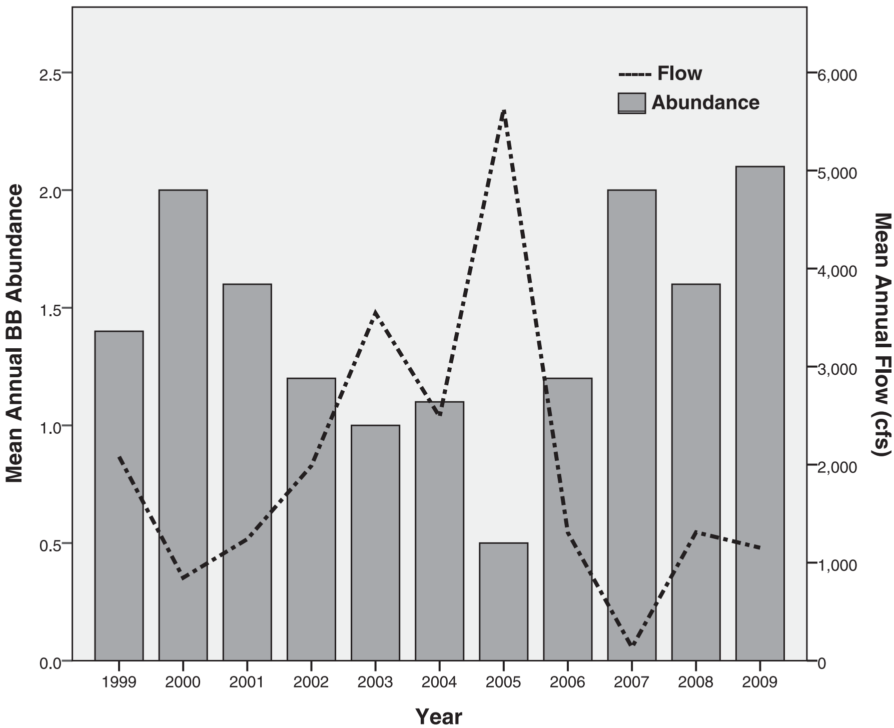

FIG. 7. Mean BB abundance of seagrasses in San Carlos Bay by year in relation to mean annual flow at the S-79 lock on the Caloosahatchee River.

Freshwater discharges from this highly managed system enter into southern Matlacha Pass and San Carlos Bay. During the study period, measured flows at the western most lock (S-79) were the highest in July 2005 (reaching 22,156 cfs), according to the SFWMD (2010). Hurricane Wilma also passed through the study area Oct. 24, 2005, generating high rainfall and flow events. As a result, southern Matlacha Pass and San Carlos Bay experienced a low fluctuating salinity environment which has been shown to be a significant factor in causing a decline in seagrass abundance (Corbett et al., 2005; Greenawalt-Boswell et al., 2006). The high flows through the S-79 lock negatively impacted seagrass abundance in San Carlos Bay (FIG. 7), as this region experienced a significant decline in abundance in 2005. Compared to other years, 2005 seagrass abundance in San Carlos Bay was found to be significantly different. Corbett et al. (2005) also related high discharge years to lower seagrass coverage due to low salinity and/or lower light availability. One particular transect in San Carlos Bay, closest to the mouth of the Caloosahatchee River, lost *T. testudinum* (a species that prefers high salinity waters) at all stations after the high flow event and the low salinity environment caused by Hurricane Wilma. Flows averaged 11,450 cfs within the 16 days after Wilma and salinity at the nearby CHAP continuous water quality station averaged 7.2 ppt ten days prior to monitoring the seagrass transect on Nov. 9, 2005 and reached as low as 3.5 ppt on Nov.4th. Doering and Chamberlain (2000) noted that *T. testudinum* is negatively impacted between salinity values of 6–12 ppt, and mortality of *H. wrightii* shoots begin below 6 ppt (Doering et al., 2002). The quantity and duration of the Caloosahatchee high freshwater flows created a low and variable salinity environment resulting in the disappearance of *T. testudinum* at this transect from 2005–2009, as well as the decrease in all San Carlos Bay's seagrass species abundance in 2005 and *T. testudinum* abundance over the study period.

Hurricane related freshwater discharges have affected seagrasses on the southeast coast of Florida (Loxahatchee River Estuary) as well. *S. filiforme* declined one month after hurricanes Frances and Jeanne in 2004 due to the high daily salinity fluctuations that resulted from the freshwater discharges (Ridler et al., 2006).

Even though San Carlos Bay has been subjected to high flow events from the Caloosahatchee River, this region has on average, from 1999–2009, the deepest growing seagrasses for all species combined (-1.91 m, FIG. 5). This may be due to the fact San Carlos Bay is a deeper waterbody compared to the other shallow estuaries of the study area, but Duffey et al. (2007) noted San Carlos Bay to have above average water clarity and found a significant increase in secchi depth (clarity) from 1998–2005.

Several environmental and anthropogenic factors negatively influence seagrass health within the CHAP other than salinity fluctuations. Nutrient over-enrichment in the water column can lead to harmful algal and epiphytic growth while frequent activities such as boating, trawling, and coastal land development (including dredging and filling) can cause an increase in turbidity (Burkholder et al., 2007; McGlathery, 2001). Together, turbidity and excess nutrients can cause a reduction in water clarity therefore leading to seagrass decline (Burkholder et al., 2007; Tomasko, 2005). Burkholder et al. (2007) and Orth (2006) explain that "other human-related changes such as increased temperatures from global warming, exotic species introductions, and trophic imbalances that lead to overgrazing may also interact with nutrient enrichment and other stressors to cause seagrass declines."

While some of the detrimental factors to seagrass are not directly manageable, such as reduced salinity due to high rainfall and storm events, others could be more effectively managed. For example, impacts from boat propeller scarring and harmful artificial releases of freshwater could be managed in an effort to support healthy and diverse seagrass beds within the CHAP.

The results from this monitoring program highlight the variability of seagrass beds found within CHAP over the study period. Specific trends in seagrass density and total abundance were dependent upon on when the parameter was first collected. In order to properly characterize long term trends, the CHAP seagrass monitoring program will continue providing a critical tool to capture annual abundance, densities, species composition, and deep edge of bed trends. These monitoring data play an integral role in assessing seagrass and estuarine health. Linking additional water quality parameters and future clarity trends to the CHAP seagrass monitoring program data will be critical to the management of the Charlotte Harbor estuarine system.

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SEAGRASS IN THE CALOOSAHATCHEE RIVER ESTUARY: THE EFFECT OF ANNUAL RAINFALL PATTERNS

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Abstract: *Seagrasses are important to estuarine health, influencing physical, chemical and biological environments of coastal waters. Seagrasses stabilize sediments, filter nutrients, and provide habitat for estuarine organisms. Understanding seagrass distribution and trends relative to freshwater input aids resource management. From 1996–2009, hydroacoustic technology was used to assess spatial and temporal fluctuations in seagrass coverage of the Caloosahatchee River Estuary relative to annual rainfall. Three estuarine areas, with different salinity regimes and species composition, were monitored three times a year for percent seagrass coverage and plant height. Sampling was performed at the beginning (spring), middle (summer), and end (fall) of the seagrass growing season. Results showed that seagrass percent coverage, percent volume infestation and plant height increased with distance downstream. All three parameters were greatest in summer, intermediate in fall and lowest during spring months. Annual rainfall influenced seagrass abundance differently. During average and wet years, seagrass measurements were greatest in summer, while in dry years measurements were similar in summer and fall. Study results indicated that seagrasses in the Caloosahatchee River Estuary are sensitive to inter-annual changes in rainfall. While there is considerable year to year variation, seagrass coverage has been relatively stable over the 13 year study period.*

Key Words: Caloosahatchee River Estuary, annual rainfall, hydroacoustic, seagrass

Seagrasses play an important part in the health and well-being of aquatic ecosystems. Seagrasses influence the physical, chemical and biological environments of coastal waters by stabilizing sediments, buffering or filtering nutrient and chemical inputs to the system, and by providing habitat, refugia, and food to many stages of estuarine organisms (Diaz et al., 2004; Zieman and Zieman, 1989). The services seagrasses provide also play an economic role in the support of coastal ecosystems. Nutrient cycling provided by seagrasses had an estimated global value of \$19,000 $ha^{-1}yr^{-1}$ in 1994 (Conservation International, 2008), and in 1998, Monroe County Florida reported an estimated 53 million dollars in total revenue of commercial and recreational fisheries, for seven seagrass-dependent species (Green and Short, 2003).

Globally, seagrass is waning at an alarming rate (Pulich and White, 1991; Waycott et al., 2009). Fifty-eight percent of the world's seagrass beds are in a state of decline, while twenty-nine percent of the known areal extent has disappeared since 1879 (Fourqurean et al., 2009). Many areas have attributed declines in seagrass abundance and distribution to changes in freshwater delivery to estuarine systems. For instance, alterations in freshwater inflow resulting from watershed development and water management practices have affected salinity and water quality within southwest Florida estuaries (Carlson et al., 2010; Keener et al., 2010; Schmidt et al., 2006). In turn, changes in salinity and other water-quality parameters affect the composition, distribution and abundance of seagrass in these systems (Greenawalt-Boswell et al., 2006; Corbett and Hale, 2006; Doering et al., 2002).

The Caloosahatchee River and Estuary, located on the southwest coast of Florida, are part of the larger Charlotte Harbor system (Fig. 1). Input to the Caloosahatchee River flows come from three main sources, Lake Okeechobee, the C-43 watershed, and the Caloosahatchee River Estuary tidal basin. Depending on the period of record considered, contributions from the C-43 watershed can range from 44–52%, while contributions from Lake Okeechobee and the Caloosahatchee River estuary tidal basin can range from 17–31% and 24–32% respectively (SFER 2012). The influence of these annual flows on salinity and water quality extend beyond the Caloosahatchee Estuary into the adjacent waters of San Carlos Bay and Pine Island Sound (Doering and Chamberlain, 1998).

There are four predominant species of seagrass found in the Caloosahatchee system, distributed along the salinity gradient from the head of the estuary out into San Carlos Bay and Pine Island Sound. The salt-tolerant freshwater species, *Vallisneria americana* (Michx) (tape grass) grows in the upper estuary (Doering and Chamberlain, 1999). The freshwater tolerant marine species *Halodule wrightii* (Ascherson) (shoal grass) is found from the lower Caloosahatchee Estuary into Pine Island Sound (Doering et al., 2002). The more obligate marine species *Thalassia testudinum* (Banks ex Konig) (turtle grass) and *Syringodium filiforme* (Kutzing) (manatee grass) are found in San Carlos Bay and Pine Island Sound. Salinity and other water quality requirements of seagrasses have been used to establish water quality targets (Corbett and Hale, 2006) for the entire Charlotte Harbor estuary system and freshwater inflow limits (Doering et al., 2002) for the Caloosahatchee River. Monitoring the resources (e.g. seagrass) upon which environmental targets are based is key to verifying the validity of these targets (Chamberlain et al., 2009).

The three methodologies for characterizing and monitoring seagrass are physical, off-water remote and on-water remote. Established manual techniques (physical) are labor-intensive and generate observations of very limited spatial extent. Off-water remote techniques, such as aerial imagery, provide large synoptic assessments of spatial patterns but are highly dependent on uncontrollable environmental factors. On-water remote techniques include boat-based methods using optical or hydroacoustic sensing devices not in direct contact with the vegetation (Sabol et al., 2002; Winfield et al., 2007).

Here we report the results of thirteen years of monitoring seagrass in the downstream portion of the Caloosahatchee system (1996–2009), using a hydroacoustic technique (Sabol et al., 2002; Chamberlain et al., 2009).

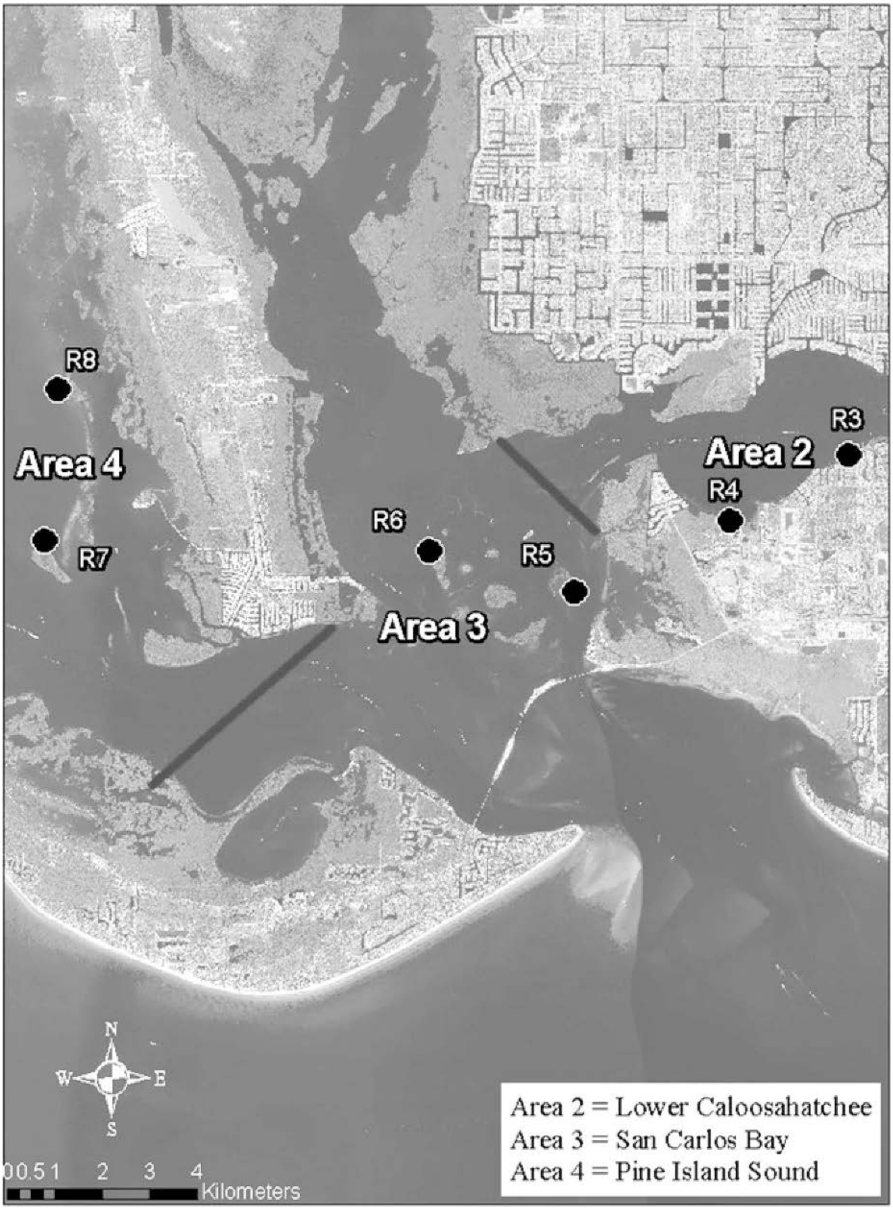

FIG. 1. Study area within the Caloosahatchee River Estuary. Lines delineate the breaks between Area 2 (the Lower Caloosahatchee River (LCR)) and Area 3 (San Carlos Bay (SCB)) and between Area 3 and Area 4 (Pine Island Sound (PIS)). Circles mark the approximate location of each reach sampled within each area (R3–R8).

Chamberlain et al. (2009) compared this technique to traditional manual monitoring and found that in the Caloosahatchee River Estuary seagrass density and canopy height were similar between the two methods.

In this study we relate spatial and temporal variations in seagrass coverage to variation in annual rainfall (i.e. average, dry and wet). We expect seagrass coverage, percent volume infestation and plant height to decrease in wetter years due to lower salinity and increased light attenuation.

METHODS—*Hydroacoustic sampling*—Hydroacoustic sampling procedures used in this study have also been used by Sabol et al. (2002) and Chamberlain et al. (2009). Equipment used in data collection was the same used by Sabol et al. (2002) and described in the aforementioned publication as a "boat-based system, referred to as the Submersed Aquatic Vegetation Early Warning System (SAVEWS)". The system used in this study consisted of a Biosonics DT4000 digital echo sounder (Biosonics, INC., Seattle Washington; Acker et al., 1999) and a Leica differentially corrected GPS linked to a Panosonic Toughbook PC. Monotone pulses or pings are generated by the 420-kHz, 6-deg single-beam transducer (echo sounder). The rate and duration of the pings is set by the user, which we set as 5 pings s^{-1} and 0.1 ms respectively, the commonly used settings. Return echoes are digitized at high frequency and dynamic range and the resultant data were stored on the Panosonic Toughbook hard drive. GPS position reports (latitude and longitude) are recorded at a slower rate (0.5 to 1.0 reports s^{-1}) and interspersed throughout the data. Horizontal accuracy of the GPS is approximately 5 m (Logsdon, 1992).

Sampling locations—Study sampling locations are shown in FIG. 1. Three areas (with 2 sites or "reaches" per area) within the Caloosahatchee River Estuary were sampled along a salinity gradient ranging from moderately mesohaline sites with only *Halodule wrightii*, to euryhaline sites containing a mix of seagrass species, including *Thalassia testudinum*, *Halodule wrightii*, and *Syringodium filiforme*. Although sites contained a mix of seagrass species the Biosonics equipment cannot differentiate between species so all seagrasses were lumped together. Each area/reach was located in the general vicinity of long-term South Florida Water Management District seagrass monitoring stations sampled using the physical method (Chamberlain and Doering, 1998a; Chamberlain and Doering, 1998b).

Each reach consisted of 10 parallel transects running perpendicular to shore. Transects were spaced 50 meters apart and varied in length between 100 and 425 meters. Sampling consisted of slowly (3–5 km hr^{-1}) driving the boat (transducer attached) along each transect using the GPS for navigation. Transects were generally sampled around high tide ($\pm$3 hrs.) to minimize danger of damaging the transducer, the sea bottom, or grounding the boat.

Sampling period—During 1996–2009, sampling was performed, with some exceptions, at the beginning (spring), middle (summer) and end of the growing season (fall) (TABLE 1).

Data processing—At the end of each sampling day, data collected from SAVEWS (one DT4 file per transect) was post processed through EcoSAV, a program developed by Biosonics specifically for this purpose. EcoSAV transforms the DT4 files into ODF files. The ODF files along with tidal information taken before and after sampling each reach, were then processed through another Biosonics program called Finalize in which bottom depth is corrected to MLW and the ODF files become one CSV file. This post processing procedure associates all data (water depth, plant height and plant percent coverage) with a reach number, transect number and state plane coordinates. The CSV file was than saved as an excel file for easy manipulation/analysis of the data.

Due to limits of the equipment, data was sorted by bottom depth and all depths shallower than 0.5 meters and deeper than 2.1 meters were deleted (BioSonics, 2004). Elimination of data in this way did not present a problem since seagrass in the Caloosahatchee has historically occurred in areas less than 2 meters deep. Of the 285,560, data points collected over the 13 year period 26,385

TABLE 1. Calendar year associated with season and survey number. S represents survey. NA designates months when surveys were not completed.

	Season		
	Spring	Summer	Fall
Year	Date (Survey #)	Date (Survey #)	Date (Survey #)
1996	March 25–28 (S1)	June 11–14 (S2)	September 10–13 (S3)
1997	NA	NA	NA
1998	NA	NA	September 1–3 (S6)
1999	March 21–23 (S7)	June 15–17 (S8)	October 12–14 (S9)
2000	March 15–17 (S10)	June 15–16 (S11)	September 25–27 (S12)
2001	March 26–29 (S13)	June 18–21 (S14)	October 1–3 (S15)
2002	March 18–20 (S16)	June 24–25 (S17)	September 3–5 (S18)
2003	March 21–23 (S19)	July 29–31 (S20)	October 6–9 (S21)
2004	March 10–13 (S22)	NA	NA
2005	NA	June 20–23 (S25)	September 12–14 (S26)
2006	April 17–19 (S27)	June 28–29 (S28)	September 19–21 (S29)
2007	March 20–22 (S30)	June 13–14 (S31)	September 18–19 (S32)
2008	April 8–10 (S33)	June 3–5 (S34)	September 22–23 (S35)
2009	NA	June 8–9 (S36)	NA

(9.2%) points were deeper than 2.1 meters while 25,516 (8.9%) points were less than 0.5 meters deep. Of these 51,901 points only 14,278 (5% of total) were reported to contain seagrass. Since limitations of the equipment put the validity of these out of bounds points in question, they were deleted. Also, when multiple records of the same latitude and longitude occurred, the data were considered spurious and were deleted.

Data analysis—The analysis of hydroacoustic data focused on percent seagrass coverage, plant height, and percent volume infestation (PVI) or biovolume. PVI describes available habitat (area of the water column filled by seagrass or the volume of biological habitat) for fish and zooplankton communities (Werner et al., 1977; Schriver et al., 1995; Perrow et al., 1999). This parameter was determined by dividing plant height by water depth and multiplying by percent cover (Canfield et al. 1984, Schriver et al., 1995).

Averages for each parameter were calculated for each transect, and transects within reaches were averaged to produce one observation per reach per survey.

Zero values were included when averaging the percent coverage since coverage is an area-based measure. However, it is not accurate to include zero values when averaging height. Therefore, when determining average plant height values along a transect, non-vegetated samples were not included.

Effects of spatial and temporal variation on percent coverage, plant height and PVI were evaluated using a mixed effect analysis of variance model. Factors were season (spring, summer, fall) area (Lower Caloosahatchee, San Carlos Bay and Pine Island Sound) and reaches (2 reaches/area). Season and area were considered fixed. Reach was considered to be a random factor. Significant main effects and interactions were evaluated with contrast statements ($p<0.05$). Prior to analyses, the dependent variable was ranked because the data and residuals violated the normality assumption. The dependent variable was ranked from smallest to largest. Average ranks were assigned in case of ties. After ranking the data, parametric ANOVA was performed on the data. This is analogous to a Friedman type of analysis (Conover and Iman, 1981; Conover and Iman, 1976).

Rainfall—Effects of rainfall variation on seagrass within the study area were investigated by dividing annual (January–December) total rainfall, into three categories (average, dry and wet).

TABLE 2. Annual rainfall categories (average, dry, wet) in inches per year, associated with survey number. S represents survey.

	Inches of Rain	Year	Surveys
Dry	≤ 49.9	2007	S30, S31, S32
		1996	S1, S2, S3
		2000	S10, S11, S12
		2006	S27, S28, S29
Average	> 49.9 and ≤ 57.5	2009	S36
		2002	S16, S17, S18
		2004	S22
		2003	S19, S20, S21
		2001	S13, S14, S15
Wet	>57.5	1998	S6
		1999	S7, S8, S9
		2008	S33, S34, S35
		2005	S24, S25, S26

Categories were developed by ranking the average annual rainfall (area-weighted Nexrad data from the South Florida Water Management District's database DBHYDRO) over the period of record, and then assigning the years in the top third wet, the years in the middle third average and the years in the lower third dry (TABLE 2). Rainfall areas used in calculations were Tidal North, Tidal South, Caloosahatchee Estuary, Telegraph Swamp and East and West Caloosahatchee Basins.

Each survey was assigned to a particular rainfall category. A mixed-effect analysis of variance model was used to analyze ranked data. Fixed factors were season, area, and annual rainfall categories. Significant main effects and interactions were evaluated with contrast statements ($p<0.05$).

Long-term, monotonic trends in percent coverage, plant height and PVI in each of the three regions were evaluated using the Seasonal Kendall Tau statistic ($p<0.05$) (Helsel et al., 2006). Data from the two reaches within each region were averaged before testing.

RESULTS—While there is considerable year to year variation, Seasonal Kendall Tau results suggest that seagrass coverage has been relatively stable over the 13 year period of observation, particularly in the Lower Caloosahatchee and San Carlos Bay. A slight declining trend may exist within the Pine Island Sound sites, but only for percent volume infestation (FIG. 2).

Analysis of variance showed a significant difference ($p<0.05$) in percent cover, plant height and PVI between the three areas (FIG. 3) and three seasons (FIG. 4) ($p = <0.0001$ for all). In general, percent cover, plant height, and PVI were greatest in Pine Island Sound, lowest in the Lower Caloosahatchee, with San Carlos Bay being intermediate. Percent cover, PVI and plant height tended to be greatest in summer and lowest in spring. The interaction between area and season was also significant ($p<0.05$) for all three of the above parameters ($p=0.0061$, $p=0.0071$, $p=0.0041$ respectively) (TABLE 3a).

The interaction between season and area was analyzed further with contrast statements. While the general patterns seen for the main effect of area held true for percent cover, plant height and PVI, in all seasons, differences were not always statistically significant. For example, while the PVI in Pine

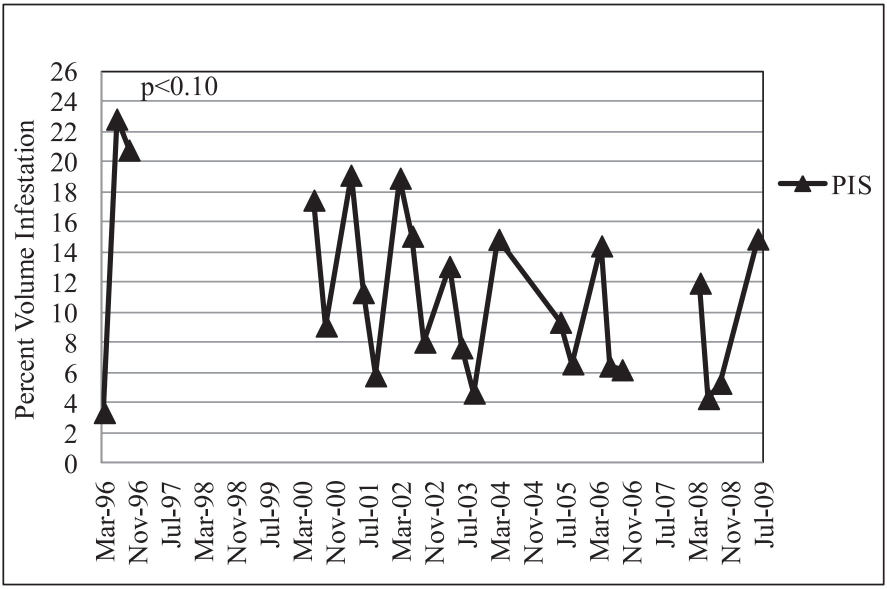

FIG. 2. Time series plot showing changes in Percent Volume Infestation over the 13 year study in Pine Island Sound (PIS).

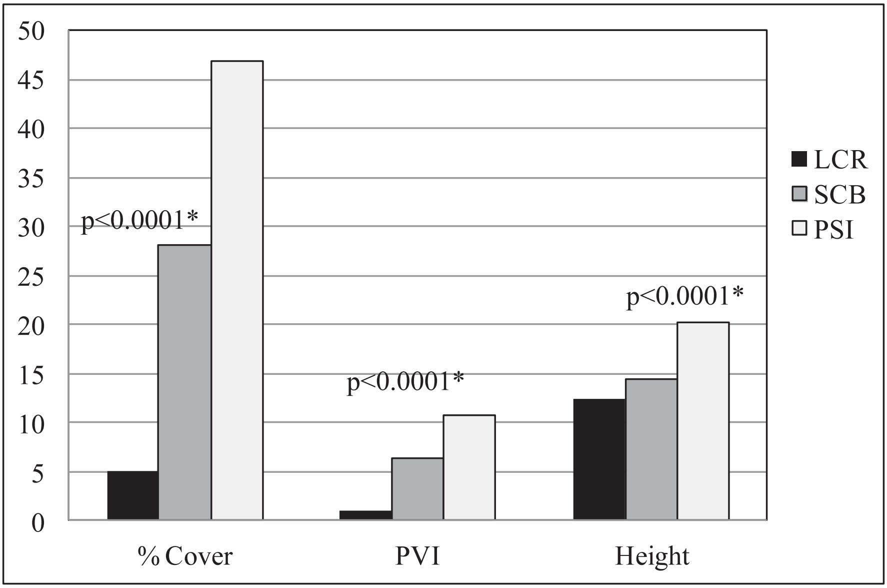

FIG. 3. Mean percent seagrass cover, mean percent volume infestation (PVI) and mean plant height by area (Lower Caloosahatchee River (LCR), San Carlos Bay (SCB), and Pine Island Sound (PIS) for the 13 year period of record. Statistical analysis of seagrass cover, PVI and plant height by area was run on ranked data. *Designates significance at $p<0.05$.

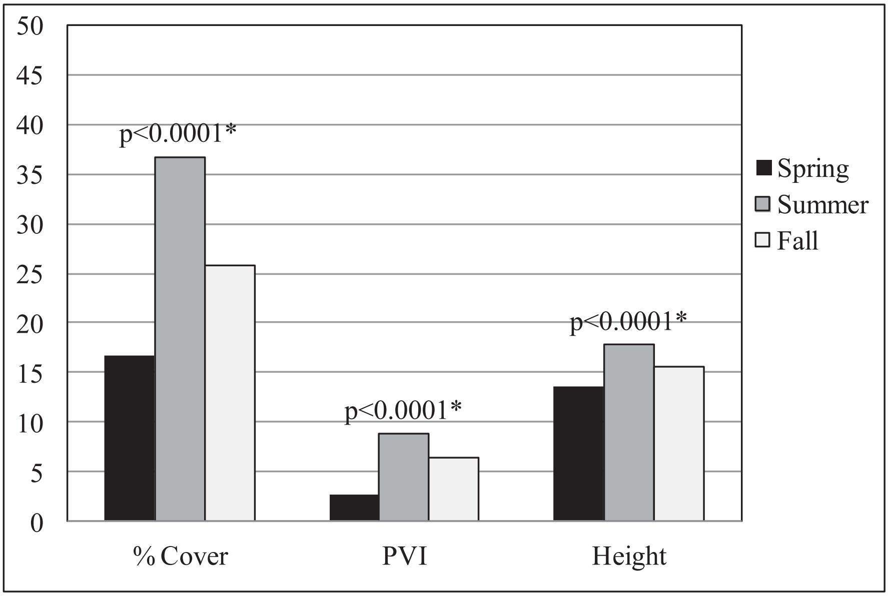

FIG. 4. Mean seagrass cover, mean percent volume infestation (PVI) and mean plant height by season (spring, summer, and fall) for the 13 year period of record. Statistical analysis of seagrass cover, PVI and plant height by season was run on ranked data. *Designates significance at $p<0.05$.

Island Sound during summer was greater than PVI in San Carlos Bay during summer, the difference was not statistically significant.

The second mixed ANOVA included an additional factor: annual rainfall category (wet, dry or average). Annual rainfall category was statistically significant for only percent volume infestations (TABLE 3b), although all three parameters were greatest in dry years and lowest in average years (FIG. 5). Percent cover, plant height and PVI all displayed the same trends between areas (Pine Island>San Carlos Bay>Lower Caloosahatchee, though the interaction of area X rainfall was not statistically significant (TABLE 3b).

Contrary to our spatial analysis, rainfall does seem to affect temporal differences. Contrast statements were used to analyze the interaction between season and annual rainfall category averaged over the 13 year period of record. Results showed that during wet and average rainfall category years, plant coverage, PVI and plant height during summer were significantly higher than in either spring or fall, which were similar (TABLE 3b). In dry years, plant coverage, PVI and plant heights in summer and fall were similar and greater than spring.

DISCUSSION—The hydroacoustic technique for monitoring seagrass was used in this study to assess spatial and temporal patterns in seagrass coverage, plant height and percent volume infestation (PVI) in lower Charlotte Harbor. There are many advantages for using the hydroacoustic technique for

TABLE 3a. Percent seagrass cover, percent volume infestation (PVI) and plant height for area (Lower Caloosahatchee River (LCR), San Carlos Bay (SCB), Pine Island Sound (PIS), season (spring, summer, fall) and the interaction of area X season. Associated p-values for each effect and parameter are included. Statistical analysis of seagrass cover, PVI and plant height by season and area was run on ranked data. *Designates significance at p<0.05.

	p<0.05 % Cover	p<0.05 PVI	p<0.05 Height			% Cover (N)	PVI (N)	Height (N)
Area	<0.0001*	<0.0001*	<0.0001*	LCR		5.0 (64)	12.0 (64)	1.0 (64)
				SCB		28.2 (63)	14.4 (63)	6.4 (63)
				PIS		47.0 (63)	20.2 (63)	11.0 (63)
Season	<0.0001*	<0.0001*	<0.0001*	Spring		16.7 (60)	2.8 (60)	13.5 (60)
				Summer		36.7 (65)	8.8 (65)	17.7 (65)
				Fall		25.8 (65)	6.4 (65)	15.5 (65)
Area x Season	0.0061*	0.0071*	0.0041*	LCR	Spring	0.5 (20)	0.1 (20)	12.3 (20)
				LCR	Summer	10.0 (22)	1.9 (22)	12.7 (22)
				LCR	Fall	4.3 (22)	0.9 (22)	12.1 (22)
				SCB	Spring	10.9 (20)	1.9 (20)	12.4 (20)
				SCB	Summer	46.1 (22)	10.7 (22)	16.4 (22)
				SCB	Fall	25.9 (21)	6.3 (21)	14.3 (21)
				PIS	Spring	38.5 (20)	6.3 (20)	15.8 (20)
				PIS	Summer	54.7 (21)	14.0 (21)	24.5 (21)
				PIS	Fall	47.2 (22)	12.0 (22)	20.2 (22)

monitoring seagrass. The method is cost effective. Whereas the system used in this study cost a one-time fee of approximately 25,000 dollars (compared to 50,000 dollars per sampling event for aerial photography), a junior system now exists for 3,000 dollars (Sabol 2012). During a typical two day monitoring event 2 people can collect upwards of 7,000 data points (several orders of magnitude greater then by typical manual monitoring). The system can be used in dark water which allows mapping of areas like that of the Caloosahatchee River, where aerial photography is normally ineffective. Also the system can be used in adverse weather conditions or in areas where putting divers in the water may be dangerous. Sampling is not limited by time of day.

The two main disadvantages of the system are canopy heights <7 centimeters cannot be detected and the system cannot distinguish between seagrass species. As with aerial photography, changes in coverage can be quantified but changes in species composition cannot. The finding that seagrass coverage in the three sampling regions has been relatively stable over the 13 year period, must not be interpreted to mean that species composition has remained constant as well. For example, after the 2004 hurricane season, degraded water quality eliminated a *Syringodium filiforme* bed in the Indian River Lagoon. Although percent coverage estimates returned to pre-hurricane conditions relatively quickly the bed endured years of successional changes (*Halophila decipens*, *Halophila johnsonii*, *Halodule wrightii*) before returning to the a Syringodium climax community (B. Orlando personal observation, Buzzelli et al., 2012).

The sampling frequency employed (3 times per year) and the fact that we did not conduct regular, coincident water quality monitoring imposed

TABLE 3b. Percent seagrass cover, percent volume infestation (PVI) and plant height for rainfall category (average, dry, wet) and the interactions of rainfall category X area and rainfall category X season. Associated p-values for each effect and parameter are included. Statistical analysis of seagrass cover, PVI and plant height by rainfall category was run on ranked dat. *Designates significance at p<0.05.

	p<0.05 % Cover	p<0.05 PVI	p<0.05 Height			% Cover (N)	PVI (N)	Height (N)
Rainfall	0.1943	0.1104	0.0016*	Average		23.8 (65)	5.1 (65)	14.7 (65)
				Dry		29.4 (72)	6.9 (72)	16.4 (72)
				Wet		26.4 (53)	6.2 (53)	15.8 (53)
Area x Rainfall	0.2303	0.5701	0.1959	LCR	Average	5.1 (22)	0.9 (22)	11.5 (22)
				LCR	Dry	4.3 (24)	0.9 (24)	13.3 (24)
				LCR	Wet	6.1 (18)	1.2 (18)	12.1 (18)
				SCB	Average	26.8 (22)	5.8 (22)	13.9 (22)
				SCB	Dry	29.3 (24)	6.8 (24)	15.0 (24)
				SCB	Wet	28.5 (17)	6.7 (17)	14.3 (17)
				PIS	Average	40.4 (21)	8.7 (21)	18.8 (21)
				PIS	Dry	54.4 (24)	12.9 (24)	20.9 (24)
				PIS	Wet	44.6 (18)	10.5 (18)	20.9 (18)
Season x Rainfall	<0.0001*	<0.0001*	0.0010*	Spring	Average	16.3 (24)	2.7 (24)	12.9 (24)
				Spring	Dry	13.6 (24)	2.1 (24)	13.4 (24)
				Spring	Wet	23.5 (12)	4.1 (12)	14.9 (12)
				Summer	Average	37.9 (23)	8.6 (23)	17.3 (23)
				Summer	Dry	35.7 (24)	8.9 (24)	18.3 (24)
				Summer	Wet	36.3 (18)	8.9 (18)	17.5 (18)
				Fall	Average	15.9 (18)	3.6 (18)	13.6 (18)
				Fall	Dry	38.7 (24)	9.7 (24)	17.5 (24)
				Fall	Wet	20.2 (23)	5.1 (23)	15.0 (23)

additional constraints, limiting our study to an assessment of seasonal, inter-annual and spatial variation. In addition the suite of possible explanatory variables was also limited. While both high and low freshwater inflows from the Caloosahatchee River are of particular concern (Doering et al., 2002; Chamberlain and Doering 1998a; Chamberlain and Doering 1998b), we chose to relate variation in seagrass parameters to variation in annual rainfall patterns. Associating responses of seagrass to specific periods of high and low freshwater inflow (and hence salinity) in the Caloosahatchee-San Carlos –Pine Island region and in other systems as well, is often based on more frequent sampling and interpretation of results is enhanced by coincident water quality data (e.g. Buzzelli et al., 2012; Ridler et al., 2006; Doering et al., 2002). Additionally, freshwater inflows to our study area are measured only at the Franklin Lock and Dam (S-79). Flows from the tidal basin downstream of S-79 are ungauged. Estimates suggest that 25–30% of the total surface water inflow to the estuary upstream of Shell Point may come from the Tidal Basin (SFWMD, 2012). Additionally, recent studies suggest that groundwater inflows to the Calooshatchee estuary may be significant ranging between 1.3×10^6 m^3 day^{-1} (530 cfs) and 3.3×10^6 m^3 day^{-1} (1300 cfs) seasonally (Loh

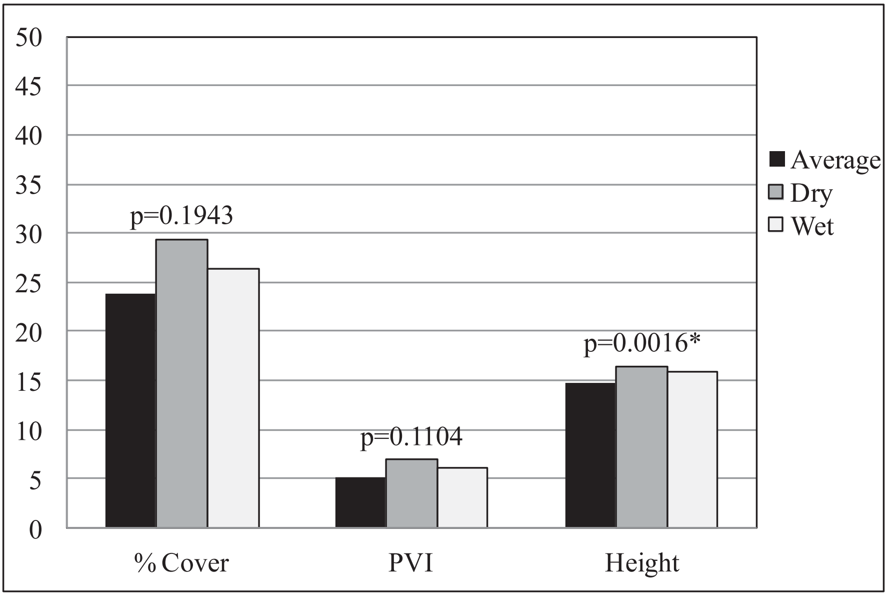

FIG. 5. Mean percent seagrass cover, mean percent volume infestation (PVI) and mean plant height by rainfall category (average, dry, wet) for the 13 year period of record. Statistical analysis of seagrass cover, PVI and plant height by rainfall category was run on ranked data. *Designates significance at $p<0.05$.

et al., 2011). Given the uncertainties in quantifying freshwater inflows to our study area, we used annual rainfall as a surrogate.

Both spatial and temporal patterns in seagrass response variables were detected. Seagrass coverage, canopy height and PVI all increased from the Caloosahatchee estuary to Pine Island Sound reflecting a gradient of exposure to freshwater. This pattern was evidenced in all three seasons and may be due to a combination of lower and more variable salinity and higher color nearer the Caloosahatchee estuary (Doering and Chamberlain 1998; Doering and Chamberlain 1999). The freshwater Caloosahatchee River has high concentrations of color and these decrease with distance from S-79 (Doering and Chamberlain 1998). Further color is major attenuator of light in the Caloosahatchee Estuary, (Doering et al., 2006), San Carlos Bay and Pine Island Sound (Ott et al., 2006). Lower light availability may also have contributed to the spatial patterns in seagrass variables that we observed. Various studies have described the influence of varying salinity and freshwater discharge on the distribution, abundance and species composition of seagrasses (Lirman et al., 2008, Irlandi 2006, Greenawalt-Boswell 2006, Bjork et al., 2008, Doering and Chamberlain 2000).

As expected, the analysis of percent seagrass cover, plant height and PVI all displayed the classic pattern with respect to season with abundances being greatest during the summer growing season. However, the results obtained from our analysis indicated that differences between seasons varied with annual rainfall.

During dry years, all three parameters measured had the greatest abundance during summer and fall which were similar. This outcome suggests that during dry years reduced rainfall results in higher salinities and clearer water: an environment more suitable for survival and growth of the seagrass found in these areas. By contrast, during average and wet years, abundance during fall was much reduced and similar to spring. Greater rainfall and the associated lower salinity and perhaps higher light attenuation may account for this pattern.

Inter-annual variation in rainfall and resultant runoff have been shown to impact seagrasses (eg. Ridler et al., 2006; Carlson et al., 2003; Carlson et al., 2010). For example, Charlotte Harbor, Tampa Bay, and Sarasota Bay showed significant declines in seagrass after major rainfall and runoff associated with the 1997–1998 El Nino event (Carlson et al., 2003, Carlson et al., 2010).

The 13 year hydroacoustic study described above was initiated in 1996 as a relatively quick and cost effective method to assess spatial and temporal variation in the abundance and distribution of seagrass within the CRE system. Given the sampling frequency employed in this study, the technique has been shown to detect spatial variation on the scale of kilometers, and temporal variation at seasonal and inter-annual scales. As employed here, the technique was also useful in detecting the influence of annual rainfall on seasonal variation in seagrass coverage, canopy height and PVI. Hydroacoustic monitoring at more frequent intervals over the spatial scales sampled here could provide valuable information on seagrass responses to different size storm events at the ecosystem level.

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DESCRIPTION OF THE BENTHIC MACROINVERTEBRATE COMMUNITIES OF FOUR TIDAL CREEKS ALONG THE EASTERN SHORE OF CHARLOTTE HARBOR

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Abstract: *The macroinvertebrate communities of four tidal creeks along the eastern shore of Charlotte Harbor were sampled bimonthly from May 2008 to June 2010. The objective of the sampling effort was to survey the benthic macroinvertebrate taxa and assess water quality characteristics and habitat conditions which may influence macroinvertebrate community structure. A total of 156 different taxa were found in these creeks, several of which were very common throughout the study area. Macroinvertebrate communities differed among tidal creeks and corresponded, in part, to benthic habitat type and diversity and to a lesser extent average salinity and other environmental conditions. These results emphasize the importance of both physical habitat and variability in environmental conditions such as long term salinity regimes and occurrences of large inflow events in supporting a diverse macroinvertebrate community.*

Key Words: Charlotte Harbor, tidal creeks, macroinvertebrates

Charlotte Harbor is a diverse and productive estuary that supports a wide range of biologically, recreationally, and commercially important species of finfish, shellfish, mammals and birds. The estuary shoreline is dominated by mangrove forest with numerous tidal inlets and creeks throughout. Tidal creeks are small tributaries that provide a link between upland watersheds and freshwater streams and the open estuary. They often exhibit unique hydrogeographic and water quality characteristics that differ from freshwater systems and from the open estuary downstream (TBTTRT, 2008). Tidal creeks are known to provide essential refuge and nursery habitat for juvenile organisms (Shenker and Dean, 1979; Holland et al., 2004; Dixon and Adams, 2010). Benthic macroinvertebrates play an important role in tidal creek biological communities—functioning as nutrient cyclers as well as providing an important food source for higher trophic level organisms (Odum and Heald, 1973; Adams et al., 2009).

The ecologic importance of tidal creeks has been of increasing interest over the past several years. A number of research efforts along the Gulf Coast of Florida have characterized fisheries populations within tidal creeks (e.g., Greenwood et al., 2008; Dixon and Adams, 2010). Estevez et al. (2010) researched a method for assessing tidal creek biological condition, and the Tampa Bay Tidal Tributary Research Team (2008) established the importance of tidal tributaries as estuarine habitat in Tampa Bay. An inventory of invertebrates from a number of open estuary and marsh habitats in the greater Charlotte Harbor estuary was conducted by Mote Marine Laboratory (2007), and macroinvertebrate studies were conducted in Charlotte Harbor habitats including the Peace and Myakka Rivers by Estevez (1986).

Although our understanding of tidal creek ecology is expanding, information on the macroinvertebrate communities inhabiting tidal creeks in Southwest Florida is limited. This study describes and compares the macroinvertebrate communities of four tidal creeks of differing hydrologic and watershed characteristics within a relatively small geographic area of the Charlotte Harbor estuary. The efficacy of using macroinvertebrates as indicators of environmental conditions has been widely documented in both freshwater and marine systems (Barbour et al., 1996; Wildsmith et al., 2011). The structure of macrobenthic communities is often reflective of factors such as dissolved oxygen concentrations, pollution levels, benthic habitats, freshwater inflow and hydrologic characteristics (Diaz and Rosenberg, 1995; Platell and Potter, 1996; Calabretta and Oviatt, 2008; Kanaya et al., 2011). Therefore, differences in macroinvertebrate assemblages among seasons and creeks may help to identify the main drivers of creek dynamics.

METHODS—*Study area*—The study was conducted along the eastern shoreline of Charlotte Harbor, a shallow, sub-tropical estuary located in southwest Florida (FIG. 1). Charlotte Harbor's eastern shoreline spans approximately 12 kilometers from Punta Gorda south to Matlacha Pass and includes portions of Lee and Charlotte counties. Nearly all of the shoreline is fringed with mangroves and supports numerous networks of tidal creeks and tributaries. Although much of the estuary's eastern shoreline remains undeveloped within the Charlotte Harbor Preserve State Park, a considerable portion of the drainage area along the corridor has been drastically altered. Extensive networks of mosquito ditches and impoundments alter sheet flow and connect tidal reaches to oligohaline ponds not historically connected directly to tidal waters. Changes in land uses and increased impervious surfaces have also altered the timing, quantity and quality of freshwater inflow to these tidal creeks.

Four creeks along the eastern shoreline of Charlotte Harbor were selected for this study (FIG. 1). North Silcox Creek and South Silcox Creek are the two northernmost creeks. They are buffered by mangrove forest within the Charlotte Harbor Preserve State Park, although their headwaters extend upstream of the park's boundaries. Despite protection within the State Park and lack of intensive development within the watershed of these creeks, hydrology has been altered from natural condition by roads and ditches in the headwaters as well as by ditches and associated spoil berms constructed for mosquito control. Yucca Pen and Durden (also known as Culvert) Creeks, located in the southern portion of the study area, are within watersheds with a higher degree of hydrologic alteration and upland development than the northern creeks. The wetland buffer adjacent to each of the southern creeks is much narrower than in the northern creeks and includes needle rush (*Juncus* sp.) in addition to the predominant mangrove fringe. The in-stream and riparian habitats of all four creeks are intact, with the exception of Durden Creek, which has been truncated and whose upstream reaches now consist of a residential canal system that is connected to the creek at a single point through a fixed weir structure. The hydrologic connection exists only during high water events, when large freshwater flows discharge to the creek.

Site Selection—To thoroughly describe the benthic fauna in each creek, and to compare the creeks to one another, efforts were focused on collecting as many different taxa as possible from as many different habitat types as were available at a selected site in each creek. The location of each sample site was selected within the upper navigable reaches of each creek where conditions have the

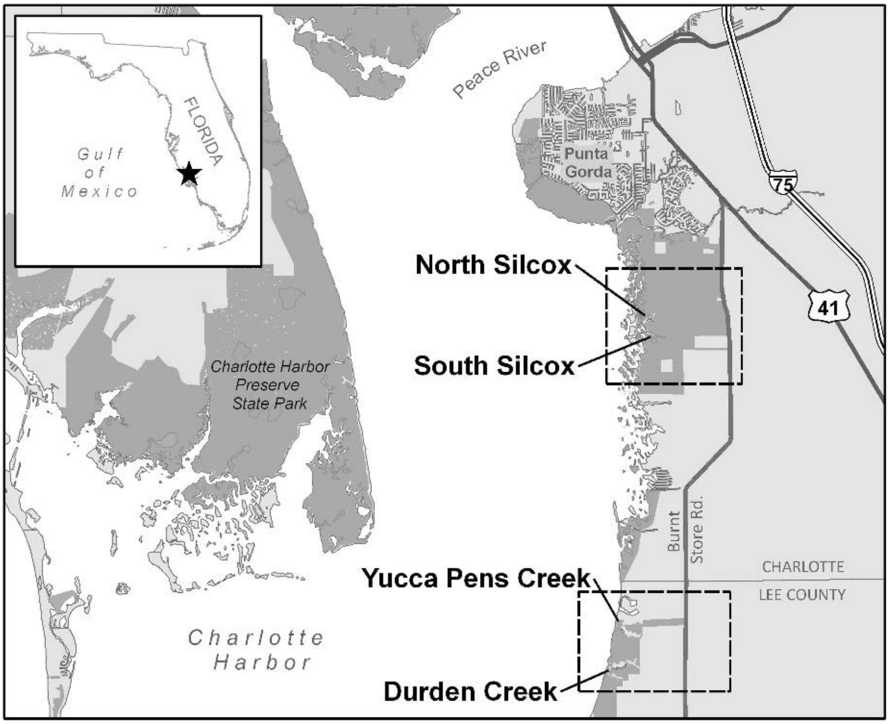

FIG. 1. Charlotte Harbor study area.

greatest potential to be influenced by both tidal exchange with the Harbor and freshwater inflow from the contributing watershed upstream. Within this transitional area, a single sample site was selected in each creek at a location with the greatest number of sampleable benthic habitats (e.g., mud/sand, mangrove roots and dead wood). The North Silcox site was located at the furthest upstream reach before the creek narrows and becomes unnavigable, about 1000 meters from the mouth. The South Silcox site was located furthest downstream, approximately 500 meters from its mouth, and as a result experiences greater currents and tidal exchange with the Harbor than other sites. The Yucca Pen Creek site was the furthest upstream, nearly 1500 meters from the creek's mouth. The larger drainage area and lower salinity observed by previous studies suggest conditions in Yucca Pen Creek are more influenced by freshwater inflow compared to the other creeks within the study area (Adams et al., 2009; Dixon and Adams, 2010). The Durden Creek site was located approximately 1000 meters from the creek's mouth and about 750 meters downstream of the water control structure. Hydrologic conditions downstream of the weir are predominately marine except during high rainfall events when freshwater inflow over the weir structure results in sudden and large fluctuations in salinity and other abiotic conditions (Adams et al., 2009; Dixon and Adams, 2010).

All four creeks had sampleable habitats of mud/sand, mangrove roots and dead wood during the majority of sampling events. Other important habitats included oyster bars in South Silcox Creek, a substantial amount of sunken logs (dead wood) in Durden Creek, filamentous algae in North Silcox Creek and sparse *Ruppia* sp. in South Silcox and Yucca Pen Creeks. The most common benthic habitats encountered at each of the sampling sites are listed in TABLE 1.

Biological sample collection and processing—Sampling methods were adapted from the Bioreconnaissance (FDEP SOP LT-7100), a rapid bioassessment methodology (FDEP, 2008). Each site was sampled 11 times approximately bimonthly between May 2008 and June 2010. Each

TABLE 1. Sample size, mean number of taxa, and total number of taxa by creek.

	Total # of Habitats					
	Avg	Max	Dominant Habitats Sampled[1]	N	Mean # Taxa	Total # Taxa
N. Silcox Creek	5	7	mud, sand, dead wood, mangrove roots, leaves	11	16	47
S. Silcox Creek	7	8	sand, dead wood, mangrove roots, oysters, SAV	11	33	96
Yucca Pen Creek	5	7	sand, mangrove roots, leaves	11	18	59
Durden Creek	5	6	mud, sand, dead wood, mangrove roots, leaves	11	22	80

[1] Habitats sampled at least 8 times throughout the study period

sampling event consisted of the collection, including sorting and counting, of individuals in the field for a total of two man hours per site, which was sufficient to sample all the habitat types present. During each sampling event benthic macroinvertebrates were collected from all available subtidal and intertidal habitat types along a 50 m stretch of the creek (TABLE 1). Sediments and leaf mats were sampled with 500 micron mesh D-frame dipnets. The upper layer (2–4 cm) of sediments and leaf mats were scooped into the net and then rinsed in the net to remove as much fine sediment as possible. The rinsed material was placed in white plastic trays where the samplers collected individuals by removing the organisms from the tray with fine forceps, sorting them into taxonomic categories, counting, and placing them in preservative for later positive identification to the lowest possible taxon. Hard substrates such as wood, mangrove roots and shell clumps were placed in the nets and rinsed vigorously to dislodge any organisms. The resulting material was also placed in a white plastic tray where organisms were removed and placed in preservative. Portions of the hard substrates themselves were also put into the plastic trays and broken apart, and firmly attached sessile organisms and burrowing organisms were removed and preserved.

As organisms were removed from the debris and collected, individuals from all habitat types were combined into a single set of sample containers for each site. The samplers identified and enumerated large taxa such as oysters, crabs and gastropods in the field. Smaller organisms were collected until a minimum of 15 individuals of each recognizable taxon was reached, and abundance categories were recorded in the field for taxa numbering between 15–49 (Abundant) as well as taxa that numbered over 50 (Dominant). Individuals of less abundant taxa (less than 15 individuals) were enumerated in the laboratory and assigned an abundance category of either Common (5–14) or Rare (1–4). Positive identifications to the lowest possible taxon were made of all taxa, including those tentatively identified and counted in the field.

Water chemistry data—Physical water quality conditions were measured during each sampling event and at each site using a YSI 600XLM sonde and 650MDS. Water quality parameters included specific conductance (later converted to salinity), dissolved oxygen, temperature, and pH. The type of benthic habitats sampled were recorded as well as tide height, direction of water flow and weather conditions.

Analyses—Statistical analyses were conducted to 1) compare differences in water quality conditions among creeks and seasons 2) compare differences in the number of taxa among creeks and seasons and 3) investigate patterns in the macroinvertebrate community structure among creeks, seasons, and in relation to environmental conditions. Differences in water quality and taxa were shown visually using box and whisker plots and were examined statistically via one-way ANOVA with Tukey's pairwise post hoc comparisons for all parameters except salinity, which used a Holm-Sidak post hoc comparison. Multivariate analyses were performed using PRIMER 6® (Plymouth Routines In Multivariate Ecological Research) software. Relative abundance data were square-root transformed prior to analysis to lessen the influence of the most common species

(Clarke and Warwick, 2001). Multidimensional scaling (MDS) and hierarchical cluster analysis were conducted based on Bray-Curtis similarity matrices. MDS plots were considered useful for interpretation if associated stress values were <0.2.

RESULTS—*Water chemistry*—Creek salinities corresponded with seasonal patterns of rainfall as well as specific rainfall events. Mean salinity was highest at Durden Creek, followed by North Silcox and South Silcox Creek sites. Yucca Pen Creek's mean salinity (10.93‰) was significantly lower ($p \leq 0.05$) than those of North Silcox, South Silcox and Durden Creeks. Yucca Pen also experienced more frequent fluctuations in salinity than all other creeks (FIG. 2). Durden Creek exhibited relatively stable and high salinity, with the exception of one occurrence of very low salinity following a large rainfall event in March, 2010, during the dry season. Freshwater inflow to Durden Creek is mediated by a water control structure which discharges from an urbanized canal system upstream during high water events.

Temperature exhibited typical seasonal patterns and was not statistically different among sites (FIG. 2). Similarly, dissolved oxygen (DO) did not vary significantly among sites, although it was generally slightly higher at the South Silcox site. Low DO conditions were often observed at all sites and the median DO for each creek was below the state criterion for marine waters of 4.0 mg/L (FDEP, 2010). Hypoxic conditions (DO $<$2.0mg/L) were most often observed at the North Silcox Creek site.

Macroinvertebrate taxa—A total of 156 taxa were collected during the study (TABLE 2). The most abundant taxa were amphipods (*Grandidierella bonnieroides* and *Apocorophium louisianum*), followed by an insect (*Rheumatobates* sp.), mollusk (*Crassostrea virginica*), and polychaete worm (*Stenoninereis martini*). Other very common taxa included *Almyracuma bacescui* (cumacean), *Balanus eburneus* (barnacle), *Geukensia demissa granosissima* (mussel), *Hargeria rapax* (tanaid), *Melita longisetosa* (amphipod), *Laeonereis culveri*, and *Leitoscoloplos robustus* (polychaetes). Of the total 156 taxa identified, 14 taxa were found in at least 50% of the samples, while 66 were collected only once. Species richness varied significantly among creeks ($p<0.05$) and showed little variation among sampling events at a given site (FIG. 3). Species richness was highest at South Silcox Creek where a total of 96 taxa were collected ($p<0.05$). Species richness was also high at Durden Creek where a total of 80 species were collected. North Silcox Creek exhibited the lowest species richness with only 47 taxa observed throughout the study (TABLE 1).

Ubiquitous taxa—Ubiquitous taxa are defined as taxa present in the majority of the samples collected at all four sites. The two most abundant taxa (the amphipods *Grandidierella bonnieroides* and *Apocorophium louisianum*) were present in at least nine of the eleven sampling events at all four sites and are considered ubiquitous to the entire study area. Because of the variable nature of estuarine environments, many estuarine species have the ability to

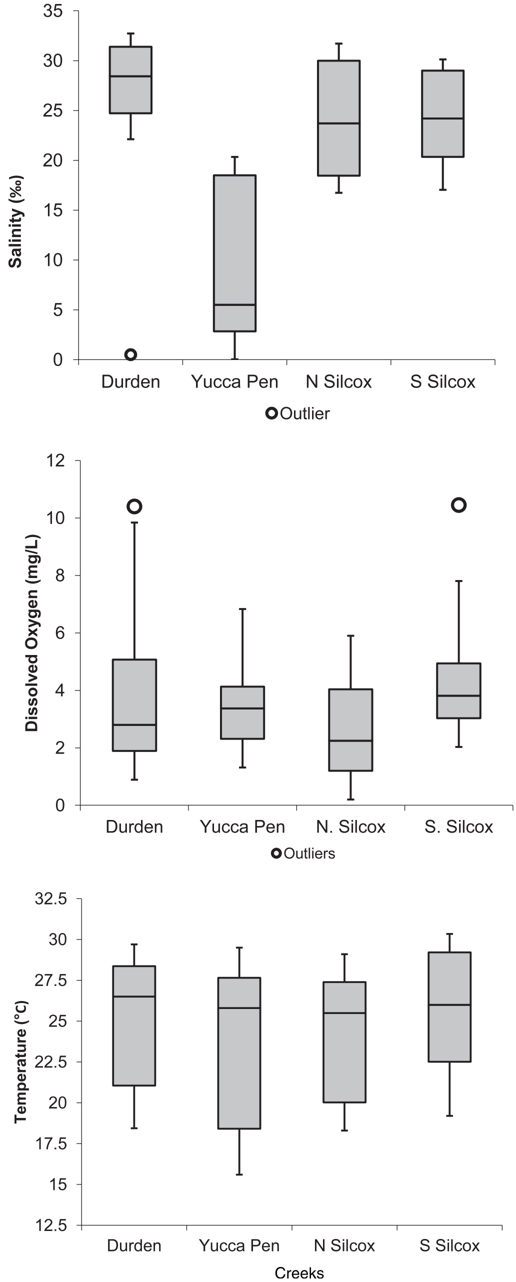

FIG. 2. Box plot of water quality results for each creek. Box and whisker plots show 2nd and 3rd quartiles, median, and whiskers equal to 1.5 times the interquartile range. Outliers are values outside the whiskers.

tolerate a wide range of environmental conditions, and may possess the ability to rapidly disburse and recolonize. The presence of these ubiquitous taxa does not necessarily indicate specific environmental conditions where they are present, but rather their absence may be an indicator of extreme or unusually stressful conditions.

Characteristic taxa—Characteristic taxa are defined as taxa that were abundant at one or two particular sites and were rare or not present at the remaining sites. Characteristic taxa were further defined as those which were either 1) four times more abundant at one site than at any other site or 2) present in over half of all samples at a particular site while being observed in less than one quarter of all samples at the remaining sites. Characteristic taxa for each site are summarized below.

North Silcox: Three taxa were characteristic of the North Silcox site, *Americorophium ellisi, Chironomus decorus* grp. and *Laeonereis culveri.* These taxa are deposit feeders which are typically tolerant of low oxygen and low salinity. Slow tidal currents and low dissolved oxygen observed at North Silcox correspond with the presence of these taxa and absence of more common filter feeders which may not tolerate lower oxygen conditions.

South Silcox: South Silcox had the highest number of characteristic taxa, which included *Crassostrea virginica, Geukensia granosissima, Crepidula* sp., *Taphromysis bowmani, Melita longisetosa, Hourstonius laguna, Erichsonella attenuata, Eurypanopeus depressus,* and *Ficopomatus miamiensis.* Conditions in South Silcox Creek are supportive of *C. virginica* (eastern oyster) in high enough abundances to form oyster bars, in turn providing hard substrate for the filter feeders *G. granosissima, Crepidula* sp., and *F. miamiensis*. These oyster bars also provide favorable structural habitat for the mud crab *E. depressus* and the amphipods *H. laguna* and *M. longisetosa*. The presence of *E. attenuata*, an epiphyte grazer common in seagrass habitats, may be attributed to the presence of *Ruppia maritima* at the site. In addition to structural habitat, less variable salinity conditions may have also contributed to the higher number of characteristic taxa at this site.

Yucca Pen Creek: Taxa considered characteristic of Yucca Pen Creek were *Palaemonetes pugio, Cassidinidea ovalis, Laeonereis culveri, Chironomus decorus* grp, and *Sphaeromopsis sanctaluciae*. The grass shrimp *P. pugio* are found at a wide range of salinities and can tolerate short periods of very low salinity conditions. Similarly, the isopods *C. ovalis* and *S. sanctaluciae*, the polychaete *L. culveri*, and the midge *C. decorus* grp. are very tolerant of fresh water conditions. The higher presence of euryhaline as well as freshwater tolerant species corresponds to lower and more variable salinity measured at the Yucca Pen Creek site.

Durden Creek: Only four taxa were identified as characteristic of Durden Creek—*Ficopomatus miamiensis, Melita longisetosa, Eurypanopeus depressus,* and *Crassostrea virginica.* These species were also characteristic of the South Silcox site which exhibited similar salinity regimes and benthic habitat types. The presence of *F. miamiensis, M. longisetosa* and *E. depressus* are likely due to

TABLE 2. List of taxa.

GROUP SUBGROUP Taxon	Creek[1]
PORIFERA (sponges)	
Porifera	S
CNIDARIA	
Anthozoa (anemones/corals)	
Actinaria	S N
Hydroidolina (hydroids)	
Hydroidolina	S
BRYOZOA (moss animals)	
Bryozoa	S Y D
PLATYHELMENTHES (flatworms)	
Euplana gracilis	S N Y D
NEMERTEA (ribbon worms)	
Nemertea	S Y D
NEMATODA (roundworms)	
Nematoda	D
ANNELIDA (segmented worms)	
OLIGOCHAETA	
Limnodrilus sp.	D
Tubificidae	S N Y D
POLYCHAETA	
Capitella sp.	D
Capitella capitata	N
Chone sp.	D
Eunicidae	D
Ficopomatus miamiensis	S N Y D
Heteromastus filiformis	Y
Hobsonia florida	S N
Laeonereis culveri	N Y D
Leitoscoloplos robustus	S N Y D
Marphysa sp.	S
Namalycastis abiuma	S
Neanthes succinea	S N Y D
Parahesione luteola	D
Polydora sp.	S D
Polydora aggregate	S
Prionospio heterobranchia	S Y
Stenoninereis martini	S N Y D
MEROSTOMATA	
XIPHOSURA (horseshoe crabs)	
Limulus polyphemus	S N
PYCNOGONIDA	
PYCOGONIDAE (sea spiders)	
Anoplodactylus sp.	S
CRUSTACEA (crustaceans)	
AMPHIPODA (amphipods)	
Americorophium ellisi	S N Y D
Ampelisca sp.	S
Apocorophium louisianum	S N Y D
Cerapus benthophilus	S
Cymadusa compta	S D
Gammarus mucronatus	N Y D
Gammarus sp. B - LeCroy	D
Grandidierella bonnieroides	S N Y D
Hourstonius laguna	S N D
Hyalella sp.	D
Melita cf. *elongata*	S
Melita longisetosa	S N Y D
Melita sp. C - LeCroy	S D
Paracaprella sp.	S
ISOPODA (isopods)	
Asellidae	S
Carpias sp.	S
Cassidinidea ovalis	S Y
Cyathura polita	S Y D
Edotea triloba	S N Y D
Erichsonella attenuata	S D
Harrieta faxoni	S
Janiridae	S
Sphaeroma quadridentatum	S N Y
Sphaeroma terebrans	S N Y D
Sphaeromopsis sanctaluciae	Y
Uromunna reynoldsi	S Y D
CUMACEA (cumaceans)	
Almyracuma bacescui	S N Y D
Oxyurostylis lecroyae	D
Oxyurostylis smithi	S D
TANAIDACEA (tanaids)	
Hargeria rapax	S N Y D
Sinelobus stanfordi	S

TABLE 2. Continued.

GROUP SUBGROUP Taxon	Creek[1]	GROUP SUBGROUP Taxon	Creek[1]
CRUSTACEA continued		INSECTA (insects)	
DECAPODA (crabs and shrimps)		DIPTERA (flies)	
Alpheus sp.	S	*Chironomus decorus* grp.	N Y D
Brachyura	Y D	*Cladotanytarsus* cf. *daviesi*	Y
Callinectes sapidus	S N Y D	*Cryptotendipes sp.*	D
Decapoda (larvae)	S	*Dicrotendipes lobus*	S Y N D
Eurypanopeus depressus	S D	*Dicrotendipes neomodestus*	Y D
Libinia dubia	S	*Dicrotendipes* sp. A- Epler	N
Majidae	D	*Goeldichironomus devineyae*	D
Neopanope packardii	Y D	*Palpomyia/Bezzia* grp.	Y
Neopanope texana	D	*Polypedilum scalaenum* grp.	SY
Palaemonetes intermedius	S	*Tanytarsus* sp. G- Epler	Y D
Palaemonetes paludosus	S N Y D	*Tanytarsus* sp. T- Epler	Y
Palaemonetes pugio	S Y D	*Tribelos fuscicornis*	Y
Panopeus obesus	D	ANISOPTERA (dragonflies)	
Petrolisthes sp.	S	*Crocothemis* sp.	Y
Sesarma cinereum	S N Y D	*Libellula* sp.	N Y D
Sesarma reticulatum	Y	TRICHOPTERA (caddisflies)	
Uca sp.	S N Y D	*Oxyethira* sp.	D
CIRRIPEDIA (barnacles)		COLEOPTERA (beetles)	
Balanus a. amphitrite	S D	*Berosus exiguus*	D
Balanus eburneus	S Y D	*Dineutus* sp.	Y
Balanus improvisus	Y D	HETEROPTERA (true bugs)	
Balanus subalbidus	S Y	*Husseyella turmalis*	D
Chthamalus fragilis	S D	*Mesovelia* sp.	D
MYSIDACEAE (mysids)		*Rheumatobates* sp.	S N Y D
Americamysis almyra	S	*Rheumatobates clanis*	Y
Bowmaniella dissimilis	S	*Rheumatobates vegatus*	D
Taphromysis bowmani	S Y D	EPHEMEROPTERA (mayflies)	
		Caenis sp.	Y
		Callibaetis floridanus	D

TABLE 2. Continued.

GROUP SUBGROUP Taxon	Creek[1]	GROUP SUBGROUP Taxon	Creek[1]
MOLLUSCA (mollusks)		GASTROPODA (snails)	
BIVALVIA (bivalves)		*Acteocina canaliculata*	S N
Amygdalum papyrium	S	*Astyris lunata*	D
Angulus tampaensis	N	*Boonea impressa*	S
Anomalocardia auberiana	S N D	*Caecum pulchellum*	S
Crassostrea virginica	S N Y D	*Cerithidea* sp.	N D
Cyrenoididae	N Y D	*Crepidula atrasolea*	S
Geukensia granosissima	S N D	*Crepidula depressa*	S D
Lioberus castaneus	S	*Crepidula maculosa*	S
Lyonsia hyalina floridana	D	Dotoidae	S
Mactrotoma fragilis	S	*Epitonium rupicola*	N
Mysella planulata	N	*Haminoea antillarum*	D
Mytilopsis leucophaeata	S N Y D	Hydrobiidae	N Y D
Parastarte triquetra	S N	*Kurtziella* sp.	S
Polymesoda caroliniana	N	*Littoridinops* sp.	Y
Tellinidae	S	*Littoraria* sp.	N Y D
Teredinidae	S	*Mangelia stellata*	S
Teredo sp.	Y	*Melampus bidentatus*	D
		Melampus coffeus	S N D
		Melanoides tuberculata	D
		Melongena corona	S N D
		Nassarius vibex	S D
		Neritina cf.*virginea*	D
		Neritina usnea	Y D
		Pilsbryspira leucocyma	S
		Pyrgophorus sp.	Y
		Sayella fusca	S
		Stellatoma stellata	S
		Turridae	S
		UROCHORDATA (sea squirts)	
		Molgula sp.	S Y D
		ECHINODERMATA	
		OPHIURIDAE (brittle stars)	S
		Ophiuridae	S
		HOLOTHUROIDEA (sea cucumbers)	
		Leptosynapta sp.	S

[1] Creeks: N=North Silcox Creek, S=South Silcox Creek, Y=Yucca Pen Creek, D=Durden Creek

substantial sunken log habitat present at the Durden site. Although *C. virginica* was considered a characteristic taxon in Durden Creek, presence was limited to scattered mangrove roots, unlike the oyster bars observed in South Silcox.

Community Structure—To further examine species assemblages within the creeks, multivariate analysis of relative abundance data was used to illuminate natural groupings among sampling events. The distances between each of the sampling events depicted on the MDS plot (FIG. 4) are a measure of similarity or dissimilarity of community structure in terms of taxa present and

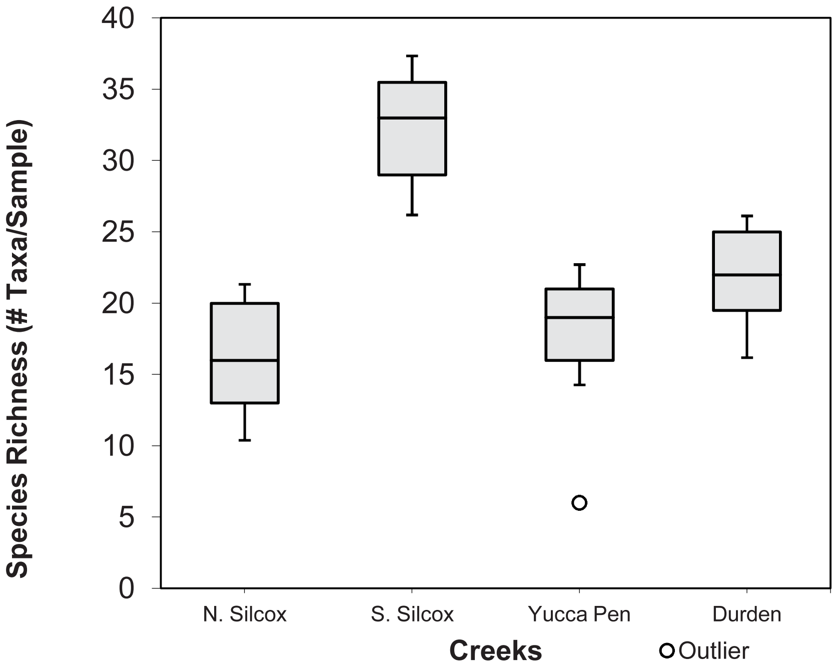

FIG. 3. Box plot of species richness (total number of taxa/sample) for each creek. Box and whisker plots show 2nd and 3rd quartiles, median, and whiskers equal to 1.5 times the interquartile range.

abundances of those taxa. The MDS plot depicts sampling events within each of the creeks grouping roughly together, indicating their similarity to each other. South Silcox Creek and North Silcox Creek showed the tightest grouping among the sampling events within each of those creeks. Durden Creek and Yucca Pen Creek also grouped roughly together. However, there is much more spread among the sampling events in each of these creeks, indicating a decreased level of similarity. The cluster analysis represents these depictions in an alternate view and demonstrates the groupings of the creeks, the levels of similarity, and the significant differences in community structure that exist among creeks as well as individual sampling events (FIG. 5). The higher level of similarity between each sample within South Silcox and North Silcox Creeks as compared to those within Yucca and Durden Creeks is apparent in the cluster dendogram.

The two sampling events showing the most distinct differences in community structure were in Durden Creek during August 2008 and Yucca Pen Creek in August 2009. The species assemblage at Durden Creek in August 2008 showed little similarity to all other sites, likely due to the absence of mobile crustaceans, specifically all amphipods, mysids and cumaceans which were present in other samples prior to and following the August 2008 sampling event. Taxa present in the August 2008 sample were predominantly attached or

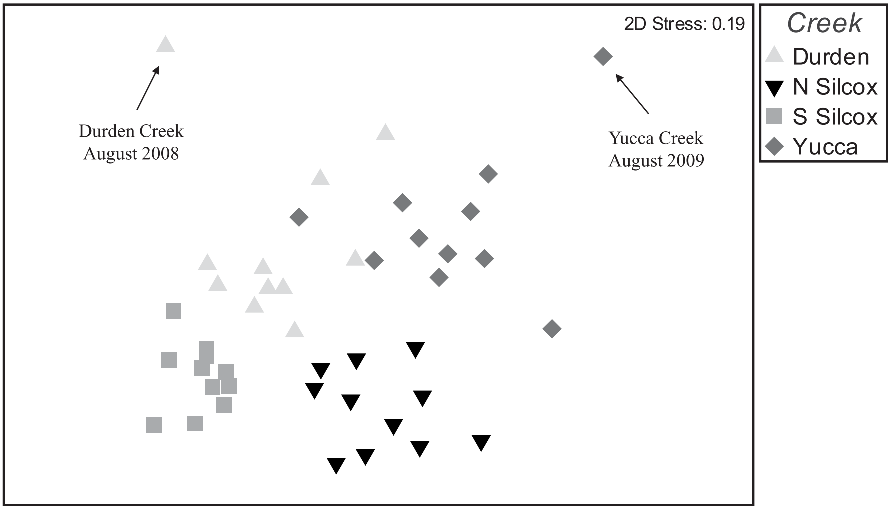

FIG. 4. Multi-dimensional scaling plot of macroinvertebrate community structure. Symbols represent creek sampling events.

burrowing organisms. Other species included a highly tolerant polychaete and freshwater tolerant insects. Similarly, the August 2009 sampling event in Yucca Pen Creek yielded a distinctly different community structure following a large rainfall event. The community structure in Yucca Pen Creek following the event lacked mobile taxa such as amphipods and isopods and was dominated by attached or freshwater tolerant taxa. Both the August 2008 and 2009 sampling events were preceded by heavy rainfall and considerable drops in salinity, although salinity at Durden Creek in August 2008 had rebounded prior to the sampling event. Distinct community shifts at the Yucca Pen and Durden sites following these high rainfall events is likely the result of displacement or mortality of amphipods and other non-burrowing organisms during periods of high freshwater inflow. Low salinity during and prior to August 2008 and 2009 may have also influenced these species shifts. However, similar community changes were not observed during other low salinity periods which may have lacked the rapid inflow or the temporal extent of low salinity associated with the high rainfall events in August of 2008 and 2009.

DISCUSSION—This study documents the benthic macroinvertebrate communities of four tidal creeks on the eastern shore of Charlotte Harbor. Each creek site had a number of taxa in common. These taxonomic similarities among creeks reflect the environmental conditions of the creeks, including species that are tolerant of low dissolved oxygen levels and species that are tolerant of a wide range of salinity conditions. Highly tolerant species have been frequently documented in similar tidal tributaries due to the wider range in environmental conditions experienced along the transition from freshwater headwaters to marine estuaries downstream (Lerberg et al., 2000). In

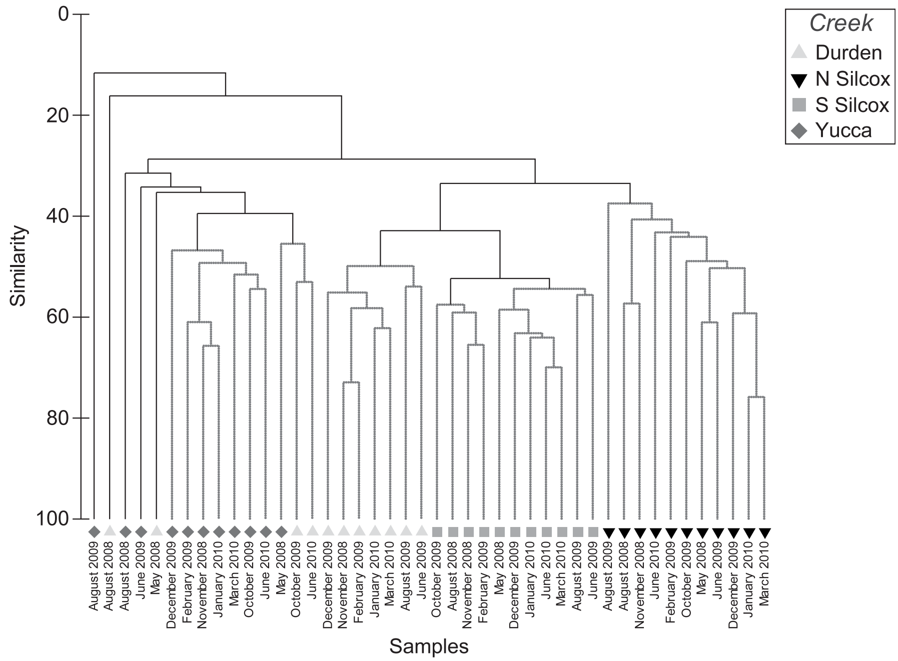

FIG. 5. Cluster analysis results of macroinvertebrate community structure. Symbols represent creek sampling events. Black lines indicate significantly distinct community structure. Clusters shown in gray are non-significant.

comparing the taxa found in this study to the taxa found by others in the region, many of the same taxa have been reported. Mote Marine Laboratory, in its benthic macroinvertebrate study of a wide variety of habitats throughout the greater Charlotte Harbor system (Mote Marine Laboratory, 2007), reported thirty-three of the fifty most abundant taxa in this study of Charlotte Harbor tidal creeks. Additionally, Sherwood et al. (2007) sampled a number of tributaries of Tampa Bay, ranging from Old Tampa Bay to freshwater sites. Although their sampling methods differed from this study, 45 of the 128 taxa they reported were also found in the Charlotte Harbor tidal creeks.

Species richness within the four creeks corresponded, in part, to benthic habitat type and diversity. The South Silcox Creek site exhibited the highest total species richness (96 total taxa), followed by the sites in Durden Creek (80), Yucca Pen Creek (59) and North Silcox Creek (47). The high species richness at the South Silcox site can be attributed to the greater diversity of benthic habitats, including habitats with high structural complexity (e.g., oysters, dead wood, barnacle clumps, and submerged aquatic vegetation (SAV)). Other researchers have also found that diversity of microhabitats is an important factor influencing the structure of macroinvertebrate communities (Johnson et al., 2003; Kon et al., 2011).

In addition, the type of taxa varied among creeks and corresponded with environmental conditions. The high abundance of euryhaline and freshwater tolerant species at the Yucca Pen Creek site can be attributed to the persistently low and variable salinity conditions. Reduced tidal circulation and corresponding depressed dissolved oxygen at the North Silcox site may explain the absence of filter feeders and abundance of deposit feeders which are tolerant of low oxygen conditions.

Differences among creeks in the temporal variability of community structure also provide insight into the influence of watershed characteristics. Variability in the structure of creek macroinvertebrate communities among within-creek sampling events reveal that the creeks with more altered watersheds are less stable over time—likely reflecting community responses to environmental stressors. Warwick and Clarke (1993) documented increases in community structure variability with increased levels of perturbation of several marine communities. Similarly, Tolley et al. (2006) found that the community structure of organisms associated with oyster reefs showed greater variability among samples at estuary stations nearest high inflow, low salinity tributaries. The results of the MDS analysis depict the North and South Silcox Creeks' macroinvertebrate communities as tightly grouped, while there is much more spread among the sampling events at the Yucca and Durden Creek sites. Similarly, the cluster analysis reveals that the level of similarity in community structure from within-creek sampling events over time is lower among sites in Yucca and Durden Creeks as compared to North and South Silcox Creeks. This increased variability in community structure indicates more frequent disturbance of the macroinvertebrate species assemblages within the creeks with more hydrologically altered watersheds.

Two notable community responses to water quality conditions further illustrate that increased variations in multivariate structure can be indicative of disturbance. Significant changes in species assemblages occurred following high inflow events in August of 2008 at Durden Creek and Yucca Pen Creek in August 2009. Reduced taxa following these events were likely the result of physical displacement or mortality during high flow, low salinity conditions.

These results emphasize the importance of both physical habitat as well as environmental conditions such as long-term salinity regimes and occurrences of large inflow events on the structure of macroinvertebrate communities. While benthic habitat diversity was a predominant factor influencing observed taxa, it is important to note that long term, antecedent hydrologic conditions are key determinants of the type and availability of benthic habitats encountered during the study. For example, oysters, submerged aquatic vegetation, and extent of leaf litter and debris which provide habitat for benthic infauna are themselves sensitive to variations in freshwater inflow, tidal flushing, water clarity, and salinity regimes (Peterson, 2000). These complex species-habitat interactions emphasize the importance of balancing both in-stream benthic habitat quality as well as longer-term environmental factors such as inflow, circulation, and salinity patterns.

Although estuarine systems including tidal creeks are inherently variable in conditions such as freshwater inflow and salinity, watershed alterations can create unnatural variability that impacts the biological community (Sklar and Browder, 1998). This effect can be seen in the fisheries data collected within these creeks (Adams et al., 2009; Dixon and Adams, 2010), which documents a higher diet diversity in predatory fish in less hydrologically impacted creeks. Similarly, this study documents a macroinvertebrate community response to large inflow events within the two creeks with more hydrologically altered watersheds.

The sampling methods for this study were designed to provide as complete an inventory of taxa as possible during each sampling event with the limited resources available. This type of methodology shares many of the same advantages as similar rapid bioassessment approaches, including the ability to maximize the number of taxa collected by sampling all available habitats. This facilitates a comprehensive account of the macroinvertebrate biota within a system without being restricted to certain habitats by sampling gear requirements (Barbour et al., 1996). Furthermore, sampling multiple benthic habitats may yield a higher rate of taxa collected compared to coring or other sediment-based macroinvertebrate collection methods (Shervette et al., 2008; Kon et al., 2011).

Building on this somewhat qualitative, yet comprehensive, inventory, future studies would benefit from a more quantitative sampling methodology, additional sampling sites along the length of the creek to gauge within creek variation, and more frequent and long term monitoring of water quality and inflow conditions. Such a framework would broaden our knowledge of the response of benthic communities to changes in water quality, available habitat, and other environmental conditions.

In conclusion, tidal creeks are uniquely positioned along the transition between uplands and the open estuary and provide critical habitat for benthic infauna and estuarine dependent species. Those benthic fauna and estuarine dependent species in turn contribute to the overall ecological importance of these systems as well as ecosystem services such as fisheries production. The impact of anthropogenic encroachment along our coasts on the ecology and ecosystem services provided by estuaries has been of increasing interest to resource managers both nationally, regionally, and locally within the Charlotte Harbor study area. The goal of this study was to supplement other research efforts in the area to support ongoing resource protection and restoration efforts by providing an inventory of macroinvertebrate communities inhabiting tidal creeks within the Charlotte Harbor study area.

Despite the limited scope of this study, we were able to detect linkages between macroinvertebrate community structure and both habitat structure and other environmental conditions. These relationships emphasize the need for resource management efforts which consider both the integrity of in-stream benthic habitats as well as factors influencing environmental conditions such as freshwater inflow and hydrologic alterations occurring within contributing watersheds. Additional monitoring including continuous and long term estimates of stream flow, runoff, and water quality conditions is critical for quantifying linkages between tidal creek fauna, watershed alterations and overall estuarine productivity.

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TEN YEARS OF THE SOUTHWEST FLORIDA FROG MONITORING NETWORK: NATURAL VARIATION AND HUMAN-DRIVEN CHANGES

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Abstract: *The Southwest Florida Frog Monitoring Network was established in 2000 to collect long-term data on frog communities of the region. Routes of 10-12 stops were monitored monthly during the rainy season (June–September). Data on all frog calls during a three minute period were recorded using a three-level intensity code. We report results from the first ten years of monitoring to examine broad trends in the frog populations of southwest Florida. We explored the abundance of all frog species, as reflected by calling intensity, to elucidate potential factors that may influence long-term changes in frog populations and communities. These factors may include: natural variations of frog populations, disappearing and altered habitats through local and global human actions, landscape context, and the impacts of invasive species. At a regional scale, it appears that most frog species are maintaining natural variations in calling levels among years, suggesting that frogs are responding to annual variation and not regional or global changes. Use of behavioral indicators, such as calling intensity of frogs, may provide understanding of the environmental implications of altered hydroperiods and other landscape perturbations in our watershed and possibly some positive responses to restoration efforts.*

Key Words: Amphibians, frog calling, frog communities, citizen science, declining amphibian populations, behavioral conservation

Amphibians have been an indicator of environmental changes beginning with the realization of declining amphibian populations in the 1980s (Collins and Storfer, 2003; Beebee and Griffiths, 2005; McCallum, 2007; Lips et al., 2008). Causes of global decreases are multifactorial and include habitat loss, human exploitation, land use change, global climate change, environmental contaminants, the introduction of non-native species, and emerging infectious diseases as well as interactions among these factors (Collins et al., 2003; Lannoo, 2005; McMenamin et al., 2008). Discovery of this global decline in amphibian populations resulted in the development of long-term monitoring initiatives to document the fate of species and communities (Dodd et al., 2007).

To ensure that as many amphibians were tracked as possible, volunteer networks were established around the country to monitor different locations, using similar protocols that would enable temporal and spatial comparisons (NAAMP, 2011; DAPTF, 2011; Frog Watch, 2011). Many of these volunteer monitoring programs have collected data on frog calls as an indicator of population trends. Frog calls are a particularly valuable gauge of population and community dynamics because the behavior is associated with reproduction. As a consequence, the presence of frog calls in an area illustrates individuals of sufficient quality to engage in reproductive displays.

Southwest Florida has experienced dramatic environmental changes over the past decades that could be expected to affect local amphibian populations. Explosive growth and development have led to habitat destruction, altered hydrology, and declining water quality, all of which might affect amphibians adversely. In addition, estimates of land use change predict an increase in urbanization by 62% between 2000 and 2025. This is expected to be accompanied by a 26.5% decrease in rangeland/upland forest and an 11.5% reduction in wetlands (SFWMD, 2008). Many of southwest Florida's waters are also listed as impaired for a variety of nutrient criteria (FDEP, 2011). Since the 1990s development regulations and best management practices have driven the construction of stormwater management ponds to retain more water across the developed landscape. However, this has still resulted in the replacement of shallow, seasonal wetlands with deeper more permanent water bodies.

In response to these potential threats to local amphibian populations, the Southwest Florida Amphibian Monitoring Network (SWF Frog Watch) was established in 2000 to document the status and population trends of local anuran species over time (Pieterson et al., 2006). To date, the network maintains a database of 10645 observations from 23 routes in Charlotte, Lee, and Collier Counties. Pieterson et al. (2006) analyzed this database after five years. They reported an increased occurrence of the exotic Cuban treefrog (*Osteopilus septentrionalis*) and an apparent shift in the anuran community toward species such as pig frogs (*Lithobates grylio*) and green treefrogs (*Hyla cinerea*). They attributed these changes to human modifications of the landscape, including the trend away from shallow, seasonal wetlands to deeper permanent stormwater retention ponds. Herein, we extend that work by analyzing more data that have accumulated, including ten total years of data. The major goal of this analysis is to explore possible changes in populations of individual frog species over our ten year study period. Specifically, we asked whether frog populations are declining in southwest Florida, and if so, which species are most concerning.

METHODS—SWF Frog Watch uses a data collection protocol similar to that used in the North American Amphibian Monitoring Program (NAAMP). Routes were established with at least ten stops, separated by at least 1 km. Stop locations along each route were established in a non-random manner to include, initially, suitable anuran habitat. All stop locations are georeferenced. Sampling occurs in the rainy season only, one night a month every June-September, and begins 15 minutes after sundown. Sampling nights are chosen arbitrarily as the third Wednesday of each month,

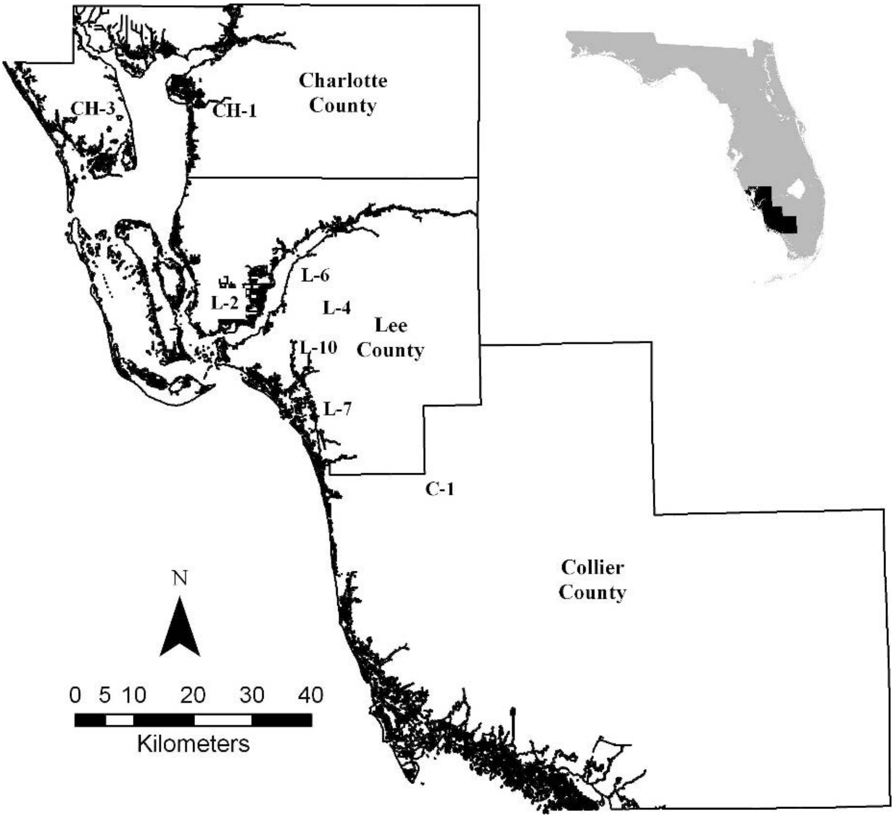

FIG. 1. Eight frog monitoring routes used in the analysis of trends over ten years, with "CH" for Charlotte County routes, "C" for Collier County routes, and "L" for Lee County routes.

which occurs randomly with respect to local weather and precipitation at each route. Volunteers listen for three minutes then record the intensity for each species' calls. Call intensities are quantified using a scale of 0–3 (0: no frogs calling; 1: individuals can be distinguished; 2: some overlap of individual calls; and 3: a chorus where individuals cannot be detected and calls are constant, continuous and overlapping).

Each route was established by a route leader, so routes and stops are not placed randomly across the landscape. In addition, sampling effort on routes varied over the ten years, with several routes 'orphaned' by leaders. For the purposes of this study, we limited the analysis to the eight most consistently monitored routes (FIG. 1), for a total of 6512 observations, including routes in all three counties within the network (Charlotte, Collier and Lee).

Calling data are summarized using *mean calling intensity*, calculated by averaging calling intensity for each species by stop or route. The percent change in mean calling intensity was compared between 2000 and 2004 and 2000 and 2009 to reexamine trends reported by Pieterson et al. (2006). Significant differences through time were tested using ANOVA on the annual mean calling intensity for each species.

RESULTS—Examination of mean calling intensity data over five year intervals (2000–2004 and 2005–2009) suggests dramatic changes in the community of anurans in southwest Florida (FIG. 2). Although most species

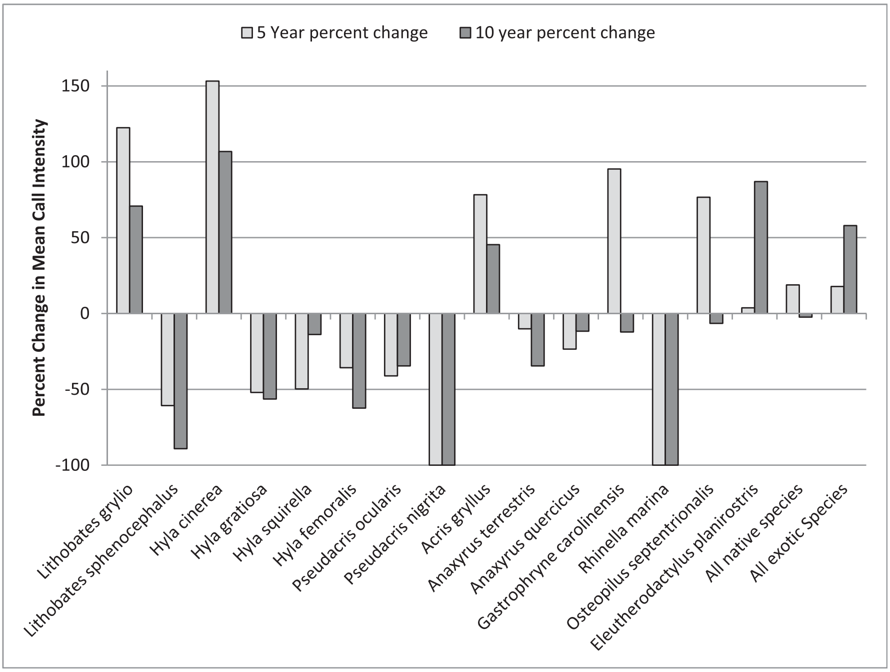

FIG. 2. Percent change in frog species mean calling intensity for the first five years, and for the total ten years, calculated by comparing mean calling intensities from year 2000 to 2004 and 2009. Nomenclature follows Collins and Taggart (2009).

exhibited a similar change in the five-year and ten-year calling intensity trends, several clear patterns exist. No native species with a positive increase after five years showed an even greater increase over the full ten years. The only species that showed consecutive increases in mean calling intensity over both the five- and ten-year intervals was the exotic greenhouse frog (*Eleutherodactylus planirostris*). When data for all native species were combined, there was an overall slight decrease over the ten years, reversing the slight increase that was observed after five years. The overall mean calling intensity for exotics increased after both five years, driven principally by the increase in Cuban treefrogs (*O. septentrionalis*), and after ten years because of the increase for *E. planirostris*. In contrast, the giant toad (*Rhinella marina*) showed a dramatic reduction in both the five- and ten-year interval. *Osteopilus septentrionalis* was one of two species that reversed its percent change from five years (positive) to ten years (negative). The other was the native eastern narrowmouth toad (*Gastrophryne carolinensis*), which showed the same trend.

While examination of mean call intensity data after five- and ten-year intervals is compelling, the dramatic negative changes in anuran populations and communities are no longer evident when annual mean calling intensity data are tracked among consecutive years. Southern cricket frog (*Acris gryllus*), pinewoods treefrog (*Hyla femoralis*), green treefrog (*H. cinerea*), and pig frog

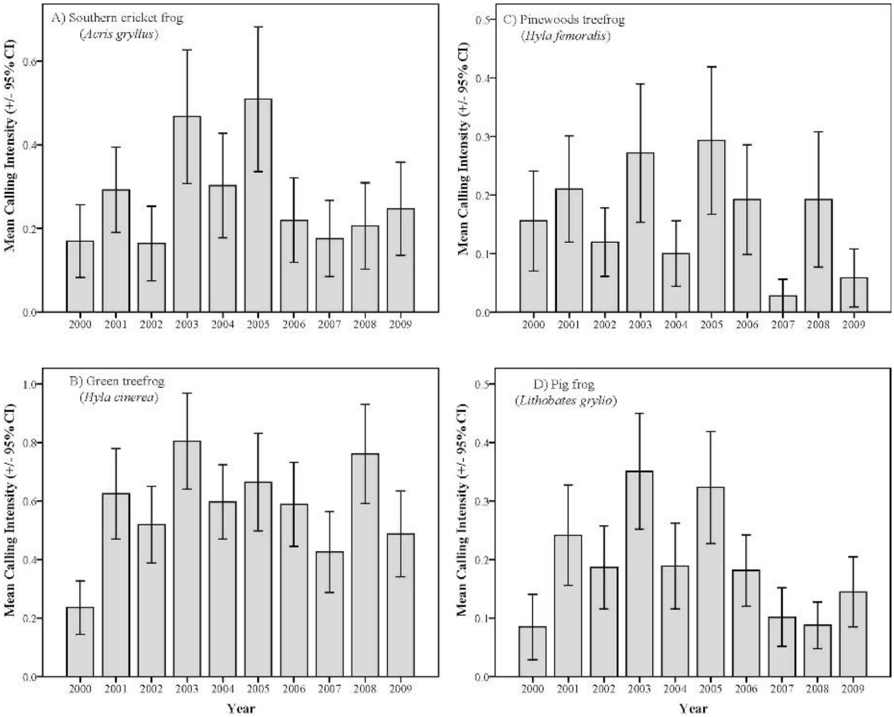

FIG. 3. Annual mean calling intensities for: A. Southern cricket frog; B. Pinewoods treefrog; C. Green treefrog; D. Pig frog. Error bars represent the 95% confidence interval.

(*L. grylio*) all showed similar trends in calling intensity among years (FIG. 3). These four species have high mean calling intensities around 2003 and 2005. In addition, these four species all showed lower mean calling intensity in 2000, 2002, 2004, and again in 2007. In particular, *H. femoralis* was one of the species to show an apparent dramatic decline after both five- and ten-year intervals but had high calling intensity in 2005, 2006, and 2008. Mean calling intensity values varied significantly among years, which is consistent with the underlying variation of amphibian populations (*A. gryllus*: $F_{9,839} = 4.1$, $p < 0.001$; *H. femoralis*: $F_{9,839} = 1.9$, $p = 0.046$; *H. cinerea*: $F_{9,839} = 4.2$, $p < 0.001$; *L. grylio*: $F_{9,839} = 6.3$, $p < 0.001$). The mean calling intensity scale (y-axis) was adjusted for each species to better emphasize annual variation (FIGS. 3–6). Among all species, the natives *A. gryllus*, *H. cinerea*, and *Anaxyrus quercicus* had the highest mean calling intensities and were all greater than any of the exotic species.

The annual mean calling intensities for eastern narrowmouth toad (*G. carolinensis*), squirrel treefrog (*Hyla squirella*), oak toad (*A. quercicus*), and southern toad (*Anaxyrus terrestris*) also showed similar patterns of annual change (FIG. 4). All species exhibited peaks in mean calling intensity in 2002 and 2008 (though *G. carolinensis* also had high mean calling intensities in 2004

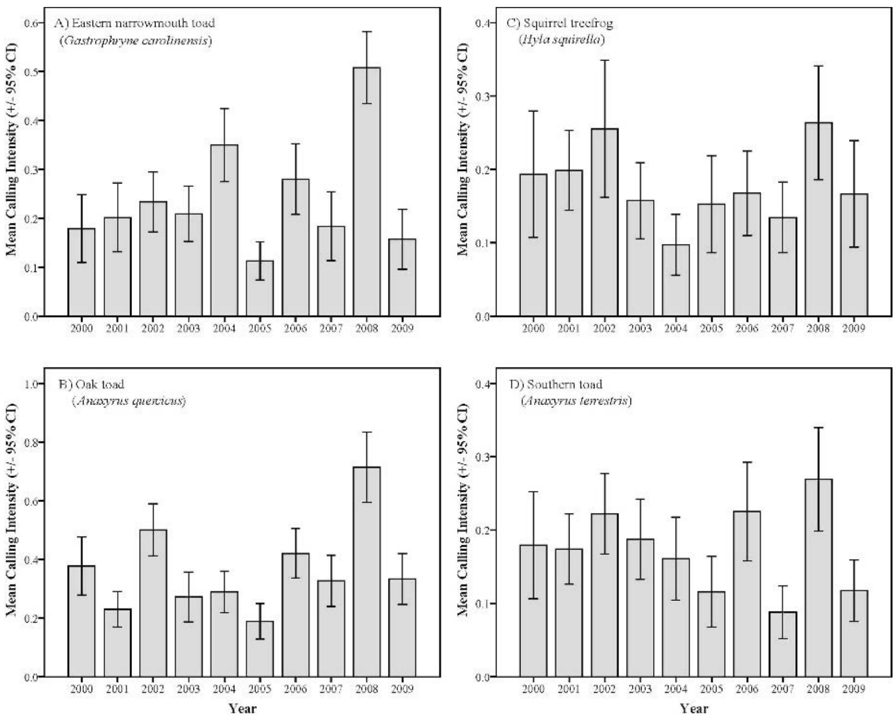

FIG. 4. Annual mean calling intensity for: A. Eastern narrowmouth toad; B. Squirrel treefrog; C. Oak toad; D. Southern toad. Error bars represent the 95% confidence interval.

and 2006). In addition, all four species showed a decline in mean calling intensity in either 2004 or 2005 and 2007. Of particular interest, *H. squirella* and *A. quercicus* both showed dramatic declines when data were examined after five- and ten-year intervals, but showed comparable increases in 2002, 2006, 2008. All four species have significantly different mean calling intensities among years (*G. carolinensis*: $F_{9,839} = 12.1$, $p < 0.001$; *H. squirella*: $F_{9,839} = 3.9$, $p < 0.001$; *A. quercicus*: $F_{9,839} = 13.0$, $p < 0.001$; and *A. terrestris*: $F_{9,839} = 4.2$, $p < 0.001$).

Some frogs were either observed rarely or showed unique patterns of call intensity data among years that were not comparable to other species (FIG. 5). The little grass frog (*Pseudacris ocularis*) showed peaks in 2001 and 2008, but apparent disappearance in 2005–2006. Both the southern leopard frog (*L. sphenocephalus*) and southern chorus frog (*Pseudacris nigrita*) may have shown declines as indicated by mean calling intensity. In fact, *P. nigrita* disappeared for 4 years from 2003–2006 as well as 2008–2009, while *L. sphenocephalus* was not heard from 2005 before returning in low activity in 2006. In each case, these three species were rarely observed over the 10 years of sampling. *Pseudacris ocularis* was documented only 29 times, *L. sphenocephalus* only 25 times, and

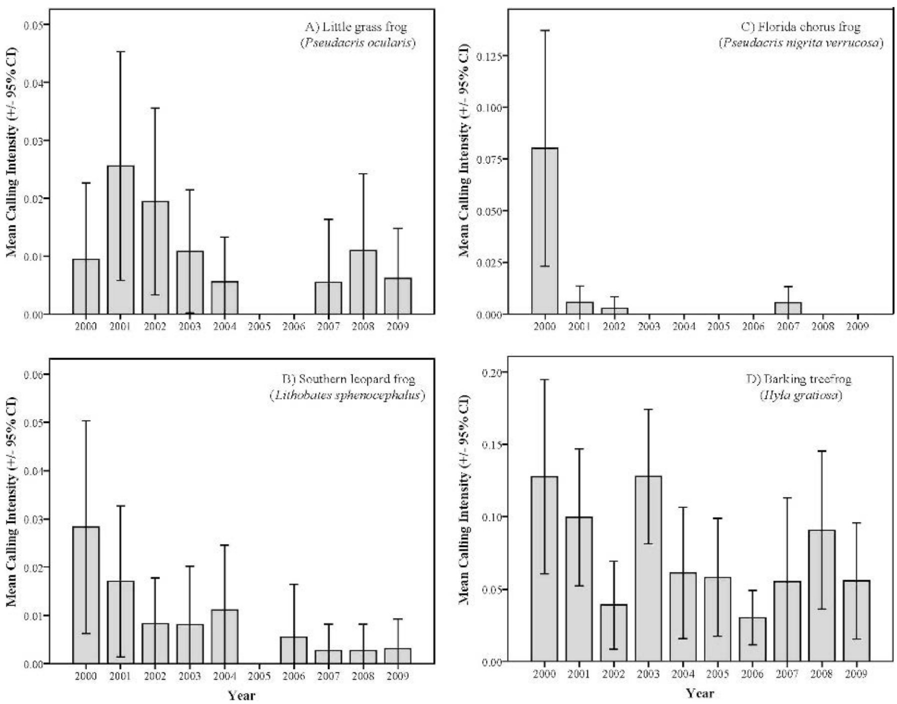

FIG. 5. Annual mean calling intensity for four of the rarer frog species: A. Little grass frog; B. Southern leopard frog; C. Florida chorus frog; D. Barking treefrog. Error bars represent the 95% confidence interval.

P. nigrita only 14 times. Barking treefrogs (*Hyla gratiosa*) showed a unique pattern of mean calling intensity over the ten-year study. *Hyla gratiosa* showed higher calling intensity in 2000, 2001, 2004, 2008 and lower calling intensity in 2002 and 2006. In addition, the overall mean calling intensity was particularly low even though these animals were heard 151 times over the study. All four species had significantly different mean calling intensities among years (*P. ocularis*: $F_{9,839} = 1.9$, $p = 0.046$; *L. sphenocephalus*: $F_{9,839} = 1.9$, $p = 0.048$; *P. nigrita*: $F_{9,839} = 11.5$, $p < 0.001$; *H. gratiosa*: $F_{9,839} = 2.2$, $p = 0.02$).

Mean calling intensity data for the exotic frog species, greenhouse frog (*E. planirostris*), cane toad (*R. marina*), and Cuban treefrog (*O. septentrionalis*) all show distinct differences in annual patterns (FIG. 6). *Eleutherodactylus planirostris* does not follow either of the annual patterns exhibited by native frogs in FIGS. 3 and 4. In addition, *E. planirostris* was the one species that did not show significant differences in mean calling intensity among years ($F_{9,989}=1.2$, $p = 0.1$). *Rhinella marina* was rarely heard (only 11 times over 10 years) and showed a spotty record of presence in some years but not in others. The sample size was too small to analyze in depth. Annual mean calling intensity for *O. septentrionalis* was significantly different ($F_{9,989}=1.3$,

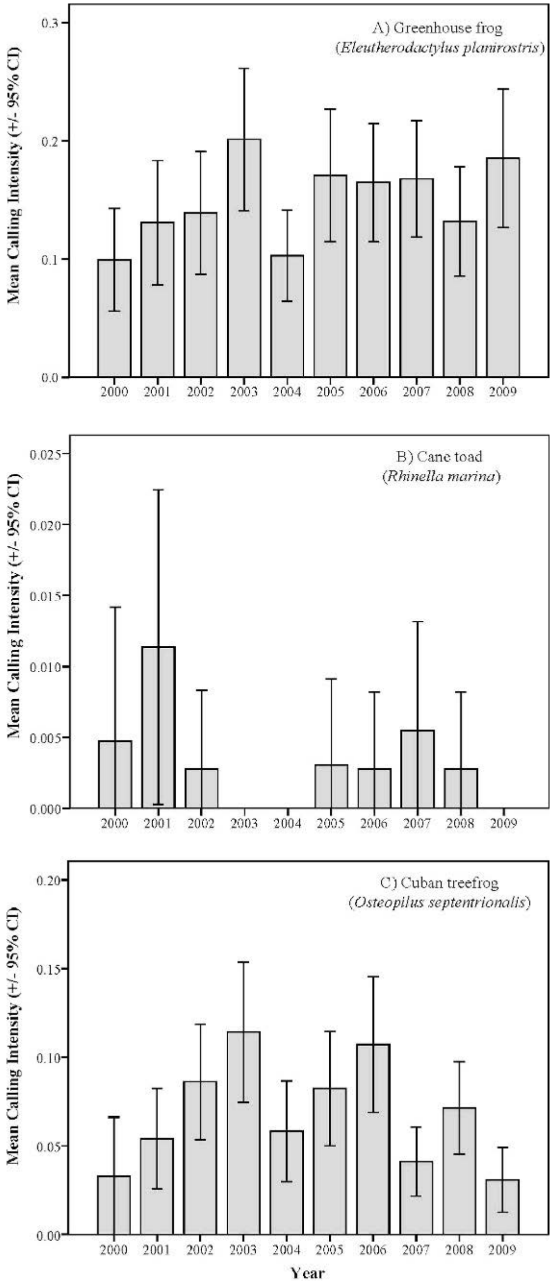

FIG. 6. Annual mean calling intensity for exotic species: A. Greenhouse frog; B. Cane toad; C. Cuban treefrog. Error bars represent the 95% confidence interval.

$p < 0.001$) among years and followed an overall pattern similar to native species in FIG. 4, with peaks in 2003, 2006, and 2008 and drops in 2004 and 2007.

DISCUSSION—The five- and ten-year percent changes in mean calling intensity documented a slight overall decline in native frog species mean calling intensity and an increase in calling of exotics. In addition, all but three native species had a negative percent change in mean calling intensity from the first to last year in the sample. The three that showed a positive percent change all had a lower percent increase after ten years compared to five years. Both the native *P. nigrita* and the exotic *R. marina* were not heard at all in the last year of sampling. In addition, the gopher frog (*Lithobates capito aesopus*) was never heard on the eight routes analyzed in this study, though they have been heard sporadically (<10 records) in the monitoring program. All three may have been extirpated from the landscape, but they may exist in areas not sampled in the monitoring network or at times outside the sampling procedure. The more resolved annual mean calling data showed an apparent downward trend for *P. nigrita* and three additional native species (*P. ocularis*, *L. sphenocephalus*, and *H. gratiosa*). With this preliminary analysis, focusing on changes in five-year periods, there appeared to be sufficient data to conclude an overall reduction of natives in the frog communities. Although some native frog species may indeed be in decline, this interpretation would be premature, or inappropriate, without more detailed study of annual call variation, habitat changes along routes, and effects of extreme environmental events (see below).

The punctuated analysis of percent change from first to fifth, and first to tenth years sampled, misrepresents trends for many of the species. For example, if we had reported on the nine-year trends (2000–2008) nine of the twelve native species would have had a *positive* percent change in mean calling intensity. Interpreting long-term anuran calling data must consider: 1) the underlying variation of anuran populations; 2) the probability of missing calling events when calling is sampled once per month; and 3) that maintenance of a population may only require one successful reproductive event in a year. The clearest signal from these data is the need to continue monitoring to differentiate background variations from long-term trends and to incorporate all annual data, instead of punctuated analyses which might misrepresent trends (Pechmann et al., 1991).

The annual variations in FIGS. 3 and 4 (and for *O. septentrionalis,* FIG. 6) may be driven by extreme wet and dry years, and particularly in the case of *O. septentrionalis* extreme cold spells, possibly leading to mortality of this tropical species. It is important to recognize that these monitoring data are not direct measures of abundance, but of calling. Changes in calling frequency and intensity are behavioral changes. During dry years, calling is expected to be reduced, even if there is no mortality associated with the drought. Wet years may stimulate reproduction, raising mean calling intensity, and this effect may carry over to the following year, with increased reproduction leading to population increases and continued high calling intensity.

Additionally, the altered hydrology associated with human development has increased the detrimental effects of drought. Development has also decreased and fragmented native habitats, and exotic plant invasions have decreased the quality of the remaining undeveloped lands. Development in southwest Florida is typically related to construction of new roads that eventually bring higher traffic levels, which has an important impact on this monitoring program limiting our ability to detect calling frogs in noisy environments. Moreover, global climate change scenarios suggest increased frequency of extreme weather events, potentially increasing the likelihood of extirpation of already reduced and stressed populations. However, some regional changes may be maintaining, or facilitating the recovery, of native frog populations. Changes in stormwater best management practices, wetland restoration, public land acquisition programs, and increased efforts to manage and control invasive exotics may all be decreasing rates of habitat loss and improving habitat in some areas.

Extreme weather events, (tropical storms and other intense rainfalls, drought, cold fronts) varied across the region over the decade of study. The impact of individual events may be obscured by our regional analyses that combined all eight routes. To better determine environmental factors that may lead to a decline, or facilitate an increase, in native frog populations, there is a need to examine changes in calling at a site-specific scale and to tie these changes to habitat characteristics. The habitat surrounding stop locations has changed in complex, site-specific ways over the period of monitoring. The behavioral response (calling) of each individual species must be interpreted in light of its unique life history requirements, in relation to these habitat changes. It is also warranted to examine biotic community interactions, particularly the role of exotics, in maintaining, or reducing, frog biodiversity. We see the opportunity to identify and categorize specific stops in terms of the degree of habitat modification through time and analyze the differential impacts on varying species.

Our citizen science monitoring effort is providing long-term data to examine overall regional trends. There is the potential for additional analyses, outside the scope of this paper, to examine more species- or site-specific patterns. The limitations of this type of monitoring effort include the tradeoff between covering larger areas, but with less intense sampling that is not tied to optimal calling conditions. Inevitably, this requires longer time periods to confirm population changes. In addition, many of the sampling locations are adjacent to private lands, with no ability to control habitat changes through time. The establishment of new routes, or networks, might include an effort to sample public conservation areas, which presumably can provide reference sites for detecting change in the anuran populations in areas undergoing habitat modification. Additionally, a citizen science monitoring network can serve as a heuristic, with the preliminary trends serving to focus additional research effort to target areas, habitats, or species within the region.

Ultimately, the frog community is a sentinel for determining the effectiveness of our efforts to maintain wetland function and water quality.

Their loss would be a measure of our failure. Implementation and continued support of citizen-science programs such as the Southwest Florida Amphibian Monitoring Network will be critical to prevent this failure.

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Biological Sciences

FRESHWATER FISH COMMUNITIES AND HABITAT USE IN THE PEACE RIVER, FLORIDA

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Abstract: *Fish communities in the Peace River were assessed biannually via electrofishing from fall 2007 through spring 2010. Habitat characteristics were quantified for each electrofishing transect to allow comparison with fish community data. We determined fish population characteristics, calculated habitat suitability indices and curves, and compared fish communities across river section, season, year, and with physicochemical parameters. Fish communities differed in each section of the river (i.e., lower, middle, upper) but did not differ across seasons or years. The strongest correlations of changes in community structure with physicochemical variables and habitat metrics occurred for measures of habitat complexity, followed by water velocity and conductivity. Species-specific habitat affinities were found to overlap substantially, but some differences were apparent and could explain broad-scale distributional differences in habitat use among the species. In conclusion, the sections used to delineate the river in this study, based primarily on differences in geology and hydrology along the course of the river, were comprised of different fish communities. The habitat affinities of fishes identified in this study should be useful to resource managers for modeling biotic responses to changes in river water levels and habitat availability.*

Key Words: Electrofishing, minimum flows and levels, Peace River, Charlotte Harbor, Florida

Florida coastal rivers have undergone significant degradation over the past 100 years from surface and groundwater withdrawal, conversion to agriculture, nutrient loading from agriculture, urban development, and phosphate mining (PBS&J, 2007). Correspondingly, the State, as required by law (Subsection 373.042(2), Florida Statutes), has developed, implemented, and will continue to revise minimum flows and levels (MFLs) to ensure adequate water supplies and prevent significant harm to ecosystems from consumptive and non-consumptive uses. Given the dramatic increases in human population growth and subsequent demand for freshwater resources that are projected in Florida, it is critical that protective measures are timely, conservative, and fully address both hydrologic and biotic considerations.

The Peace River, located in the rapidly developing region of SW Florida, has experienced natural (e.g., drought and hurricanes) and anthropogenic (e.g., phosphate mining and urbanization) disturbances over the last 150 years

(SWFWMD, 2002). However, the river flows unobstructed (e.g., no water control structures) from its headwaters and still contains significant flood plains, wetlands, and forested corridors that provide critical habitat for many wildlife species and nurseries for juvenile fish (DEP, 2007). As part of the State's MFL process, the Peace River was divided into three distinct sections based primarily on differences in geology and hydrology. Because significant differences in physical characteristics occur among these sections, it follows that biotic communities should also differ. However, for fish communities, this assumption has never been explicitly tested. Comparing habitat utilization by fishes to habitat availability in each of the river sections can identify thresholds and ultimately predict population and community level responses to changes in flow (Warren et al., 2008). Although general surveys of fish communities are available for fixed stations along the Peace River (Champeau, 1990; Champeau et al., 2009), new approaches are needed to sample riverine habitats and to evaluate affinities of the dominant fish species with quantified habitat, the results of which will be useful to water managers for modeling biotic responses to changes in water levels and habitat availability. Our objectives were to 1) determine fish community metrics (abundance, diversity, richness, composition) in the freshwater portions of the Peace River using stratified-random surveys, 2) identify any differences in fish communities among different sections of the river, and 3) evaluate fish species association with quantified habitat.

MATERIALS AND METHODS—*Study site*—The Peace River (~187 km) is a tannin stained system with its headwaters stemming from the confluence of Saddle Creek and the Peace Creek Drainage Canal in central Polk County. The Peace River watershed encompasses over 5,959 km^2 (2,300 mi^2) and flows southward through four counties. Along its course, the river is fed by several tributaries and drops over 70 meters in elevation from the headwaters to discharge in the Charlotte Harbor Estuary (DEP, 2007). The Peace River was divided into three distinct sections (SWFWMD, 2002) defined as upper (Bartow to Zolfo Springs), middle (Zolfo Springs to just above Nocatee), and lower (Nocatee to Charlotte Harbor; FIG. 1).

The upper Peace River section begins in Polk County just south of Lake Hancock near State Road 60 and flows south to Zolfo Springs, Florida. The terrain and geology of the upper Peace River is of karst origins that form direct connection between the river and the Floridan Aquifer, resulting in total loss of surface water flows during drought conditions. Flow rates in the upper portion of the Peace River are the most variable of the three river sections, due to groundwater withdrawal and the cessation of supplemental base flows. Additionally, the change in composition and quality of upland habitats, urbanization, agriculture, and flood control has adversely modified the upper Peace River system (PBS&J, 2007). A prolonged drought during the study period affected the upper Peace River resulting in extended periods of dry flood plains, exposed sand bars and limestone shelves, and variable flow rates.

The middle Peace River (Zolfo Springs to Arcadia) narrows more frequently than the other sections of the river, contains tree-lined bluffs and has a substrate that originates primarily from upland sandy soils. The middle Peace River basin is the least developed section of the river. Furthermore, the middle Peace River receives more inflow from tributaries than the rest of the river, resulting in a more consistent hydrological environment. Although agriculture accounts for the largest land use (56%) in the middle section, over 40% of the basin still remains as native upland habitat and wetlands. Large decreases in native upland habitats due to conversion to agriculture (e.g., improved pasture, 46% of basin) may have the greatest effect on this section of the river (PBS&J, 2007). Increases in agricultural ground water discharges within the middle basin as well as upstream may have contributed to long term increases in pH, total alkalinity, sodium, and chloride concentrations (PBS&J, 2007).

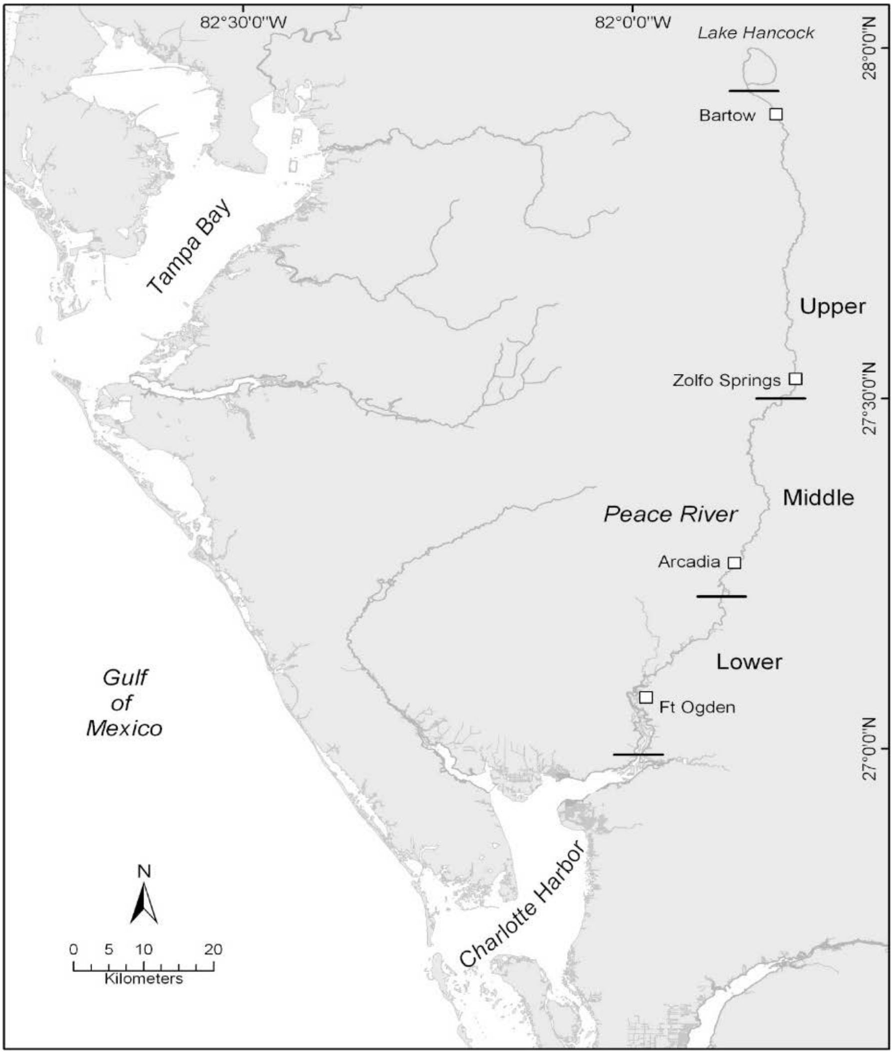

FIG. 1. The upper, middle and lower sections of the Peace River as delineated by the Southwest Florida Water Management District.

The lower section of the Peace River begins in Arcadia below State Road 70 and flows through Hardee, DeSoto and Charlotte counties. The lower section contains a transition area from freshwater to an estuarine system. The increased biological productivity (Odum, 1971; Norris et al., 2010) and additional cover provided by macrophytes make these areas ideal nursery habitats for both freshwater and marine fish species (Gunter, 1967; Houde and Rutherford, 1993). However, there are increased risks associated with physiological challenges resulting from fluctuating salinity (Norris et al., 2010). Due to the proximity of the cities of Port Charlotte and Punta Gorda to the mouth of the Peace River, the lower section of the river has had substantial urban land use. The Peace River Water Treatment Facility is located about 30 km from the mouth of the river, and surface water withdrawals provide 400,000 people in three counties with drinking water (PBS&J, 2007; DEP, 2007).

TABLE 1. The number of transects sampled biannually from fall 2007 to spring 2010 for each river section (upper, middle, and lower) and each sampling type (standard and low water).

	Number of Transect			
Sample Period	Upper	Middle	Lower	Sampling Type
Fall 2007	4	35	39	Low Water Sampling
Spring 2008	22	22	38	Low Water Sampling
Fall 2008	39	36	31	Standard Sampling
Spring 2009	21	21	43	Standard Sampling
Fall 2009	30	42	29	Standard Sampling
Spring 2010	20	40	35	Standard Sampling

The fish communities of the Peace River were affected by the passage of hurricanes during the past decade, but the communities have recovered sufficiently to address the objectives of this study. The first intensive fish surveys of the Peace River were conducted 1983–1992 (Champeau, 1990). Fish sampling in 2005 and 2006, following a low-dissolved oxygen event associated with the passage of consecutive hurricanes over the region, found that fish community structure changed substantially from that of the earlier period (Champeau et al., 2009). Despite these changes, fish community metrics such as species richness and diversity remained stable. A recent update repeating the methods of Champeau et al. (2009) found that fish community structure in 2009 had again become similar to that of the historic period (Appendix A).

Field and laboratory methodology—Sampling was conducted biannually in the fall (September through December) and spring (February through May) from 2007 through 2010 (TABLE 1) to determine if temporal trends in fish community structure correlated with habitat utilization. ArcView Geographic Information System (GIS) software (ESRI, 1999–2009) was used to create a centerline for the entire Peace River. Potential sampling points were located every 0.1 km along the centerline, and random sites for sampling were selected using these points at an intensity of three sites for every four river kilometers (km). Shoreline sampling was also randomized (e.g., east or west bank of the river).

An alternative sampling strategy was developed for low water conditions. Random sampling in the middle and upper sections of the river was at times not possible because of many shallow, exposed shoals and limestone riffles that obstructed boat travel. At boat ramps along the river, transects were completed covering small stretches of river that were representative of river stretches in the vicinity. Three sites were randomly selected per one river kilometer during low water periods.

A 5.5 m aluminum electrofishing boat equipped with a Smith-Root^{TM} 9.0 GPP or a 4.9 m aluminum electrofishing boat equipped with a Smith-Root^{TM} 5.0 GPP was used for all sampling events and locations. Boat type and electrofishing unit selection were determined based on water depth, water flow, conductivity, and the accessibility of sampling areas. A new sampling protocol was developed to compare fish assemblages and habitat utilization at a microscale (e.g., 8 m^2) versus large scale sampling (e.g., 100 m or greater). Conductivity and temperature were measured to standardize the electrical power needed to transfer required wattage and amperage levels for effective electrofishing (Bonvechio, 2009). Each electrofishing transect was marked with two floating buoys placed approximately 4 m apart and 2 m from the shoreline prior to sampling. The 4×2 m transect was sampled using a three pass depletion method collecting all fish. Fish were identified to lowest possible taxa, measured in total length (mm), weighed (g), and returned to the water. Smaller fish (<100 mm) collected in large numbers were measured in one centimeter groups and batch weighed. Any unidentifiable species were placed on ice and returned to the laboratory for identification. Species richness, diversity (Shannon index), Pielou's evenness index, and electrofishing catch-per-unit-distance were determined for each section of the river by season and year.

Microhabitat measurements of structural and water quality parameters were recorded within each 4×2m transect subsequent to fish sampling. Microhabitat structural measurements included counts of woody debris, aquatic macrophyte coverage, and substrate type. Woody debris (WD) (e.g., snags, root wads, cypress knees) was counted according to size class (debris diameter <2.5 cm, 2.5–9.9 cm, 10–25 cm, and >25 cm). Up to twenty pieces of WD were counted for each size-class, and if counts exceeded 20, a range class was used (21–50 pieces, 51–100 pieces, and >100 pieces). Aquatic macrophyte coverage was recorded as a percentage of the total transect covered, and all plants were identified to the lowest possible taxon. Plants were grouped by category (submersed, floating, or emergent). Substrate type was categorized generally (e.g., limestone, mud, sand, and detritus). Water quality parameters included temperature (°C), salinity (ppt), conductivity (μS/cm), and dissolved oxygen (DO; mg/l) and were measured using a Yellow Springs Instrument YSI Model 556™ at the surface and on the bottom if water depth was >1 m. In addition, water velocity (m/s) was measured within each transect at a depth of 0.5 m using a Marsh-McBirney Model 2000 Flo-Mate™, turbidity (ntu) was measured using a WQ770 Turbidimeter at the surface and bottom, and Secchi depth (m) was recorded.

Development of Habitat Complexity Index—A Habitat Complexity Index (HCI) equation was developed to determine species association with habitat type and complexity. An HCI similar to the one described by Dutterer and Allen (2008) was created for each transect of the Peace River. The HCI includes a macrophyte category, a woody debris (WD) category, average depth, and weighted habitat score. The macrophyte and WD categories represent the midpoint of a range of percent coverages (i.e., 0–20% = 10, 21–40% = 30.5, 41–60% = 50.5, 61–80% = 70.5, and 81–100% = 90.5). Using categories helped to eliminate observer bias by allowing for some disagreement between observers. The habitat metrics for the entire river HCI were standardized as a proportion of the maximum value for each parameter across the entire river by year and season. The standardized metrics were shared to create the HCI by river section, year, and season for each transect as:

$$\mathrm{HCI} = \mathrm{AvgD}/\mathrm{Max\ AvgD} * \mathrm{MM}/\mathrm{Max\ MM} * \mathrm{WDM}/\mathrm{Max\ WDM} * \mathrm{HS}/\mathrm{Max\ Hs}$$

where AvgD = average depth, MM = macrophyte midpoint, WDM = woody debris midpoint, HS = weighted habitat score, where HS = $\sum WD\ x\ SD$, where WD is the number of pieces in each size category and SD is the size of the pieces. The size of the pieces was standardized using the midpoint of specific size ranges (i.e., <2.54 cm = 2.54, 2.54–9.9 cm = 7.62, 10–25.3 cm = 17.78, and >25.4 cm = 25.4). The denominators signify the maximum values across each section of river by year and season. The HCI values range from 0, being the least complex habitat, to 1, being the most complex habitat.

Community analyses—Spatial patterns in fish community structure were analyzed using multivariate techniques (PRIMER v. 6; PRIMER-E Ltd., Plymouth, UK). Sample abundance indices for each species (fish per transect) were square-root transformed to reduce the influence of highly abundant species. After calculating Bray-Curtis similarity (Bray and Curtis, 1957) matrices on data averaged by sampling event and river section, non-metric multidimensional scaling (MDS; Clarke and Warwick, 2001) was used to determine if fish community structure varied among river sections. In addition, a two-way analysis of similarity (ANOSIM; Clarke and Warwick, 2001) was used to compare fish communities by sampling event and river section. Similarity percentage analysis (SIMPER) was used to identify species representative of dissimilarities between river sections. Lastly, correlations between fish community change and environmental variables (up to five habitat measures) over the study period were assessed using the BIO-ENV routine (Clarke and Warwick, 2001) for each river section.

Species habitat affinities—Habitat Suitability Curves (HSC) were constructed for species contributing to differences in fish community structure within the Peace River and also for any exotic species for which habitat use information is lacking. The HSCs for each species were constructed to characterize patterns of habitat selection along seven gradients: woody debris percentage, macrophyte percentage, depth, conductivity, water velocity, habitat complexity index, and weighted habitat scores. Habitat suitability curves constructed for largemouth bass *Micropterus salmoides*, bluegill sunfish *Lepomis macrochirus*, redear sunfish *Lepomis microlophus*, and spotted sunfish *Lepomis punctatus* were segregated according to juvenile and adult life stage based on length frequencies for each sampling event. Suitability curves allow for an examination of resource use that accounts for non-uniform sampling across gradients of environmental variability and provide valuable information regarding patterns of habitat selection across species and life stages (Aadland, 1993; Dutterer and Allen, 2008). For each univariate habitat suitability analysis, environmental data were subdivided into equal intervals; interval ranges for each variable were chosen to produce the smoothest habitat suitability curves possible. For each interval, habitat suitability values were then calculated as:

$$S = P(EF)/P(E)$$

where P(EF) represented the proportion of samples within which a species occurred (resource used) and that fell within a specific environmental interval, and P(E) represented the proportion of all samples collected, regardless of whether a given species occurred (resource availability), that fell within the same environmental interval (Baltz et al., 1990). Suitability values for each analysis were then standardized by dividing by the greatest observed suitability so that values ranged from zero (intolerable) to one (optimal) (Switzer et al., 2009). Data from the spring of 2010 was excluded from consideration for blue tilapia *Oreochromis aureus* and armored catfish *Pterygoplichthys* spp. due to zero catch rates, resulting from record cold fish kills.

Principal component analysis (PCA) was used to help visualize differing habitat affinities among species contributing to differences in fish community structure. A PCA was conducted that weighed fish relative abundance against seven correlated environmental covariates: river location (upstream to downstream, based on latitude), river flow, water depth, conductivity, percent woody debris, percent macrophyte cover, and HCI. The seven covariates were separated into orthogonal components based on the correlation matrix. PCA was conducted using the Factor procedure and SAS software (SAS, 2002), and principal components were rotated using the varimax option to facilitate the interpretability of each respective component. Variable loadings and principal-component scores were calculated independently for each sample.

RESULTS—*Community analyses*—A total of 545 transects were completed during the 6 seasonal sampling events, ranging from 77 to 105 transects per event, from fall 2007 to spring 2010 (TABLE 2). Community metrics for each sampling event ranged from 24 to 39 for species richness, 2.31 to 2.97 for species diversity, and 0.72 to 0.81 for evenness. Fish community structure associated with seasonal sampling events differed spatially but not temporally in the Peace River (FIG. 2). The results of the two-way ANOSIM support the relationship stated above: fish communities differed by river section ($p = 0.05$, $R = 0.42$; a moderate value indicating that fish communities were fairly well separated across river sections) but not by year. One-way ANOSIM by season was not significant.

The species distinguishing differences in fish communities between river sections were spotted sunfish *Lepomis punctatus*, Seminole killifish *Fundulus seminolis*, bluegill *Lepomis macrochirus*, eastern mosquitofish *Gambusia holbrooki*, shiners *Notropis* spp., largemouth bass *Micropterus salmoides*, and redear sunfish *Lepomis microlophus* (FIG. 3). Each of these species contributed >5% to the total average dissimilarity in river section comparisons using

TABLE 2. The Peace River sampling summary from fall 2007 through spring 2010. Samples equal the number of electrofishing transects completed. Richness equals the total number of species, diversity equals the Shannon diversity index, and evenness equals the species evenness related to diversity. Data for river sections was pooled by sampling event to compare with historical sampling.

Year	Season	Samples	Total Fish	Richness	Diversity	Evenness
2007	Fall	78	1076	35	2.86	0.80
2008	Spring	82	1984	39	2.97	0.81
2008	Fall	106	1527	34	2.57	0.73
2009	Spring	85	1529	37	2.73	0.76
2009	Fall	101	1072	30	2.45	0.72
2010	Spring	95	1440	24	2.31	0.73

SIMPER. Spotted sunfish, shiners, and largemouth bass generally increased in abundance from the lower to upper river. Seminole killifish, bluegill, and redear sunfish generally decreased in abundance from the lower to upper river. Eastern mosquitofish abundance was lowest in the middle section of the river.

Correlations of changes in fish community structure and sampling events with changes in habitat characteristics for each river section were moderate to high (BIO-ENV ρ = 0.333–0.820 (values close to 1 indicate strong relationships; TABLE 3). The strongest correlations occurred for the lower and middle sections ($\rho > 0.745$). In the upper section, macrophyte cover and water velocity best correlated with changes in fish community structure. In the

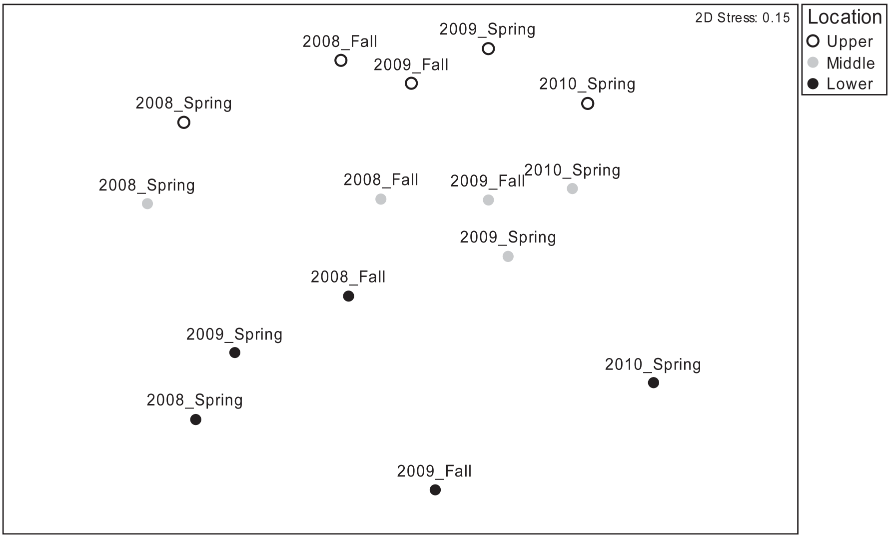

FIG. 2. Two-dimensional nonmetric scaling (MDS) ordination of fish community structure captured in electrofishing samples from three sections of the Peace River, Florida. Each point represents a sampling event (year season) in each of the river sections.

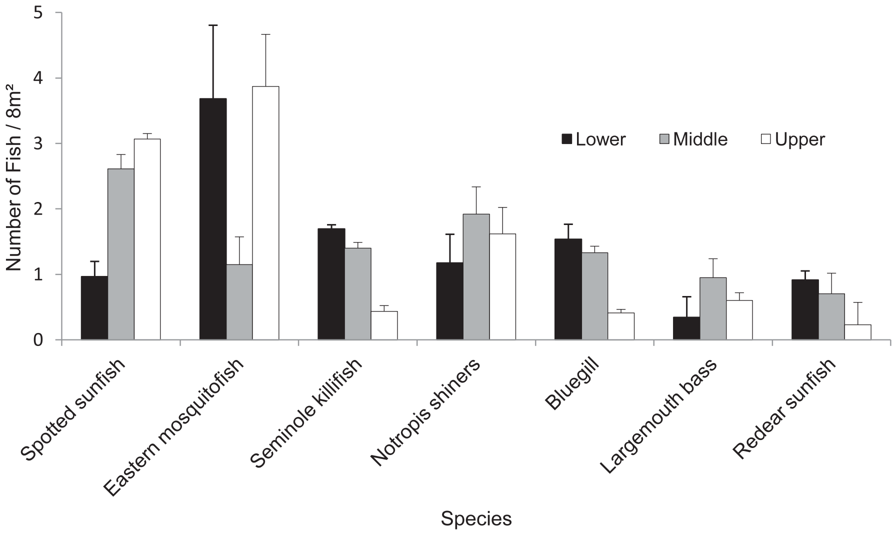

FIG. 3. Species distinguishing differences in fish communities among the lower, middle, and upper river sections of the Peace River, Florida. The species shown are those that contributed >5% to the total average dissimilarity in section comparisons using similarity percentage analysis (SIMPER).

middle section, four variables best correlated with changes in fish community structure: HCI, woody debris, depth, and water velocity. In the lower section, woody debris was correlated best with changes in fish community structure among sampling events.

Species habitat affinities—In general, juvenile and adult sportfish habitat suitability curves showed very similar use of habitat complexity (i.e., intermediately complex habitats). A great degree of similarity was found in HSCs among the seven distinguishing species identified by SIMPER (e.g., see FIG. 4). Nevertheless, there were some differences in habitat suitability curves among species that deserve mention. Shiners did not select for woody debris

TABLE 3. Summary of results correlating change in fish community structure between sampling events with changes in habitat characteristics in three sections of the Peace River, Florida. HCI refers to habitat complexity index.

River Section	BIO-ENV (best correlation)	BIO-ENV (next best correlation)
Lower	0.830 (woody debris)	0.758 (woody debris, macrophyte cover, conductivity)
Middle	0.770 (HCI, woody debris, depth, water velocity)	0.745 (HCI, woody debris, water velocity)
Upper	0.358 (macrophyte cover, water velocity)	0.333 (macrophyte cover, depth, water velocity)

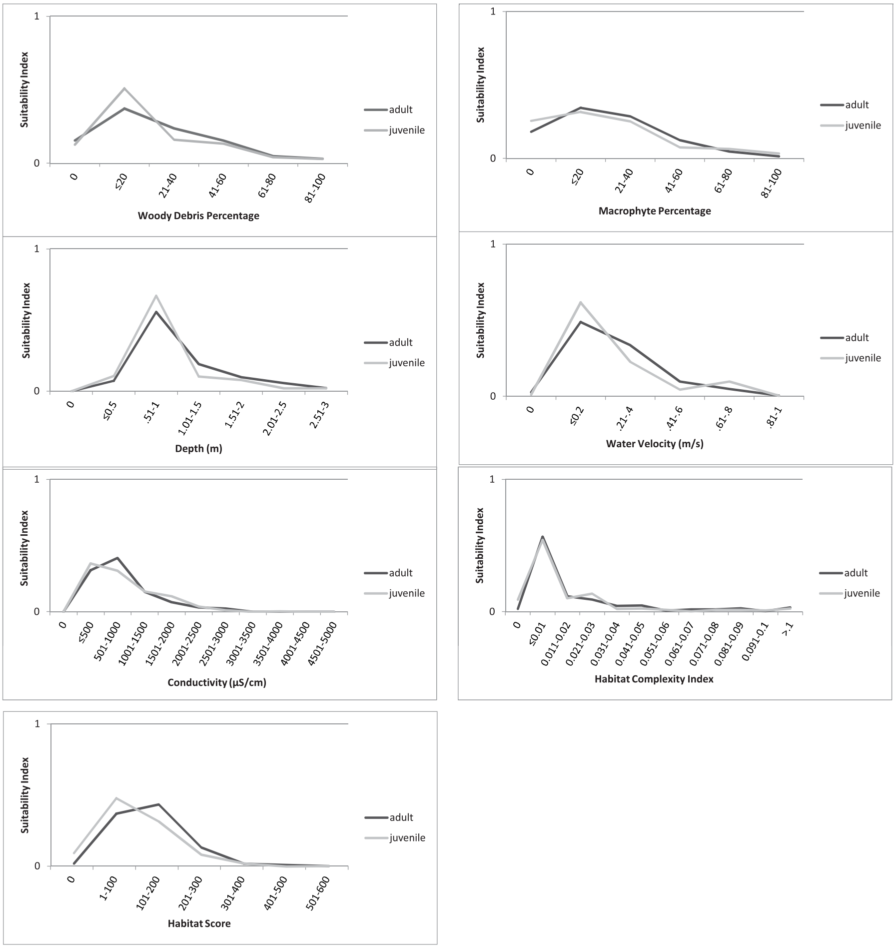

FIG. 4. Habitat suitability curves for juvenile and adult spotted sunfish derived for each of a number of structural and water quality measures of habitat.

coverage. Redear sunfish exhibited a higher tolerance to conductivity and selected less habitat complexity (i.e., woody debris, macrophytes) when compared to the other centrarchids. Conversely, spotted sunfish exhibited use of lower conductivity when compared to the other species. Seminole killifish used conductivities between 500 and 2000 µS/cm. The exotic species blue tilapia and *Pterygoplichthys* spp. selected for habitat of intermediate complexity. Blue tilapia, however, had the highest tolerance to conductivity (2500 µS/cm) of all fishes for which suitability curves were constructed.

Principal component analysis allowed for the visualization of different habitat affinities among the seven species contributing to differences in fish community structure among river sections (FIG. 5). The PCA of combined

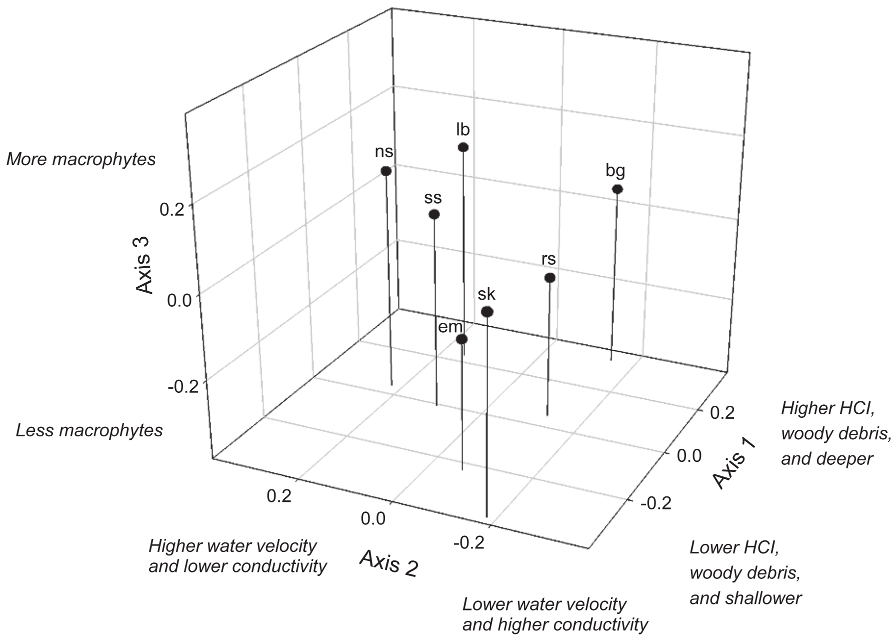

FIG. 5. Principal component analysis showing habitat affinities of the seven species contributing to fish community differences among river sections: spotted sunfish (ss), eastern mosquitofish (em), Seminole killifish (sk), Notropis shiners (ns), Bluegill (bg), largemouth bass (lb), and redear sunfish (rs). Each axis arrays correlated environmental variables as a gradient (e.g., shallower to deeper, higher water velocity to lower water velocity).

environmental data identified three major axes of environmental variability (eigenvalues >1) that together explained 57% of total variability. Axis 1 described a gradient of HCI, woody debris, and water depth with higher scores representing higher HCI, greater percent coverage of woody debris, and deeper water depths. Axis 2 described a gradient of water velocity and conductivity. Axis 3 described a gradient of macrophyte cover. Largemouth bass was associated with greater structural complexity (HCI, woody debris, and macrophytes) and deeper depths. Eastern mosquitofish and Seminole killifish were associated with lower structural complexity, higher conductivity, and shallower water. Shiners were associated with the highest water velocities and lowest conductivity of the species analyzed. Of the sunfishes, bluegill was associated with the highest structural complexity, and spotted sunfish was associated with the highest water velocity and lowest conductivity.

DISCUSSION—A stratified-random sampling approach was developed that used fine-scale electrofishing transects and quantified habitat in an attempt to better analyze fish communities and their associations with riverine habitat. Although broad-scale sampling (e.g., 100 m transects) that incorporates habitat variability across a given transect can be useful for assessing overall fish community structure (Champeau et al., 2009), the use of numerous, discreet (8 m^2) transects had the advantage of relating fish collections to detailed habitat assessments, allowing for the development of habitat suitability curves and a comparison of habitat affinities among species. Despite the methodology being different from previous fixed station sampling in the Peace River (Champeau et al., 2009), community metrics were similar. The stratified-random sampling approach allowed for a greater number of samples to be collected along the entire mainstem of the river, rather than having to exert greater effort into each transect that necessitates the selection of specific reaches or fixed stations for study. It was necessary, however, to limit sampling to river stretches close to access points during low water. Even so, site selection was still randomized and representative habitats were sampled allowing for characterization of long stretches of river.

Fish community structure and species-specific abundances differed in each section (i.e., lower, middle, and upper) of the Peace River, which suggests that the geologic and hydrologic differences used to treat each of the river sections separately for the purposes of setting minimum flows and levels did indeed lead to differences in fish habitat. The upper Peace River was dominated by large percentages of aquatic macrophytes and dramatic ranges in water velocity, both of which were correlated with changes in fish community structure. There is likely an interaction between macrophyte coverage and water velocity because, during high water velocities, conditions may be unfavorable for macrophytes, resulting in displacement or uprooting. The abundant macrophytes consisted of water hyacinth *Eichhornia crassipes*, hydrilla *Hydrilla verticillata*, and spatterdock *Nuphar lutea*. Studies have shown that structurally complex coverage, such as woody debris and macrophytes, provides forage and refuge locations for stream fish communities (e.g., Dutterer and Allen, 2008). The greatest mean abundances recorded during the study were for eastern mosquitofish and spotted sunfish in the upper section of the river. Habitat affinities (HSCs, PCA) of spotted sunfish suggested greater use of higher water velocity and lower conductivities than other centrarchids, which explains their great abundance in the upper river section. Eastern mosquitofish can be abundant when high water levels flood adjacent wetlands and forests, especially if hypoxic conditions have occurred. For example, populations of eastern mosquitofish increased dramatically after the passage of Hurricane Charley through the study area in 2004 (Champeau et al., 2009). Affinities for higher water velocity and greater macrophyte coverage, however, were not apparent in eastern mosquitofish HSCs and in the PCA. These results may be due to lower catchability of small-bodied species by electrofishing in heavily vegetated areas (Killgore et al., 1989; Gelwick and Matthews, 1990; Miranda and Pugh, 1997; Chick et al., 1999).

Primary literature indicates that fish species select different habitats based on the interaction of a variety of dynamic factors such as availability of food resources (Grenouillet et al., 2002), predation risk (Werner and Mittelback, 1981), habitat complexity (Lewin et al., 2004), and water depth (DeVries, 1990; Jordan et al., 1998). Fish species in the middle section of the Peace River appear to follow this trend. Several physicochemical variables and habitat characteristics were correlated with changes in fish community structure among sampling events, probably as a result of the complex habitat dynamics and interactions that occur in this river section. One of the top predators of the river system, largemouth bass, was most abundant in the middle section of the river. Largemouth bass are generalists (Schlosser, 1982; Irvine, 1985), and this was apparent in their central position in the PCA.

For the sunfishes, competition and/or subtle differences in habitat requirements may have been contributing factors that led to broad-scale distributional differences among the species. For example, spotted sunfish decreased in abundance from the upper to the lower section, whereas bluegill increased in abundance. Results from PCA indicate that, of the sunfishes, bluegill was associated with the highest structural complexity and specific conductivity, which may explain its greater abundance in the lower section of the river. Spotted sunfish, in contrast, was associated with higher water velocity, greater macrophyte cover, and lower conductivity, reflecting its greater abundance in the upper section. The middle section of the river is where these two species have the most interaction, based on catch rates and overlapping HSCs. A greater understanding of how habitat and water level fluctuation influence competition among sunfishes could be gained though intensive study of the middle section of the Peace River.

In the lower section of the Peace River, the physicochemical variables and habitat characteristics that were correlated with changes in fish community structure included habitat complexity indices (woody debris, macrophyte cover) and conductivity. Tidal influences and creek inputs in the lower section of the river contribute to highly variable conductivity ranges that apparently affect fish community structure. Species that were most abundant in the lower section of the river compared to other sections were bluegill, redear sunfish, eastern mosquitofish, and Seminole killifish. Results from PCA indicate that these species exhibited the greatest affinities for higher conductivity. Bluegill and redear sunfish are known to tolerate high conductivity and can be found in the oligohaline (0–5 ppt) zones of rivers (Peterson and Meador, 1994). Eastern mosquitofish and Seminole killifish are known to tolerate even wider ranges in conductivity (Nordlie, 2006). The PCA showed that these two species occupied shallower depths than the sunfishes, which is consistent with life history strategies associated with wetland vegetation (Nordlie, 2006).

In agreement with other studies (e.g., Winemiller and Jepsen, 1998), the Peace River supports fish communities in which species use similar types of habitat, as apparent in the high degree of overlap in the HSCs and relatively subtle differences depicted in the PCA. Warm-water-river game fish species are facultative rather than obligate riverine species and may have generalized flow preferences (Aadland, 1993). However, loss of floodplain habitat during low water level may result in reduced cover from aquatic and terrestrial predators and reduced food resources (e.g., invertebrates) or accessibility to food resources. This study helped to identify environmental factors related to habitat that influence fish community structure along the course of the Peace River. Availability of complex habitat (e.g., higher percentage of woody debris) greatly influenced fish community structure in the entire river. Woody debris is an important component in streams that plays a role in physical, chemical, and biologically processes (Angermeir and Karr, 1984). Large pieces of woody debris trap particles as well as provide detritus and organic material needed for many aquatic organisms (Shearer, 1972; Reice, 1974; Anderson et al., 1978; Naiman and Sedell, 1979; Bilby and Likens, 1980; Triska et al., 1984; Benke et al., 1984). In addition to habitat complexity, it appears that water velocity and macrophyte coverage influence fish community structure to a large extent in the upper river section, and conductivity (from both tributary input and tidal influences) influences fish community structure in the lower river. Seasonal fluctuation in hydrology directly affects fish community structure and associated species-specific habitat preferences (Travnichek et al., 1995; Petts, 1996; Warren et al., 2008). The species that tended to be best suited to habitats in the upper river (based on abundance and habitat affinities) was spotted sunfish, and species that tended to be best suited to habitats in the lower river were bluegill, redear sunfish, eastern mosquitofish, and Seminole killifish. The middle section of the river likely represents the area where interspecific competition is most intense, particularly for the sunfishes.

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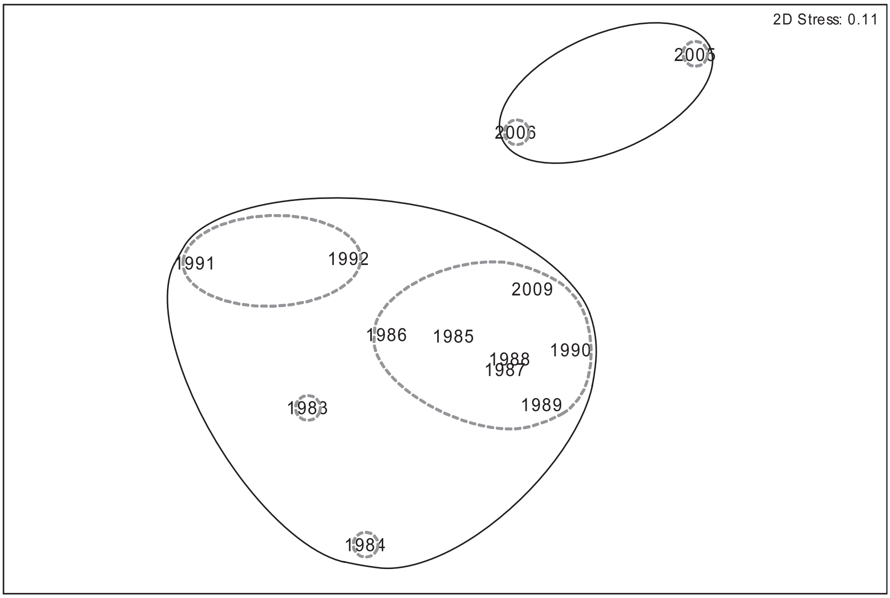

Appendix A. Nonmetric multidimensional scaling ordination of fish community samples collected during fixed-station electrofishing in the Peace River, Florida from 1983 to 2009. Ellipses denoting different groups were determined using Bray-Curtis similarity percentages of 55 (solid lines) and 65 (dashed lines) from hierarchical agglomerative cluster analysis. Fixed-station sampling during 2009 has been added to Fig. 5 of Champeau et al. (2009). Fish community structure during the post-hurricane period (2005 and 2006) differed from the historic period (1983–1992) and from more recent sampling conducted in 2009.

COMPARATIVE ECOLOGY OF EURYHALINE AND FRESHWATER PREDATORS IN A SUBTROPICAL FLOODPLAIN RIVER

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Abstract: *Euryhaline predatory fishes use river systems in south Florida, but the degree to which they compete with resident freshwater fish predators is unknown. The objectives of this study were to determine the abundance, distribution, habitat, and diet of a large diadromous euryhaline predator (common snook* Centropomus undecimalis*) in a southwest Florida river relative to large freshwater predators. Fish were electrofished in the mainstem of the Peace River 2007–2010, and gastric lavage was used to acquire stomach contents. Common snook and resident freshwater predators were found throughout the river. Common snook were most abundant in the lower section of the river, whereas the two dominant freshwater species, Florida gar* Lepisosteus platyrhincus *and largemouth bass* Micropterus salmoides, *were most abundant in the upper section. Both euryhaline and freshwater predators had affinities for structure (e.g., snags) and ate similar prey (predominantly crayfish* Procambarus *spp. and brown hoplo* Hoplosternum littorale*). Separate niches likely resulted from innate differences in seasonal movement patterns (e.g., spawning locations), habitat preferences (e.g., water depth and flow), and tolerance of environmental conditions (e.g., temperature) that either spatially separated common snook from freshwater predators or reduced competition for resources.*

Key Words: Fish, electrofishing, distribution, habitat, diet, Peace River, Florida

A variety of larger euryhaline predatory fishes from various families (e.g., Centropomidae, Sciaenidae, Megalopidae, Elopidae) move in and out of coastal rivers, where they reside and function as natural components of freshwater systems (Winemiller, 1983; Winemiller and Leslie, 1992). The often complex life cycles and movements of these transient, euryhaline predators can have major effects on coastal river fish populations and food web dynamics as these fish compete with resident freshwater predators on various spatio-temporal scales (Winemiller and Jepsen, 1998).

In southwest Florida, the primary euryhaline predator that penetrates well into coastal rivers is the common snook *Centropomus undecimalis* (Champeau, 1990). This species inhabits tropical and subtropical waters of the western Atlantic (34°N–25°S latitude; Rivas, 1986). Because of its complex life history traits (i.e., diadromous, obligate marine spawner, protandric hermaphrodite; Taylor et al., 2000), its ecology can vary greatly between bodies of water (e.g., river vs. estuary, upper river vs. lower river), depending on available habitat types (Taylor et al., 1998; Blewett et al., 2006; Blewett et al., 2009; Stevens et al., 2010; Winner et al., 2010). Common snook can be found in large numbers around barrier islands and passes, in the estuary proper, in tidal and nontidal freshwater sections of rivers, and, in some systems, as far as 100 km upriver (Champeau, 1990). Common snook are present in the tidal section of rivers year-round; however, abundances can double during fall and winter as a result of complex movement patterns that are not well understood (Blewett et al., 2009). The species is an opportunistic predator that consumes size-specific prey that is abundant in its environment (Blewett et al., 2006), and its diet has been used as a proxy for determining post-hurricane changes in riverine fauna (Stevens et al., 2010). Little is known about how the ecology of this euryhaline predator compares with that of resident freshwater predators that occur in the same river systems. The objective of this paper is to define the distribution, seasonality, habitat, and diet of common snook and associated large freshwater predators—specifically largemouth bass *Micropterus salmoides*, Florida gar *Lepisosteus platyrhincus*, longnose gar *Lepisosteus osseus*, and bowfin *Amia calva*—in a subtropical floodplain river system.

MATERIALS AND METHODS—*Study area*—The Peace River is 182 river-kilometers (rkm; distance along the river's centerline) long, descends 30 m at an average gradient of 0.2 m km^{-1}, drains a 5,959-km^2 watershed, and has an average annual discharge of 32.7 m^3 s^{-1} (Estevez et al., 1981). The highest river flows typically occur from July through September, coinciding with the summer rainy season in south Florida (Kelly and Gore, 2008). Strip mines, agriculture, and urban development within the Peace River basin have altered the hydrology and degraded water quality (PBS&J Inc., 2007), but the course of the river and the majority of its main tributaries remain unaltered and unimpounded. The river's banks are also relatively pristine, and there are extensive floodplains, backwater sloughs, wetlands, and forested corridors throughout its course.

The river was sampled from 20 to 156 rkm upstream of the mouth, for a total coverage of 136 rkm (FIG. 1). Within the sampling area, the width of the river ranged from 10 m at the upstream boundary to 160 m at the downstream boundary. The farthest downstream area is influenced tidally for 15 rkm and is characterized predominately by freshwater throughout the year; however, during drier years a portion of this area (~7 rkm) becomes brackish in winter or spring. In this seasonally brackish stretch of the river, shorelines are dominated by mangroves, emergent marsh grasses, and leather fern, transitioning upstream to large overhanging trees (oak, cypress, and willow), shrubs, and snags, where the river becomes permanently fresh.

Fish sampling—Apex predator fishes were sampled in the Peace River, Florida using electrofishing during three seasons from July 2007 through January 2010: fall (21 September–20 December), winter (21 December–20 March), and summer (21 June–20 September) (FIG. 1). A stratified-random sampling design was used to collect most electrofishing samples in the river. For the purpose of site selection, the river was divided into zones to distribute samples evenly; each zone was approximately 4 rkm long. One site in each zone was randomly selected from points spaced 0.1 km apart along the center line of the river. At each random site, a 200-m transect was completed along the shoreline. Due to low water, fall and winter random sampling in the middle and upper sections of the river was not possible because of the many shallow and exposed shoals and limestone riffles that obstructed boat travel. In order to sample these sections, all access points along the river were used. At each access point, multiple transects (4–6) were completed covering small stretches of river. These stretches were representative of the rest of the river, as they were comprised of deeper pools separated by shallow shoals and riffles. Length of transects within the low-water universe varied from 50 m to 300 m (average 150 m), depending on availability of water

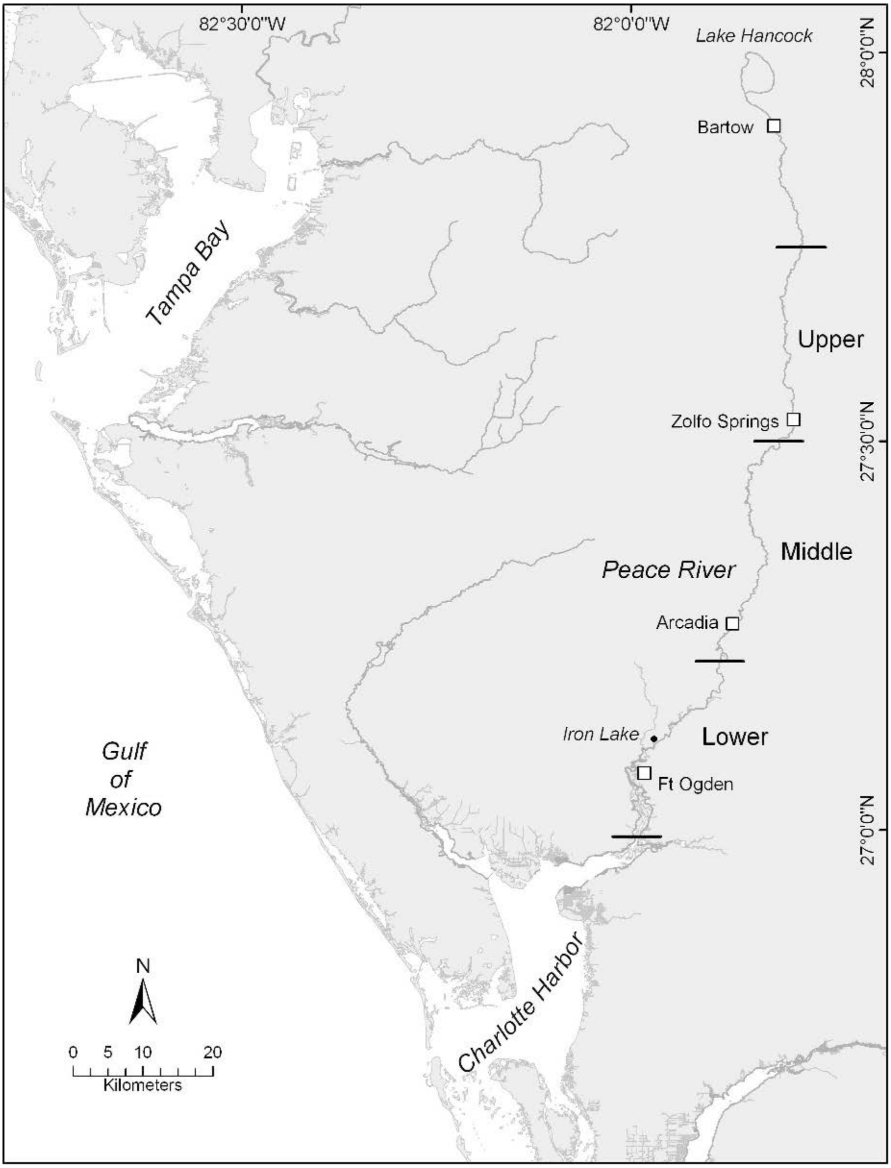

FIG. 1. Map of the study area showing the different sections of the river sampled by electrofishing.

depths ≥1.0 m. No sampling was completed in spring (typically the driest season) during this study due to extreme shallow water in the nontidal areas of the river and increased salinity in the lower tidal reaches. All sampling was completed between 0700 and 1900 EST. Water temperature (°C), salinity (ppt), conductivity (µS/cm), dissolved oxygen (mg/l), and pH were measured at the surface and bottom at each sampling site using a YSI 600QS sonde (YSI Inc.). In addition, each January, water temperatures were monitored every hour near the bottom of the river with temperature dataloggers (HOBO Water Temp Pro v2, Onset Computer Co.). Dataloggers were stationed at Fort Ogden and Zolfo Springs in the mainstem of the river and in Iron Lake, a deep man-made lagoon off the mainstem of the lower section of the river (FIG. 1).

Fishes were collected from an electrofishing boat that used pulsating direct current regulated by a Smith-Root model 5.0 or 9.0 electrofisher. The aluminum boat hull served as the cathode and a pair of bow-mounted boom electrode arrays served as the anodes. Almost all sampling was conducted in freshwater (only four samples in salinities >1 ppt, and no samples exceeded 2.2 ppt). Water depth was recorded at the beginning and end of each transect and was monitored throughout. Water depth of 1.0–4.0 m was required for boat maneuverability and gear efficiency; if water was not deep enough at a site, an alternate, adjacent site was selected by heading in a randomly chosen preselected direction (upstream or downstream). Electrofishing transects were run as the boat moved into the current along the shoreline. Stunned common snook, largemouth bass, and bowfin were captured with dip nets by three research staff, two stationed at the bow of the boat and one at the stern. Stunned Florida gar and longnose gar were not collected with dip nets but were counted by staff. All dip-netted fish were measured to the nearest mm total length (TL).

Diet samples were taken from common snook and largemouth bass using gastric lavage and clear plexiglass inspection tubes, and fish were released alive (procedure described in Stevens et al., 2010). Because lavage was unsuccessful for bowfin, this species was culled and its stomachs dissected. Both species of gar were excluded from diet analysis because lavage was difficult and stomachs were often empty. Stomach contents were preserved in 10% buffered formalin in the field and taken to the laboratory where they were identified to the lowest possible taxon. Pieces of prey items were counted as one unless discrete countable parts such as otoliths or claws were found. The volume of each prey item was determined by measuring the volume (mm^3) of water it displaced in a graduated cylinder. Specimens less than 100 mm^3 in volume were measured by a cylindrical geometric model using a microscope-mounted micrometer. An elliptical geometric model was used to measure volume of small fragmentary items that had to be slide-mounted.

Data analysis—The effect of sampling event, river section, water conditions, and habitat characteristics on the variability of abundance (fish 100 m^{-1} shoreline) was examined using a generalized linear model (SAS Institute Inc., Cary N.C., U.S.) for the three most abundant predators (common snook, largemouth bass, and Florida gar) collected from summer 2007 through winter 2010. The Poisson or the negative binomial distribution was fit to species abundance data when deemed most appropriate. Sampling event (season and year), river section (lower, middle, upper; FIG. 1), and snag and overhanging vegetation coverage along shorelines (recorded in 10% increments) were used as categorical variables. Covariates were water depth (average of start and end depth) and water temperature; both variables were $\ln(x+1)$ transformed. The most parsimonious model was selected using Akaike's information criterion (AIC). The full generalized linear model was tested against reduced models by removing combinations of variables and examining the effects on the AIC value. The best fit model was chosen as the combination of variables that minimized AIC. The sizes of each species collected in this study typically represented adults, except for largemouth bass, which included high numbers of juveniles and adults (juveniles <300 mm; Clugston 1964). Because both size groups of largemouth bass followed the same general seasonal and annual abundance patterns, they were treated as one in the model.

Principal component analysis (PCA) was used in visualizing distinct niches for the most abundant predators in the river. A PCA was conducted that weighed predator relative abundance against five correlated environmental covariates: water temperature, river flow, depth, river location (upstream to downstream), and percentage snags and overhanging vegetation. The five covariates were separated into orthogonal components based on the correlation matrix. The PCA was conducted using the Factor procedure and SAS software, and the principal components were rotated using the varimax option to facilitate the interpretability of each respective component. Variable loadings and principal-component scores were calculated independently for each sample.

Stomach contents of common snook, largemouth bass, and bowfin are reported as percent numerical abundance (A), the number of individuals of each food type as a percentage of the total number of identifiable prey items; percent volume (V), the percentage of the total volume of all prey

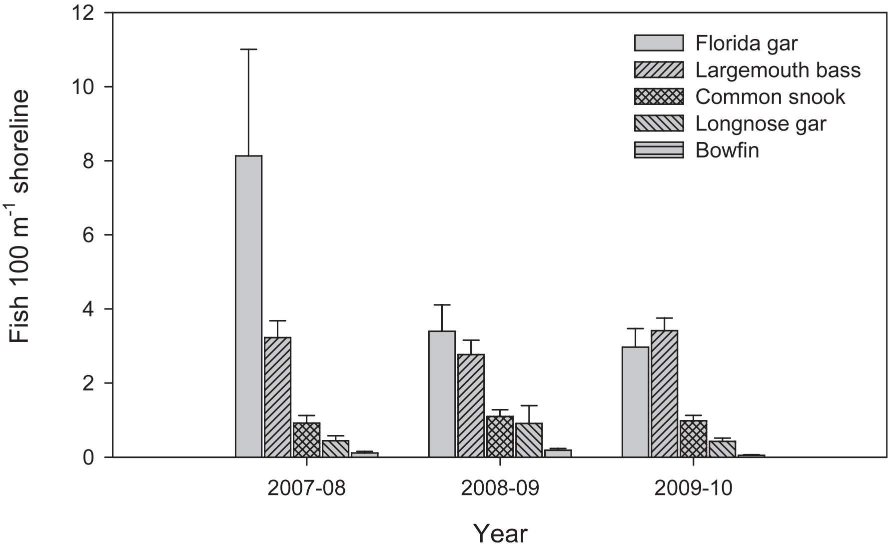

FIG. 2. Annual abundance (± standard error) of five predatory fish species collected from the Peace River, Florida.

items; and percent frequency of occurrence (F), the percentage of stomachs containing prey in which a particular prey taxon occurred. Nonparametric multivariate techniques were used to analyze diet for each of the three species among sampling events and river sections based on prey numeric data and for largemouth bass between juvenile and adult life stages (Clugston, 1964; Wheeler and Allen, 2003). These techniques were also used to analyze any differences among the diets of the three predators. Mean prey abundance indices for each species (no. prey per stomach) were square-root transformed to reduce the influence of highly abundant taxa. Nonmetric multidimensional scaling (MDS; Clarke and Warwick, 2001) and hierarchical agglomerative cluster analysis (CLUSTER; Clarke and Warwick, 2001) were used to group spatio-temporal samples containing similar prey assemblages. Similarity percentage analysis (SIMPER; Clarke and Warwick, 2001) was used to identify taxa representative of dissimilarities between groups determined from MDS. Taxa that were considered distinguishing were those that contributed greater than 5% to the total average dissimilarity between groups. All multivariate analyses were conducted with PRIMER v.6 (PRIMER-E Ltd., Plymouth, UK).

RESULTS—We collected 4,348 predators from 317 collections during three years of sampling in the Peace River. Florida gar was the most abundant predator collected (n = 1,850), followed by largemouth bass (n = 1,489), common snook (n = 603), longnose gar (n = 342), and bowfin (n = 64). Abundance of all five predators was consistent among the three years of the study; however, one large catch of Florida gar (190 fish 100 m^{-1}) in the upper section of the river increased its mean abundance during 2007–2008 (FIG. 2).

Sampling event (season and year), river section, depth, and the percentage of snags or overhanging vegetation contributed significantly to variation in abundance of Florida gar, largemouth bass, and common snook (TABLE 1). The temporal (sampling event) and spatial (river section) factors were significant for all three species. Florida gar and largemouth bass exhibited

TABLE 1. Results of generalized linear model analyses of the effect of sampling event, habitat, and environmental characteristics on three top predators in the Peace River, Florida. Significance level was set at 0.05. Model variables that were eliminated from the final model are identified as nonsignificant (NS). Poisson or negative binomial distributions were fit to species abundances.

Model Variable	df	F	P
Common snook (Poisson)			
Sampling event	9	13.90	<0.0001
River section	3	11.37	<0.0001
Water depth	1	9.05	0.0029
% snags	10	2.31	0.0058
% overhanging vegetation	10	2.55	0.0126
Temperature	1	56.43	<0.0001
Largemouth bass (Poisson)			
Sampling event	9	22.58	<0.0001
River section	3	28.70	<0.0001
Water depth	1	25.67	<0.0001
% snags	10	2.55	0.0058
% overhanging vegetation	10	3.10	0.0009
Temperature	1		NS
Florida gar (negative binomial)			
Sampling event	9	4.03	0.0001
River section	3	15.70	<0.0001
Water depth	1	8.19	0.0045
% snags	10	3.20	0.0006
% overhanging vegetation	10		NS
Temperature	1		NS

the same seasonal trends; both were typically least abundant in the summer and most abundant in the winter (FIG. 3). No obvious seasonal trend was apparent for common snook. Spatially, largemouth bass and Florida gar increased in abundance from the lower section to the upper section of the river. During winter, common snook was most abundant in the lower river and absent from the upper section. During the coldest part of winter (January) water temperatures in the upper river were 1.0–3.5 °C colder than in the lower section (mainstem or backwater Iron Lake; FIG. 4).

Habitat factors (water depth, percent snags, and overhanging vegetation) also significantly affected abundance of the top predators (TABLE 1). Both Florida gar and largemouth bass were most abundant from collections in shallower water (average depth, 1.0–1.3 m), and abundance decreased as depth increased, whereas common snook were collected mostly from deeper water (>1.3 m). The presence of snags at a site significantly affected abundance of all three predators; abundance increased as the percentage of snags increased. The amount of overhanging vegetation significantly affected abundance for two of the predators. Largemouth bass were most abundant at sites with intermediate

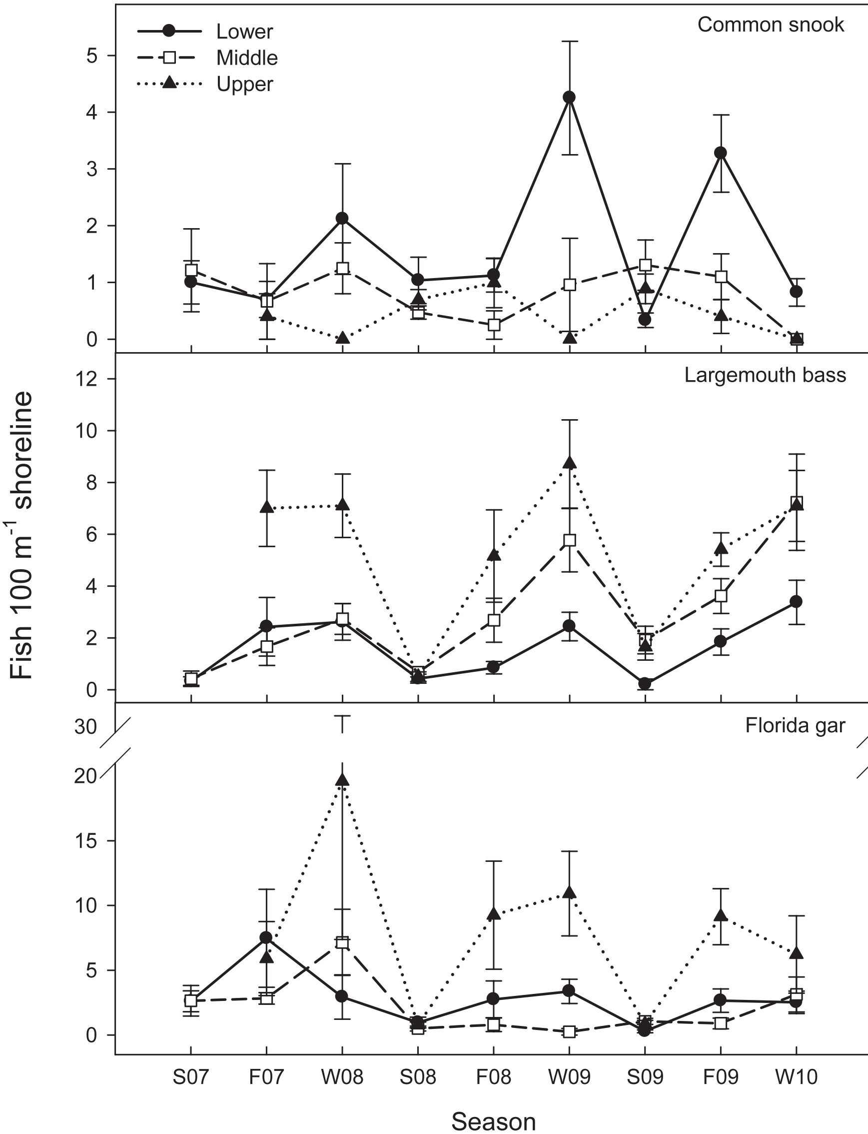

FIG. 3. Abundance (± standard error) of the top three predators in the Peace River, Florida, by season and river section (note scale difference). The letters on the x axis represent the season (e.g., F = fall) and the numbers represent the year (e.g., 07 = 2007).

overhanging vegetation coverage (20–40%), and common snook were most abundant at sites with high coverage (>50%).

Principal component analysis helped in visualizing distinct niches for the most abundant predators in the river (FIG. 5). The PCA of combined environmental data identified three major axes of environmental variability

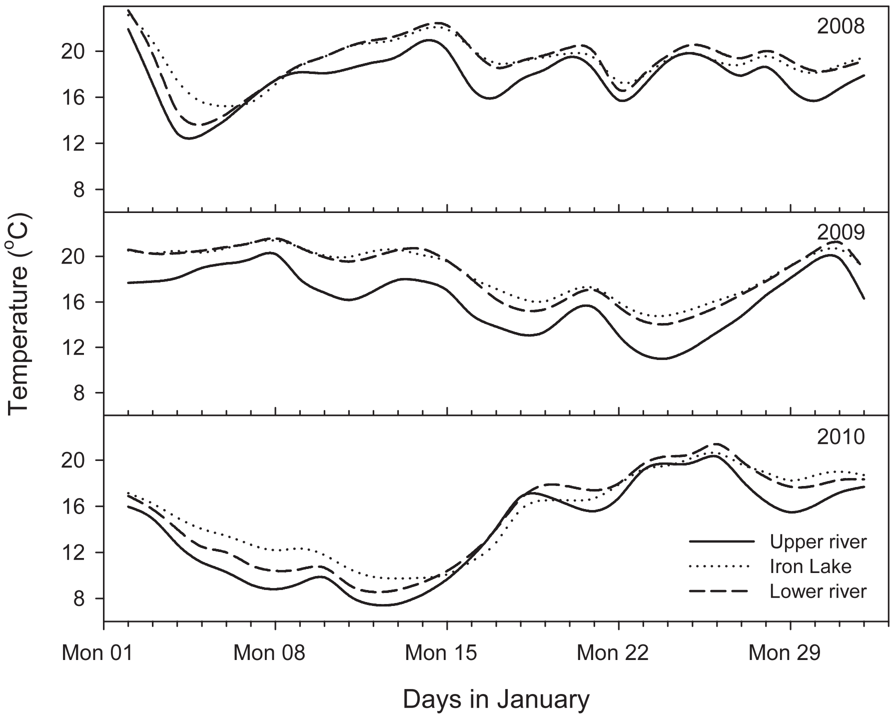

FIG. 4. The average daily water temperatures (°C) from three locations in the Peace River, Florida. General locations of temperature loggers: upper river = Zolfo Springs, Iron Lake = north end of a deep backwater area in the lower river, lower river = Ft. Ogden.

(eigenvalues >1) that together explained 78% of the variability. Axis 1 described a gradient of water temperature and river flow; higher scores represented warmer water and higher river flows. Axis 2 described a gradient from downstream to upstream and from shallow to deeper depths. Axis 3 described a gradient of habitat complexity (snag and overhanging vegetation). In general, common snook were collected at sites that were warmer, deeper, and farther downstream than those at which largemouth bass and Florida gar were collected. Common snook were also collected more often during higher river flows compared with the other two predators. All three species used complex habitats, but those used by largemouth bass were the most complex.

Each predator species examined for diet (common snook, largemouth bass, and bowfin) had a high incidence of prey items (48–62%) and a wide variety of prey was found (Appendices; common snook, 41 taxa; largemouth bass, 100 taxa; bowfin, 20 taxa). The average number of prey items found per stomach in adult freshwater predators was similar among seasons, whereas for common snook the amount of prey in winter was one-quarter of that found in summer (TABLE 2). Comparisons of diet among the three predators (by species and season) revealed that the prey assemblages of common snook and adult

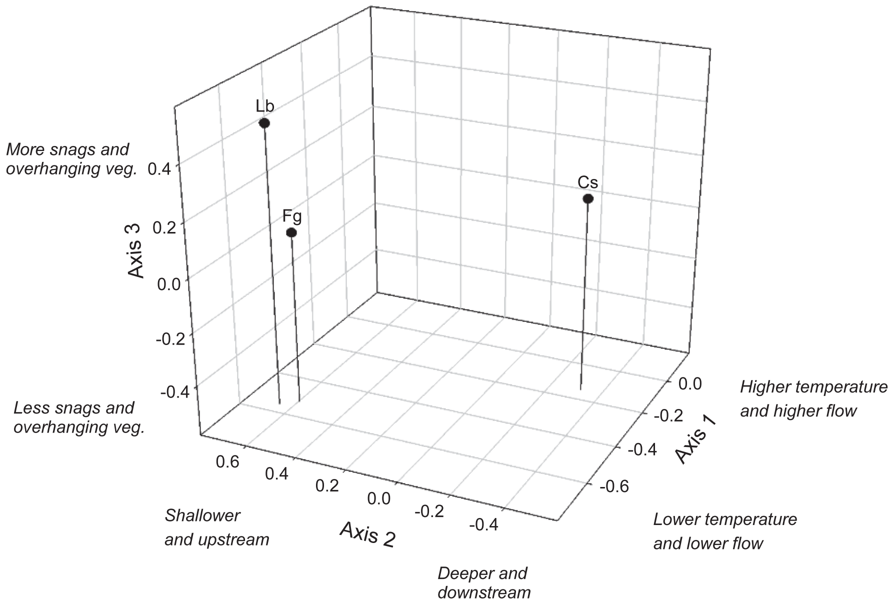

FIG. 5. Principal component analysis showing predator habitat affinities along environmental gradients in the river. Each axis arrays correlated environmental variables as a gradient (e.g., shallower to deeper, higher to lower temperature). Lb = largemouth bass, Fg = Florida gar, and Cs = common snook.

largemouth bass were similar, but the prey assemblages of bowfin (winter and summer) and juvenile largemouth bass (winter and fall) each differed from that of the common snook and adult largemouth bass prey assemblages at 45% similarity (FIG. 6). Distinguishing taxa of the prey assemblages (contribution to total average similarity) of common snook and adult largemouth bass were crayfishes *Procambarus* spp. (56.8%), sunfishes *Lepomis* spp. (14.8%), brown hoplo *Hoplosternum littorale* (13.1%), and grass shrimp *Palaemonetes* spp. (7.8%). Compared with common snook and adult largemouth bass, bowfin ate greater numbers of crayfishes and grass shrimp, and fewer sunfishes and brown hoplo (FIG. 7). Juvenile largemouth bass ate more grass shrimp, mayflies Ephemeroptera, sailfin molly *Poecilia latipinna*, and jewelfish *Hemichromis letourneuxi*, and fewer crayfishes, sunfishes, and brown hoplo. The diets of the predators did not differ by river section (MDS stress >0.19).

DISCUSSION—Although habitat use and diet in common snook were generally similar to those for resident freshwater predators, differences over time and space suggest separate ecological niches. Use of habitats on a broad scale showed that the top river predators studied preferred snag habitats and must coexist to some degree along river shorelines. Snags in riverine environments serve several functions as fish habitat. They provide habitat for invertebrates (primarily insects) that are prey for many species of fish (Benke

TABLE 2. The average number (± standard error) of prey items found in the stomachs of common snook *Centropomus undecimalis*, largemouth bass *Micropterus salmoides* (adults), and bowfin *Amia calva* 2007–2010.

Season	Common Snook	Largemouth Bass	Bowfin
Summer	2.3 (0.38)	1.6 (0.45)	2.1 (0.65)
Fall	1.8 (0.28)	1.4 (0.19)	2.2 (0.65)
Winter	0.5 (0.08)	1.2 (0.16)	2.9 (1.04)

et al., 1985), provide cover as an effective ambush point for capturing prey (Angermeier and Karr, 1984), and provide relief from swimming against high-velocity currents (Lobb and Orth, 1991). Among snag habitats, both largemouth bass and Florida gar were more abundant at shallow depths (<1.4 m). In the case of largemouth bass, biases of electrofishing may have affected abundance estimates at different depths. Largemouth bass come to the surface more slowly than the other predator species surveyed (D. Blewett, personal observation) and may not be as readily observed by dip netters collecting in deeper turbid waters (average water clarity of study samples, 1.0 m). For common snook, relative abundance was greater at deeper depths (>1.3 m). A sensitive lateral line system and proficient vision in low levels of light (Eckelbarger et al., 1980) may allow them to ambush prey effectively and

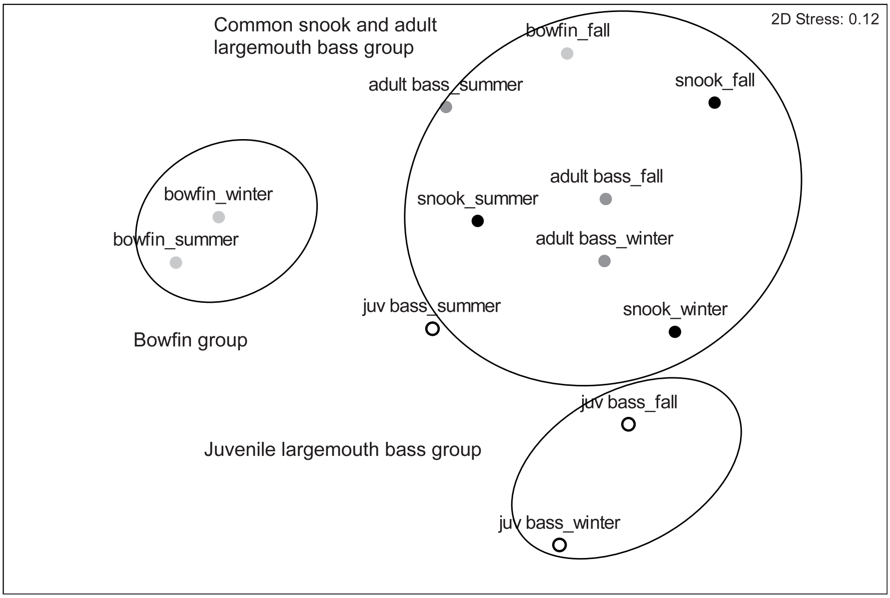

FIG. 6. Two-dimensional non-metric scaling ordination (MDS) of prey assemblages collected from the stomachs of apex predators in the Peace River, Florida (2007–2010). The prey assemblages of each predator are shown separately by season. Ellipses denote groups that were identified using a Bray-Curtis similarity percentage of 45 from hierarchical agglomerative cluster analysis (CLUSTER).

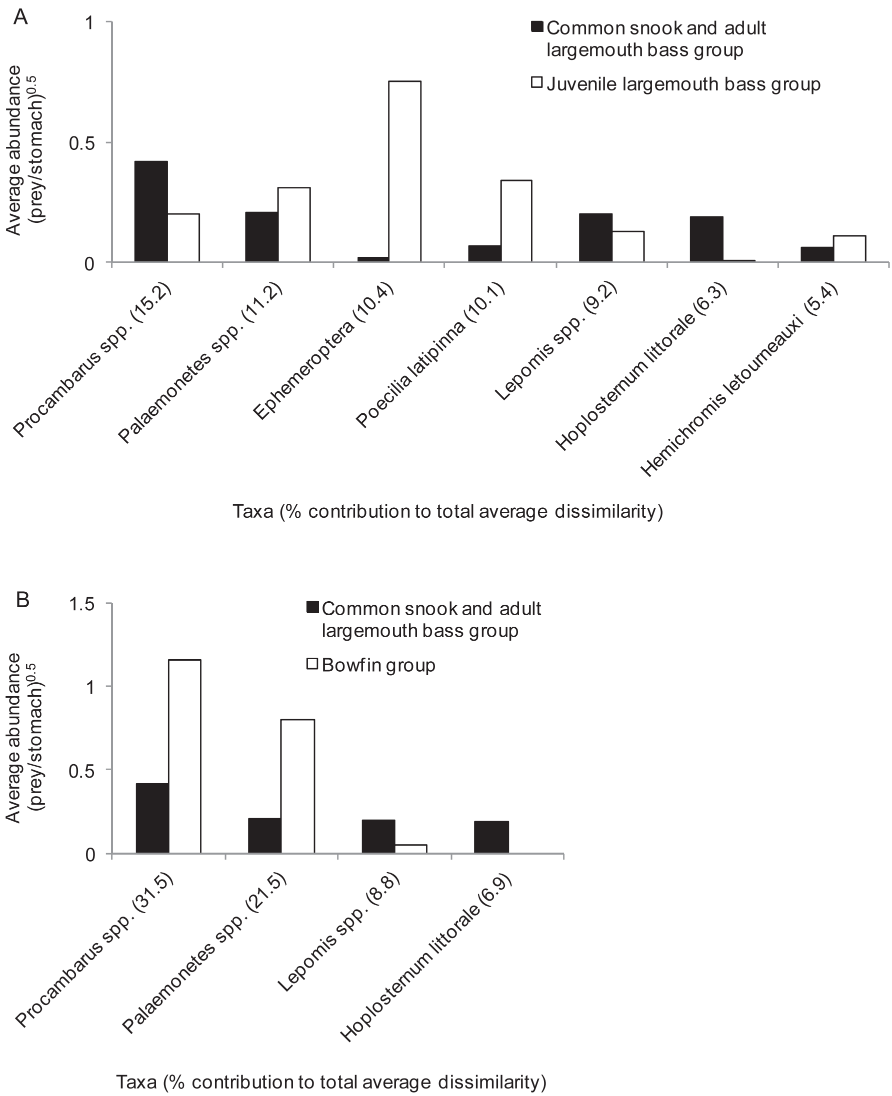

FIG. 7. Comparisons of distinguishing prey items consumed by juvenile largemouth bass (panel A) and bowfin (panel B) relative to those of common snook and adult largemouth bass as determined by similarity percentage analysis (SIMPER).

avoid predators in deep, tannin-stained waters and in the shade of overhanging vegetation.

The different predator species typically ate prey of the same taxa, primarily brown hoplo and crayfishes. These prey items are known to reside and reproduce in flooded riverine habitats. Brown hoplo is an exotic catfish native to South America that constructs bubble nests in shallow swamp and floodplain habitats during its spawning season (Winemiller, 1987; Nico and Muench, 2004). Crayfishes are most abundant in backwater areas of rivers, such as oxbow sloughs, but are also found in channel habitats (Peterson et al., 1996; Jordan et al., 2000). Although the diet of bowfin differed from that of common snook and of largemouth bass, the differences resulted from changes in the proportions of common prey items rather than exploitation of different prey taxa. Only juvenile largemouth bass regularly consumed prey taxa in addition to brown hoplo and crayfish (i.e., insects and small-bodied fishes), but as fish matured, their diets became dominated by crayfishes, which is consistent with studies in other rivers (Wheeler and Allen, 2003).

Although the adult predators ate similar prey, physiological differences between species likely affected seasonal feeding rates. Common snook was the only species in the study that occurred at the northern limits of its range, and cold winter temperatures are known to greatly slow its metabolism and affect feeding (Shafland and Foote, 1983; Howells and Sonski, 1990). In this study, the amount of prey consumed by common snook in winter was one-quarter that of summer; whereas seasonal differences in prey consumption by freshwater predators were minimal.

Although predators were widely distributed throughout the river, their centers of abundance in the river differed over time and space. Large coastal floodplain rivers such as the Peace River present fishes with a diversity of options for habitat selection, first within the lateral gradient of the floodplain and second along its longitudinal axis, which presents a gradient of habitats that span different landscapes, topographies, productivities, and abiotic conditions (Junk et al., 1989). Common snook, a euryhaline predator that can take advantage of ecological opportunities in freshwater or saltwater, was most abundant in the lower section of the Peace River. In contrast, largemouth bass, a freshwater predator better suited for slow currents (i.e., lentic habitats, backwater or pool habitats in lotic situations; Wheeler and Allen, 2003), was most abundant in the upper section and during winter. Roughly half of the lower section is tidal, and during dry periods a portion of the normally freshwater tidal area becomes brackish, which discourages establishment of largemouth bass. During the dry season, deeper sections of the upper river become separated by riffles 0.3 m deep or less. Larger fish such as largemouth bass and Florida gar can thus be restricted to these deeper habitats. As the dry season progresses, conditions promote an increase in aquatic vegetation, creating ideal habitat, foraging, and reproduction opportunities for fish species better suited for lentic habitats (Neumann et al., 1996; Winemiller et al., 2000; Robertson et al., 2008). For common snook, the dry season brings a stranding risk in the upper section, which is farther north and more inland, resulting in water temperatures that are 1.0–3.5 °C colder than in the lower section. This tropical species must move downstream as the cold and dry season progresses; consequently, it is absent from the upper river section during winter when the other freshwater predators there are most abundant.

The summer and fall seasons present different opportunities for predators in the river. As the summer rainy season approaches, many common snook leave the river to spawn. Those that stay will eventually gain access to upriver mainstem habitats as water in the river deepens. Meanwhile, large expanses of ephemeral floodplain and backwater habitats are created and are preferred by many resident freshwater fishes (i.e., largemouth bass and gar; Wydoski and Whitney, 1979; Snedden et al., 1999, Robertson et al., 2008). So it appears that during summer, many large freshwater fishes could be moving off of the river mainstem into backwater habitats as common snook are moving upriver.

As water recedes from the floodplain in fall, interactions between common snook and freshwater predators and prey species should increase. The gradual drying of river floodplain and backwater areas causes fish interactions to intensify within the river channel (Winemiller and Jepsen, 1998). For example, the tigerfish *Hydrocynus vittatus*, a predator native to Africa that resides primarily in river channels, preys more often on floodplain species as river levels drop (>40% of the diet, Winemiller and Kelso-Winemiller, 1994). The Peace River might also be expected to experience an influx of prey from the floodplain to the river channels in the fall; however, our study did not detect major seasonal shifts in prey (i.e., from channel species to floodplain species). One reason could be that common snook and largemouth bass do not reside solely in channel habitats during times of high water but most likely feed in or near oxbow sloughs and along the interface of the floodplain, which would account for the large numbers of floodplain species in their diets in both the wet (summer) and transitional (fall) periods. Another reason could be that a floodplain species is difficult to categorize in this subtropical river system, and the species that use the floodplain may be somewhat adept at using both the floodplain and channel habitats.

In conclusion, we documented that Florida gar, largemouth bass, and common snook were the most abundant large predators in the Peace River. Common snook, a euryhaline species, were found throughout the river but were most abundant in the lower section. Florida gar and largemouth bass were most abundant in the upper portion of the river. Both freshwater and euryhaline predators used similar habitats (e.g., snags) and ate similar prey, but sufficient differences were found in relation to water depth and seasonal and spatial use to suggest distinct niches within the river. Separate niches likely resulted from innate differences in seasonal movement patterns (e.g., spawning locations), habitat preferences (e.g., lotic vs. lentic), and tolerance of environmental conditions (e.g., temperature) that either spatially separated common snook from many of the freshwater predators or reduced competition for resources (e.g., differences in seasonal prey consumption).

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APPENDIX 1. Prey items found in stomachs of *Micropterus salmoides* (largemouth bass; juvenile n = 260, adult n = 301) in the Peace River, Florida. A total of 1,176 fish sampled for diet from electrofishing (48% of stomachs contained prey items). UID, unidentified; A, percent numerical abundance; V, percent volume; F, frequency of occurrence.

	Juvenile			Adult		
Prey Species	A	V	F	A	V	F
Animalia						
Annelida UID	0.04	< 0.01	0.38	.	.	.
Clitellata						
Hirudinea	.	.	.	0.28	< 0.01	0.66
Arthropoda						
Arachnida						
Araneae UID	0.09	0.02	0.77	0.71	0.04	1.66
Lycosidae	0.09	0.27	0.77	.	.	.
Trombidiformes						
Hydracarina UID	0.13	< 0.01	1.15	0.14	< 0.01	0.33
Branchiopoda						
Diplostraca						
Bosminidae						
Bosmina spp.	0.04	< 0.01	0.38	.	.	.
Insecta UID	0.72	0.05	5.77	1.13	0.01	2.66
Coleoptera, imago UID	0.04	< 0.01	0.38	0.14	< 0.01	0.33
Coleoptera, larva UID	0.09	< 0.01	0.77	0.14	< 0.01	0.33
Elmidae, imago UID	0.04	0.02	0.38	0.14	< 0.01	0.33
Haliplidae UID	.	.	.	0.14	< 0.01	0.33
Hydrophilidae, larva UID	0.04	0.16	0.38	.	.	.
Gyrinidae						
Dineutus carolinus	.	.	.	0.42	0.01	0.33
Dineutus spp.	.	.	.	0.14	< 0.01	0.33
Diptera, imago UID	2.59	0.02	0.38	0.14	< 0.01	0.33
Diptera, larva UID	0.22	< 0.01	0.77	.	.	.
Diptera, pupa UID	0.27	< 0.01	1.15	0.85	< 0.01	0.66

APPENDIX 1. Continued.

	Juvenile			Adult		
Prey Species	A	V	F	A	V	F
Chironomidae, larva UID	3.04	< 0.01	2.31	5.80	< 0.01	1.99
Chironomidae, pupa UID	0.13	< 0.01	0.77	.	.	.
Chironominae, larva UID	0.22	< 0.01	1.15	0.14	< 0.01	0.33
Chironominae, pupa UID	0.18	< 0.01	0.38	.	.	.
Paratanytarsus spp.	0.04	< 0.01	0.38	.	.	.
Tanypodinae, pupa UID	0.04	< 0.01	0.38	.	.	.
Orthocladiinae UID	0.04	< 0.01	0.38	.	.	.
Simuliidae						
Greniera longicornis	0.09	< 0.01	0.38	.	.	.
Tanyderidae UID	0.04	< 0.01	0.38	.	.	.
Ephemeroptera UID	0.89	0.04	2.69	2.40	0.04	0.66
Baetidae UID		13.10	0.07	5.77	.	.
Baetis intercalaris	0.31	< 0.01	0.38	.	.	.
Callibaetis floridanus	2.42	0.02	2.69	.	.	.
Callibaetis spp.	0.13	< 0.01	0.77	.	.	.
Centroptilum spp.	0.13	< 0.01	0.77	.	.	.
Procloeon rufostrigatum	2.46	0.02	3.46	.	.	.
Procloeon viridoculare	5.68	0.03	3.46	.	.	.
Pseudocentroptiloides usa	33.41	0.47	8.46	0.71	< 0.01	0.66
Pseudocloeon ephippiatum	0.04	< 0.01	0.38	.	.	.
Pseudocloeon frondale	.	.	.	0.14	< 0.01	0.33
Caenidae						
Cercobrachys spp.	0.04	< 0.01	0.38	.	.	.
Caenis diminuta	0.04	< 0.01	0.38	.	.	.
Ephemeridae UID	0.04	< 0.01	0.38	.	.	.
Hexagenia spp.	0.36	0.11	1.15	2.4	0.04	0.66
Heptageniidae UID	0.04	< 0.01	0.38	.	.	.
Maccaffertium exiguum	0.09	< 0.01	0.38	.	.	.

APPENDIX 1. Continued.

Prey Species	Juvenile			Adult		
	A	V	F	A	V	F
Hemiptera UID	0.04	< 0.01	0.38	0.28	< 0.01	0.66
Corixidae UID		0.13	< 0.01	1.15	.	.
Mesoveliidae						
Mesovelia musanti	.	.	.	0.14	< 0.01	0.33
Pentatomidae UID	.	.	.	0.14	0.01	0.33
Reduviidae UID	.	.	.	0.14	< 0.01	0.33
Hymenoptera, imago UID	0.04	< 0.01	0.38	0.42	0.01	1.00
Formicidae, imago	0.18	< 0.01	1.54	0.14	< 0.01	0.33
Megaloptera						
Corydalidae						
Corydalus cornutus	0.04	0.14	0.38	0.14	0.02	0.33
Odonata UID	0.18	< 0.01	0.77	0.28	< 0.01	0.66
Anisoptera	0.09	0.30	0.77	0.28	< 0.01	0.66
Corduliidae		0.00				
Epitheca princeps regina	.	.	.	0.14	0.02	0.33
Gomphidae						
Stylurus plagiatus	.	.	.	0.14	< 0.01	0.33
Libellulidae	0.09	0.71	0.77	0.28	0.03	0.66
Zygoptera	0.04	< 0.01	0.38	0.14	< 0.01	0.33
Coenagrionidae						
Neoerythromma cultellatum	.	.	.	0.14	< 0.01	0.33
Orthoptera						
Acrididae						
Leptysma marginicollis	.	.	.	0.14	0.02	0.33
Schistocerca spp.	.	.	.	0.14	0.09	0.33
Plecoptera UID	0.13	< 0.01	0.38	0.14	< 0.01	0.33
Trichoptera UID	0.09	< 0.01	0.77	0.14	< 0.01	0.33
Hydropsychidae UID	0.31	< 0.01	0.77	1.70	< 0.01	0.66

APPENDIX 1. Continued.

Prey Species	Juvenile A	Juvenile V	Juvenile F	Adult A	Adult V	Adult F
Cheumatopsyche spp.	0.13	< 0.01	0.77	.	.	.
Hydropsyche mississippiensis	0.04	0.01	0.38	.	.	.
Hydroptilidae UID		0.04	< 0.01	0.38	.	.
Leptoceridae						
Triaenodes spp.	0.04	< 0.01	0.38	.	.	.
Malacostraca						
Amphipoda						
Gammaridae						
Gammarus cf. tigrinus	1.07	0.09	0.38	.	.	.
Gammarus spp.	1.48	0.07	1.54	0.42	< 0.01	0.33
Isaeidae						
Photis spp.	.	.	.	0.14	< 0.01	0.33
Decapoda UID	0.04	< 0.01	0.77	0.14	< 0.01	0.33
Mithracidae UID	.	.	.	0.14	0.01	0.33
Panopeidae						
Panopeus spp.	0.04	0.05	0.38	.	.	.
Cambaridae						
Procambarus alleni	0.04	0.16	0.38	0.85	1.56	1.99
Procambarus fallax	0.22	1.83	1.54	0.28	0.08	0.33
Procambarus paeninsulanus	0.22	3.55	1.92	0.99	1.10	2.33
Procambarus spp.	2.24	11.38	16.15	16.40	9.69	28.90
Caridea UID	0.94	0.06	3.08	0.42	< 0.01	1.00
Palaemonidae UID	0.13	0.03	0.77	.	.	.
Palaemonetes intermedius	0.13	0.30	1.15	0.28	0.03	0.33
Palaemonetes paludosus	3.53	2.19	8.46	1.56	0.09	2.33
Palaemonetes pugio	0.89	1.40	5.00	1.27	0.03	1.00
Palaemonetes spp.	0.49	0.33	3.46	1.98	0.14	3.99
Palaemonetes vulgaris	0.04	0.07	0.38	.	.	.

APPENDIX 1. Continued.

Prey Species	Juvenile			Adult		
	A	V	F	A	V	F
Processidae						
Processa spp.	.	.	.	0.14	< 0.01	0.33
Mysida UID	.	.	.	0.14	< 0.01	0.33
Mysidae						
Taphromysis bowmani	0.27	0.01	1.15	.	.	.
Americamysis almyra	0.04	< 0.01	0.38	.	.	.
Maxilopoda						
Cyclopoida UID	0.22	< 0.01	1.15	.	.	.
Chordata						
Vertebrata UID	0.04	< 0.01	0.38	0.14	< 0.01	0.33
Actinopterygii UID	4.43	7.67	33.85	17.96	6.13	28.24
Atheriniformes UID	.	.	.	0.28	0.18	0.66
Atherinidae UID	0.09	0.48	0.77	.	.	.
Atherinopsidae						
Labidesthes sicculus	0.36	0.85	1.92	0.28	0.05	0.66
Menidia spp.	0.18	0.16	1.15	0.14	0.02	0.33
Clupeiformes						
Engraulidae						
Anchoa spp.	0.04	0.07	0.38	.	.	.
Cypriniformes						
Cyprinidae UID	0.98	1.95	5.77	0.57	0.09	1.33
Notropis petersoni	0.04	0.18	0.38	.	.	.
Notropis spp.	0.04	0.09	0.38	.	.	.
Cobitidae						
Misgurnus anguillicaudatus	0.04	0.46	0.38	0.42	0.61	1.00
Cyprinodontiformes						
Cyprinodontidae						
Cyprinodon variegatus	.	.	.	0.14	0.06	0.33

APPENDIX 1. Continued.

Prey Species	Juvenile			Adult		
	A	V	F	A	V	F
Floridichthys carpio	0.04	6.86	0.38	.	.	.
Fundulidae	0.45	1.35	3.46	0.28	0.08	0.66
Fundulus seminolis	0.22	4.39	1.54	0.28	0.51	0.66
Fundulus similis	0.04	0.46	0.38	0.14	0.34	0.33
Fundulus spp.	0.31	2.24	2.69	0.57	0.61	1.00
Lucania goodei	0.09	0.04	0.77	.	.	.
Lucania parva	0.09	0.21	0.38	.	.	.
Lucania spp.	0.13	0.25	0.77	.	.	.
Poeciliidae						
Gambusia holbrooki	5.32	5.68	14.23	3.39	0.23	3.65
Heterandria formosa	0.04	0.01	0.38	.	.	.
Poecilia latipinna	0.98	4.35	5.77	3.25	1.27	2.33
Mugiliformes						
Mugilidae						
Mugil spp.	0.27	0.27	0.38	.	.	.
Perciformes						
Gerreidae UID	.	.	.	0.28	0.08	0.33
Eugerres plumieri	.	.	.	0.14	0.02	0.33
Centrarchidae						
Lepomis auritus	.	.	.	0.14	0.27	0.33
Lepomis gulosus	0.04	0.23	0.38	.	.	.
Lepomis macrochirus	0.04	0.69	0.38	1.27	11.13	2.66
Lepomis microlophus	.	.	.	0.14	0.57	0.33
Lepomis punctatus	0.36	5.70	3.08	1.13	6.61	2.66
Lepomis spp.	0.72	7.49	5.77	4.95	9.28	10.96
Cichlidae						
Hemichromis letourneauxi	1.21	14.69	8.46	3.39	2.19	4.65
Oreochromis aureus	.	.	.	0.42	2.34	0.66

APPENDIX 1. Continued.

Prey Species	Juvenile			Adult		
	A	V	F	A	V	F
Oreochromis spp.	0.36	0.71	1.54	3.68	18.55	7.31
Gobiidae						
Microgobius spp.	0.04	0.23	0.38	.	.	.
Percidae						
Etheostoma fusiforme	0.31	1.05	1.92	0.42	0.05	0.66
Etheostoma spp.	0.04	0.11	0.38	0.14	0.02	0.33
Siluriformes						
Callichthyidae						
Hoplosternum littorale	0.18	0.30	1.54	3.82	16.88	8.97
Ictaluridae UID	0.67	2.21	3.85	2.40	1.89	3.32
Ictalurus punctatus	0.04	1.60	0.38	0.28	0.53	0.66
Ictalurus spp.	.	.	.	0.42	2.83	1.00
Noturus gyrinus	0.04	1.03	0.38	.	.	.
Noturus spp.	0.04	0.09	0.38	0.14	0.02	0.33
Loricariidae UID	0.04	0.01	0.38	1.56	2.20	3.65
Amphibia						
Anura UID	0.04	0.01	0.38	0.42	1.18	1.00
Hylidae	.	.	.	0.14	0.06	0.33
Ranidae						
Rana spp.	.	.	.	0.14	0.08	0.33
Reptilia						
Squamata						
Polychrotidae						
Anolis spp.	0.09	1.14	0.77	0.14	<0.01	0.33
Serpentes UID	.	.	.	0.14	<0.01	0.33
Testudines						
Kinosternidae						
Sternotherus ordoratus	0.04	0.69	0.38	.	.	.

APPENDIX 2. Prey items found in stomachs of *Centropomus undecimalis* (common snook; n = 272) in the Peace River, Florida. A total of 530 fish sampled for diet from electrofishing (48% of stomachs contained prey items). UID unidentified, A percent numerical abundance, V percent volume, F frequency of occurrence.

Prey Species	A	V	F
Arthropoda			
Malacostraca			
Decapoda UID	0.1	< 0.1	0.4
Cambaridae			
Procambarus alleni	1.4	1.8	2.9
Procambarus fallax	0.8	0.4	1.5
Procambarus paeninsulanus	1.5	1.2	3.7
Procambarus spp.	7.4	3.7	18.4
Palaemonidae UID	1.3	< 0.1	4.0
Palaemonetes intermedius	0.1	< 0.1	0.4
Palaemonetes paludosus	8.5	0.3	2.6
Palaemonetes pugio	2.1	0.2	1.5
Palaemonetes spp.	4.3	0.1	9.6
Panopeidae			
Rhithropanopeus harrisii	0.3	0.1	1.1
Portunidae UID	0.5	0.2	1.5
Callinectes sapidus	0.2	1.1	0.7
Xanthidae UID	0.1	< 0.1	0.4
Chordata			
Actinopterygii UID	26.5	12.4	57.4
Ariidae UID	0.3	0.4	0.7
Atherinopsidae			
Menidia spp.	0.6	0.1	0.4
Callichthyidae			
Hoplosternum littorale	8.6	30.5	16.9
Catostomidae			
Erimyzon sucetta	0.1	3.6	0.4
Centrarchidae			
Lepomis macrochirus	0.2	5.4	0.7
Lepomis microlophus	0.1	< 0.1	0.4
Lepomis punctatus	0.3	3.8	1.1
Lepomis spp.	5.0	7.8	11.8
Micropterus salmoides	0.1	0.6	0.4
Cichlidae			
Hemichromis letourneauxi	0.8	1.0	2.2
Oreochromis spp.	1.2	15.7	3.7
Cobitidae			
Misgurnus anguillicaudatus	0.7	0.2	0.7
Clupeidae UID	0.1	1.4	0.4
Cyprinidae UID	1.6	0.1	1.8
Notropis petersoni	1.3	0.3	0.4
Cyprinodontidae			
Cyprinodon variegatus	0.2	0.1	0.4
Engraulidae			
Anchoa mitchilli	2.8	0.5	0.4
Anchoa spp.	8.6	0.9	2.2

APPENDIX 2. Continued.

Prey Species	A	V	F
Fundulidae UID	0.1	< 0.1	0.4
Fundulus chrysotus	0.1	< 0.1	0.4
Fundulus seminolis	0.1	< 0.1	0.4
Fundulus spp.	3.4	0.5	2.6
Lucania goodei	2.2	0.2	0.4
Lucania parva	0.1	< 0.1	0.4
Lucania spp.	0.1	< 0.1	0.4
Gobiidae UID			
Microgobius gulosus	0.2	< 0.1	0.7
Ictaluridae UID	1.6	0.9	4.0
Ameiurus catus	0.1	0.1	0.4
Ictalurus punctatus	0.2	1.7	0.7
Ictalurus spp.	0.1	0.1	0.4
Lepisosteidae UID	0.1	1.4	0.4
Loricariidae UID	0.9	0.5	2.9
Mugilidae			
Mugil spp.	0.2	0.1	0.7
Pleuronectidae UID	0.2	< 0.1	0.7
Poeciliidae			
Gambusia holbrooki	0.2	< 0.1	0.7
Poecilia latipinna	1.4	0.3	1.5

APPENDIX 3. Prey items found in stomachs of *Amia calva* (bowfin; n = 37) in the Peace River, Florida. A total of 60 fish sampled for diet from electrofishing (62% of stomachs contained prey items). UID, unidentified; A, percent numerical abundance; V, percent volume; F, frequency of occurrence.

Prey Species	A	V	F
Annelida			
Oligochaeta UID	1.5	< 0.1	5.4
Arthropoda			
Arachnida			
Araneae UID	0.8	< 0.1	2.7
Malacostraca			
Decapoda UID	0.8	< 0.1	2.7
Cambaridae			
Procambarus alleni	1.5	2.1	5.4
Procambarus fallax	0.8	< 0.1	2.7
Procambarus paeninsulanus	0.8	0.3	2.7
Procambarus spp.	29.0	18.2	51.4
Palaemonidae UID	4.6	0.0	8.1
Palaemonetes intermedius	0.8	< 0.1	2.7
Palaemonetes paludosus	8.4	0.2	5.4
Palaemonetes pugio	2.3	< 0.1	5.4
Palaemonetes spp.	13.7	1.3	16.2
Panopeidae			
Rhithropanopeus harrisii	1.5	0.3	2.7

APPENDIX 3. Continued.

Prey Species	A	V	F
Portunidae UID	0.8	0.1	2.7
Chordata			
Vertebrata UID	1.5	1.0	5.4
Actinopterygii UID	9.2	1.6	18.9
Ariidae UID	0.8	0.3	2.7
Callichthyidae			
Hoplosternum littorale	8.4	43.7	24.3
Centrarchidae UID	0.8	3.2	2.7
Lepomis spp.	3.1	6.7	8.1
Cichlidae			
Hemichromis letourneauxi	6.1	3.6	5.4
Clariidae			
Clarias batrachus	0.8	5.0	2.7
Ictaluridae			
Ameiurus spp.	0.8	4.1	2.7
Loricariidae UID	0.8	0.9	2.7
Amphibia			
Ranidae			
Rana spp.	0.8	7.3	2.7

THE EFFECTS OF ENVIRONMENTAL DISTURBANCE ON THE ABUNDANCE OF TWO RECREATIONALLY-IMPORTANT FISHES IN A SUBTROPICAL FLOODPLAIN RIVER

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Abstract: *Seasonal abundances of two large fishes, common snook* Centropomus undecimalis *and largemouth bass* Micropterus salmoides, *were surveyed in the lower portion of the Peace River, Florida, 2004–2010. During and just prior to the study period, a number of environmental disturbances occurred: the passage of five hurricanes over the river's basin in 2004 and 2005, resulting in high river flows and one large-scale hypoxic event, and extreme cold temperatures in 2010. The two species responded differently to these events. In the year following hurricane-induced hypoxia (2004–2005), adult largemouth bass were absent from our collections. Common snook, however, were up to three times more abundant than during subsequent years (2007–2010); increased river flow, abundance of prey, and lack of interspecific competition may have contributed to the high abundance. A record cold winter in 2010 initially reduced the abundance of common snook in the lower river, but abundance quickly returned to pre-event levels. Largemouth bass abundance appeared unaffected by the extreme cold event. These results illustrate how disturbance events can temporarily affect the community structure of aquatic systems and create challenges for managers striving to track long-term trends in species populations.*

Key Words: Electrofishing, Peace River, Florida

In ecology, disturbance is defined as "a relatively discrete event in time that disrupts the ecosystem, community or population structure and changes the resources, substrate availability or physical environment" (White and Pickett, 1985). Along the coast of Florida, disturbances often take the form of hurricanes (Greenwood et al., 2006), red tides (Gannon et al., 2009), and freezes (Gilmore et al., 1978). For scientists and resource managers attempting to understand changes in an ecosystem, a community or a population, such events can greatly influence the interpretation of long-term trends (Adams, 2001; Flaherty and Landsberg, 2011). Although ecosystem modeling allows for the inclusion of disturbance effects (i.e., switches that can exert control on specific components; Odum et al., 1998), applying commonly used methods to identify the specific physical, chemical, or biological variables that lead to changes in the abundance or distribution of a species can be challenging. If environmental changes occur gradually, they can be correlated with changes in a species population (Rijnsdorp et al., 2009). In contrast, changes in variables associated with environmental disturbances can be very short-lived (e.g., hours to days), with values often returning to pre-event conditions by the time of population sampling (Stevens et al., 2006). If long-term data from multiple sites are available, a before-after-control-impact approach can be used to assess the effects of an event (Smith et al., 1993). However, in many cases, the investigator is left only with population trends in a particular area and a timeline of events to interpret. Nevertheless, for long-term assessments it is important to document the effects that environmental disturbances have on population trends, especially when they are catastrophic and obvious.

Examples of two fishes whose populations were likely affected by environmental disturbances are illustrated in a river of southwest Florida. During a six-year period in the lower Peace River, we examined seasonal abundances of common snook *Centropomus undecimalis* and largemouth bass *Micropterus salmoides*. During and just prior to the study period, a number of disturbance events occurred. First, in August 2004, Hurricane Charley passed directly over the Peace River, roughly following the path of the floodplain and resulting in extremely high river flows and a large hypoxic event that affected most of the river (Tomasko et al., 2006). During the following months the hypoxia was prolonged (2–3 mo) by the passage of two other hurricanes (Frances and Jeanne) over the watershed. High river flows continued through 2005 with the passage of two additional hurricanes (Wilma and Arlene) over the watershed. Then during winter 2010, extreme cold temperatures occurred throughout south Florida, drastically affecting the flora and fauna of the region (Rehage et al., 2010). The large number of disturbance events impacting the Peace River over a relatively short period of time, coupled with the existence of a fish monitoring program, provided a unique opportunity to study the responses of two large fishes with very different life histories (common snook: tropical, euryhaline, obligate marine spawner; largemouth bass: temperate, freshwater). The objective of this paper is to describe how disturbance events influence abundance patterns of common snook and largemouth bass in the lower reaches of the Peace River, Florida.

Materials and Methods—Abundance data for common snook and largemouth bass in the lower Peace River were collected by electrofishing (lower river study area described in Blewett et al., 2013 and Call et al., 2013). Data from two previous studies (Blewett et al., 2009; Blewett et al., 2013) were combined with additional monitoring data from the river to produce a relatively long-term data set. The lower river section was sampled seasonally 2004–2010: fall (21 September–20 December), winter (21 December–20 March), spring (21 March–20 June), and summer (21 June–20 September). A gap in sampling occurred from fall 2006 through spring 2007. As a result of increased salinity in the lower tidal reaches of the river, sampling was not conducted in spring 2008 and 2009 (electrofishing is inefficient in deep saline waters). In addition, Iron Lake, a deep backwater area connected to the mainstem of the Peace River, was sampled during winter, spring, and summer of 2006 and winter 2010.

A stratified-random sampling design was used to collect electrofishing samples in the river. For the purpose of site selection, the lower river was divided into zones, each approximately 4–6 river kilometers (rkm) long. One site in each zone was randomly selected from points spaced 0.1 km apart along the center line of the river. From 2004 through 2006, a 5-min transect was completed at each site, and the distance traveled in meters was measured (average transect, 150 m). Starting in

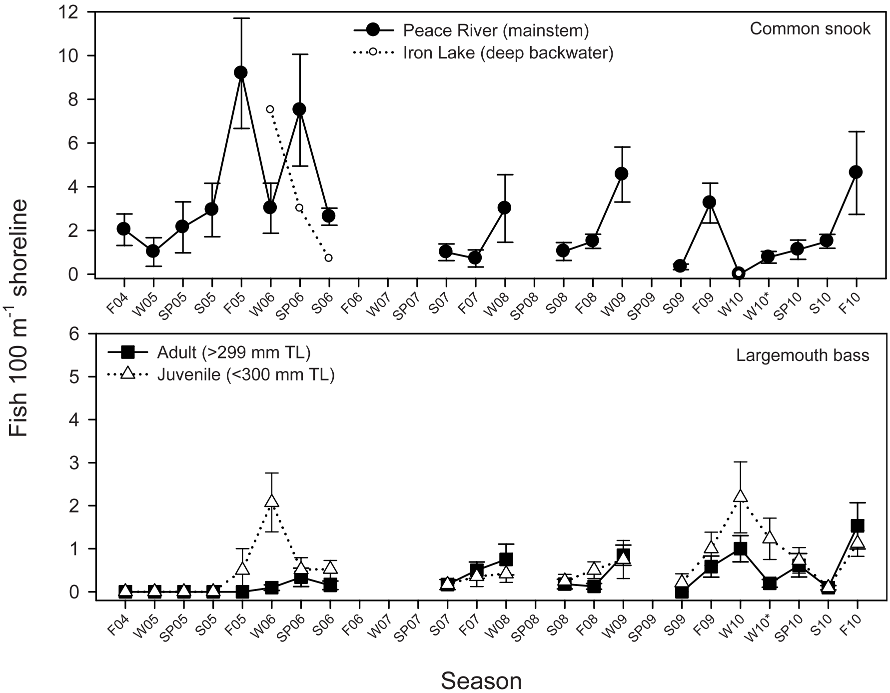

FIG. 1. Abundance (± standard error) of common snook *Centropomus undecimalis* (top) and largemouth bass *Micropterus salmoides* (bottom) in the lower Peace River, Florida 2004–2010. Abundance of common snook in Iron Lake, a deep backwater area, is also shown (four sampling events). Letters on the x-axis represent season, numbers represent year, and the asterisk denotes a second winter sampling event in 2010. Note scale difference in abundance between species. Hurricane Charley passed directly over the Peace River during August 2004, four additional hurricanes passed over the Peace River watershed during 2004 and 2005, and a severe freeze occurred in January 2010.

2007, transects were standardized at 200 m. All sampling was completed between 0700 and 1900 EST.

RESULTS—Disturbance events greatly affected sport fish abundance patterns (FIG. 1). After the hypoxic event in 2004, largemouth bass were absent from the mainstem of the lower Peace River and remained so for more than a year. A seasonal pattern of high winter and low summer abundance for largemouth bass was evident thereafter. Catch rates of common snook increased during the year following the 2004 hurricane, reaching a maximum in fall 2005 at 9 fish 100 m^{-1} shoreline; this was approximately three times greater than catch rates for common snook later in the study. Sampling events that occurred just days after the passage of cold fronts, in winter 2005 and winter 2006, detected a lower abundance of common snook in the mainstem of the river compared with the preceding and following seasons. This pattern of abundance (high fall, low winter, high spring) in the mainstem was most pronounced during 2005 and 2006. Although abundance was low in the mainstem during winter 2006, abundance in the backwater area of Iron Lake was high and reflected the overall annual trend of the mainstem. Following an extreme freeze event in January 2010, no common snook were collected in the mainstem or in Iron Lake during regularly scheduled winter sampling. Sampling a month later showed that common snook were again present in the mainstem of the lower river, and by summer and fall they were as abundant as before the freeze. Based on trends from two previous years, largemouth bass abundance appeared to be unaffected by the extreme cold event.

DISCUSSION—Disturbance events had a major effect on the distribution and abundance of common snook and largemouth bass in the Peace River. Abundance of largemouth bass was reduced after the hypoxic event associated with Hurricane Charley (Champeau et al., 2009). Largemouth bass, an obligate freshwater species, was confined to the river during this event and likely experienced high mortalities as a result. In contrast, common snook, a euryhaline species, had the capability to leave the hypoxic river and upper estuary during this event and either moved into more oxygenated brackish and high-salinity waters in the lower estuary or to adjacent rivers where there was no hypoxia (i.e., Myakka River). In addition, many common snook using the river had already moved into the lower portion of the estuary to spawn. These two factors appear to have minimized the effect of hypoxia on the common snook population, whereas freshwater fish populations in the river were greatly reduced (Champeau et al., 2009). During the passage of additional hurricanes, high flow and prolonged inundation of the floodplain may have provided a substantial prey base that led to greater use of the river by common snook. Piscivores intercept fishes as they move from drying floodplains to permanent channels (Winemiller and Jepsen, 1998). The concentration of food resources in the mainstem of the Peace River following extensive floodplain inundation and the lack of interspecific competition for these resources were likely major contributing factors to the increase in common snook abundance during late 2005.

Normal cold-weather patterns (i.e., frontal systems) decreased the abundance of common snook in winter, while cold conditions had no observable effect on the abundance of largemouth bass. Cold weather appeared to prompt small-scale movements of common snook from the mainstem of the river to deep backwater areas such as Iron Lake. For example, when water temperatures dropped to 15°C during winter sampling in 2005 and 2006, abundance of common snook in the mainstem was much less than that in the preceding and following seasons. The greater abundance in Iron Lake, however, suggested that the fish had simply moved off the mainstem. When winter sampling was conducted during warmer (18–20°C) weather in 2008 and 2009, catch rates in the mainstem were higher than in preceding and following seasons. Thus, drops in water temperature just prior to sampling appeared to have affected abundance of common snook in the mainstem river.

Following an extreme cold-weather event during winter 2010, abundance of common snook initially decreased in the mainstem of the river but had returned to pre-freeze levels by summer and fall. Common snook likely left the study area during the cold weather to seek deeper water downstream or outside of the mainstem areas sampled. Deep water is available downstream in the tidal portions of the mainstem river and its meanders as well as in man-made canals adjacent to the river mouth (these areas are too deep and typically too saline to electroshock). These deep, wind-protected waters near the river mouth are probably warmer during cold events than waters farther upstream that originate in colder inland locations (Blewett et al., 2013). Also, these areas tend to form haloclines where saline water on the bottom is prevented from mixing with freshwater at the surface. During cold events haloclines can provide thermal refuge (Stith et al., 2010). The fact that only 35 common snook were found dead after an extensive search of the mainstem river, from the mouth to 160 rkm upstream, supports the premise that they had moved downriver and had not been killed by the freeze. Although common snook may exhibit downriver movements during cold conditions (Blewett et al., 2013), leaving the deep river shorelines and moving to the shallow, more exposed shorelines of the open estuary could be risky during the most extreme freeze events. Searches for dead common snook in the estuary proper after the freeze found widespread kills of 100–2,000 individuals at each of 24 discrete locations (D. Blewett, unpublished data). Initial analyses of fisheries monitoring data in Gulf coast estuaries indicated that the relative abundance of common snook decreased substantially after the freeze, and so emergency fishing-season closures were put in place for the species. Differential survival of common snook in river systems clearly has implications for the populations in those systems, and evaluating the long-term effects of a cold event on this species will necessitate an approach that uses a variety of available datasets (e.g., fisheries-dependent monitoring, fisheries-independent monitoring in estuaries, river electrofishing).

The natural disturbances that occurred during the study mark major changes in the abundance patterns of large predators in the Peace River. The abundance of largemouth bass decreased so much after the 2004 hypoxic event that none were collected throughout the following year. A seasonal pattern of high winter and low summer abundance for largemouth bass was evident thereafter. Abundance trends for common snook were more dynamic. Common snook appeared to take advantage of post-hurricane conditions (high flows and lack of interspecific competition): abundance during 2005 was as much as three times greater than during subsequent years. These findings illustrate the acute effects of environmental events on the abundance of sport fishes and highlight how fishes may respond differently to events in a highly dynamic coastal river system.

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FISH COMMUNITY STRUCTURE OF STREAMS AND CANALS AT BABCOCK RANCH, CHARLOTTE AND LEE COUNTIES, FLORIDA

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Abstract: *Fish communities were surveyed three times each year at 6 lotic habitats (small streams and manmade drainage ditches and canals) at Babcock Ranch, Charlotte and Lee Counties, Florida. Sampling methods were adopted from the baseline assessment of the Picayune Strand Restoration Project. Breder traps were deployed for one hour at each site for each sampling event. Active fish sampling with D-frame dip nets was also conducted for one hour at each location while traps were set. Data from passive and active sampling methods were combined for statistical analyses. A total of 26 fish species, representing 14 families were collected, including seven non-native fish species from four families. The most abundant fishes were native* Gambusia holbrooki *and non-native* Hemichromis letourneuxi*, respectively. Fish assemblages in canals were significantly different than in natural streams. Native sunfishes of the genus* Lepomis *were more abundant in streams than canals. Canals served as thermal refugia for non-native cichlids during severe cold weather events in 2009 and 2010. The removal of anthropogenic drainage canals and ditches, that serve as thermal refugia and pathways for dispersal for* H. letourneuxi *should benefit native fish species over time.*

Key Words: Babcock Ranch, non-native fishes, thermal refugia, streams, creeks, drainage ditches, canals, *Hemichromis*, *Gambusia*, *Lepomis*

In South Florida, the natural hydrologic cycle is typically expressed as flooding and overland sheet-flow in the wet season (June through November) when most of the annual rainfall occurs, and wetland drawdown and lower surficial aquifers during the dry season (December through May). The hydrology of this region is dynamic, but generally predictable with varying degrees of drawdown and flooding from year to year. Aquatic fauna populations in South Florida have evolved or acclimated to survive to these seasonal extremes through various life history and behavioral adaptations that allow some individuals of a species to survive within aquatic refugia during drought or seasonal dry-down. In Florida, undisturbed isolated wetlands serve as important aquatic refuges for the survival of numerous species of fish and wildlife (Hart and Newman, 1995). In the Everglades ecosystem, Fury et al. (1995) reported shifts in species abundance in response to long-term rainfall patterns and hydrologic conditions in Everglades Water Conservation Areas. In streams, pool depth can be an important characteristic in drying stream pools for fish communities (Capone and Kushlan, 1991; Magoulick, 2000) and therefore serve as aquatic refugia during droughts. Anthropogenic disturbances such as canals, dams, and roadways can exacerbate the negative effects of drought and flooding through the amplification of natural hydrologic extremes and as barriers to fish movement.

When an ecosystem is impacted or restored, its biological integrity can be altered. Baber et al. (2002) found that ditching from agriculture and cattle ranching in the Kissimmee Prairie had an adverse effect on wetland fish communities by altering hydroperiod and connectivity. Furthermore, hydroperiod and water depth were found to control local populations while on a larger scale, connection to permanent waters determined species assemblages. In southwest Florida, fish communities of large drainage canals were found to be similar to nearby borrow pit ponds; the dominant species were non-native cichlids and large predaceous native fishes (Ceilley et al., 2007). Together, the fish assemblages in these anthropogenic habitats were different from those of natural refugia such as popash ponds and willow ponds (Ceilley, 2007). Aerial wading bird surveys in this same area found that natural refugia (e.g., cypress domes and strands) were preferred over deep canals and borrow pits for foraging (Ceilley et al., 2007). Rehage and Trexler (2006) found that large (mostly predaceous) fishes were abundant in the canal systems, but only moved short distances into the adjacent marshes using deeper airboat trails for migration. It is clear that deep-water canals provide aquatic refugia for non-native fishes in South Florida (Harvey et al., 2010). Canals and borrow pit lakes may also provide thermal refugia for cichlids in areas where air temperatures might exceed lower lethal limits for brief periods during winter cold fronts. Shafland and Pestrak (1982) and Schofield et al. (2010) found that the most important factor regulating non-native fishes in Florida was limited cold tolerance and they identified lower lethal temperatures for several cichlids including species of *Cichlasoma*, *Tilapi*a and *Hemichromis*.

The landscape changes and associated effects to fish communities described above are relevant at the Babcock Ranch, one of the largest contiguous parcels remaining in Florida, totaling ~37,000 ha and located in Charlotte and Lee Counties, north of the Caloosahatchee River (FIG. 1). Freshwater creeks or streams connect with the Caloosahatchee River and freshwater and estuarine fishes can move through these systems during much of the year. Historically, portions of the ranch were poorly drained and generally unmanageable because of seasonally-high flood waters. As ditching and canal excavation occurred on the ranch, many wet prairie areas and historic hydric pine communities were converted into upland agricultural uses. In addition to the conversion of wetlands for agricultural purposes, the remaining wetlands are likely influenced by the ditching. In the Picayune Strand Restoration Project in Collier County, Florida the drainage canals (uniformly 4 m deep) were found to decrease water levels and hydroperiods in adjacent wetlands up to 3.2 km away (Duever, unpublished data). Hydrologic connection between canals and wetlands had also been severed by berms and

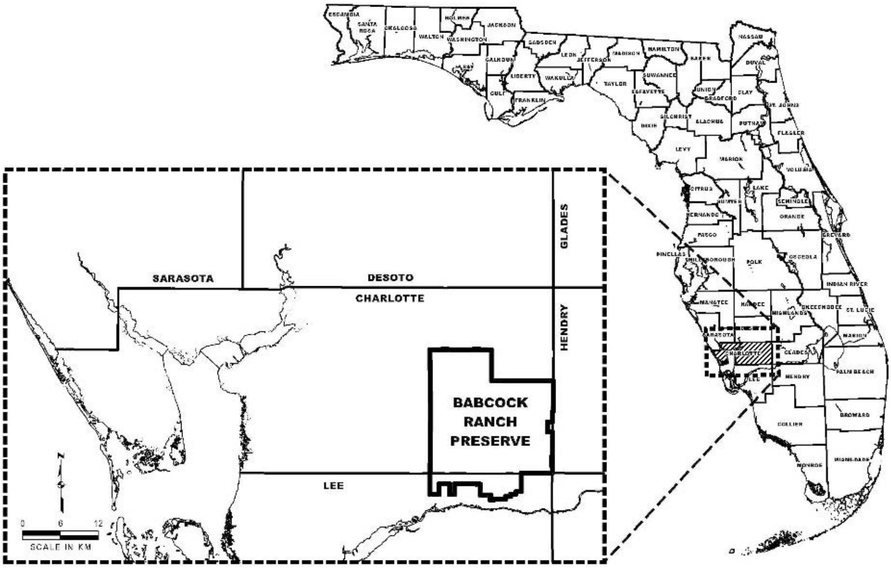

FIG. 1. Map of Babcock Ranch in Charlotte and Lee Counties, Florida.

road construction which prevented sheet-flow and migration of fishes between habitats during the wet season (Ceilley, 2008). The system of ditches and shallow drainages at Babcock Ranch allows manipulation of water levels for ranching and agriculture, but has likely impacted the hydrology of wetlands and natural creeks. These drainage features have had undetermined impacts on the aquatic ecosystem trophic structure.

During 2005 and 2006, the State of Florida, Charlotte County, and Lee County worked collaboratively with Kitson and Partners, LLC to form a public-private partnership which resulted in: 1) the public purchase of 29,775 ha of the Babcock Ranch, to be held by the State of Florida and Lee County in perpetuity for conservation purposes; and 2) the private acquisition of the remaining 7,198 ha of the Babcock Ranch, to be developed into a self-sustainable, environmentally conscious, mixed-use community now known as the Babcock Ranch Community. The portion of the land owned by the State of Florida and Lee County is called the Babcock Ranch Preserve. Hydrologic restoration projects are currently underway to help rehydrate drained wetlands within the Babcock Ranch Preserve and also to restore base flows to Trout Creek. This study evaluated the fish community structure of natural creeks/streams, an impacted stream, and agricultural drainage ditches at Babcock Ranch Preserve as part of a larger baseline assessment of aquatic and wetland fauna, prior to any restoration actions or the development of Babcock Ranch Community. We hypothesized that fish species assemblages would be different between natural and anthropogenic aquatic habitats and that agricultural canals would have more non-native species.

METHODS—*Field sampling*—Fish communities were monitored across Babcock Ranch between October 2006 and February 2011. Fishes were collected at 31 sites, three times per year for a total of 14 sampling events to document species distribution and seasonal fluctuations in aquatic fauna communities (Ceilley and Johnson Eng. Inc., 2010). Study sites included cypress strands, cypress domes, marshes, small streams, and manmade drainage ditches and canals. As part of the larger baseline study of the aquatic fauna from across Babcock Ranch, we decided to only compare the lotic fish assemblages of the agricultural drainage canals and natural streams. The six lotic systems included two natural streams (Owl Creek and Telegraph Creek), three drainage canal sites (Curry Canal, Big Island Canal and Big Island Weir), and one impacted stream (Trout Creek) that receives direct discharge from upstream Curry Canal (FIG. 2). All of these lotic systems are hydrologically connected downstream to the Caloosahatchee River in Lee County. The canals are linear drainage ditches that bisect upland and wetland habitats with berms constructed from side-cast spoil. Canal depths ranged from 0.5 to 1.5 m during the sampling periods, depending on season and rainfall. They were typically devoid of submergent and emergent vegetation, but contained isolated patches of *Ludwigia repens* and filamentous algae on snags and rocky riffle areas where cattle and road crossings exist. The natural streams were sinuous, closed canopy systems with little if any macrophytes or algal growth except in open crossings. Streams were approximately the same depth as canals but were narrower, with more snags and roots along their banks. Sampling methods for fishes were adopted from the baseline assessment of aquatic fauna for the Picayune Strand Restoration Project (Ceilley, 2008).

Ten clear plastic fish traps (Breder, 1960) were deployed in a stratified manner along streams and canals to sample all available habitat, but avoiding fast flowing areas that could dislodge traps. Pools, undercut banks, snags, and vegetated areas were sampled to increase the likelihood of successful collections. Traps were set during daytime hours between 0900 and 1600 Eastern Standard Time for a period of 1 h at each site to sample fish communities. Fish sampling was also conducted using dip nets for a period of 1 h while traps were deployed. Supplemental sampling used seines, funnel traps, and cast nets in order to build a complete species list for each site. The supplemental sampling was not standardized and therefore not included in the statistical analyses. We sampled three times during the hydrologic year to capture changes in fish assemblages from August through February as water levels increase and then decline. We assessed the impacts of two unusual cold events in both the winters of 2009 and 2010 to evaluate seasonal changes in fish community structure that may have resulted from fish kills. Water quality was not sampled on a regular basis, but some daytime temperature data was collected following unexpected cold events in January 2009 and February 2010 using a YSI model 85 temperature/DO/salinity meter.

Statistical analysis—Univariate and multivariate data analyses were conducted using Primer v6 (Clarke and Gorley, 2006). Abundance data from the combined 10 Breder traps and dip netting for all sampling events were fourth-root transformed to down weight the importance of extremely abundant species. A Bray-Curtis similarity matrix was then used as a basis for comparison of the fish communities between sites and lotic habitat types. Hierarchical agglomerative clustering of the similarity matrix was employed to construct cluster diagrams of percent similarity between groups using the group-average method. The similarity-profile random permutation test (SIMPROF) was used to identify statistically significant linkages in the cluster diagram. In addition, similarity percentage analysis (SIMPER) was used to describe species contributions to the similarity within and dissimilarity between groups. Two-dimensional non-metric multidimensional scaling (MDS) was also used to graphically display spatial and temporal similarities (groupings) and dissimilarities (distances) in fish communities.

RESULTS—A total of 9,059 fish were collected, including 26 fish species, representing 14 families including seven non-indigenous (exotic) fish species from four families (TABLE 1). The most dominant species in terms of total abundance were the native *Gambusia holbrooki* and the invasive exotic *Hemichromis letourneuxi*.

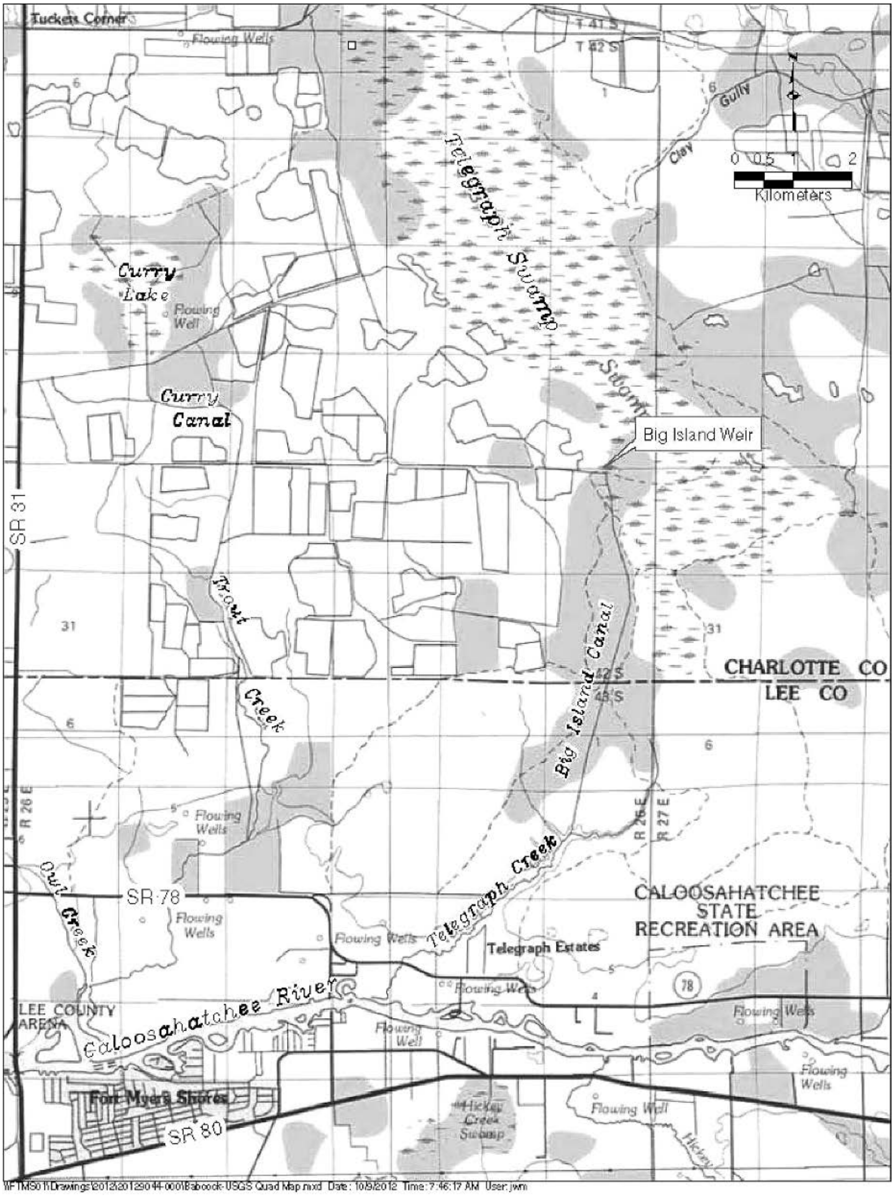

FIG. 2. Lotic habitats for fish sampling on Babcock Ranch and proximity to the Caloosahatchee River.

Fish assemblages in natural streams differed from those in canals and impacted streams. A cluster analysis of all sampling events combined for each location identified two significantly different fish communities (FIG. 3). Even though geographically far apart and in different sub-basins, Owl Creek and

TABLE 1. Fish species collected at Babcock Ranch from all aquatic habitats, 2006–2011. Fishes are sorted by phylogeny. Asterisk indicates non-native species.

Family	Genus	Species	Common Name
Lepisosteidae; Gars	*Lepisosteus*	*platyrhincus*	Florida gar
Cyprinidae; Minnows	*Notropis*	*petersoni*	coastal shiner
Ictaluridae; Bullhead Catfishes	*Ameiurus*	*natalis*	yellow bullhead
Clariidae; Labyrinth Catfishes*	*Clarias*	*batrachus*	walking catfish
Callichthyidae ; Armored Catfishes*	*Hoplosternum*	*littorale*	brown hoplo
Loricariidae; Suckermouth Catfishes*	*Pterygoplichthys*	spp. (juv)	Ptero
Fundulidae; Topminnows	*Fundulus*	*chrysotus*	golden topminnow
	Fundulus	*confluentus*	marsh killifish
	Lucania	*goodei*	bluefin killifish
Poeciliidae; Livebearers	*Gambusia*	*holbrooki*	Eastern mosquitofish
	Heterandria	*formosa*	least killifish
	Poecilia	*latipinna*	sailfin molly
Cyprinodontidae; Minnows	*Jordanella*	*floridae*	flagfish
Centrarchidae; Sunfishes	*Lepomis*	*gulosus*	warmouth
	Lepomis	*macrochirus*	bluegill
	Lepomis	*marginatus*	dollar sunfish
	Lepomis	*microlophus*	redear sunfish
	Lepomis	*punctatus*	spotted sunfish
	Lepomis	spp. (juv)	sunfish (juv.)
	Micropterus	*salmoides*	largemouth bass
Percidae; Darters	*Etheostoma*	*fusiforme*	swamp darter
Elassomatidae: Pygmy Sunfishes	*Elassoma*	*evergladei*	Everglades pygmy sunfish
Cichlidae; Cichlids*	*Cichlasoma*	*bimaculatum*	black acara
	Cichlasoma	*urophthalmus*	Mayan cichlid
	Hemichromis	*letourneuxi*	African jewelfish
	Oreochromis	*aureus*	blue tilapia
Soleidae	*Trinectes*	*maculatus*	hogchoker

Telegraph Creek fish communities shared 82% similarity and were significantly different ($P<0.05$) than the canal and impacted stream fish communities. Curry Canal, Big Island Canal, Big Island Weir and Trout Creek fish communities grouped together at just more than 72% similarity and were not significantly different from each other. Trout Creek and Curry Canal fish communities grouped together at 78% similarity and were different ($P<0.10$) than the Big Island Canal and Big Island Weir sites. An MDS ordination of these six sampling locations further illustrates the community differences between natural stream sites and impacted streams and canals (FIG. 4). The two-dimensional stress level of 0.01 indicates that it is an excellent ordination with no real prospect of misleading interpretation (Clarke and Gorley, 2006). The high similarity of fish communities at natural streams sites Owl Creek and Telegraph Creek is represented by a tight grouping in the ordination while the disturbed stream site at Trout Creek grouped in the center of the three canal sites.

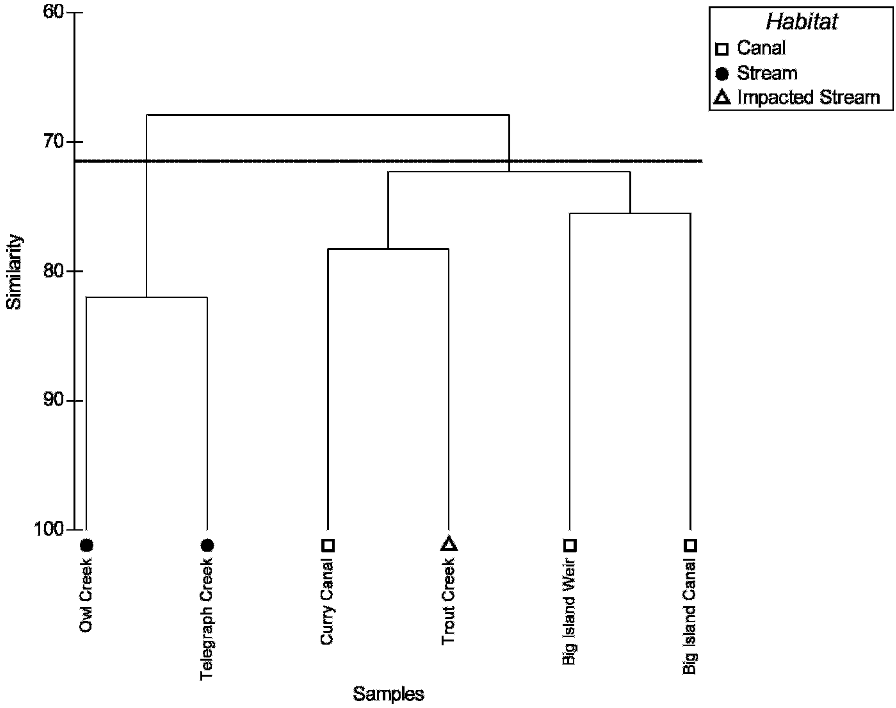

FIG. 3. Cluster diagram based on Bray-Curtis similarity of fish communities from lotic sites (streams, canals and impacted stream) with slice inserted at 72% similarity to illustrate two different groups (P<0.05).

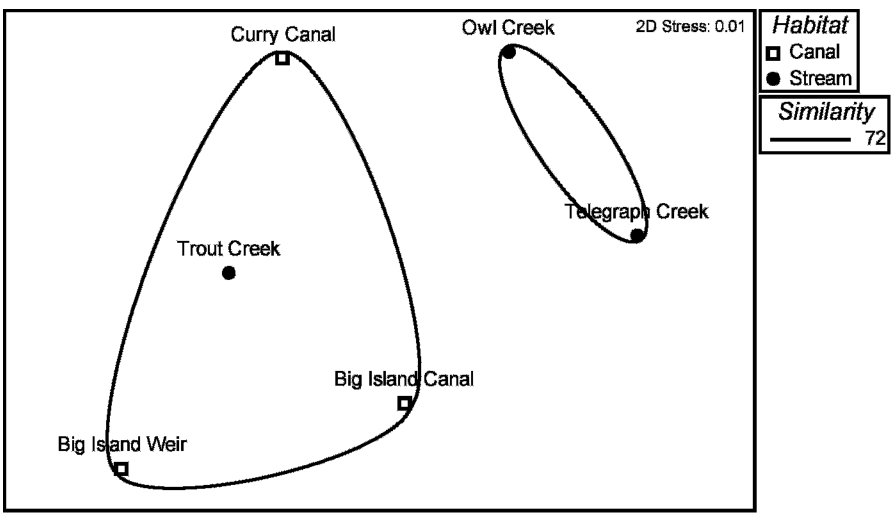

FIG. 4. MDS ordination of fish communities from lotic sites with overlay of significant groups (P<0.05) from the SIMPROF test.

TABLE 2. SIMPER results showing average abundance (fourth-root transformed) of fishes collected from streams, and species contributions to the similarity between stream sites (total average similarity = 82.01%).

Species	Avg. Abundance	Contribution %	Cumulative %
Hemichromis letourneuxi	2.50	14.99	14.99
Gambusia holbrooki	2.50	14.61	29.60
Poecilia latipinna	1.41	8.12	37.72
Lepomis marginatus	1.26	7.69	45.41
Jordanella floridae	1.38	7.38	52.80
Lepomis punctatus	1.14	6.79	59.59
Lepomis gulosus	1.16	6.42	66.01
Lepomis macrochirus	1.14	6.21	72.21
Fundulus chrysotus	0.92	5.4	77.61
Lepomis sp. (juvenile)	0.93	4.72	82.33
Etheostoma fusiforme	0.68	4.22	86.55
Lepomis microlophus	0.82	3.82	90.37

The SIMPER analysis produces a ranking of individual species contributions to the overall similarity between sites within a habitat type, which provides an indication of the relative importance of that fish species contribution to community structure. The SIMPER analysis identified that Owl Creek and Telegraph Creek were 82% similar in fish community structure with twelve species contributing to 90% of the similarity (TABLE 2). *H. letourneuxi*, and *G. holbrooki* each contributed 15% to the overall similarity, followed by *Poecilia latipinna* (8%), *Lepomis marginatus* (8%), and *Jordanella floridae* (7%) for a total of 53% of the similarity. Excluding the impacted stream site at Trout Creek, the canal sites shared 71% overall similarity in fish communities with only eight species responsible for 93% of that similarity (TABLE 3). *G. holbrooki* contributed 24%, followed by *H. letourneuxi* with 21% and *P. latipinna* with 12% for a total of 57% of the total similarity of canal fish communities between the three sites.

The dissimilarity information provided in the SIMPER analysis was useful for determining which species accounted for the differences between the

TABLE 3. SIMPER results showing average abundance (fourth-root transformed) of fishes collected from canals, and species contributions to the similarity between canal sites (total average similarity = 70.96%).

Species	Avg. Abundance	Contribution %	Cumulative %
Gambusia holbrooki	2.84	24.33	24.33
Hemichromis letourneuxi	2.41	20.77	45.10
Poecilia latipinna	1.60	11.81	56.91
Jordanella floridae	1.30	9.99	66.90
Lepomis gulosus	0.80	7.75	74.65
Lucania goodei	0.72	6.73	81.38
Fundulus chrysotus	0.84	6.38	87.76
Heterandria formosa	0.85	6.05	93.81

TABLE 4. SIMPER results showing average abundance (fourth-root transformed) of fishes collected from canals and streams, and species contributions to the dissimilarity between canals and streams (total average dissimilarity = 32.74%).

Species	Avg. Abundance Canals	Avg. Abundance Streams	Contribution %	Cumulative %
Lepomis marginatus	0.17	1.26	10.00	10.00
Heterandria formosa	0.85	0.00	7.95	17.95
Lepomis punctatus	0.41	1.14	6.81	24.75
Ethostoma fusiforme	0.00	0.68	6.16	30.92
Lepomis microlophus	0.23	0.82	5.59	36.50
Cichlasoma bimaculatum	0.00	0.60	5.40	41.90
Lepomis macrochirus	0.61	1.14	5.31	47.21
Lucania goodei	0.72	0.26	4.26	51.47
Cichlasoma urophthalmus	0.17	0.47	4.21	55.68
Lepomis sp. (juvenile)	0.53	0.93	4.12	59.80
Trinectes maculatus	0.20	0.61	4.07	63.87
Poecilia latipinna	1.60	1.41	3.93	67.80
Gambusia holbrooki	2.84	2.50	3.85	71.65
Ameiurus natalis	0.31	0.26	3.60	75.25
Lepomis gulosus	0.80	1.16	3.28	78.53
Micropterus salmoides	0.17	0.52	3.26	81.79
Jordanella floridae	1.30	1.38	3.14	84.93
Hemichromis letourneuxi	2.41	2.50	3.05	87.97
Hoplosternum littorale	0.23	0.32	2.97	90.94

natural stream and canal fish assemblages. The dissimilarity between canal and unimpacted stream fish communities for the entire study period was 33% (TABLE 4). Although *G. holbrooki* and *H. let*ourneuxi were the most abundant species in the study, their abundances in the two habitats were virtually tied, and so they did not contribute to the dissimilarity. *Lepomis* species were identified by SIMPER as major contributors to the dissimilarity between habitat types, streams and canals. *Lepomis marginatus* was more abundant on average in the stream habitats and the most important contributor (10%) to the overall dissimilarity. Three other *Lepomis* species were identified by SIMPER as major contributors to the difference in fish communities including, *L. puntctatus* (7%), *L. microlophus* (6%), and *L. macrochirus* (5%) which were more abundant in the streams than canals. *Etheostoma fusiforme* (6%) and *Cichlasoma bimaculatum* (5%) were also important contributors to the dissimilarity and were not collected from the canal sites during the entire study. Conversely, *Heterandria formosa* contributed 8% to the overall dissimilarity and was collected only from the canal sites. *Lucania goodei* was more abundant in the canals and contributed 4% to the dissimilarity in fish communities. Eleven other fish species contributed between 3% and 4% to the overall dissimilarity.

Observations associated with record cold weather—In January 2009, there was a record-setting severe cold front in southwest Florida, but it was short in duration. On January 21, 2009, the nighttime air temperatures at Punta Gorda,

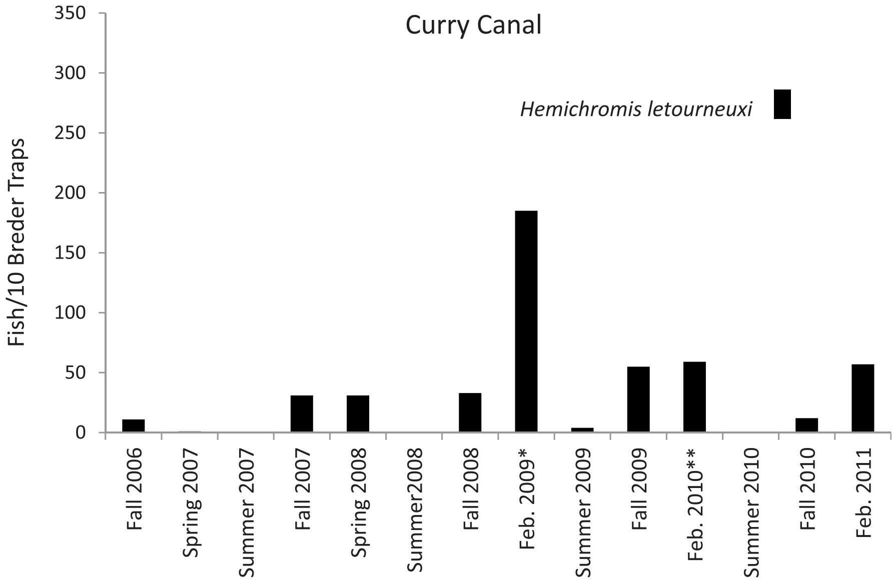

FIG. 5. Abundance of *Hemichromis letourneuxi* at a Curry Canal site over time from November 2006 through February 2011 (*post 2009 freeze event; **post 2010 freeze event).

Florida dropped to 0.6 °C, and on January 22 and 23, 2009 air temperatures dropped to −3.3 °C and 0.6 °C, respectively (NCDC 2011). Average low air temperature in January 2009 was 9.2 °C. Fish sampling followed that cold event, and at most lotic sites we observed varying degrees of fish kills. In all cases the fish kills were non-native cichlids, and predominantly *H. letourneuxi.* The fish kills were most visible at the natural stream sites. While canal sites also had small numbers of dead cichlids along the banks and shallows there were large and active schools of *H. letourneuxi* observed in the deeper pools of Curry Canal and Big Island Canal. The capture rate of *H. letourneuxi* at Curry Canal in Breder traps increased from a total of 33 fish in the November of 2008 to 185 fish in February 2009 (FIG. 5). At the same time, the *H. letourneuxi* population decreased in all of the natural streams after the 2009 cold event. At Telegraph Creek, the capture rate decreased from 306 *H. letourneux* in November of 2008 to two *H. letourneuxi* following the 2009 cold event (FIG. 6) with dozens of dead fish observed on the stream bottom. Similar results were observed at the other streams sites with decreases from 185 to 3 individuals at Trout Creek, and 163 to 32 individuals at Owl Creek following the 2009 cold event. During the period of January 27 and February 3, 2009, water temperatures taken during fish collections ranged 11–13 °C in the streams (Owl Creek, Telegraph Creek, and Trout Creek), and 15–21 °C in the canals.

Beginning on January 2, 2010, South Florida experienced a prolonged cold weather period that lasted for two weeks. January 10, 2010 was the coldest day throughout South Florida since December 24, 1989 (NCDC, 2011). In January 2010, nighttime low air temperatures averaged 7.7 °C in nearby Punta Gorda.

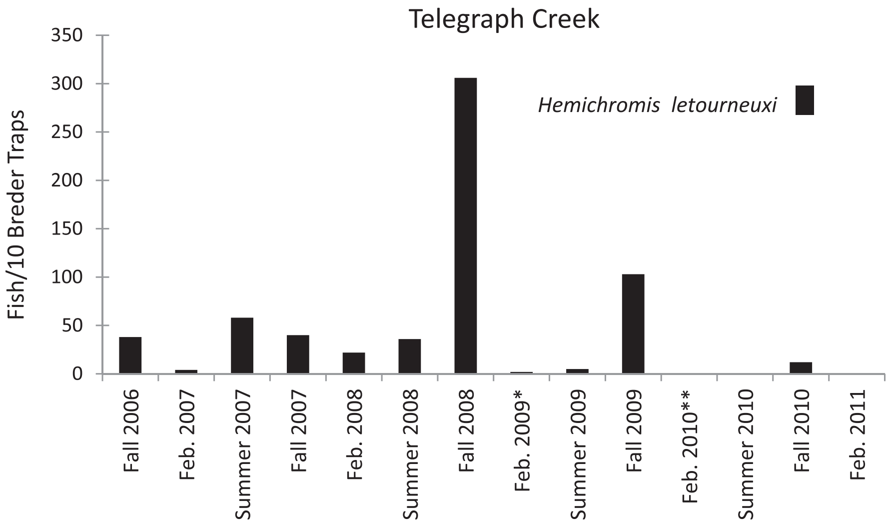

FIG. 6. Abundance of *Hemichromis letourneuxi* at Telegraph Creek over time from November 2006 through February 2011 (*post 2009 freeze event; **post 2010 freeze event).

However, during the two week cold event the nighttime low air temperatures averaged 1.9 °C and at least five times dropped below 0.0 °C. Water temperatures were recorded on one day at all locations after fish sampling was completed in February. The natural creek systems had daytime temperatures ranging 11–15 °C while the canal systems temperatures on the same day ranged 21–24 °C. The water temperatures in the canal habitats were typically 6–7 °C warmer than in the natural streams during the daytime when fish sampling was conducted. Again, we observed massive die-offs of *H. letourneuxi* at Owl Creek, Trout Creek, and Telegraph Creek sites, but not in the canals.

We used percent composition of fish species to evaluate changes in community structure before and after two severe cold events in 2009 and 2010. These freeze events resulted in changes at the community level, which are illustrated by changes in percent composition, particularly for *H. letourneuxi* and native livebearers. Average percent composition of *H. letourneuxi* at the canal sites from 2006 to 2008, ranged from 7.3% to 63.9% (TABLE 5). Percent composition of *H. letourneuxi* following the cold event in 2009 and later in 2010 increased ~ 6× at Big Island Weir (from 7.3% to 43.6%) and almost 3× at Curry Canal from (7.3% to 21.0%). Percent composition of *H. letourneuxi* at the Big Island Canal site decreased slightly from an average of 63.9% (2006–2008) to 45.4% (2009–2011) composition. Greater representation of *H. letourneuxi* in the canals was even more evident when examining individual sampling events pre- and post-cold events. The paucity of other species in the canal habitats is represented by the low percent composition for all species, with the exception of the livebearers. *Gambusia holbrooki* ranged from 21.5% to 90.1% composition while *Heterandria formosa* and *Poecilia latipinna* were

TABLE 5. Percent (%) composition of the 26 fish species collected at Babcock Ranch in the canal sites before (2006–2008) and after (2009–2011) 2009 and 2010 cold events.

		Big Island Weir		Big Island Canal		Curry Canal	
Genus	Species	Before	After	Before	After	Before	After
Ameiurus	*natalis*	0.0	0.0	1.3	0.0	0.0	0.0
Hoplosternum	*littorale*	0.0	0.0	0.0	0.0	0.0	0.2
Lepisosteus	*platyrhincus*	0.0	0.0	0.0	0.0	0.0	0.0
Cichlasoma	*bimaculatum*	0.0	0.0	0.0	0.0	0.0	0.0
Cichlasoma	*urophthalmus*	0.0	0.0	0.0	0.0	0.1	0.0
Hemichromis	*letourneuxi*	7.3	43.6	63.9	45.4	7.3	21.0
Fundulus	*chrysotus*	0.0	0.2	0.0	1.9	0.6	0.2
Fundulus	*confluentus*	0.0	0.0	0.0	0.0	0.0	0.0
Lucania	*goodei*	0.9	0.0	0.6	0.1	0.0	0.2
Gambusia	*holbrooki*	90.1	41.5	21.5	47.4	73.6	66.0
Heterandria	*formosa*	0.0	8.2	0.1	0.5	0.1	0.0
Poecilia	*latipinna*	0.0	3.7	8.6	2.1	10.4	9.5
Jordanella	*floridae*	0.9	1.9	0.9	2.3	6.5	2.4
Lepomis	spp. (juv)	0.6	0.0	1.2	0.2	0.0	0.0
Lepomis	*gulosus*	0.3	0.9	0.6	0.1	0.0	0.3
Lepomis	*macrochirus*	0.0	0.0	0.6	0.0	1.2	0.0
Lepomis	*marginatus*	0.0	0.0	0.0	0.0	0.0	0.1
Lepomis	*microlophus*	0.0	0.0	0.4	0.0	0.0	0.0
Lepomis	*punctatus*	0.0	0.0	0.2	0.0	0.0	0.1
Micropterus	*salmoides*	0.0	0.0	0.0	0.0	0.1	0.0
Etheostoma	*fusiforme*	0.0	0.0	0.0	0.0	0.0	0.0
Trinectes	*maculatus*	0.0	0.0	0.0	0.0	0.1	0.0
Elassoma	*evergladei*	0.0	0.0	0.0	0.0	0.0	0.0
Pterygoplichthys	spp. (juv)	0.0	0.0	0.0	0.0	0.0	0.0
Clarais	*batrachus*	0.0	0.0	0.0	0.0	0.0	0.0
Notropis	*petersoni*	0.0	0.0	0.0	0.0	0.0	0.0
Oreochromis	*aureus*	0.0	0.0	0.0	0.0	0.0	0.0

less common at some sites ranging from 0.9% to 10.4% composition. At the stream sites, changes in community structure were also evident. Before the extreme cold events (2006–2008), the percent composition of *H. letourneuxi* ranged from 33.3% at Trout Creek to 42.3% at Owl Creek and 64.5% at Telegraph Creek (Table 6). After the cold event of 2009, the percent composition of *H. letourneuxi* decreased at all stream locations to 10.1% at Trout Creek, 14.2% at Owl Creek, and 26.2% at Telegraph Creek.

DISCUSSION—Breder traps were found to be an effective sampling gear for most of the freshwater species found in wetlands, ditches, canals and shallow streams at Babcock Ranch (Ceilley and Johnson Eng. Inc., 2010). Sargent and Carlson (1987) reported that Breder traps had the least amount of sampling bias when compared to other methods in tidal marsh habitats. Breder traps were also found to be to be a cost-effective, non-destructive and repeatable method for assessing fish communities in isolated wetland habitats across South Florida (Main et al., 2007) and for detecting the impacts of hydrologic disturbance on an Everglades restoration Project in SW Florida (Ceilley, 2008). While Breder traps function very well in shallow-water systems with dense emergent vegetation, they have many limitations in deep waters or rapidly flowing systems. These activity traps favor collection of gregarious and motile species over secretive or less active species and tend to under represent species richness and diversity. Water temperature did not appear to impact the effectiveness of Breder traps, but that was not part of the scope of this study. It seems clear that dilution of fish density occurs early in the rainy season while a concentration of fishes occurs later in the hydroperiod cycle through reproduction and receding water levels. Together, the dilution effect, reproduction rates, and dry-down concentration of fishes explains the capture rates by season. Three fishes not collected by Breder traps were *E. fusiforme*, *Trinectes maculatus*, and *Elassoma evergladei*. Active sampling with dip nets was successful in collecting these species, but at no locations were they found to be common or abundant. The combination of sampling methods appears to be an effective way of sampling fish communities in shallow canals and streams and together resulted in identifying significant differences between the habitat types.

Dunson and Travis (1991) emphasized the integrative role of biotic and abiotic factors in the development of faunal community structure. In some cases, species have broad environmental tolerances and can be ubiquitous throughout an area, which was the case for *G. holbrooki* in this study. Nevertheless, after sampling lotic sites over a five-year period, we found significant differences in fish communities between natural streams and manmade drainages (canals) that appear to be a function of physical features including depth, dry season volume, and hydrology. Since the Curry Canal discharges directly into the impacted stream, Trout Creek, it is not surprising to see similarity in fish assemblages at these sampling sites. The physical attributes of manmade canals at Babcock Ranch include linear uniform width

TABLE 6. Percent (%) composition of the 26 fish species collected at Babcock Ranch in the stream sites before (2006–2008) and after (2009–2011) 2009 and 2010 cold events. Fishes are sorted by phyogeny.

Genus	Species	Trout Creek		Owl Creek		Telegraph Creek	
		Before	After	Before	After	Before	After
Lepisosteus	*platyrhincus*	0.0	0.0	0.0	0.0	0.0	0.2
Notropis	*petersoni*	0.0	0.0	0.0	0.0	0.0	0.0
Ameiurus	*natalis*	0.0	0.0	0.0	0.0	0.0	0.2
Clarais	*batrachus*	0.0	0.0	0.0	0.0	0.0	0.0
Hoplosternum	*littorale*	0.0	0.0	0.3	0.0	0.0	0.0
Pterygoplichthys	spp. (juv)	0.2	0.0	0.0	0.0	0.0	0.0
Fundulus	*chrysotus*	0.0	1.4	0.0	2.0	0.0	1.7
Fundulus	*confluentus*	0.0	0.0	0.0	0.0	0.0	0.0
Lucania	*goodei*	0.0	0.8	0.0	0.2	0.0	0.0
Gambusia	*holbrooki*	49.0	47.2	38.8	55.1	27.1	46.7
Heterandria	*formosa*	0.3	0.2	0.0	0.0	0.0	0.0
Poecilia	*latipinna*	11.5	22.3	4.8	5.1	1.2	6.9
Jordanella	*floridae*	5.0	16.0	7.8	2.0	1.4	3.7
Lepomis	*gulosus*	0.2	0.2	1.3	4.6	0.4	2.8
Lepomis	*macrochirus*	0.0	0.0	2.0	3.2	0.9	1.5
Lepomis	*marginatus*	0.0	0.2	0.0	6.4	0.6	6.0
Lepomis	*microlophus*	0.0	0.0	0.3	2.2	0.0	0.4
Lepomis	*punctatus*	0.2	0.6	0.8	3.4	1.9	1.1
Lepomis	spp. (juv)	0.0	0.4	0.5	0.0	1.7	1.5
Micropterus	*salmoides*	0.0	0.0	0.0	0.2	0.0	0.2
Etheostoma	*fusiforme*	0.0	0.2	0.0	0.5	0.1	0.4
Elassoma	*evergladei*	0.0	0.0	0.0	0.0	0.1	0.0
Cichlasoma	*bimaculatum*	0.0	0.0	0.0	0.5	0.1	0.0
Cichlasoma	*urophthalmus*	0.0	0.0	1.0	0.3	0.0	0.0
Hemichromis	*letourneuxi*	33.3	10.1	42.3	14.2	64.5	26.2
Oreochromis	*aureus*	0.0	0.0	0.0	0.0	0.0	0.0
Trinectes	*maculatus*	0.3	0.4	0.3	0.2	0.0	0.2

and shorelines with consistent depth profiles. The substrate is comprised of soft bottom sediments in many areas presumably as a result of soil erosion created by direct surface water flows from smaller ditches that bisect farm fields and upland flatwoods. The canal vegetation can include patches of emergent grasses along the banks, submerged vegetation (*Ludwigia repens*), and algal mats on the bottom. These features may explain the greater abundance of *H. formosa* and *L. goodei* in the canals relative to stream habitat. These species are highly wetland-dependent and they are associated with dense vegetation and soft substrates (Lee et al., 1980; Main et al., 2007). *Lucania goodei* is reported to feed on periphyton and sometimes vascular plants (Lee et al., 1980). Loftus (2000) found only trace amounts of algae, periphyton and plant detritus in *L. goodei* stomachs and copepods, cladocerans and amphipods were dominant prey items. Both periphyton and small crustaceans were much more abundant in the canals than streams (Ceilley and Johnson Eng. Inc., 2010).

Natural streams and creeks are sinuous with riffles, pools, and snags with greater variability in width and depth than canals, which likely contributed to differences in fish community structure between the two habitat types. The greater abundance of *Lepomis* spp. in streams can be attributed to the features of the stream habitat. For example, *Lepomis* spp. are abundant in rivers with dynamic water level conditions and their presence correlates well with amount of snag habitat (Dutterer and Allen, 2008). Streams and creeks at Babcock Ranch have sandy bottoms in many areas, and contain limestone outcrops and ledges carved out by high current velocities. The sandy-bottom habitat found in the natural streams also provides more suitable nesting habitat for centrachids than the soft-bottom habitat of canals. Page and Burr (1991) report *E. fusiforme* to be common in coastal streams and associated with slow-flowing water over mud or sand. As a darter, they are a demersal species and feed along the bottom of waters they inhabit. *E. fusiforme* are predaceous and feed on small live animals including aquatic insects and crustaceans (Hoyer and Canfield, 1994) especially midge larvae and copepods and cladocerans (Boschung and Mayden, 2004). While the genus *Etheostoma* is reported to be the most speciose genus of fishes in North America with 132 described species (Boschung and Mayden, 2004), *E. fusiforme* is the only darter know from South Florida, and as such, may serve as an indicator of stream quality. The higher current velocities, greater diversity of habitat, and sandier substrate in the natural streams, relative to canals, may explain the greater abundance of *E. fusiforme* there (Lee et al., 1980, Boschung and Mayden, 2004). The non-indigenous *C. bimaculatum* was present in low numbers in the streams, but not in the canals. This finding is difficult to explain, but the absence of *C. bimaculatum* from the canals could occur from interspecific competition with another non-indigenous species (*H. letourneuxi*) which was very abundant in the canals and is known to be quite aggressive.

Results and observations before, during and after two severe cold events indicate that manmade canals serve as thermal refugia for non-native cichlids during cold weather and seasonal dry-down. During freezes that occurred in

2009 and 2010, daytime water temperatures in natural streams (11–13 °C) were recorded below the lower lethal limit for *H. letourneuxi*, as identified by Shafland and Pestrak (1982) and Schofield et al. (2010). Where a stream habitat was connected to canal habitat like at Trout Creek, the cold weather events may have triggered a migration to the warmer canal waters. Such a migration could account for the increased abundance of *H. letourneuxi* that occurred at canal sites just after the freeze. However, the Owl Creek and Telegraph Creek sites were not connected to canals and direct migration during periods of low water was not possible. We observed massive die-offs of *H. letourneuxi*, during and after the cold events in 2009 and 2010. Temperatures were consistently 6 to 7°C higher at the canal sites and we attribute that to greater exposure to sunlight, greater depth, larger volume, and surface water runoff through a series of agricultural ditches. We observed non-native fish mortality in the native stream habitats during and after prolonged cold weather events in 2009 and 2010. Prior to cold events in 2009 and 2010, there appeared to be a strong negative correlation between native *G. holbrooki* and non-native *H. letourneuxi* that should be explored further (Ceilley and Johnson Eng. Inc., 2010, Rehage et al., in press). Such a negative correlation could be attributed to direct predation, which we observed on numerous occasions, and possibly learned avoidance of predators (Rehage et al., 2009). We also observed direct predation and tail-biting of other species of native fishes by *H. letourneuxi* and what appeared to be secondary bacterial infections as a result of this tail-biting.

Based on five years of observation and fish sampling it appears that native fishes use natural streams as dry-season refugia, especially in the absence of *H. letourneuxi*. We suggest that the scarcity of native Centrarchidae (and other native fishes) within the canal systems at Babcock Ranch is an indication of disturbance, both hydrological and ecological. The removal of drainage canals that serve as thermal refugia and pathways for dispersal for *H. letourneuxi* should benefit native fish species over time, since they are well adapted to the natural hydrologic and seasonal cycles of SW Florida.

A public and private partnership currently exists at Babcock Ranch Preserve for land management purposes and the property is managed by Babcock Ranch Inc., a Florida not-for-profit corporation. The mitigation plan for development of the privately owned Babcock Ranch Community includes hydrologic restoration projects in the Curry Canal and Trout Creek flow-way (Johnson Eng. Inc., 2009). As restoration of flow-ways and wetland habitats begins at Babcock Ranch there will be opportunities for resource managers and policy makers to develop aquatic habitat restoration plans that favor the recovery of native fish communities over non-indigenous species. A comprehensive approach to aquatic habitat restoration should include: 1) the removal of non-critical drainage systems on the landscape; 2) restoration of headwater wetland hydrology to restore natural base-flows to streams and creeks; 3) creation of riparian buffers and restoration projects to enhance existing natural streams and creeks and prevent further degradation; and 4) enhancement of drainage systems that remain to mimic natural flow-ways and streams and protect all aquatic habitats from agricultural and residential degradation.

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MONITORING COLONIAL NESTING BIRDS IN ESTERO BAY AQUATIC PRESERVE

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ABSTRACT: *Estero Bay Aquatic Preserve is a shallow-water estuary which contains a diverse array of natural communities that make it an attractive environment for wading and diving birds to forage and nest. Nest surveys conducted in Estero Bay for the last four decades detect trends in wading and diving bird populations while engaging and educating the public through volunteerism. Brown pelican (*Pelecanus occidentalis*) nest counts conducted in May show a significant decrease in nesting pairs in Estero Bay; a loss of approximately 5 nesting pairs per year across the record of study. Comparisons of historic April nest counts to modern nest counts for great blue heron (*Ardea herodias*) show an increase of 158 percent, which represents a gain of 32 nesting pairs between the two time periods. Other species showing increasing trends in nest counts are yellow-crowned night heron (*Nyctanassa violacea*) and double-crested cormorant (*Phalacrocorax auritus*), while anhinga (*Anhinga anhinga*) showed a decreasing trend. Species-level analyses of more recent standardized monitoring provide a more detailed view of population trends in the bay including shifts in species composition and peak nesting times. Future analyses should include nesting data collected by other agencies to assess nesting success on a larger geographical scale.*

Key Words: Estero Bay Aquatic Preserve, colonial nesting birds, wading birds, diving birds, volunteerism

ESTERO Bay was designated as Florida's first aquatic preserve in 1966 and is managed under the Florida Department of Environmental Protection's Office of Coastal and Aquatic Managed Areas. The Estero Bay Aquatic Preserve consists of approximately 11,000 acres of sovereign submerged lands and is located in southwest Florida, extending from Fort Myers Beach to Bonita Springs (FIG. 1). The shallow estuary is designated as an Outstanding Florida Waterbody by the Florida Department of Environmental Protection (FDEP, 1997a) and is fed by five freshwater tributaries and four passes connecting to the Gulf of Mexico. Estero Bay contains mangrove islands, nineteen of which have been documented as breeding colonies for a variety of bird species, including 10 species of wading birds and three species of diving birds. Long-term monitoring data of wading and diving bird populations is an important resource for aquatic preserve managers who are tasked with preserving the bay in its "*essentially natural or existing condition so that its aesthetic, biological and scientific values may endure for the enjoyment of future generations*" (FDEP, 1997b).

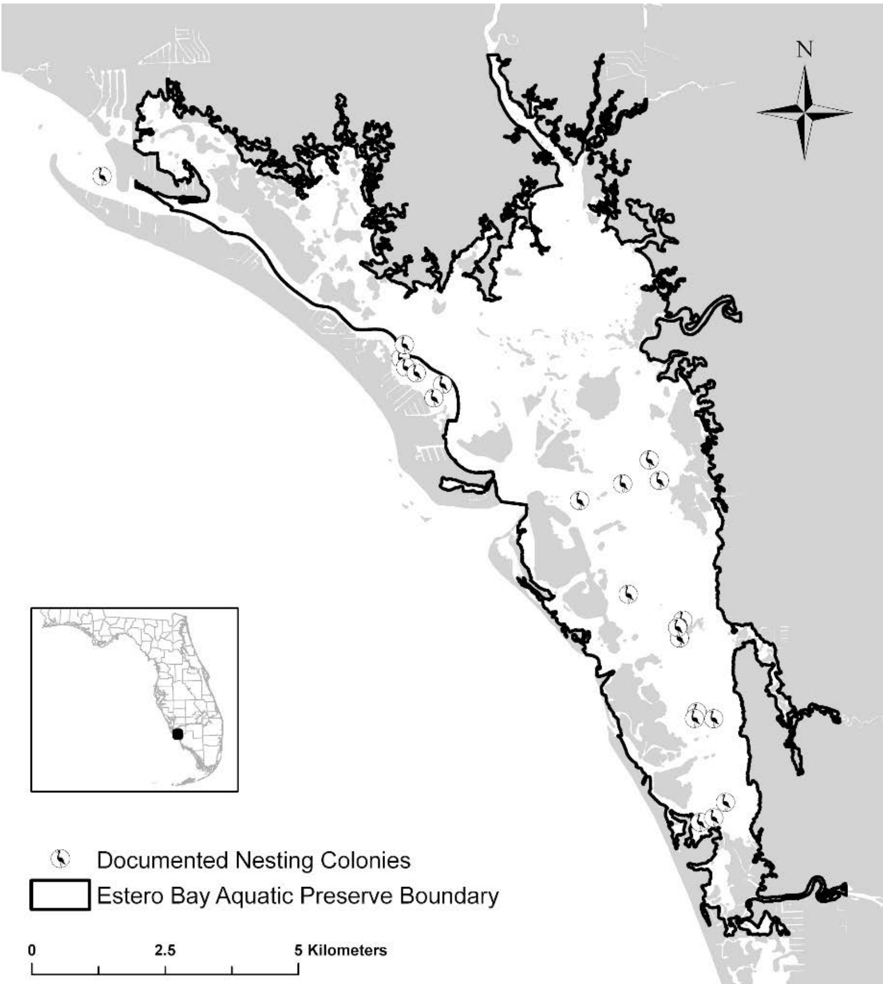

FIG. 1. Extending from Fort Myers Beach to Bonita Springs, Estero Bay Aquatic Preserve consists of approximately 11,000 acres of sovereign submerged lands located in southwest Florida. Nineteen nesting islands have been documented in Estero Bay since the induction on surveys in 1977; 13 of these islands are located within the aquatic preserve boundary and all consist primarily of mangroves.

In the late nineteenth century, after 40 years of plume hunting, wading birds became a focal point for conservation. In the 1970s, extensive colonial nesting bird surveys were initiated along the North American Atlantic and Gulf coasts (Kushlan, 1997). Governmental and non-governmental agencies began collecting data on wading bird populations and decades of data set the standard for their use as an indicator species (Kushlan, 1993). Wading birds maintain a high aesthetic and recreational value to humans and their reproductive performance is a crucial aspect of their population dynamics

(Kushlan, 1993). During the nesting season, breeding adults are using a limited portion of their environment and populations are concentrated (Kushlan, 1997; Urfi et al., 2005), making them an ideal subject for volunteer-based management studies. Nesting surveys in Estero Bay began in 1977 and the program implemented a variety of survey techniques throughout its history. Since 2008, Estero Bay Aquatic Preserve staff and volunteers have conducted monthly nest counts throughout the nesting season. This program provides peak estimates of nesting effort for each species of colonial nesting bird, monitors population trends, maintains a current atlas of historic and active colonies, documents human disturbance, documents the number of entanglements and fatalities due to fishing-line and trash, and increases community involvement through volunteerism and by engaging and educating the public.

Colonial nesting wading birds are particularly susceptible to local human disturbances (Parnell et al., 1988). This is a concern in Estero Bay since boats bring tour groups to select colonies several times a day, and other recreational activities such as photography, camping, kayaking, paddle boarding and personal watercraft use frequently take place within the 100 meter buffer suggested for nesting wading birds (Erwin, 1989; Rodgers and Smith, 1995; Burger, 1998; Carney and Sydeman, 1999). Disturbances in early nest building and incubation periods can cause nest desertion (Steinkamp et al., 2003) and frequent disturbance may cause a reduction in clutch size and hatching success (Schreiber and Risebrough, 1972). Predation of eggs by fish crows (*Corvus ossifragus*) when adult birds are flushed from the nest due to disturbance was noted by Schreiber and Risebrough (1972) as the leading cause of egg loss.

MATERIALS AND METHODS—*Historic surveys*—Annual nesting surveys conducted between 1977 and 1982 were performed in May (TABLE 1); file notes indicate that perimeter counts were employed and only brown pelican (*Pelecanus occidentalis*) nests were recorded. Counts were conducted once annually in late May 1983 through 1987 using ground survey methods; colonies were entered on foot to document active wading and diving bird nests. May 1989 counts were conducted using a perimeter count method to survey all active nests. No written protocols have been documented for May surveys conducted between 1977 and 1989; all method descriptions are based on field notes and written communications. Surveys conducted on 03 April 1989 and 06 March 1998 did not include all active nesting islands in the area, or data were lost, so results are not included in these analyses. Surveys conducted on 29 April 1998, 24 April 2001 and 18 April 2007 employed a direct count method as described in the National Audubon Society's Project Colony Watch, later published as Audubon of Florida (2004): "*Colonies on coastal islands can be circled repeatedly by boat until counts are complete…ask multiple observers to compare counts, and use cooperation to arrive at reasonable estimates.*" Surveys conducted in 2002 on 05 June, 01 July, 04 July and 14 July were conducted by canoe using a perimeter count method; however, surveys were only conducted in the southern portion of the bay, so results were not used in these analyses. No nesting surveys were conducted in 1988, 1990 through 1997, 1999 through 2000, or 2003 through 2006.

Modern surveys—Surveys between 2008 and 2011 were conducted once mid-month throughout the nesting season. Each year, surveys were initiated when birds were observed carrying nesting materials and concluded when all chicks had fledged. Surveys were conducted using a direct count method as described by Audubon of Florida (2004). A 17-foot boat with an outboard and trolling motor was used to circle each island at a distance of 30 to 45 meters. Two

TABLE 1. Colonial nesting bird surveys conducted in Estero Bay and used for analysis in this paper; including survey methods employed as described by Steinkamp and co-workers (2003) and species counted. Surveys were conducted once per month at each known active nesting colony for years and months listed. (BRPE=brown pelican, DCCO=double-crested cormorant, ANHI=anhinga, GBHE=great blue heron, GREG=great egret, SNEG=snowy egret, LBHE= little blue heron, TRHE=tri-colored heron, REEG=reddish egret, CAEG=cattle egret, YCNH=yellow-crowned night-heron, BCNH=black-crowned night-heron, GRHE=green heron).

Year	Month(s)	Method Employed	Species Counted	Comments
1977	May	Perimeter	BRPE	No protocol available
1978	May	Perimeter	BRPE	No protocol available
1979	May	Perimeter	BRPE	No protocol available
1980	May	Perimeter	BRPE	No protocol available
1981	May	Perimeter	BRPE	No protocol available
1982	May	Perimeter	BRPE	No protocol available
1983	Late-May	Ground	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG	No protocol available; written corres-pondence indicates "walked through the islands"
1984	Late-May	Ground	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG	No protocol available; written correspondence indicates "I walked through the islands"
1985	Late-May	Ground	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG	No protocol available; written correspondence indicates "I walked through the islands"
1986	Late-May	Ground	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG	No protocol available; written correspondence indicates "I walked through the islands"
1987	Late-May	Ground	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG	No protocol available; written correspondence indicates "I walked through the islands"
1989	May	Perimeter	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG	No protocol available; only BRPE were included in analyses
1997	May	Perimeter	BRPE	No protocol available
1998	29, April	Perimeter	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG, YCNH, BCNH, GRHE	Direct Count Method
1998	May	Perimeter	BRPE	No protocol available
2001	24, April	Perimeter	BRPE, DCCO, ANHI, GBHE, GREG, SNEG	Direct Count method LBHE,TRHE, REEG - species were present but nest counts were not conducted

TABLE 1. Continued.

Year	Month(s)	Method Employed	Species Counted	Comments
2007	18, April	Perimeter	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG, YCNH, BCNH, GRHE	Direct Count method as described by Audubon of Florida (2004)
2008	March–August	Perimeter	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG, YCNH, BCNH, GRHE	Direct Count method as described by Audubon of Florida (2004)
2009	February–August	Perimeter	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG, YCNH, BCNH, GRHE	Direct Count method as described by Audubon of Florida (2004)
2010	February–September	Perimeter	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG, YCNH, BCNH, GRHE	Direct Count method as described by Audubon of Florida (2004)
2011	January–September	Perimeter	BRPE, DCCO, ANHI, GBHE, GREG, SNEG, LBHE,TRHE, REEG, CAEG, YCNH, BCNH, GRHE	Direct Count method as described by Audubon of Florida (2004)

observers counted the number of nesting pairs, or nests, by species and nesting stage. The primary observer, an aquatic preserve staff member, was consistent throughout the study period, and trained volunteers conducted secondary observer counts. The average of the two observers' counts was reported; empty nests were not included in calculations. Monthly counts from 2008 through 2011 are compared only with corresponding monthly counts from historic datasets.

Data analysis—Statistics were calculated using IBM SPSS 19.0 and the USGS's KTRLine 1.0. Analyses were run for all individual species, canopy nesters, and interior nesters, as described by Burger (1978), McCrimmon (1978) and Spendelow et al. (1989), and total counts. Canopy nesters include great blue heron (*Ardea herodias*), great egret (*Ardea alba*), brown pelican, double-crested cormorant (*Phalacrocorax auritus*) and anhinga (*Anhinga anhinga*). Spendelow et al. (1989) separated great egrets from other canopy nesting species and McCrimmon (1978) found that their nesting patterns were significantly different from interior nesting species. However, Burger (1978) grouped great egrets with great blue herons as a canopy nesting species, and based on personal observations of nesting in Estero Bay great egret are classified as a canopy nesting species for these analyses. Interior nesters include snowy egret (*Egretta thula*), little blue heron (*Egretta caerulea*), tricolored heron (*Egretta tricolor*), reddish egret (*Egretta rufescens*), cattle egret (*Bubulcus ibis*), yellow-crowned night-heron (*Nyctanassa violacea*), black-crowned night-heron (*Nycticorax nycticorax*), and green heron (*Butorides virescens*).

Assumptions of normality were tested using the Jarque & Bera LM test. The assumption of homoscedasticity was tested using the Breusch-Pagan test. In addition, assumptions of linearity were examined using plots of observed versus predicted values and residuals versus predicted values. Assumptions of independence were assessed using the Durbin-Watson d test. An analysis comparing historic surveys to modern surveys used independent samples t-tests if assumptions were met, and Mann-Whitney U with Hodges-Lehmann estimator (for the confidence interval, CI) if assumptions were violated. In addition, analyses comparing "decades" (i.e., 1977–1979=1970s; 1980–1982=1980s; 1989–1998=1990s; and 2008–2011=2000s) for the month of May (brown pelican only) were conducted using ANOVA along with Gabriel's *post hoc* test if assumptions were met, and Kruskal-Wallis H with Mann-Whitney U and Hodges-Lehmann if assumptions were violated. If the transformed data did not satisfy all assumptions, robust nonparametric regressions (Theil-Kendall regression; KTRL) were used for detection of trends. However, if assumptions were met, ordinary least squares (OLS) regression was used. The data were further screened using ANOVA (on transformed data) in order to determine if there were statistically significant differences between: historic ground surveys (1983–1987) and historic perimeter surveys (1977–1982); historic brown pelican data for the month of May (1977–2001) and modern surveys (2008–2011); and historic data for all species for the month of April (1998 and 2001) and modern data (2007–2011). ANOVA by year was used on the modern data (2008–2011) to detect significant differences due to the different sampling periods, and to assess shifts in peak nesting by month, both 2-way factorial ANOVA and repeated measures mixed ANOVA were used.

RESULTS—All of the raw data were found to be non-normal, and most were heteroscedastic. However, assumptions of independence and linearity were rarely violated. Therefore, the data were transformed, and analyses were run on the transformed data. The historic ground survey data were significantly different from the historic perimeter surveys ($p<0.01$) and were removed from further analyses.

Comparison of historic and modern data—Historic brown pelican data for the month of May were significantly different from modern data ($p<0.01$). However, it is unclear if this is due to sampling technique or if this is because of a declining trend. Therefore, these data were included for analyses. The transformed brown pelican data for the month of May, from 1977–2011, did not violate assumptions, and were regressed using OLS. Comparisons of historic surveys to modern surveys for brown pelicans during the month of May show a mean decrease of 56.9 percent (SE=16.6, $p<0.01$, 95% CI=57.27–130.51), representing a significant decrease in nesting pairs of brown pelicans in Estero Bay for this time period (FIG. 2). Robust KTRL regression of the raw data showed this trend to be a loss of approximately five nesting pairs per year. In addition, when comparing decades, there was a 62.1 percent mean decrease between the 1970s and the 2000s (SE=21.4, $p<0.01$, 95% CI=61.29–171.38), a 54.3 percent mean decrease between the 1980s and the 2000s (SE=11.9, $p=0.01$, 95% CI=50.61–118.06), and a 53.3 percent mean decrease between the 1990s and the 2000s (SE=20.5, $p=0.02$, 95% CI=28.32–133.68). There were no statistically significant differences between the 1970s, 1980s, and the 1990s.

Analysis of April surveys show that nest counts for some species differed substantially between historic and modern periods (TABLE 2). Nest counts of double-crested cormorant ($p=0.05$), great blue heron ($p=0.05$), and yellow-crowned night heron ($p=0.04$) increased between 110 and 204 percent.

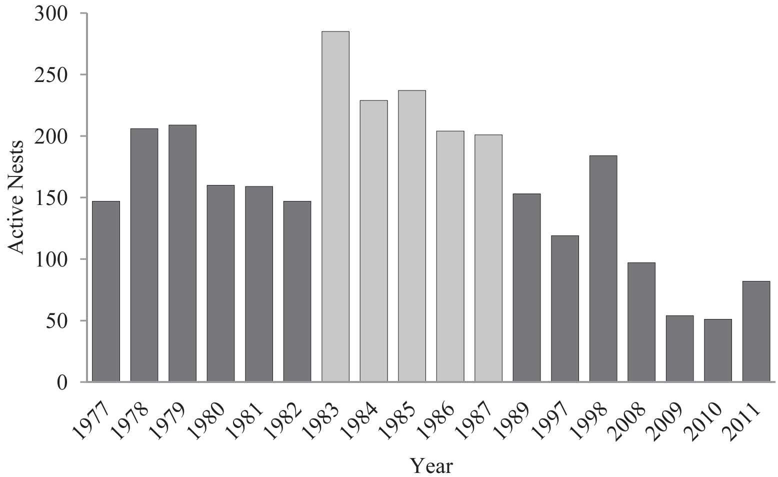

FIG. 2. May brown pelican (BRPE) surveys were conducted using perimeter surveys (black bars) and ground surveys (grey bars). Ground surveys conducted between 1983 and 1987 were significantly different from the historical perimeter surveys ($p<0.01$) and were removed from analyses. Comparison of historic perimeter surveys (1977–2001) and modern perimeter surveys (2008–2011) show a loss of 94 nesting pairs.

Modern data analysis—April nest counts were used to analyze annual trends in nesting success by species. All transformed data for the month of April, from 2007 to 2011, were non-normal; therefore, KTRL regression of the raw data was used. Regressions showed that black-crowned night-heron, little blue heron, and tricolored heron were increasing; there was no trend for cattle egret, green heron, or reddish egret; and brown pelican, double-crested cormorant, anhinga, great blue heron, great egret, snowy egret and yellow-crowned night-heron were declining. However, none of the KTRL regressions were significant.

In addition to testing for annual trends in nest counts by using April surveys only, an analysis of all monthly nest counts available in the modern surveys was conducted to detect any long-term trends. There were no significant differences between monitoring season length (i.e., nesting season) between years during the modern surveys, with the exception of great egret and brown pelican ($p=0.02$ and $p=0.02$, respectively); therefore, the entire dataset was analyzed. Transformed data of the modern surveys were non-normal; therefore, KTRL regression of the raw data was used. In the March 2008 through September 2011 modern survey record, there were declining trends in brown pelican, double-crested cormorant, great egret, canopy nesters, and total counts. However, these were not significant. There were no other significant trends for this time period.

Combining peak nest counts of all species during each year of the modern surveys revealed a 38 percent reduction in overall nesting activity: March through July 2008 ($N=534$), February through August 2009 ($N=428$), February through August 2010 ($N=424$), and January through August 2011

TABLE 2. Mean nest count, standard error, and percent mean difference, by species, for April historic surveys (1998 and 2001) and modern surveys (2007–2011). For the historic period, standard error that is null indicates that nest counts for that species only occurred in one of the two years, and SE that is equivalent to its respective mean results from a species that had a positive nest count in one year and a count of zero in the other. See Table 1 for abbreviations.

	Historic Surveys		Modern Surveys		Percent Difference
	Mean	Std. Error	Mean	Std. Error	
BRPE	158.0	32.0	83.2	28.6	−47.3
GBHE	23.5	11.5	60.6	6.7	157.9
TRHE	95	–	2.8	1.5	−97.1
LBHE	6	–	3.2	1.3	−46.7
SNEG	46.0	39.0	6.8	3.1	−85.2
GREG	35.0	20.1	39.8	8.7	13.7
REEG	9	–	2.2	1.2	−75.6
CAEG	50.0	50.0	1.4	1.0	−97.2
YCNH	2.5	2.5	7.6	1.0	204.0
BCNH	6.5	6.5	3.0	1.0	−53.9
GRHE	0.0	0.0	0.6	0.4	–
DCCO	27.0	4.0	56.8	9.7	110.4
ANHI	4.5	0.5	0.4	0.4	−91.1

(N=351). A shift in peak nesting time (FIG. 3) from March (2008) to April (2009) to June (2010), then back to April (2011), was observed in Estero Bay; this shift, however, was not statistically significant.

Shifts in species composition among canopy nesters, specifically brown pelican and great blue heron, were observed between 2008 and 2011 (FIG. 4). Brown pelican and great blue heron combined represent 43 to 51 percent of documented nests annually. The decline in brown pelican nesting success between 2008 and 2010 coincided with an increase in great blue heron nesting. Peak nest counts indicate that brown pelican were the dominant nesting species in 2008, 2009 and 2011. In 2010, great blue heron was the dominant nesting species.

DISCUSSION—*Methodology and scale*—Variation in survey methods, lack of written protocols, and gaps in data collection throughout the monitoring program in Estero Bay made interpretation of historic data difficult. Perimeter surveys have been the dominant survey technique implemented in Estero Bay over the past four decades. There are several limiting factors to using perimeter counts including detection probability and double counting (Nichols et al., 2000; Steinkamp et al., 2003). However, the reliability of nest counts using this method to assess populations is reasonably high (Urfi et al., 2005). May brown pelican surveys 1977–1982, 1989 and 1997 were conducted using a boat to circle islands and at least one experienced birder to count nesting pairs. However, detailed protocols and raw data are not available for these surveys. Multiple observer perimeter counts have been employed in Estero Bay since April 1998 when members of National Audubon Society began conducting and

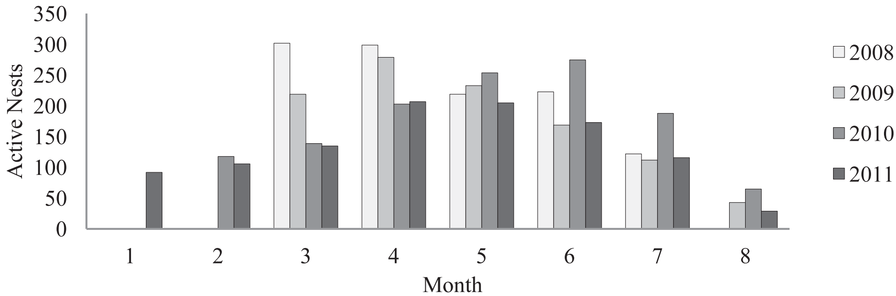

FIG. 3. Monthly nest counts of all species combined (March–August 2008, February–August 2009, February–August 2010 and January–August 2011). Peak nesting time shifted from March (2008) to April (2009) to June (2010), then back to April (2011); this shift however was not statistically significant.

assisting with surveys using the Project Colony Watch protocol. In 2008, the Project Colony Watch protocol was modified slightly when the program was expanded- instead of having multiple observers discuss nest counts and come to a conclusion, nest counts from two observers are recorded and averaged in an effort to reduce errors produced by one of the observers under counting or double counting nests, which were identified by Steinkamp et al. (2003) as problems with perimeter counts. Ground surveys were conducted between 1983 and 1987 by walking through the colonies to document nests. Surveys during this time period provided statistically higher nest counts (FIG. 2); however, it is unclear if this difference is due to survey method or increased nesting activity since a comparison of methods during that time period does not exist. Steinkamp et al. (2003) state that ground surveys provide the most accurate counts at nesting colonies and Urfi et al. (1997) recommend testing multiple census techniques to provide data for comparison of survey methods. Due to the level of disturbance and limitations associated with ground surveys,

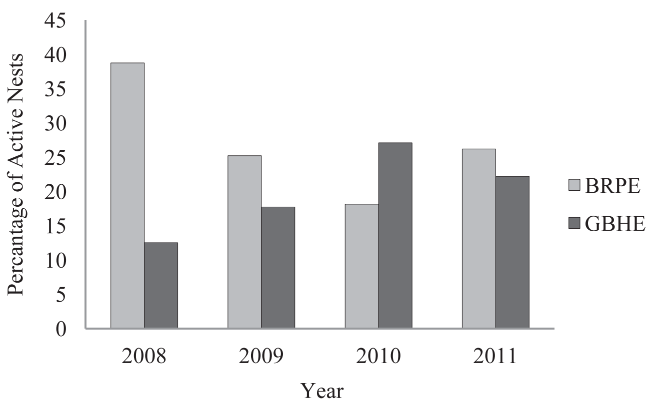

FIG. 4. Percentage of active brown pelican (BRPE) and great blue heron (GBHE) nests, based on peak annual nest numbers for each species, within Estero Bay colonies.

perimeter counts will continue to be the primary survey methods employed by aquatic preserve staff. However, periodic ground surveys may be employed in the future so that more nesting data are available for future analyses. Detection probability for the primary observers, at minimum, should be calculated as outlined by Nichols et al. (2000) annually to allow for additional analysis of data.

Expanding the colonial bird monitoring dataset to include a wider geographical range would allow resource managers to conduct analyses of population trends on a region-wide basis. In the present study, there were few significant trends; this is most likely due to the small size of the dataset. Understanding of trends in these nesting colonies may be bolstered with a larger dataset using standardized methods. Most species of colonial waterbirds are long-lived and decades of data are needed to evaluate significant changes in populations (Steinkamp et al., 2003). Some species of colonial waterbirds have complex annual cycles and wide geographic ranges (Kushlan, 1993), so continuing the monthly monitoring in Estero Bay in conjunction with data collected by Charlotte Harbor Aquatic Preserves and J.N. "Ding" Darling National Wildlife Refuge, who currently use the same survey techniques, would enable managers to examine nesting success on a larger geographical scale and assess whether colonies are moving to other areas.

Comparison of historic and modern data—Data comparisons between time periods provided gross trends for species currently listed by the state as species of special concern which are now under consideration for changes in protection status. Five decades of data collected on the brown pelican in Estero Bay show a significant decreasing trend in nesting for the species while comparisons of historic and modern April surveys showed the decrease in nesting activity was not significant during the shorter time period. In 2011, the Florida Fish and Wildlife Conservation Commission (FWC) reviewed the status of the brown pelican statewide and are recommending its removal from the Florida imperiled species list (FWC, 2011). Several species nesting in Estero Bay are being recommended for state listing as threatened. These include little blue heron, reddish egret, snowy egret, and tricolored heron. Although not significant, these species showed a decrease in nesting activity of between 46 and 97 percent from the historic to modern periods.

Trends in the number of nesting pairs for each species can result in overall changes in the species composition at the colonies, which can influence colony dynamics. For example, night-heron populations have shown an increase from April historic surveys to current surveys, which may affect the nesting success of other species since they have been documented as preying upon avian nests (Kelly et al., 1993) including other wading birds (Hall and Kress, 2008). Hall and Kress (2008) found that 58% of boluses collected from black-crowned night-heron nestlings contained bird remains; however, this may be a specialized feeding behavior for a small subset of the population. Tern chicks were the most common bird prey species identified (Hall and Kress, 2008), which may also have implications for the tern colonies at Little Estero Island Critical Wildlife Area on Fort Myers Beach and at Lovers Key State Park.

Analysis of modern data—Although there were no significant changes in peak nest counts across the more recent sampling period, a declining trend over the past four years (61 nests per year) may reflect real losses of nesting pairs that warrants further monitoring or it may be an artifact caused by an extended nesting period. Peak nest counts are calculated by taking the highest nest count for each species at each of the colonies and adding them to obtain the total peak nest count for the season. Peak nest counts may exclude nests that are not occupied during the peak of the nesting season and therefore may exclude more nests when nesting seasons are spread out. This calculation does not take into account the duration the nest is occupied, which varies widely among species, from nest building to fledging. Using monthly surveys to calculate peak nest counts may provide a more accurate representation of the nesting population than annual surveys. Monthly surveys also provide the opportunity to better track peak nesting times which shifted annually over the four-year monthly survey period; peak nest counts were recorded once in March and June, and twice in April. Changes in nesting time could represent shifts in food availability (Keith, 1978; North American Bird Conservation Initiative, U.S. Committee, 2010), including feeding by humans at fish cleaning stations (Perrins, 1970), or shifts in age composition of the population since younger birds tend to nest later in the season (Perrins, 1970). Keith (1978), Perrins (1970) and Schreiber (1979) showed that brown pelicans and other species that nested earlier in the season were more successful at producing fledglings because young produced later in the season often starved.

Changes observed in species composition could result from changes in nesting time and species dominance. Body size, nest placement, and dominance during interactions influence species co-occurrence at colonies (Spendelow et al., 1989; Burger, 1978). Shifts in species dominance between the brown pelican and great blue heron, both large-bodied canopy nesting species, should be examined in conjunction with arrival time at nesting sites and initiation of nest building in future analyses.

Additional observations at the colonies—Disturbance on nesting islands has been documented through the history of Estero Bay monitoring. Repeated observations of commercial and recreational boat traffic prompted DEP biologists to post one active colony in 1998 with "closed" signs. In 2010, a local eco-tour operator began bringing boat tours to an active colony multiple times a day. Nest counts for this specific island have been documented during April surveys since 1998 and annual counts have averaged 55 nests with 77 active nests in April 2010. The boat tours were observed flushing birds from nests throughout the nesting season, and in 2011 the colony contained only three nests, including one great blue heron and two night-heron. Discarded fishing line also poses a threat to colonies within the bay. In 2009, 30 birds were documented entangled in fishing line; in 2010, 17 birds were documented; and in 2011, 9 birds were documented. Brown pelican accounted for 56–59 percent of fatalities. The decline in entanglement fatalities may be due in part to annual fishing line cleanups conducted by Estero Bay Aquatic Preserve staff and volunteers, as well as the efforts of other local agencies including Keep Lee County Beautiful.

ACKNOWLEDGMENTS—Monthly monitoring would not be possible without the hard work and dedication of a wonderful group of volunteers. The Town of Fort Myers Beach and Estero Bay Preserve State Park provided staff and vessels for monitoring and cleanup efforts. Fish-Tale Marina, Lovers Key State Park, and Lee County Parks and Recreation allowed us to utilize their launching and parking facilities. National Audubon Society and Lee County contributed in the collection of data between 1977 and 2007. I am particularly grateful to Heather Stafford and Philip Stevens, PhD, for reviewing this paper and providing valuable comments.

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SALINITY REGRESSION MODELING: NOVEL FLOW WEIGHTING AND USE FOR BIOLOGICAL EVALUATIONS IN THE MYAKKA RIVER

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Abstract: *The Southwest Florida Water Management District funded salinity regression modeling for the determination of minimum flows of the tidal Myakka River. Selected isohaline (1, 2, 4, 8, 16, 20, and 24 PSU) locations were modeled as a function of weather terms, tidal conditions, and a variety of weighted daily flows of the Myakka River near Sarasota, FL, the tributary Myakkahatchee Creek, and the Peace River near Arcadia. Peace River flows were included due to potential influence on downriver boundary conditions. The most influential variables were variably weighted flow terms, weighted over periods based on both flow and the riverine volume upstream of the computed isohaline location. Adjusted multiple* r^2 *of the regressions for 8 PSU and lower isohaline locations were 0.80 or better (n~70–100). Regressions were validated with a subset of data withheld from regression development. Regressions were used to simulate time series of isohaline positions under both existing flows and potential flow reductions. Changes in isohaline locations could then be used to evaluate amounts of critical habitats (benthic, freshwater marsh, forested wetlands) exposed to salinity intervals, to compute the projected changes in isohaline locations under reduced flow scenarios, and to balance projected flow reductions with potential habitat alterations.*

Key Words: Salinity modeling, regression modeling, Myakka River, minimum flow, flow weighting

The lower Myakka River, located in Sarasota and Charlotte Counties (Fig. 1), is a valuable natural resource with portions variously designated as an Outstanding Florida Water, a Florida Wild and Scenic River, and an Aquatic Preserve. Tidal along much of its length, the lower river supports a diverse oligohaline and estuarine biota (Flannery, et al., 2011), and has a shared boundary with the waters and biological resources of Charlotte Harbor. To fulfill missions of both water supply and resource protection, the Southwest Florida Water Management District (SWFWMD) is charged with determining the minimum flows and levels (MFL), defined as "…the limit at which further withdrawals would be significantly harmful to the water resources or ecology of the area" (F.S. Chapter 373.042). The SWFWMD determinations take place in the context of discernible climatic change over the period of record and substantial anthropogenic alterations to watershed hydrology. An important tool for evaluating the impacts of both past and future flow alterations was salinity regression modeling, which gave SWFWMD the ability to simulate historical salinity conditions and to quantify the potential impacts of a variety

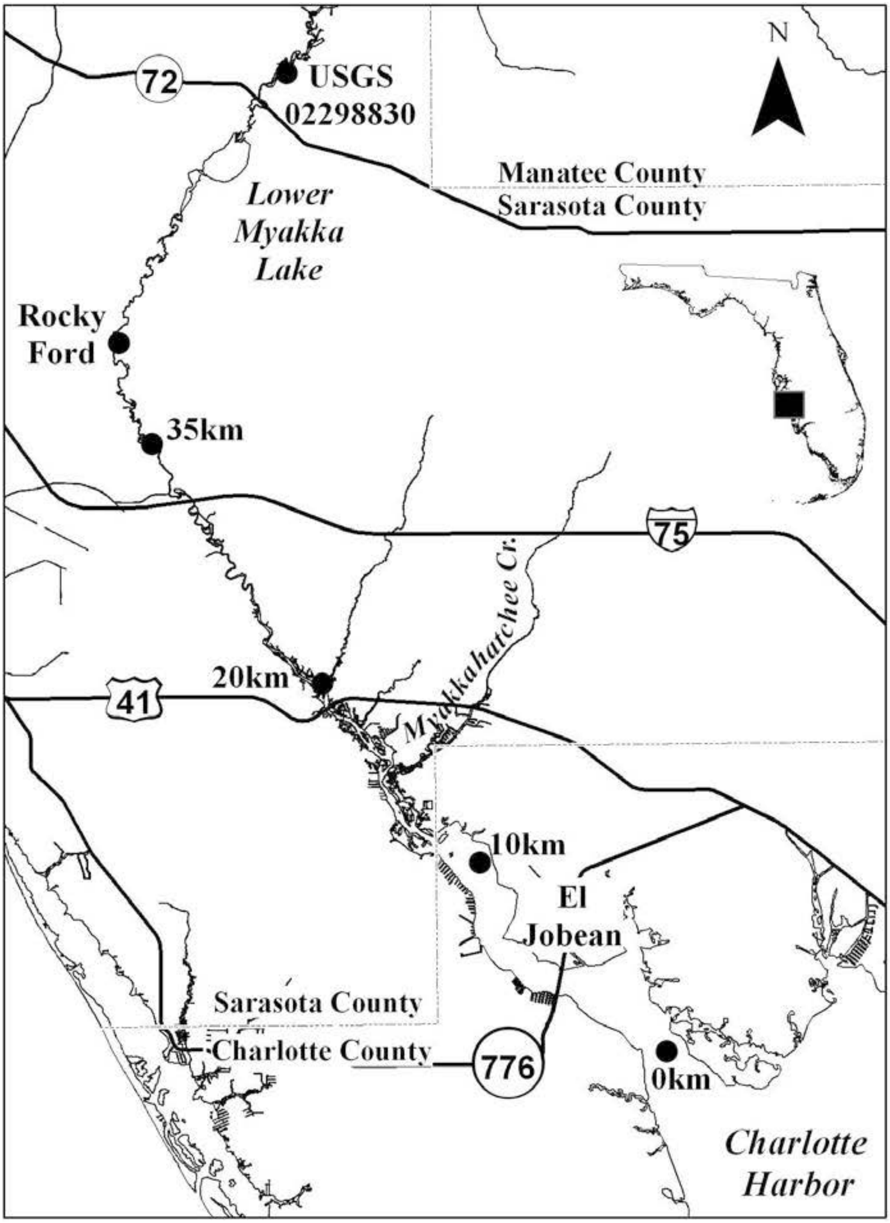

FIG. 1. Lower Myakka River study area.

of withdrawal scenarios by comparing model outputs under both altered and baseline conditions with mapped biological resources.

METHODS—The dependent variables sought by modeling were time series of river kilometer positions of specific, biologically relevant isohalines (*sensu* Bulger et al., 1993; 1, 2, 4, 8, 12, 16, 20, and 24 PSU). Salinity input data were obtained from sampling conducted by the U.S. Geological Survey, the Florida Department of Environmental Protection, Sarasota County, SWFWMD, Mote Marine Laboratory, and others. Stations were assigned a river kilometer and both surface and bottom isohaline positions were calculated as linear interpolations between the salinity data of adjacent stations. Mid-depth data from shallow stations (less than 1.0 m deep) were assigned to surface categories.

Input flows were the daily values of the Myakka River (U.S.G.S. 02298830) and Myakkahatchee Creek flows (entering the Myakka River near river kilometer 15), provided by SWFWMD (Flannery, et al., 2011). Flows from the Peace River at Arcadia (U.S.G.S. 02296750; the largest gauged watershed within the Peace basin) were also used as an indicator of boundary conditions affecting the lower Myakka River (i.e. salinity in Charlotte Harbor). All three flow records were subjected to a variety of flow weighting techniques. Lagged flows of 1 to 10 days were computed, as well as triangular and exponentially weighted (Berthouex, et al., 1978) terms over a range of intervals (3, 5, and 7 days). Rates of change in flow were calculated to allow the capture of any differences in relationships between the ascending or descending hydrograph limbs.

A variable length flow weighting technique was also developed. The number of days over which flow weightings were performed, DAYS, was calculated as the river volume between the isohaline position and 51.0 km, divided by the daily flow, i.e. the number of days required to 'fill' the river to the specified location if the flow remained constant. River volume was computed at mean tide level from an empirical approximation of the river volume:kilometer file developed by SWFWMD based on recent bathymetric data (Wang, 2004). As a result, the DAYS value varied both by day, spatial location, and salinity. On a given date, DAYS was smaller for upriver isohalines than for positions downstream, consistent with the upper river being influenced by a shorter period of antecedent flows than the lower river. Similarly, the quantity DAYS was smaller for high flow conditions than for low flow. Once the DAYS parameter was computed, flow weighting was performed over the minimum of either DAYS or 45 days as both a declining linear and an exponential weighting. The 45 day maximum was selected based on observations of hydrograph and salinity response to rainfall events. Variably weighted flows over a minimum of DAYS and either 30 or 15 days were also computed with similar approaches for Myakkahatchee Creek and the Peace River. Additionally, natural log transformations of most flow parameters resulted in 63 potential flow variables.

The method for the variable flow weighting follows. On date_i, with a Myakka flow of Q_i and a given isohaline at position *km*, the volume of the river upstream of *km* (Vol_i), in cubic meters, is empirically computed from data supplied by SWFWMD as:

$$\text{If } km < 13,\ \text{Vol}_i = 1000 * \left(74.405 * (km)^2 - 3109.8 * (km) + 30342\right)$$

$$\text{If } km \geq 13,\ \text{Vol}_i = 1000 * \left(10837 * e^{(-0.1107 * km)}\right)$$

The variable flow weighting period, DAYS, and the resulting variably weighted flow, VWT45, are computed as:

$$\text{DAYS}_i = \text{Integer}\left(\text{Minimum}\left((\text{Vol}_i/Q_i) * \left((35.31\ \text{ft}^3\ \text{m}^{-3}) * (86{,}400\ \text{sec day}^{-1})\right), 45\right) + 0.5\right)$$

$$\text{VWT45} = \left(\sum_{n=0}^{(\text{DAYS}_{i-1})} (\text{DAYS}_{i-n}) * Q_{i-n}\right) \Big/ \left(\sum_{n=0}^{(\text{DAYS}_{i-1})} (\text{DAYS}_{i-n})\right)$$

For instance, when $Vol_i/Q_i = 4.1$, $DAYS_i$=Integer(Minimum(4.1,45) + 0.5) = 4, and

$$VWT45 = (Q_i * 4 + Q_{i-1} * 3 + Q_{i-2} * 2 + Q_{i-3} * 1)/(4+3+2+1)$$

Much of the study area was tidal with mixed, semi-diurnal tides, even to river kilometer 28, where salinities were predominantly less than 1 PSU. Tidal influences were investigated in regression model development using predicted tidal heights to separate the effects of astronomical tides from stage elevations due to increased flows. A 30 day period of hourly data (U.S.G.S. 02299496 Myakka River at El Jobean, FL) during the lowest flow period available (5/31/1985–6/30/1985; flows less than 0.003 m^3s^{-1} [0.1 ft^3s^{-1}]) was identified. Seasonal variation in sea level was removed based on linear interpolations of monthly values of sea level variations at Fort Myers (NOAA/NOS CO-OPS 8725520, http://tidesandcurrents.noaa.gov/sltrends/seasonal.shtml?stnid=8725520&name=Fort+Myers&state=Florida). Tidal harmonics were abstracted (Boon and Kiley, 1978), predictions of hourly stage were generated for the period of record, and seasonal variations in sea level were returned to the predicted record. Correspondence between observed and the resulting predicted timing and tidal elevation was excellent, with an average RMS error ranging from 0.05 to 0.10 m during other low flow periods.

Predicted tidal variables for potential model variables included stage and rate of change for the time of each isohaline location (data from the nearest hour), as well as the mean, minimum, and maximum values of the prior 3 hours, 6 hours, or entire day. Lag times of from one to three hours were also investigated based on the range of lags in stage timing reported by Hammett (1992). A total of 26 potential tidal variables were available.

Potential independent weather variables were derived from hourly data recorded at Venice, FL, approximately 10 km to the west of the Myakka River (National Data Buoy Center VENF1, http://www.ndbc.noaa.gov/station_history.php?station=venf1). Using weather data captured the impacts of strong winds and departures of water levels from predicted tides that were associated with frontal passages. The differences between observed and predicted tides were plotted as a function of wind direction during low flow periods to determine that positive residuals were maximized when wind direction was from 230°, and were the most negative when wind direction was from 50°. Wind direction data were therefore transformed as the negative cosine of (wind direction minus 50°) resulting in a value of 1.0 when the wind direction was 230° M. Wind stress was approximated by multiplying the transformed wind direction by the wind speed squared. In addition to the hourly values, the averages of the prior three and six hours of barometric pressure, wind direction, and wind stress were also computed, giving nine potential weather variables.

Regression input data were limited to the 99th percentile (59 m^3s^{-1} [2115 ft^3s^{-1}]) and below of variably weighted Myakka flow and to isohaline positions at or above river kilometer 0.0 to emphasize low flow, upriver conditions. Data were further restricted to isohalines computed from raw station data separated by no more than 6 km and 7 PSU as a compromise between the uncertainty of computed isohaline positions and the number of retained data. Lastly, data were limited to a single value per month-year to reduce serial correlation. Data which passed all of the former criteria but were not used as the one value per month-year were reserved for regression model verification.

Models of surface and bottom isohaline position were developed as forward interactive regressions with a constant term, using $p \leq 0.05$ as the criterion for inclusion and maintenance in the model. Once a flow term from a river or creek was included, no other flow term (other than rate of change in flow) of the same river was included. Weather and tide variables were included subsequent to flow terms and limited to one parameter of each category. The sign of the individual regression coefficients, constancy of signs with the inclusion of additional variables, and significance of individual retained variables were all examined to prevent spurious correlations. Due to the inclusion of wind and tide terms, the constant term was not synonymous with isohaline position at zero flow.

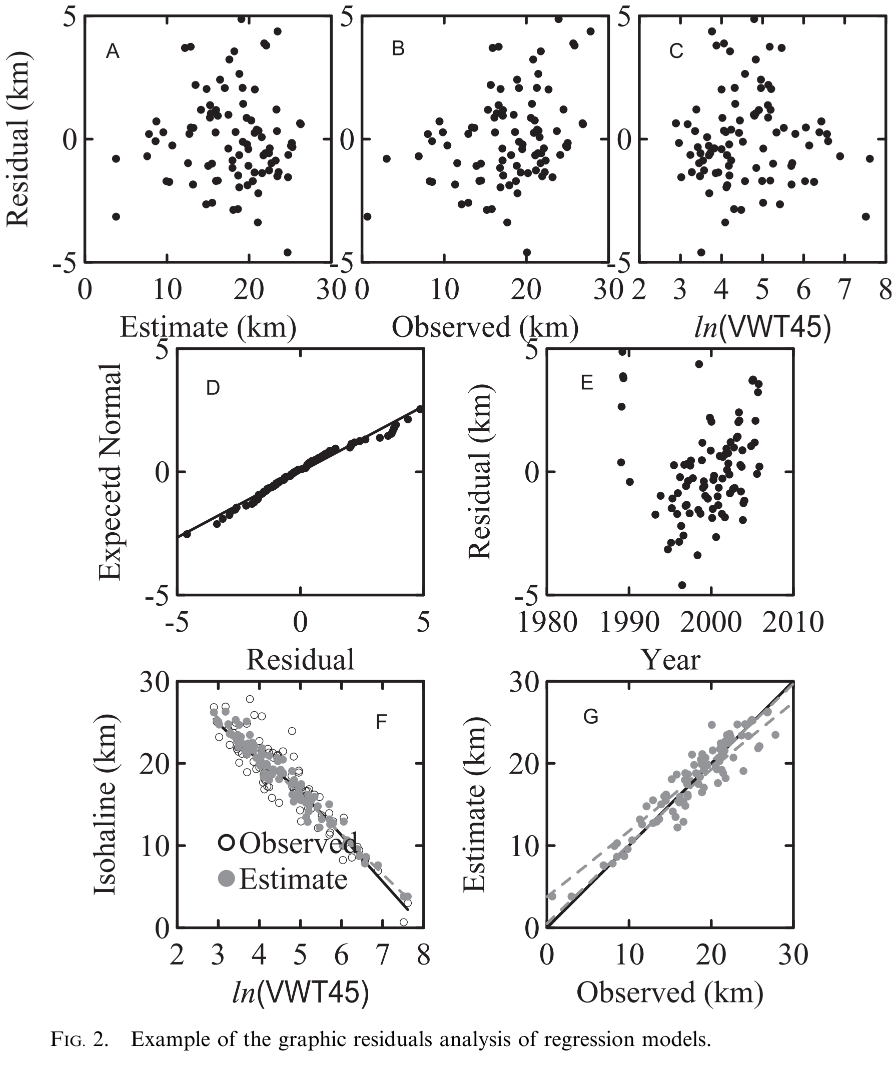

FIG. 2. Example of the graphic residuals analysis of regression models.

All regression models were subjected to residuals analysis and verification. Residuals analysis (FIG. 2) included graphic analysis of residuals as a function of both the predicted dependent variable or regression estimates (A), the observed dependent variable (B), and of the dominant independent flow term (C). Residuals were graphically examined for normality (D) and for trend over time (E). The distributions of estimated and observed isohaline position were also illustrated as a function of the dominant flow term (F) Lastly, the regression estimates and the associated 95% confidence intervals were illustrated as a function of observed isohaline position (G), with inclusion of the 1:1 slope within the confidence interval indicating the best agreement of modeled with observed data. Outliers to the regression relationship were examined for reasonableness, but generally were not removed from consideration as data often represented an end-member condition (highest flow of one of the secondary flow variables, highest tide conditions, etc.).

Regression verification consisted of applying the equations derived using initiating data to observed flow, tide, and weather variables not used in the development of regressions (non-initiating

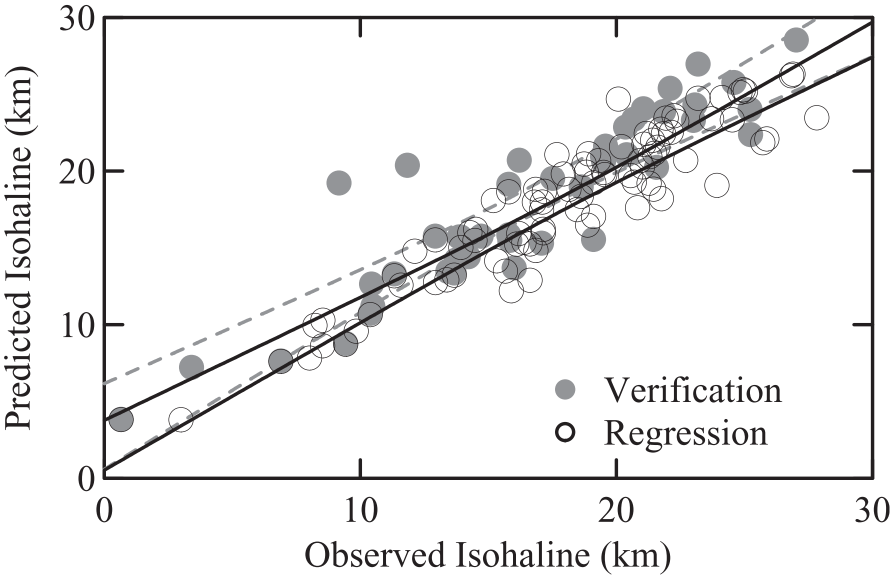

FIG. 3. Verification of regression models by the overlap of the 95% C.I. of initiating (regression) and non-initiating (verification) data.

data). The 95% confidence interval of estimated as a function of observed values was computed for both the initiating and non-initiating data. The overlap of the two confidence intervals (FIG. 3) indicated robustness of regression coefficients.

Regression models were developed using isohaline locations at a given time of day, using time-specific weather and tide data and inclusion of weather and/or tide variables in the regression models nearly always enhanced regression significance. Diurnal variations in tides and weather have been observed to produce diurnal salinity variations and extreme, weather-driven events undoubtedly affect riverine biological resources. Flows used as independent variables, however, were available as daily averages and weather data were not available for the entire simulation period (1980–2004). Management of flows on less than a daily increment and management of extreme weather events were not anticipated.

Accordingly, in order to highlight the potential salinity differences from managed flow alterations, weather and tidal variables were set to constant values for all simulations using the mean conditions observed in the initiating data. The approach provided weather-neutral simulations of daily average isohaline locations to isolate the effects of flow manipulations, and was not designed to simulate extremes in isohaline location produced by the diurnal or seasonal range in tides, storms or unusual tide and wind events.

The modeled distributions of weather-neutral isohaline positions over the simulation period undoubtedly included fewer outliers than had daily wind and tide been used. However, given the regression model formula, the *difference* between baseline and proposed flow regimes would be nearly identical, regardless of the weather and tide treatment, with the result that the weather-neutral simulations were a useful evaluation tool.

An additional model application note was that when modeling isohaline locations, if variably weighted flows were a significant variable, the isohaline location was needed to determine the number of days (DAYS) over which flows were to be weighted, resulting in an iterative process to simulate locations. The DAYS parameter was calculated as described, using the isohaline position from the day prior and a new daily flow, and a new isohaline position was calculated. Recalculation of DAYS and isohaline position was repeated until the difference between successive isohaline

TABLE 1. Significant terms for multiple linear regression modeling of isohaline positions on the Myakka River. All were highly significant ($p < 0.001$). (W – wind, T – tide, M – Myakka R. flow, P – Peace R. flow, H – Myakkahatchee Creek. flow, R – Rate of change of Myakka R. flow).

Isohaline	Signif. Variables	Adj. Mult. R^2	N=	S.E.E. (km)
1S	W, -, M, P, -, -	0.80	96	2.3
1B	W, T, M, P, -, -	0.84	97	2.0
2S	W, T, M, P, -, -	0.85	88	2.1
2B	W, T, M ,P, -, -	0.88	89	1.9
4S	W, T, M, P, -, -	0.87	84	2.2
4B	W, T, M, P, -, -	0.87	81	2.1
8S	W, T, M, P, H, R	0.86	84	2.2
8B	W, T, M, P, -, R	0.86	73	2.2
12S	W, T, M, P, -, R	0.81	83	2.7
12B	-, T, M, P, -, R	0.78	116	3.0
16S	W, T, -, P, -, -	0.74	71	3.1
16B	-, -, -, P, -, -	0.64	118	3.3
20S	-, T, -, P, -, -	0.66	84	2.9
20B	-, T, -, P, -, -	0.63	68	3.3
24S	-, T, -, P, -, -	0.49	30	3.2
24B	-, -, -, P, -, -	0.34	39	3.5

positions was less than 0.1 km, there was no change in the DAYS parameter, or until iterations had reached 15. Convergence within specifications was generally achieved within two to three iterations.

Regression models, together with flows and mean weather conditions, were used to generate a daily record of average surface and bottom isohaline positions from October 1980 through December 2004. The simulation performed using observed flows and mean weather conditions was termed the 'baseline' condition. Additional simulations performed for the MFL used a variety of flow manipulations, typically seasonal or year-long percentage reductions (Flannery, et al., 2011). In the absence of detailed quantitative information on the duration and extent of salinity stress on biota, distributions of isohaline positions under baseline conditions (whether assessed with weather-neutral or weather-specific techniques) were assumed sufficient to maintain exiting biota. The baseline positions and subsequent translocations of isohaline positions under flow alterations were superimposed on the location and amounts of various biological communities to estimate potential biological impacts. A variety of simulations could then be used to select the amount and timing of water withdrawals which would minimize encroachment of selected salinities on biological resources. The work presented here does not evaluate between scenarios, but rather provides examples of the methodological approach.

RESULTS—Resulting models of isohaline position were highly significant ($p<0.001$) with adjusted multiple r^2 of 0.8 or better and standard error of the estimates (S.E.E.) 1.9 to 3.0 km (TABLE 1) for the 8 PSU and lower isohalines. Correlation coefficients were somewhat lower and S.E.E. somewhat larger for the higher salinity isohalines. All significant flow terms were the natural log transformed, linear variably weighted flows as described above with the exception of the Myakkahatchee flows for the 8 PSU surface isohaline, where the natural log transform of the exponential variably weighted flows provided the best fit.

Notably, the Peace River flows, as *ln* transformed variably weighted flows, were significant for models of all isohalines. Myakka flows, again as *ln* transformed variably weighted flows, were included in the model only for the 12 PSU and lower isohalines. The Myakkahatchee flows were significant only for the surface 8 PSU isohaline, and rate of flow change of the Myakka River was significant for both the 8 and 12 PSU isohalines. A wind variable was generally significant for the 16 PSU and lower isohalines, while a tide variable was included in all but three (bottom for 8, 16, and 24 PSU) of the 16 models. Based on distribution of input data, modeling results were limited to isohaline positions above 0 km, and weighted flows of Myakka and Peace Rivers less than 59 and 224 m^3s^{-1} (2115 and 8000 ft^3s^{-1}), respectively, or approximately the 99%-ile of daily flows between 1980 and 2004.

Graphic analysis of all models indicated no strong relationships of residuals with predicted or observed isohaline positions, or with the dominant flow term. Residuals for all models were essentially normally distributed, with little overall trend over time. Both observed and predicted isohaline positions produced very similar distributions in relation to the dominant flow variables. For all models, however, the 95% confidence interval of the relationship between the estimated and the observed positions failed to include the 1:1 line (G, FIG. 2, above). The result indicated a small bias of under prediction of kilometer position for upriver locations and an over prediction of kilometer position at downriver locations, and is quantified in the SEE of each model. Similar bias but somewhat larger existed for the models of higher isohalines. As observed weather and tide variables were used both for model development and verification, the bias cannot be attributed to unusual input data, tides, or winds.

Despite this bias, regression verifications (FIG. 3, above) of all models were successful in that the 95% confidence intervals (of the estimated as a function of observed positions) of regression input data overlapped the confidence interval developed from data reserved from regression development. The verification indicated that models were robust to a wide variety of input conditions and that different data used in model development would not produce materially different model coefficients. An example of modeled bottom isohaline position for the 2 PSU, together with interpolated observed positions, appears in FIG. 4 and further indicates that the model closely describes both the range and variation of isohaline positions under high and low flow conditions.

In model applications for MFL determinations, isohaline positions were computed using mean tide and wind conditions. In the selected example (FIG. 5), the existing distribution of tidal freshwater marsh is plotted as a histogram by river kilometer. The lower horizontal box plot illustrates the distribution of the 2 PSU isohaline position under baseline flows and, under weather-neutral conditions, is considered to be a distribution which will maintain the existing marsh. The distribution of isohaline positions under reduced flows (upper box plot) resulted in an upriver displacement of median isohaline location of about 1 km, while the maximum upriver penetration remained similar. Provided marshes are sensitive to the median presence of 2 PSU, approximately 0.5 ha of marsh may be affected by this flow alteration

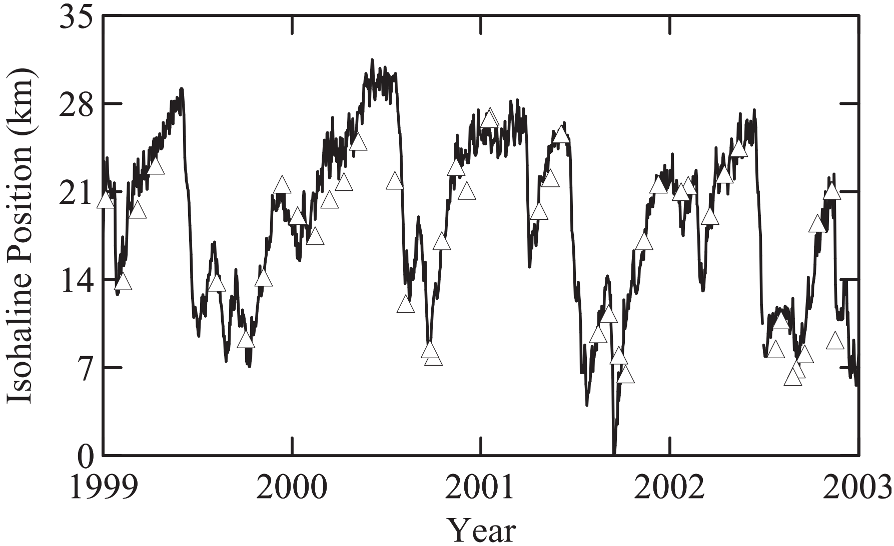

FIG. 4. Observed isohaline positions interpolated from field observations (triangles) and modeled isohaline position (line); 2 PSU, bottom.

scenario. Other percentiles of isohaline position and other isohalines are similarly evaluated.

DISCUSSION—Regression models of isohaline position benefited from a large data set collected over many years under a variety of climatic conditions. The wealth of data allowed for both model development and verification which indicated that differing input data would produce very similar regression coefficients. Significance of resulting models was uniformly high ($p<0.001$),

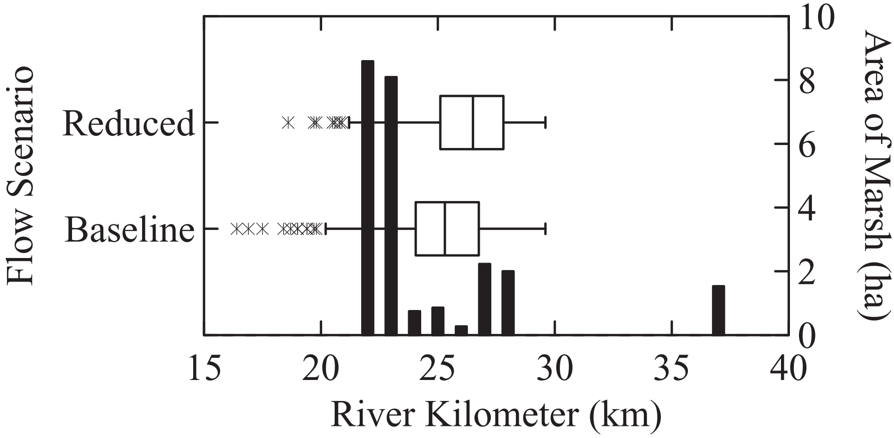

FIG. 5. The distribution of the surface 2 PSU isohaline under baseline and a reduced flow condition, superimposed on the quantity and location of tidal freshwater marsh.

particularly for low salinity, upriver isohalines where any flow reductions during the dry season were expected to have the greatest potential impact on habitat. The bias noted for some under-prediction of upriver locations during low flow and over-prediction of downriver locations during higher flows could be consistent with the computation of the DAYS parameter failing to account for the difference in riverine volume under differing stage (as it was computed based on mean tide level), or with the DAYS parameter being computed based on a single daily flow (rather than the time history of flows in the antecedent days). The bias also resulted in an apparent trend over time in residuals during later portions of the record as a result of higher flows in the initiating data during 2000–2005. Additional sophistication in applying the flow weighting might be a useful area to explore in future work.

The significance of variably weighted flow parameters in the models indicated that the salinities in the Myakka River responded to other than fixed antecedent periods. The variable flow weighting, while empirical, was effective at incorporating the varying antecedent periods. The relevant period of influence was longer during low flow conditions and at downriver locations than at high flow and/or upstream locations. The inclusion of Peace River weighted flows in all models, even that for the 1 PSU isohaline, was surprising, as was the absence of Myakka River flows for the 16 PSU and higher isohalines. The result was a clear statement that Peace River flows, and potential flow reductions in that system, should be expected to have an effect on salinity in the Myakka River.

In addition to flow dependence, positions of most isohalines were also significantly related to tidal heights and wind conditions. While models were developed with observed weather and predicted tides, models were applied using fixed weather and tide variables in order to more directly compare the effects of potential flow alterations and to simulate a longer period of record and range of climatic conditions. Had specific weather and tide been included, the distribution of modeled isohaline positions would have undoubtedly been larger under all flow scenarios. Due to the form of the linear regression model, however, the difference between baseline and a flow alteration would be nearly identical under weather-neutral or weather-specific simulations.

The application of models provided some results that were counter-intuitive. Occasionally, as daily flows declined, the variably weighted flow parameter would increase briefly (FIG. 6). Similarly, when comparing flow scenarios, a flow reduction occasionally resulted in a temporary downstream migration of an isohaline position relative to baseline conditions. The phenomenon is product of the definition and method of calculation of weighted flow parameter. As flow declines, the DAYS parameter increases the antecedent period used for flow weighting, therefore increasing the potential to include flows from a prior high flow event, if present. If high flows were encountered during the antecedent period, the weighted flow parameter would increase and isohaline position would move downstream in response. The effect was infrequent, and occurred only when extreme dry periods followed immediately after very wet periods with a rapid transition in flows between

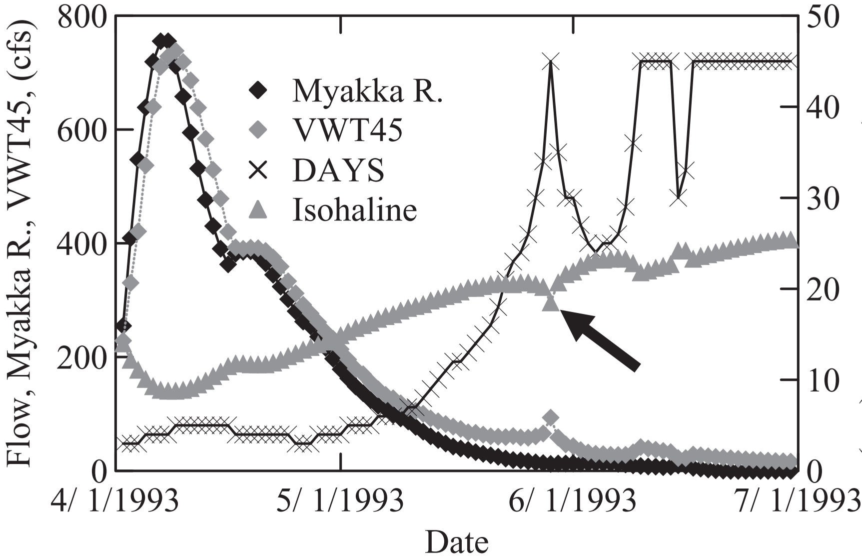

FIG. 6. Illustration of the smoothing of daily flows resulting from the variable flow weighting, and an instance (arrow) where declining daily flow results in an increase in DAYS and weighted flow (VWT45).

the two conditions. The results were accepted as regressions were developed based on the flow weighted parameter calculated as described.

Viewed as a whole, other unusual model results were that modeled isohalines could occasionally be inverted, i.e. the 4 PSU position computed as upstream of the 2 PSU. The phenomenon resulted from the different methods of including tide and wind variables in the various models, as well as slightly differing regression input data. (The randomly selected single monthly value may not have been from the same date for each isohaline.) Again, the result was atypical and did not materially affect the distributions of isohaline locations.

In summary, the regression models developed were of high significance, high adjusted multiple r^2, and robust for all isohalines examined. Peace River flows cannot be neglected when examining salinity in the Myakka River. Simulating isohaline locations under weather-neutral conditions compressed the range of simulated locations but facilitated the comparison of changes in location due to flow alterations. The flow weighting technique developed for this work was a novel approach to computing the variable extent of influential antecedent conditions under both high and low flows.

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Biological Sciences

A WATER CLARITY EVALUATION AND TRACKING TOOL FOR THE ESTUARINE WATERS OF LEMON BAY, CHARLOTTE HARBOR AND ESTERO BAY, FLORIDA

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Abstract: *The Charlotte Harbor National Estuary Program (CHNEP) has established a foundation of objective, science-based tools to guide management of estuarine water quality and is furthering its goals under its Comprehensive Conservation and Management Plan (CCMP) to protect and restore water quality through rigorous assessment of these tools. In this study, we furthered the objectives of the CCMP by developing a water clarity evaluation and tracking tool for CHNEP estuarine waters. Benchmark points along a reference period distribution of water clarity were identified as management level criteria for evaluating water clarity and the binomial test was used to establish a scoring system to identify deviations from reference water quality conditions. The scoring system uses a rating scale that varies between -2 and 2. The resulting scores are tabulated and the numerical values are reported as color coded grades (i.e., red, yellow, or green). The grades are then compiled for each estuarine segment of the CHNEP into a table that can be easily integrated into public media formats including the new CHNEP Water Atlas.*

Key Words: Charlotte Harbor, seagrass, water clarity, estuaries, optical model, light attenuation

The Charlotte Harbor National Estuary Program (CHNEP) has been developing criteria to monitor and report on the condition of its estuarine waters which range from Lemon Bay to Estero Bay and include all of Charlotte Harbor from the Peace River to San Carlos Bay (Fig. 1). Seagrass is a principal natural resource in these waters and a critical indicator of estuarine condition. The CHNEP recently identified seagrass acreage targets to either maintain or restore seagrass to its historic extent and designated each segment as either preservation or restoration areas based on these seagrass targets (Janicki Environmental, 2009). The targets allow managers to evaluate seagrass areal extent through time relative to historical benchmarks and will help evaluate the effectiveness of management actions within the watershed to minimize impacts to estuarine seagrasses as established under the CHNEP's Comprehensive Conservation and Management Plan (CCMP). Because seagrasses are known to be primarily limited by the quantity and quality of light available for photosynthesis, the CHNEP identified the need to develop management level indicators of water clarity; a significant factor in determining the condition of seagrass in the estuarine waters of the CHNEP.

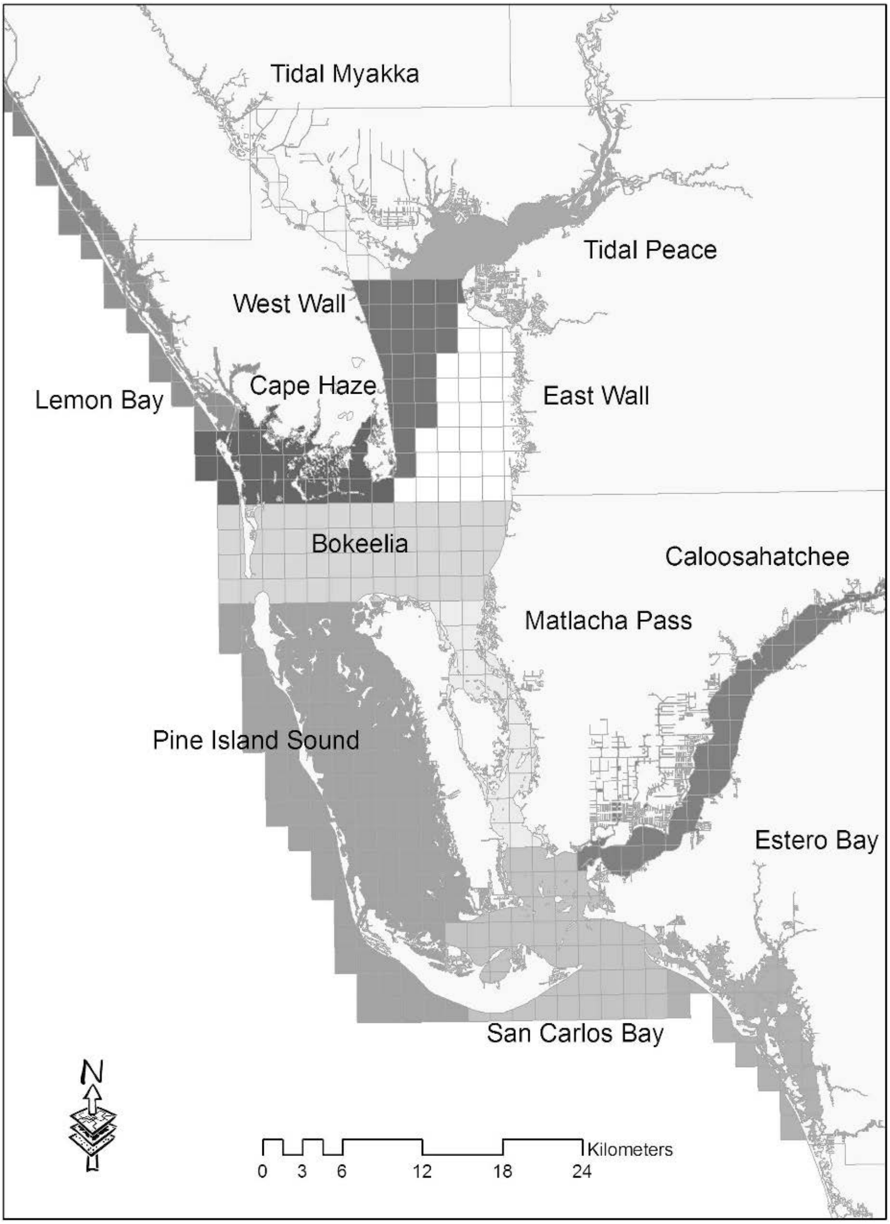

FIG. 1. Segmentation scheme used by the Coastal Charlotte Harbor Monitoring Network for water quality sampling.

There have been substantial efforts by the CHNEP and others to link water clarity to the principal attenuators of light in the water column; namely colored dissolved organic matter (CDOM), turbidity and chlorophyll a concentrations, (McPherson and Miller, 1994; Corbett and Hale, 2006; Dixon and Kirkpatrick 1999). However, an evaluation comparing model-based and empirical estimates of light attenuation revealed broad discrepancies between current model-based estimates and field measurements of light attenuation (Wessel and Corbett, 2009). Most recently, Dixon et al. (2010) refined the approach to modeling light attenuation by accounting for wavelength specific attenuation of light through the water column and by accounting for the spectral narrowing of the light field as a function of water depth. This model is still being tested in the southern CHNEP estuarine segments. The CHNEP is currently funding efforts to refine the methods for the field collection of light, and to establish an optical model that will allow for the site-specific calculation of the photosynthetically usable radiation to seagrass. While the CHNEP has recognized that mechanistic relationships between ambient/antecedent water quality conditions and the living resource requirements of seagrass are currently not fully understood, stability in measures of seagrass areal extent over recent history suggests that ambient water quality conditions from recent history are sufficient for success of seagrasses (expressed as aerial extent).

The objective of this study was to develop a water clarity evaluation and tracking tool to identify potential deviations from the reference period conditions that resulted in stable or increasing seagrass areal extent throughout CHNEP estuarine waters without explicitly identifying the light requirements of seagrass. The evaluation tool was developed such that it could be adapted to work with any index of light attenuation (either empirically measured K_d or model based estimates) and provide a convenient format for reporting on the condition of water clarity in CHNEP estuarine waters to natural resource decision makers and the general public.

MATERIALS AND METHODS—The calculation of the light attenuation coefficient K_d is derived by defining the slope of the natural log-transformed sub-surface photosynthetically active radiation (PAR) values as a function of water depth to obtain an average change in irradiance with depth. This method, based on the Lambert Beer law, was adapted from the equation for downward irradiance described by Kirk (1994).

$$K_d = [-\mathrm{Ln}(E_d(z_1)/E_d(z_2))]/z_2 - z_1 \tag{1}$$

where: E_d = downward irradiance at depth d and Z= depth below the water surface in meters.

While different types of sensors are used in different portions of CHNEP estuarine waters (the South Florida Water Management District (SFWMD) requires the use of a spherical light sensor that measures light captured from all angles while the Southwest Florida Water Management District (SWFWMD) requires the use of a 2π cosine sensor that is intended to capture only downwelling irradiance), the irradiance values were used identically in the equation above.

The 2003–2007 time period was chosen as a reference period and a cumulative distribution function (CDF) of light attenuation (K_d) data over the reference period was generated for each estuarine segment. The 30^{th} and 70^{th} percentile values from the reference distribution were chosen as benchmarks from which to evaluate K_d data on an annual basis. The rationale for choosing these particular points along the CDF curve included: achieving internal consistency in reporting among segments by choosing consistent benchmarks points among segments; that the 30^{th} and 70^{th}

TABLE 1. Benchmark values for segment specific water clarity targets.

Harbor Segment	P30 Exceedance criteria K_d values	P70 Exceedance criteria K_d values
Charlotte Harbor Proper	0.62	1.17
Dona And Roberts Bays	0.62	1.03
Estero Bay	0.91	1.6
Lemon Bay	0.73	1.13
Matlacha Pass	0.79	1.52
Pine Island Sound	0.64	1.17
San Carlos Bay	0.71	1.18
Tidal Caloosahatchee River	1.65	3.04
Tidal Myakka River	1.59	2.72
Tidal Peace River	1.08	2.57

percentiles were in general proximity to previous estimates of the light requirements of seagrasses (Corbett and Hale, 2006) and the reference period averages, respectively; that the 30th and 70th percentiles are not likely to be effected by extreme observations, and that the 30th and 70th percentiles are likely to vary independently of one another relative to points closer together on the distribution curve.

The binomial test (Wackerly et. al., 1996) was used to establish a scoring method to evaluate yearly water quality data for each segment at the benchmark points on the distribution for each estuarine segment. For example:

- if more than 30% of the K_d measurements were below the 30th percentile benchmark with statistical significance (alpha=0.05), then water clarity was considered to be improving relative to the 30th percentile benchmark and a value of +1 was assigned for that evaluation.
- If less than 30% of the values were below the benchmark with statistical significance (alpha=0.05), then water clarity was considered to be degrading relative to the 30th percentile benchmark and a value of −1 was assigned for that evaluation.
- Otherwise the value was 0.

This scoring was performed on both benchmark points (i.e. the 30th and 70th percentile) for each estuarine segment and the sum of these scores was used as the segment score of water clarity for that segment. Therefore, the distribution of potential scores ranges from −2 to 2.

RESULTS—The K_d values for the 30th and 70th percentiles for each estuarine segment are provided in TABLE 1 and the CDF's of K_d are provided in FIG. 2 with the benchmark points appearing as filled circles in these figures. The shape of the curve between the 30th and 70th percentile values tended to be relatively flat in the CDF plots for both the restoration and preservation segments. This indicates that substantial changes in the frequency and duration of acceptable conditions as identified by the benchmark values may be realized with management actions that resulted in small changes in the K_d value.

Once the targets were identified, the next step was to apply a decision rule to the evaluation process. The development of segment specific management designations for seagrass as either preservation or restoration segments (Janicki Environmental, 2009) were used to establish the decision rule. The decision rule was formulated such that:

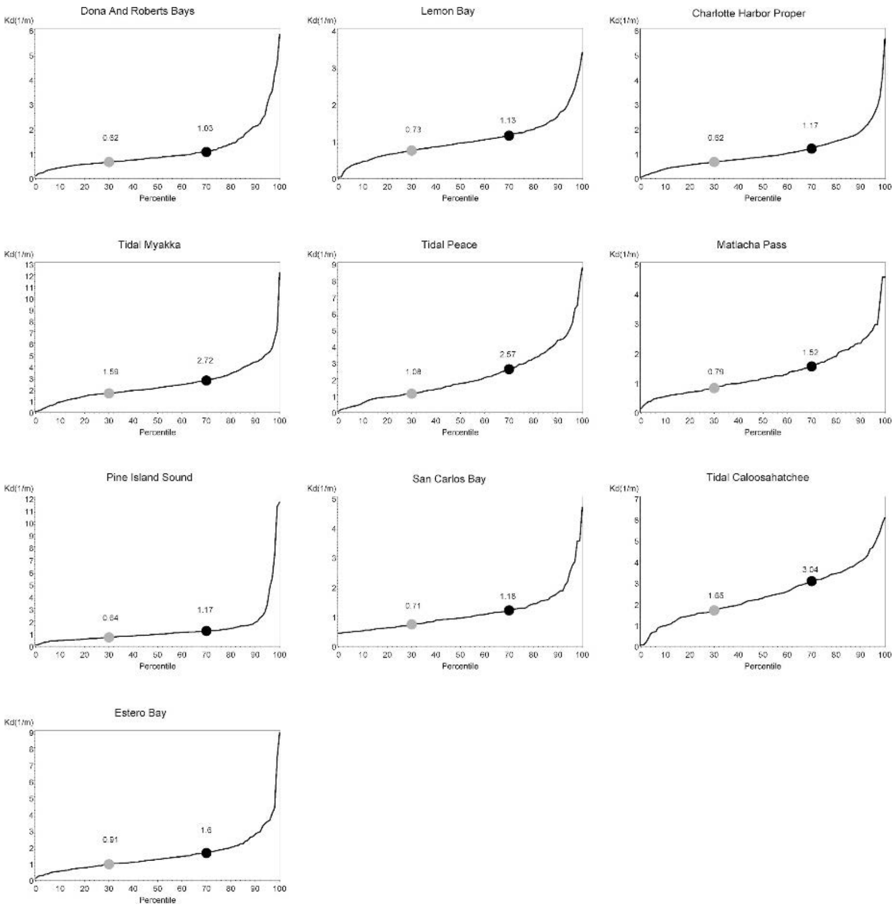

FIG. 2. Reference distribution CDF's for each CHNEP estuarine segment with benchmark points highlighted as circles on the curve.

- If a segment was designated as a preservation segment for seagrass, then the water clarity target was designated as a "maintenance" strategy to maintain ambient conditions experienced over the reference period.
- If a segment was designated as a restoration segment for seagrass, then the water clarity target was designated as an "improvement" strategy to improve water clarity in this segment relative to its reference period condition.

Based on this decision rule, a color coded grading system was established as summarized in TABLE 2 and described in the paragraphs below.

For the preservation segments, scores greater than -1 were given a green color grade indicating "stable" water clarity conditions. A score of -1 or less was given a yellow color grade indicating "cautionary" water clarity

TABLE 2. Grading system for water clarity based on the sum of the 30th and 70th percentile scores and the segment designation as either preservation or restoration for seagrass acreage.

	Grading System				
Segment Designation	−2	−1	0	1	2
Preservation	red/yellow	yellow	green	green	green
Restoration	red	yellow	yellow	yellow	green

conditions. A score of −2 for consecutive years was given a red color grade indicating "degrading" water clarity conditions.

For the restoration segments, the following color coded grading system was established. Scores greater than +1 were given a green color grade indicating " improving" water clarity conditions relative to the reference period. Scores between −1 and +1 were assigned a yellow color grade indicating "cautionary" water clarity conditions. A score of −2 was given a red color grade indicating "degrading" water clarity conditions.

Based on this grading system, the restoration segments have more stringent water quality criteria than the preservation segments. Stability in scores relative to the benchmark period is considered sufficient for the preservation segments but not for the restoration segments. Therefore, scores between −1 and 1 are given a "cautionary" score for the restoration segments and a stable score for the preservation segments. The final reporting format is provided in FIG. 3 with scores and grades updated through 2009. The tracking tool allows the

Estuary Segment	Dona & Roberts Bays	Lemon Bay	Tidal Myakka	Tidal Peace	Charlotte Harbor	Pine Island Sound	Matlacha Pass	San Carlos Bay	Tidal Caloosa-hatchee	Estero Bay
Goal	Restore	Restore	Preserve	Restore	Preserve	Preserve	Restore	Preserve	Restore	Restore
Year										
1998	Data Un-available	-2	-2	Data Unavailable						
1999		-1	-2							
2000		-1	0							
2001		0	-2							
2002		0	0	-1	-2		-1	0	-1	
2003	0	0	-2	-2	-2	1	-1	0	0	0
2004	0	0	0	0	-2	0	0	0	-1	0
2005	0	0	0	-1	-1	0	-1	-1	-1	0
2006	0	0	2	0	2	0	0	0	1	0
2007	0	0	1	2	2	0	2	0	1	-1
2008	0	0	1	1	0	0	1	-1	0	-2
2009	0	0	0	-1	-1	0	0	0	1	0

Color Key:	Stable/ Improving	Caution	Degrading
	GREEN	YELLOW	RED

FIG. 3. Final reporting tool established for the CHNEP with associated scores and color grades for each CHNEP estuarine segment.

reader to easily assess trends in water clarity over time for each estuarine segment.

Discussion—The Intergovernmental Task Force on Monitoring Water Quality (1995) recommended that estuarine indicators be: quantifiable over time; sensitive to potential impacts; cost effective; easily measured and interpreted; linked to an assessment endpoint, and benchmarked against reference values. The tool developed in this study has applied those principles to develop a useful mechanism to track a principal indicator of seagrass success in CHNEP estuarine waters. The goal of this water clarity evaluation and tracking tool was to identify an effective measurement of changes in water clarity that were relevant to the success of seagrasses without directly specifying an explicit model or estimating the specific light requirements of seagrasses in CHNEP waters. By using a reference period approach, water clarity conditions that have resulted in stable or increasing seagrasses over the reference period were used to identify benchmark points for evaluating annual reporting data on water clarity. A color coded grading system and reporting format was developed to convey the results of annual water quality grades to managers and the public in a convenient format that can be easily incorporated into CHNEP publications and electronic media such as the new CHNEP Water Atlas. The measurement of light and the specific light requirements of seagrass in CHNEP estuarine waters are areas of continuing research. However, the ability to report on annual water clarity conditions in a format readily available to the public and decision makers is an important goal for the CHNEP while the mechanistic relationships between water quality, light attenuation, and seagrass success are still being evaluated.

The tidal tributary seagrass segments including the Tidal Peace, Tidal Caloosahatchee, Tidal Myakka and Dona and Roberts Bay contribute highly colored, tanic waters making detection of seagrass within these systems difficult using aerial photography. The tidal tributary segments were assigned as restoration or preservation segments by CHNEP staff based on local expertise but are not considered for evaluation of trends in seagrass acreage based on aerial photography (Janicki Environmental, 2009). However, the grading system for water clarity was applied to these segments based on their preservation or restoration status and is reported in an identical manner to the other estuarine segments of the CHNEP.

While the scores and grades used in this study were based on empirical light attenuation data, the tools developed for evaluation and reporting are easily transferable to model based estimates if desired by the CHNEP provided historic water quality data can be used to adequately hindcast model estimates for the reference period. Currently, the water quality monitoring scheme is based on monthly sampling and it is recommended that this temporal frequency be maintained since the water clarity evaluation tool would be sensitive to changes in temporal sampling frequency. It is recommended that the evaluation tool be re-evaluated after data are collected through 2012 to assess the sensitivity and concordance of the grades with additional data collected on recent trends in seagrass acreage.

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QUADRAT VS. VIDEO ASSESSMENT OF MACROALGAE COVER: A METHODS COMPARISON

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Abstract: *In 2004–2007, the beaches of Sanibel Island, Florida, were affected by large drifts of macroalgae that accumulated on shore. The adjacent seafloor is composed of three ecoregions: inshore, nearshore and offshore. To measure attached and unattached macroalgae on various substrata, two field survey methods were compared from 2008 to 2010 along thirteen 100 m long transects. One method used visual assessments of macroalgae in 1 m^2 quadrats along the transects and one used an underwater camcorder to capture video footage along the transects. Results of each method were converted to percent cover of the transect and compared for each sampling date. The results showed that digital video observations were not significantly different than diver assessments for two of three ecoregions. Sparse amounts of macroalgae were underestimated nearshore using visual SCUBA, due to the inconsistent, but high percentage, of algae cover. Uses, advantages, disadvantages, and time-effectiveness of the two methods were compared. Natural resource managers can choose which survey method meets their scientific, management and budget needs. Based on these results, it is suggested that video analysis of bottom cover is a practical method for rapid, widespread assessment of macroalgae abundance surrounding Sanibel Island.*

Key Words: Macroalgae, community assessment, video vs. SCUBA, methods, Southwest Florida

The need for technically sound, cost effective methods of assessing macroalgae coverage was identified in response to recent large accumulations of macroalgae on the beaches of Sanibel Island, in southwest Florida. During 2004–2007, undesirable quantities of macroalgae accrued on beaches throughout Pine Island Sound (Dawes, 2004; Lapointe et al., 2005), adversely affecting both economically important tourism and ecological balances (LaPointe et al., 2007). Increases in human development and associated cultural eutrophication in southwest Florida have been hypothesized to exacerbate the standing biomass of macroalgae blooms regionally (Lapointe et al., 2007). Additionally, stable isotope analysis (Heaton, 1986) and nutrient stoichiometry (Lapointe et al., 2005) have been used to determine nutrient sources for coastal primary producers, such as macroalgae. Understanding coastal macroalgae population dynamics could aid natural resources staff in managing the beaches of Sanibel for ecological and economic sustainability. However, macroalgae cover is widespread and sometimes patchy, and large sampling area size limits the opportunities for frequent monitoring of the changing benthic cover, especially in regards to cost.

Video analysis of benthic cover has been utilized as a quick assessment option for large scale sampling (Edmunds and Witman, 1991; Leonard and Clark, 1993). Video transect methodology has the advantage of capturing every quadrat on a given transect in a small amount of time, whereas diver collection relies on sub-sampling for the same amount of time. Video observations thus allow for a larger total area sampled within the same time constraints when compared with diver observation. This has the added advantage of capturing macroalgae conditions at a large number of sites concurrently, on the same sampling day. Video recording is used extensively for the study of coral reefs (Meese and Thomich, 1992; Kohler and Gill, 2006) as well as for macroalgae and seagrass studies (Leonard and Clark, 1993; Bernhardt and Griffing, 2001; Miller et al., 2003; Ninio et al., 2003; McDonald et al., 2006; Bucas et al., 2007). Using video to capture an accurate assessment of species that may drift ashore is unique to Sanibel Island. A seafloor characterization effort for benthic areas surrounding Sanibel was undertaken in a two-year study. This analysis is only part of the total effort, and focuses on a review of two methods for basic macroalgae cover surrounding Sanibel: quadrat and video analysis. Both analyses were grouped into three ecoregions, consisting of a total of 13 sites, sampled over a two year period.

MATERIALS AND METHODS—*Transect locations*—As shown in FIG. 1, a total of 13 100 m transects divided into three ecoregion: inshore, nearshore and offshore ecoregions. The ecoregions were defined by location, bottom characteristics and substrate type (see TABLE 1). Inshore sites were located on the Pine Island Sound side of Sanibel and dominated by seagrass cover. Nearshore sites were located in the Gulf of Mexico and had varied substrate, with areas of sand and silt, abundant shell fragments, and small ledges and hardbottom outcroppings. Offshore sites were also located in the Gulf of Mexico but in deeper water further from shore, with substrates ranging from featureless sand to abundant soft coral and hard rock outcroppings. Inshore, nearshore and offshore sites included 2 (CES11 and GOM16), 6 (GOM01, GOM02, GOM03, GOM04, GOM06 and GOM07) and 5 (GOM05, GOM08, GOM10, GOM11 and GOM12) transects respectively (FIG. 1). The details of the transects are given in TABLE 2. The inshore sites ranged from 2.0 to 3.7 m deep, the nearshore sites ranged from 5.0 to 8.0 m deep, and the offshore sites were from 9.7 to 13.5 m deep.

Transects were outlined using a 100 m leaded line with two dive flag marker buoys at each end. The line was deployed from a small boat at the assigned GPS locations (TABLE 1).

Sampling dates—Eleven sampling events were conducted between September 2008 and June 2010, as shown in TABLES 2 and 3. During each sampling event, the video survey was conducted along the entire length of the deployed transect line, followed by the quadrat survey along the same line.

Macroalgal species—Macroalgae characterized here included attached algae and drifting algae on the substrate in the quadrat. It did not include actively drifting algae in the water column. Cover of macroalgae was determined for the video post-field day in the laboratory. The identification of individual macroalgal species was generally not possible from the underwater video but generalized morphological types or functional groups were distinguishable. Quadrat analysis included active collection of algae for specific identification of species types and biomass measurements (data not presented here).

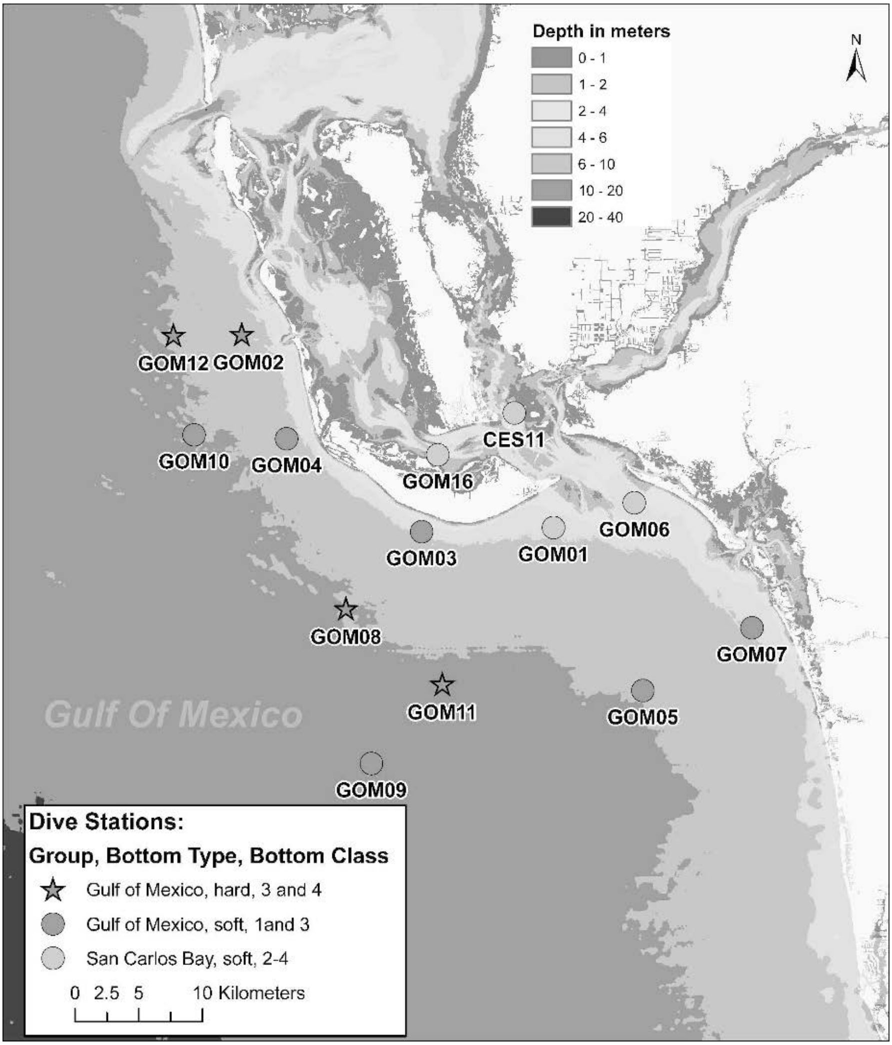

FIG. 1. Map of the study area, Lee County, FL. The figure legend denotes the current and discontinued stations. Sampling began in June 2008 and continued bimonthly until July 2010. Station GOM09 was discontinued in November 2008. GOM12 was added in September 2008 and GOM16 was added in January 2009.

Video survey of macroalgae—A video of each transect was recorded by a diver using a mini DV camcorder (model # DCR-TRV 950 Sony Corp.) with a fisheye lens and housed in an underwater housing (Amphibico Dive Buddy Plus 950). This setup was equipped with actinic lights (Oceanoptics Inc., Cape Coral) and a laser which was 15 cm wide (Beam of Light Technologies, Inc., Clackamas, Oregon). The laser allowed for calibration of distance by providing a reference length, as often used in fish population studies (Meuller et al., 2006). A 53 cm long aluminum rod was attached to the camera housing to maintain the lens at a steady vertical distance to the seafloor (Lam et al., 2006). The SCUBA diver holding the camcorder swam the length of the transect in

TABLE 1. Descriptions of stations visited during this study.

Station	Macroalgae	Mean Depth (m)	Depth of Overlying Sediment (cm)	Prominent Benthic Feature	Latitude (N)	Longitude (W)
CES11	Abundant	3.7 (± 0.9)	>75*	*Diapatra cuprea*	26.4989	82.0456
GOM16	Abundant	2.0 (± 0.8)	>75*	patchy seagrass	26.4660	82.0998
GOM03	Rare	5.7 (± 2.3)	67 (± 33)	live bottom	26.4163	82.2067
GOM04	None	8.0 (± 0.5)	>75*	pen shells, urchins	26.4507	82.206
GOM06	Rare	5.5 (± 0.7)	>75*	pen shells, urchins	26.3049	81.954
GOM07	Rare	5.0 (± 1.0)	>75*	pen shells, urchins	26.3277	81.8832
GOM05	None	7.6 (± 0.6)	>75*	featureless	26.3046	81.9545
GOM08	Rare	10.0 (± 0.9)	34 (± 14)	featureless	26.3628	82.1644
GOM10	Rare	9.7 (± 0.9)	50 (± 33)	large sand waves	26.4839	82.2710
GOM11	Abundant	11.3 (± 1.3)	10 (± 12)	live bottom	26.3091	82.0973
GOM12	Abundant	13.5 (± 1.6)	2 (± 3)	live bottom	26.5545	82.286

* Depth of overlying sediment was deeper than length of measuring rod

approximately 20 minutes. Two divers minimum would be in the water at a time, for safety. Thus, the total time for each video transect was approximately 40 minutes. The camera positioning projected a 1 × 0.7 m^2 rectangular image, broken into frames totaling the length of the transect, with an average of 142 frames per 100 m transect.

Staff and boat time—TABLE 3 shows the estimated staff and boating days needed to complete macroalgae surveys at the 13 sites using the 2 methods. The quadrat survey required up to 5 days to complete the 13 sites, using 4 field researchers per day, totaling approximately 20 field staff days and 5 boat use/rental days. The video sampling, if conducted alone without the quadrat surveys, would require about 2 days to complete the 13 sites, using 2 field staff, totaling 4 staff days and 2 boat use days. Both methods required approximately 2 weeks post-survey time in the laboratory to sort and identify algae from the quadrat or process images from the video footage. The total costs of each method would vary depending on field researcher and laboratory scientist salaries and boat use/rental/maintenance costs. However, general estimates of field costs for the 13 sites, including

TABLE 2. Average percent cover m^2 from video analysis.

Station	Date										
	Sept. 2008	Nov. 2008	Jan. 2009	Mar. 2009	May 2009	July 2009	Sept. 2009	Nov. 2009	Jan. 2010	Mar. 2010	June 2010
CES11	30	*	*	34	0	*	*	15	15	34	*
GOM16	—	–	–	75	*	*	*	0	5	34	*
GOM01	0	0	0	1.4	0	2	*	0	10	29	21
GOM02	2	*	7	0	0	0	0	9	0	0	1
GOM03	0	4	0	2	0	*	*	6	0	0	5
GOM04	*	*	1	*	0	3	0	0	0	*	0
GOM06	*	6	9	6	*	2	0	14	0	34	*
GOM07	0	0	*	4	0	1	*	0	1	9	0
GOM05	0	1	*	1	0	0	*	0	0	0	*
GOM08	6	0	0	23	0	0	19	0	*	0	*
GOM10	*	*	0	0	0	1	0	29	0	*	3
GOM11	4	*	4	9	7	*	11	6	0	0	5
GOM12	–	–	4	5	8	5	16	13	8	*	26

* Poor visibility, therefore video was not interpreted

TABLE 3. Average percent cover per m^2 from quadrat analysis.

Station	Sept. 2008	Nov. 2008	Jan. 2009	Mar. 2009	May 2009	July 2009	Sept. 2009	Nov. 2009	Jan. 2010	Mar. 2010	June 2010
CES11	7	0	2	13	16	15	0	0	12	7	7
GOM16	0	0	0	3	25	0	1	0	1	0	42
GOM01	0	0	0	1	0	0	0	0	0	0	<1
GOM02	0	0	0	0	0	0	0	0	0	0	16
GOM03	0	0	0	0	0	0	5	6	0	0	0
GOM04	0	0	0	0	0	0	0	0	0	0	0
GOM06	1	0	0	1	0	0	0	0	0	0	1
GOM07	0	0	0	0	<1	0	0	0	0	0	0
GOM05	0	1	0	<1	0	0	0	0	0	0	0
GOM10	>1	0	0	0	0	5	0	<1	0	<1	0
GOM11	13	0	0	0	3	12	4	1	0	0	4
GOM12	3	>1	1	3	7	56	34	5	3	0	22

salaries, boat use and insurance (boat and workman's comp) range for \$2,500 per day for the video method to \$3,500 per day for the quadrat method. Total field costs for the 13 sites are estimated to range from \$5,000 for the video method (2 days) to \$17,500 for the quadrat method (5 days). Each of the 13 sample sites was visited once every two months for two years.

Quadrat survey of macroalgae—Divers' direct visual observations of the macroalgae were recorded within a series of 1 m^2 quadrats along each transect. Twenty random quadrat locations (10 each for 2 divers) were surveyed during each transect during each sampling event. Quadrat locations were found using numbered tags on the deployed transect line. A collapsible quadrat was carried with each diver and deployed once divers found the location. The quadrat was subdivided in a grid of 100 squares (10×10 cm^2). Presence/absence of macroalgae was noted within each square to determine the percentage present by m^2 (see TABLE 3). The time to cover a transect is approximately 1 hour, using 4 divers for a total of approximately 4 hours per transect.

Video data processing—Upon returning to the laboratory, the videos were transferred to a computer hard drive via Pinnacle Studio (Ver. 12) software. Exported files were date and time stamped. The camera positioning projected a 1×0.7 m^2 image. The identification of individual macroalgal species was generally not possible from the underwater video but generalized morphological types or functional groups were distinguishable.

TABLE 4. Time comparison of quadrat vs. video methods for macroalgae surveys of 13 sites.

Staff and Boat Days	Quadrat	Video
Field Days/13 Sites	5 Field Days	2 Field Day
Researchers/Field Day	4 Researchers/Day	2 Researcher/Day
Field Researcher Days/13 Sites	20 Researcher Days	4 Researcher Days
Laboratory Analysis Days/13 Sites	10 Scientist Days	10 Scientist Days
Total Staff Days/13 Sites	30 Staff Days	14 Staff Days
Boat	5 Boat Days	1 Boat Day
Insurance (Boat/Workman's Comp)	5 Days	1 Day

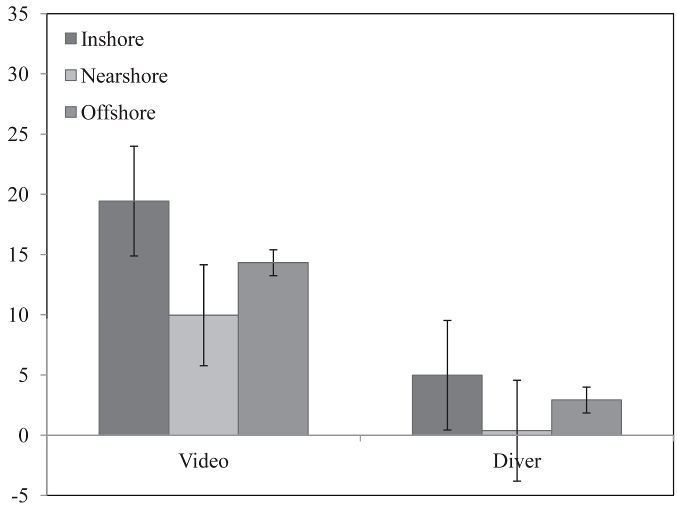

FIG. 2. Average percent cover by m^2 split by ecoregion. Ecoregions include inshore, offshore and nearshore.

Analyses of collected transect video—The analysis of the underwater video footage was performed with Coral Point Count with Microsoft Excel Extensions (CPCE) from the National Coral Reef Institute (NCRI). Using Pinnacle (Ver. 12) software, a reviewer split the video into an average of 142 still images per transect and isolated it into a JPEG file. A total of 21,600 still images were analyzed overall. Also included was an analysis by morphological type, grouped into three observations.

Data analyses—Average percent cover of macroalgae over each transect by m^2 by date was computed using the video and quadrat survey methods, as summarized in TABLES 2 and 3, respectively. Quadrat analysis was computed for 10 quadrats per diver, totaling 20 quadrats per 100 m. Divers collectied quadrat data after the video was taken for each date. Video analysis used the average of the entire sample set, over the same deployed transect line. Percentage covers were arc-sin transformed prior to the test (Fabricius, 1996), using SigmaStat ®. Several comparisons were not considered because of poor water visibility (as indicated TABLE 2 with *). A Kruskal Wallis ANOVA on ranked cover between regions was performed. Data tested were considered significantly different if $p<0.05$.

RESULTS—*Macroalgae percent cover*—Average percent cover estimates of macroalgae for each sampling event and transect for the video method are shown in TABLE 2 and for the quadrat method in TABLE 3. The average percent cover for each ecoregion for each sampling event is shown in Fig 2A for the quadrat method and FIG. 2B for the video method. Results of the video analysis indicated that macroalgae abundance for the inshore stations peaked in March 2009 and March 2010. Nearshore stations had peaks in November 2009 and June 2010, and offshore stations peaked in September 2009, November 2009 and June 2010. Quadrat surveys showed somewhat similar results, with inshore peaks in May 2009 and June 2010; a nearshore peak in June 2010; and offshore peaks in July 2009, September 2009 and June 2010.

When the results of the two methods were compared for each transect and sampling event, there were no significant differences between the video analysis versus the diver collection for two of the three ecoregions - inshore and offshore ($p>0.05$). However, nearshore percent cover was significantly higher for the video transects compared to the diver collection, showing median percent cover for video analysis of 8% vs. quadrat cover of <1% cover ($p<0.001$). Interestingly, the count of the still images containing algae indicate that only ≤50% of the frames contained macroalgae. Divers sub-sampling this same transect missed some of the areas of high algae biomass, especially on transects where percent cover varied greatly along the length of the transect. Results varied by season as well (see TABLES 2 and 3). Trends also indicate that algae abundance was higher for most of the year offshore (FIG. 2). Average percent cover (m^2) grouped into ecoregions for the entire two year period also indicates that video images recorded higher percent cover in all ecoregions (FIG. 3), while overall trends were the same. The greatest abundance of macroalgae was seen offshore and inshore.

DISCUSSION—*Percent cover determination*—While there were no significant differences between overall percent cover between divers and video for most ecoregions, divers overlooked macroalgae that video recorded for nearshore areas on several dates (FIG. 1). This can be attributed to the difficulty in quantitatively sampling large pieces of drift macroalgae from a known area of the seafloor (Bioavailability, 2011). A more thorough analysis of each transect using the underwater video revealed higher percent covers by m^2. This result is similar to studies using comparable video point intercept (PIT) techniques vs. divers in areas with little coral cover (Lam et al., 2005). Results indicated a greater accuracy in areas with little to no coral cover when using video footage.

Macroalgae species determination—Individual species determination is only possible using diver collection and laboratory identification and documentation (collected by group, data not shown here, see Bioavailability 2011; Technical Report). While macroalgae seen in the video frames could be broken down into morphological groups, individual species identification was not possible, with the exception of some *Caulerpa* sp.

Potential uses, advantages/disadvantages of each survey method—Using videography for a benthic macroalgae survey enables researchers to survey a larger area in a smaller time during field operations. Videography also allows observations of many sites in one day to capture aerial extent of macroalgae coverage over a large area concurrently. While processing time between methods is similar, field operations are often more expensive. Therefore, a video approach may be more desirable for researchers with limited budgets. Also, the video approach could be advantages when it is important complete field surveys as soon as possible. One benefit of divers collecting samples is that other biological indices such as the Shannon-Weiner diversity index can be

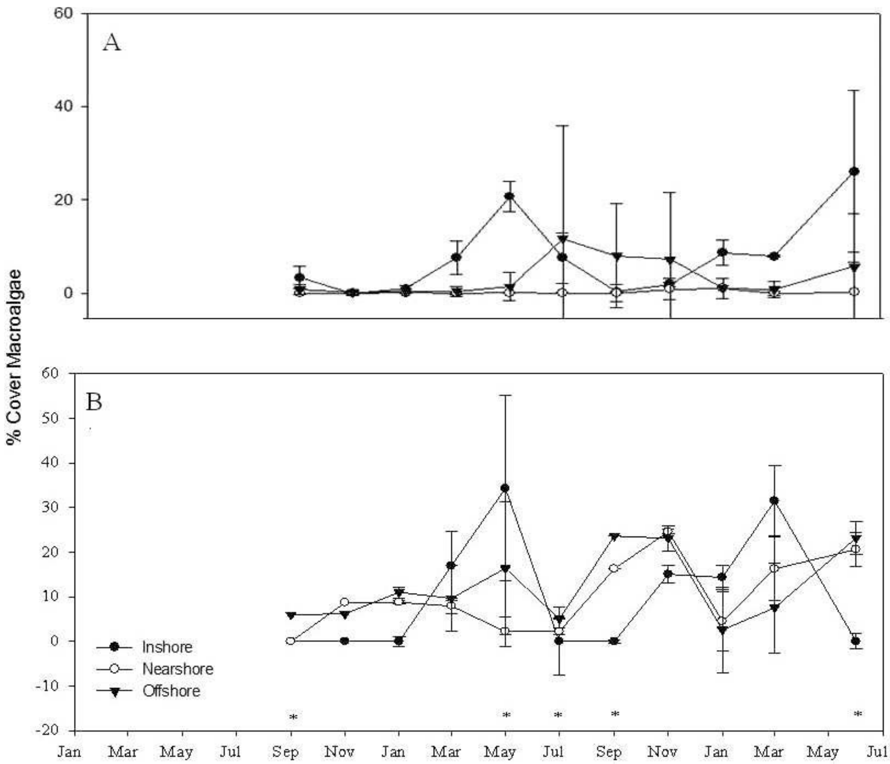

FIG. 3. Percent cover per m^2 estimates for the three regions by date, derived from the quadrat (A) and video (B) analysis. Inshore sites include inshore GOM16 and CES11; nearshore sites include GOM01, GOM02, GOM03, GOM04, GOM06 and GOM07; and offshore sites include GOM11 and GOM12. *Indicates dates where some transect videos were not available, as referenced in TABLE 1.

applied to the field data collected (Preskitt et al., 2004). A disadvantage of diver surveys is that percent cover estimates may differ between observers. (Meese and Thomich, 1992; Preskitt et al., 2004). A total of five divers were employed for this study, potentially increasing variability among results. Using the video analysis approach, one observer/videographer would interpret the results of each quadrat, thus standardizing the data. Video has the enhanced value of creating a permanent record of benthic cover for future studies. Ideally, an approach that utilized both methods would be able to have a mix of benefits, with a permanent record to analyze later, and a greater level of detail from diver surveys.

Time-effectiveness—The estimated field research, laboratory scientist and boat time needed to complete the macroalgae surveys at 13 sites using the 2 methods are shown in TABLE 4. The actual costs would vary depending on field researcher and laboratory scientist salaries, boat rental/use and boat/workman's comp insurance rates. The quadrat method required over twice as much staff time as the video methods because it required more staff per day and more days to complete the 13 sites. This would also necessitate 5 times as many boat use/rental days and associated insurance costs. The laboratory analyses of the quadrate macroalgae samples and the video footage was similar, approximately two weeks for the 13 sites. Research institutions could benefited from the reduced field sampling days time needed to complete the video survey. Less field time also needs less boat and equipment rental and maintenance, less captain rental and less staff training time.

Conclusions—Local resource managers would benefit most from using the video survey method when wide-scale, large area sampling is required. This could benefit smaller laboratories with smaller budget because of the fewer field days required. Contrastingly, diver surveys with collection and attention to quadrats would be able to provide more specific detail, including species analysis and biomass accumulation data. Macroalgae is becoming increasingly common in historically coral-dominated areas, and may continue to become part of managers' assessments of ecosystem health (Miller et. al., 2003). The different techniques satisfy different needs, depending on budget, time and desired specificity of data and bathymetry (flat vs. high relief bottom). Methods for quantification of cover in patchy areas of preferred algae habitat, of both drifting and attached macroalgae biomass will become more important for estuary areas like Pine Island Sound as managers make choices for indicator species and events that deal with increased eutrophication, watershed runoff concerns and hydrologic alterations, such as Caloosahatchee River releases from the lock and dam structures. For drift macroalgae determination along varied ecoregions, video sampling along an entire transect captures the most accurate depiction of percent cover. It is a useful tool for identifying large mats of floating algae, which divers might not encounter on a sub-sampled transect. In contrast, diver collection provides the opportunity for biomass collection and species identification, as well as detailed analyses. Ideally, both methods would be used together under a pre-determined scientifically sound sampling regime designed to capture the information of most benefit to the local resource managers.

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EFFECTIVENESS OF ANTI-FOULING COATINGS IN SOUTHWEST FLORIDA'S ESTUARINE AND MARINE WATERS, WITH EMPHASIS ON REAL-TIME OBSERVING SYSTEMS

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Abstract: *Biological fouling is the accumulation and growth of aquatic organisms on submerged surfaces. Fouling can reduce the operation time and quality of data from aquatic real-time sensors. The SCCF Marine Laboratory currently has seven 'River, Estuary and Coastal Observation Network' (RECON) real-time sensor arrays deployed in the waters throughout southwest Florida. This study's goal was to compare eight commercially available anti-fouling coatings at three RECON stations (Redfish Pass, Gulf of Mexico and Shell Point). At all locations, PVC frames holding six plates (each ~10.2cm^2) with various treatments were deployed. At the RECON sites, plates were deployed for four months and sampled monthly using digital photography. Plate images were analyzed using image analysis software Coral Point Count, for percent cover of organisms such as biofilm and barnacles. Four copper-based coating types were the most effective at preventing fouling, particularly by damaging barnacles and amphipod tubes, especially at high fouling locations. Using the correct coating type at a given RECON site is essential in preventing fouling while also minimizing down-time for redeployment.*

Key Words: Biofouling, coatings, barnacle, biofilm, anti-fouling

Biofouling organisms, such as bacteria, barnacles and tunicates can attach and grow on any surface that stays in the water for a period of time, including boat hulls, docks and navigation pilings. The build-up of biofouling on ships can cause an increase in drag and thus an increase in fuel consumption (Morley et al., 2003; Tang et al., 1998). It can also weigh down floating navigational equipment, cause the erosion of wooden structures and cause damage to certain types of water quality monitoring equipment. Billions of dollars are spent every year in the prevention, maintenance and removal of fouling organisms (Zardus et al., 2008).

A biofilm formation is the first stage of the biofouling process and is an accumulation of bacteria, diatoms and other microorganisms that can attach to a surface within hours of it being submerged. Biofilms provide a more textured surface that is easier for larvae to attach to than the original surface, which may be very smooth and can cue the settlement of larval organisms, such as barnacles (Zardus et al., 2008; Wahl, 1989; Patil and Anil, 2005). There are two main types of sessile fouling organisms: hard (barnacles and oysters) and soft (algae and solitary and colonial tunicates, sponges, etc.). While still in the larval stages, an organism will screen potential sites for attachment suitability, proper chemical cues (such as pH), water flow for filter feeding, electrostatic properties, water depth, and food supply for both the larval and adult forms are necessary (Wahl, 1989; Garcia et al., 1998). If a larvae lands on a site that is deemed unsuitable, it can move back into the water column (Dahms et al., 2004). Once a suitable site has been found the larvae will attach itself to the substratum using one of many methods, the most common of which is a form of chemical adhesion (Abelson and Denny, 1997). It takes approximately one to three weeks after immersion in marine water for a surface to become colonized by a multi-dimensionally structured microbiotic community (Wahl, 1989).

The most common method used to prevent larvae from attaching is the application of anti-fouling coatings. There are two main types of coatings currently used: ablative and hard. Ablative coatings work by flaking away in minute amounts over time, as the boat moves through the water, and are thus constantly revealing a new toxic surface. This prevents anything from attaching and growing on the surface. Hard coatings slowly leach a chemical from the coating into the water that repels organisms and prevents attachment.

Most marine coatings contain a metal component, many of which are considered to be harmful to the environment. Formerly, most marine anti-fouling coatings contained tributyltin ($(C_4H_9)_3Sn$) or TBT because it was found to be the most successful in preventing fouling. However, TBT was found to be toxic to marine organisms (can cause imposex in mollusks and harm organ systems of vertebrates) and has since been banned from use around the world (Radke et al., 2008; Bryan et al., 1986; Tang et al., 1998). Since the ban of TBT, the most common metal used in anti-fouling coatings is cuprous oxide or copper (Cu_2O). Biofilms have been found to develop more slowly and contain fewer healthy cells when growing on these coatings (Tang et al., 1998). There is concern though, that copper leaching into the water may be almost as harmful as TBT to marine organisms and habitats (Perret et al., 2006). There are other types of anti-fouling coatings that are more environmentally friendly and these include E-paint and silicone-based coatings. E-paints release hydrogen peroxide and dissolved oxygen from the water to repel organisms. Silicone based coatings create a slick surface that is more difficult for organisms to attach to.

The Sanibel Captiva Conservation Foundation (SCCF) Marine Lab has seven River, Estuary and Costal Observation Network (RECON) units. The purpose of the RECON units is to collect real-time water quality data accessible anytime from the internet, through the use of a sensor network that measures temperature, salinity, depth, turbidity, CDOM (colored dissolved organic matter), nitrate, dissolved oxygen and chlorophyll *a*. The units are deployed throughout the Caloosahatchee River and estuary at Moore Haven, Ft. Myers, Shell Point, Redfish Pass, Blind Pass, Gulf of Mexico and Tarpon Bay. The Moore Haven unit is constantly in freshwater and the Ft. Myers unit is in freshwater the majority of the time. However, the other five units are constantly in a brackish to marine environment and are therefore subject to extremely high rates of fouling, due to continuous submersion throughout the year. The RECON units come factory equipped with many anti-fouling measures, including copper plates, copper-cladding, a bleach injection system to minimize bacteria and a TBT impregnated 'doughnut' (SCCF RECON, 2010). The lab also takes additional measures to decrease fouling. First, each component of the sensor equipment is wrapped in a plastic tape. Several layers of anti-fouling coating are then applied on top of the taped areas and the frame that is not taped. Cables that are used to connect the various sensors are wrapped in a copper foil tape that also helps to prevent fouling. This entire process can take several days and only helps to minimize, but not eliminate, the amount of fouling that occurs. The RECON units are currently being protected from fouling with E-paint. E-paint is a copper free coating that is used in part because the RECON units' frames are made of aluminum. Using a copper based coating with the aluminum would cause problems with electrolysis in the water (Tang et al., 1998). However, the coating has not been as effective as hoped in the high fouling, high salinity areas where most of the units are located. In the future, SCCF may change the RECON frames to fiberglass, allowing the use of copper-based anti-fouling coatings.

Therefore, the goal of this project was to determine if any of the available types of anti-fouling coatings from Interlux, SeaHawk or Pettit are more efficient at three sites (Gulf of Mexico, Shell Point and Redfish Pass RECON), as compared to E-paint. A more efficient coating will have a lower percentage of hard fouling organism growth, over a longer period of time, in comparison with e-Paint. This study is unique because it is testing the anti-fouling coatings for use on water quality sensors, which do not move in the water and are located in high water flow areas, as opposed to boats that move through the water or docks in a marina where water flow may be restricted. Each of these sites has a different salinity level, suite of fouling species, amount of water flow, etc. The outcome of this research project is to reduce time and effort (materials versus labor costs) associated with maintaining RECON at the current locations.

MATERIALS AND METHODS—*Site design*—The three RECON locations used were Redfish Pass, Shell Point and Gulf of Mexico (FIG. 1). The goal of the study was to determine what coating would maximize sensor performance at several representative locations throughout the RECON deployment area. The study was designed to maximize replicates of each paint type throughout the lower estuarine system. At each of three RECON locations, a total of six 84×20.3cm PVC frames, each holding six 10.2 cm^2 plates, were deployed (n=36 plates/site). At each site, all of the coating types were represented by two replicates (n=8 coatings × 2 plates=16 plates). Frames were deployed in September 2010, which is the start of the three month recruitment peak for the barnacle *Balanus trigonus*, a commonly occurring barnacle in Florida waters (Werner, 1967). The four experimental frames at each site contained one taped, unpainted control plate and five plates that were taped and then painted with anti-fouling coatings. Of the five painted plates, one from each

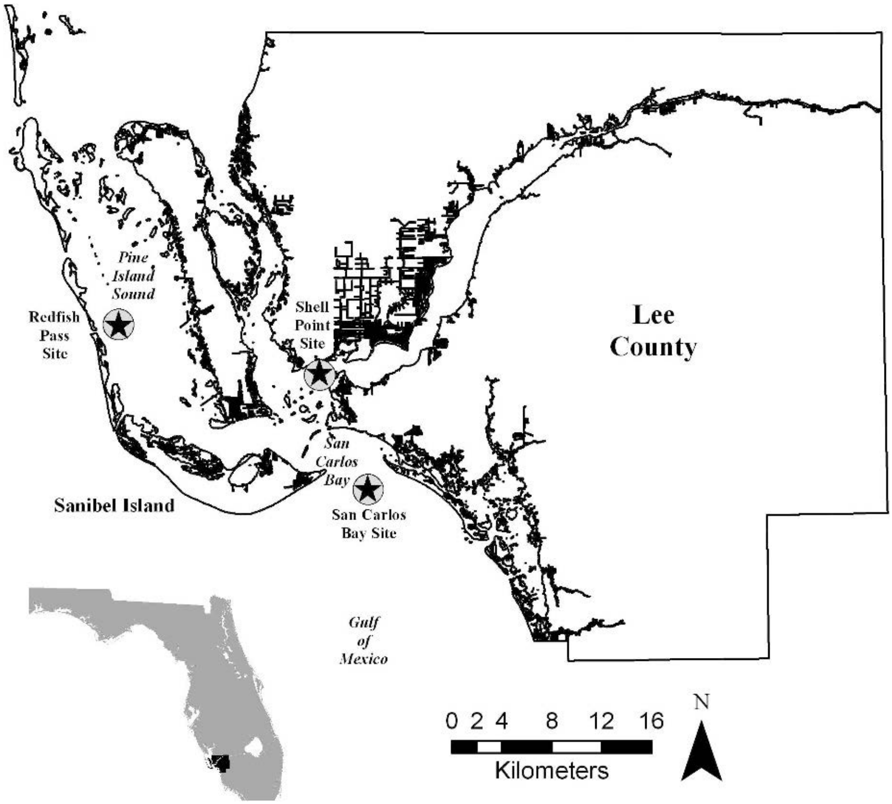

FIG. 1. Sanibel Island and the three RECON sensor locations, which are represented by stars. Redfish Pass is located in Pine Island Sound, Gulf of Mexico is off Ft. Myers beach, and Shellfish Point at the mouth of the Caloosahatchee River.

frame was randomly placed as a 'blank' and therefore was not used in the results (n=24 plates/site: 4 control plates, 16 plates with coatings and 4 'blank' plates). The final two frames each contained six, untaped and untreated plates, that will be used to help assess the organisms comprising the fouling community (n=12 plates/site). The coatings were from three different companies and both hard and ablative coatings were represented. A total of eight coatings were tested (TABLE 1). The

TABLE 1. The eight different paint types and their properties.

Paint	Brand	Copper	Hard/Ablative
SN-1	e-Paint	0%	Hard
Mission Bay	Sea hawk	0%	Hard
Vivid	Pettit	25%	Hybrid
Pacifica	Interlux	0%	Ablative
Ultra	Interlux	50–75%	Hard
Micron CSC	Interlux	25–50%	Ablative
Micron 66	Interlux	25–50%	Ablative
Experimental	Interlux	Unknown	Unknown

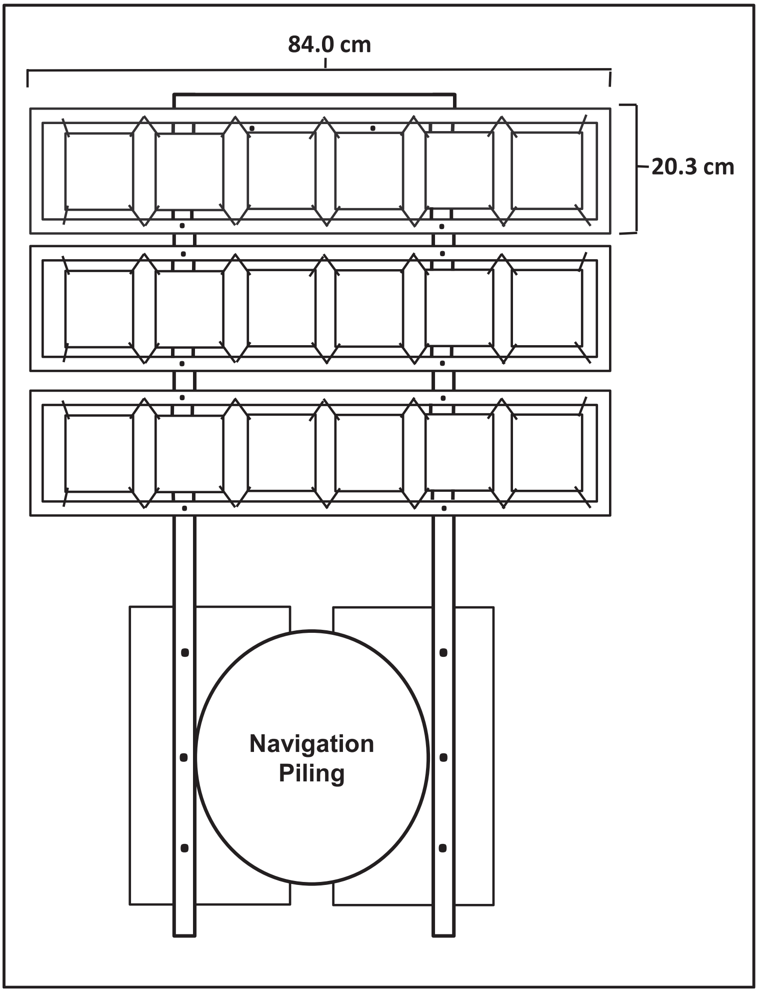

FIG. 2. The design for attaching plates/frames to the navigational pilings at each of the three RECON study sites, viewed as looking from above, down towards the sea-floor bottom. A second set of frames was located 0.45m below the top frames.

painted plates were arranged on the four frames randomly. These frames were pulled out of the water once a month and each plate was photographed. The last sampling visit occurred in January 2011.

The four experimental frames were oriented horizontally in the water and attached to U-shaped PVC mounting structures that were attached to the RECON navigation pilings. The frames were approximately 4.2 m deep at Shell Point, 3.5 m deep at Redfish pass and 5.7 m deep at Gulf of Mexico, depending on the current tides. This positioned them at 1.3 m off the bottom at Shell Point, 1.2 m at Redfish Pass and 2.6 m at Gulf of Mexico. The variation in depths is due to attaching the frames to the already placed, permanent RECON mounting blocks. Two of these U-shaped mounting structures were used at each site, situated about 0.45 m apart, one above the other. Two experimental plate-frames and one untreated frame were attached to each mounting structure and the frames sat next to each other along their longest edge (FIG. 2).

Photographic analysis—All photographs taken of sample plates were analyzed using the computer program Coral Point Count with Excel extension or CPCe. CPCe was used to determine the percent coverage of different organisms on each plate, with the Area/Length Analysis function. Six main categories of fouling were created: biofilm, barnacles, amphipod tubes, tunicates, macroalgae, and other invertebrates. Other invertebrates included hydroids, bryozoans and mollusks. The edges of each plate were outlined first in order to determine an accurate area for each plate. The individual organisms on a plate were identified and then their shape was outlined to determine total coverage area per plate. Area data was exported into Microsoft Office Excel (ver. 2007) spreadsheets. The area of the plates and organisms was used to calculate percent coverage of a type of organism for each plate.

Statistics—A square-root transformation was performed on percent coverage data, excluding the biofilm category, in order to attain equal variances in the data. The biofilm category was excluded because biofilm does not cause damage to RECON sensors and is easily removed.

Differences in percent cover between paint treatments, sites, and sample dates were analyzed using a three-way general linear model ANOVA (GLM) (Minitab 13.2). Interaction effects were analyzed between the paint treatment, site, and sample date factors. After significant differences were found, Tukey pairwise comparisons among levels for location, treatment, and the interaction term location*treatment were made. Significance levels for all tests were $p < 0.05$.

RESULTS—*Organisms Surface Coverage*—The frames containing unpainted plates were used to determine the fouling community at the three study locations. At all of the study locations except Redfish Pass, for all coating types, the biofilm category was found to comprise the highest percentage (generally >40%) of coverage, throughout the course of the study. At Redfish Pass barnacles (≤68.17%) and amphipod tubes (≤85.16%) were the most prevalent for the course of the study. It was often found that the barnacles grew on the plates first, and then the amphipod tubes formed on top of the barnacles, providing a higher than 100% total coverage. Redfish Pass saw a continual increase in barnacle growth over the four months for all coating types except Micron 66, Micron CSC, Ultra and the Experimental. Tunicates were only present at Redfish Pass (≤2.47%) and GOM (≤3.09%). A high percentage of growth at Shell Point, aside from biofilm, consisted of macroalgae (≤30.67%) and barnacles (≤12.07%).

Location—The GLM indicated statistically significant differences between locations (TABLE 2; FIG. 3). Evaluation of the location effect with the Tukey

TABLE 2. General Linear Model: Response of Adjusted Cover (cover relative to control) versus Location (3), Treatment (9), Time (4).

Source	DF	Adj MS	F	p
Location	2	2.023	87.37	0.00*
Treatment	8	1.29	55.96	0.00*
Time	3	0.245	10.61	0.00*
Location*Treatment	16	0.154	6.69	0.00*
Location*Time	6	0.075	3.25	0.01*
Treatment*Time	24	0.065	3.08	0.00*
Error	156	0.023		
Total	215			

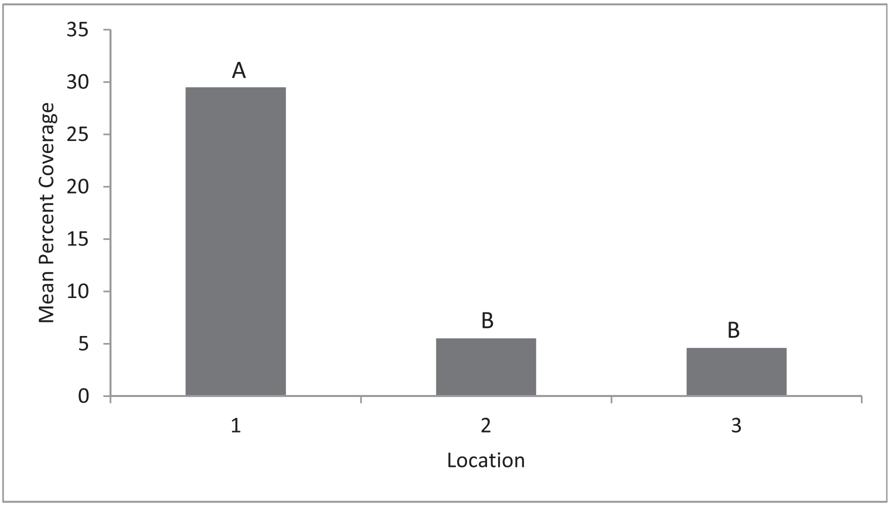

FIG. 3. Mean percent coverage for each location and results of the Tukey Pairwise Comparison test, 3-way GLM, Minitab 13.2. Locations with the same letter are not significantly different. The locations are 1=Redfish Pass, 2=Gulf of Mexico and 3=Shell Point.

pairwise test revealed that Redfish Pass had a significantly higher amount of fouling (29%) than the Gulf of Mexico (T-Value = −11.10; $p<0.00$) and Shell Point (T-Value = −11.77; $p<0.00$). Fouling at the Gulf of Mexico and Shell Point was similar at about 5% and no statistical difference was detected (T-Value −0.628; $p=0.78$) (FIG. 3). The three sites had varying salinity levels throughout the course of the study. Redfish Pass experienced the most stable salinity levels throughout the course of the study, usually ranging between 31–35 ppt. Shell Point experienced the lowest and most variable salinity ranges, usually between 15–33 ppt (FIG. 4).

Coating—The GLM also indicated differences between coating types. For all locations, the control treatment had significantly higher fouling than other coated treatments, as expected ($p<0.00$, FIG. 5). The four copper-based coatings (Micron 66, Ultra, Micron CSC, and Experimental) were not significantly different from each other and exhibited the lowest degree of fouling (FIG. 5). The three copper-free coatings, e-Paint, Pacifica, and Mission Bay had significantly higher fouling than all copper-based coatings except Vivid (FIG. 5; $p=0.00$). The copper-based coating Vivid performed similarly to the copper free coatings, exhibiting a higher degree of fouling than other copper-based paints. Vivid had significantly greater fouling than all other copper-based coatings but significantly lower fouling than non-copper coatings. Interlux Pacifica coating was the best performing of the copper-free coatings and performed better than some copper-based coatings.

When the coating types from all locations are expressed as a percentage of e-Paint, it becomes evident which coatings performed the most effectively in

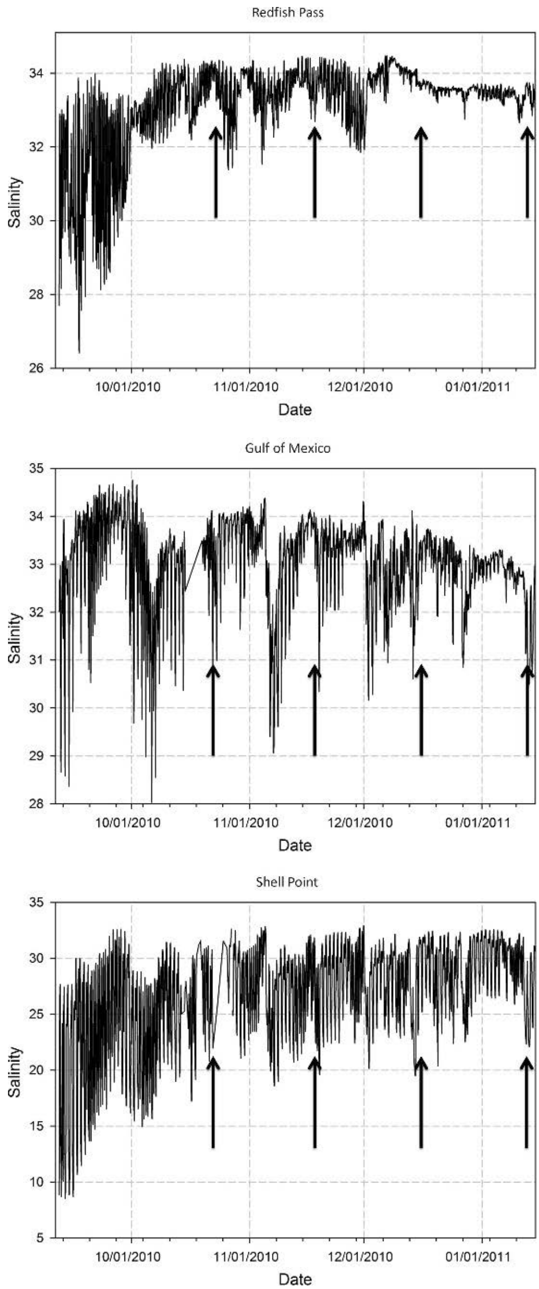

FIG. 4. Salinity ranges for the three RECON study locations, for the four months of the study. Top is Redfish Pass, middle is Gulf of Mexico and bottom is Shell Point. Black arrows point to exact dates of sampling.

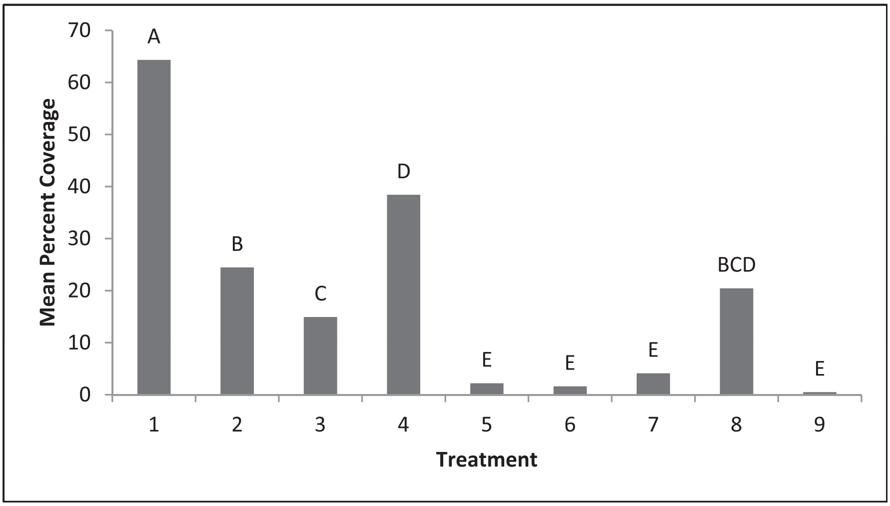

FIG. 5. Mean percent coverage for each treatment and results of the Tukey Pairwise Comparison Test. Treatments with the same letter are not significantly different at $p<0.05$. The treatments are 1=Control, 2=e-Paint, 3=Pacifica, 4=Mission Bay, 5=Micron 66, 6=Ultra, 7=Micron CSC, 8=Vivid, and 9=Experimental.

the deployment area over time. Interlux Micron 66 had the lowest overall percentages of fouling of all eight coating types and experienced only 0–55.06% of the e-Paint fouling over the course of the study. In comparison to e-Paint, the Interlux Experimental paint performed the best, with only 0–11.19% of the fouling that was present on the e-Paint over the course of the study, followed closely by Interlux Ultra, which had 0–13.17% of the fouling of e-Paint (leaving out the January Gulf of Mexico's 161.42% of the e-Paint fouling, where the total Ultra fouling was 2.05% compared to e-Paint fouling of 1.27%). SeaHawk Mission Bay coating was the least effective compared to e-Paint, experiencing up to 3200% of the fouling for e-Paint (TABLE 3).

Coating Performance by Location—Results of the GLM indicated an interaction between coating and location (TABLE 2, FIG. 6). Under conditions of high fouling (e.g. Redfish Pass, TABLE 4) some of the copper free coatings (e-Paint, Mission Bay) did no better than the control. As was the case for the main effect of coating, copper-based coatings outperformed copper-free coatings, with the exception of Vivid. Under low fouling conditions (e.g. Shell Point and Gulf of Mexico), all coatings outperformed the control but differences between copper-based and copper-free coatings were less pronounced and not detected statistically (FIG. 5, TABLES 5–6). This may have been due to a combination of low fouling and the fact that there were only 2 replicates per coating. Given that there were only 2 replicates per coating per location, additional interpretation and comparisons of coating performance in the lower fouling locations should be cautiously approached.

TABLE 3. The amount of fouling at all three study sites and for the seven different coating types (excluding biofilm), over the four months of the study, expressed as a percentage of the e-Paint fouling. For October at Shell Point, e-Paint had 0.00% fouling, therefore the other coating types cannot be expressed as a percentage of the e-Paint.

	October	November	December	January
Redfish Pass				
Pacifica	18.48	64.72	64.64	93.42
Mission Bay	133.18	183.43	109.02	103.71
Micron 66	3.45	55.06	0.31	0.00
Ultra	9.24	13.17	5.33	1.07
Micron CSC	8.75	66.87	0.34	0.25
Vivid	160.26	214.55	32.64	42.09
Experimental	1.94	2.37	0.21	0.29
Gulf of Mexico				
Pacifica	54.89	4.48	220.54	35.43
Mission Bay	285.14	746.77	1829.19	3213.39
Micron 66	0.00	45.27	0.00	9.45
Ultra	1.61	7.21	0.00	161.42
Micron CSC	7.90	161.69	0.00	649.61
Vivid	12.99	143.03	28.65	69.29
Experimental	0.80	11.19	0.00	0.00
Shell Point				
Pacifica	NA	3.09	79.97	8.94
Mission Bay	NA	0.00	73.85	45.21
Micron 66	NA	7.32	0.26	0.00
Ultra	NA	0.00	8.89	5.30
Micron CSC	NA	12.47	2.44	73.73
Vivid	NA	84.78	5.93	11.80
Experimental	NA	2.29	7.97	6.68

DISCUSSION—In the course of the study, the goal was to find an effective anti-fouling coating for use on water quality monitoring sensors around Sanibel Island, FL. There were certain aspects of this study that made it unique, when compared to other studies, such as one conducted by Practical Sailor in 2011 in Sarasota, FL. In their study, they compared 65 anti-fouling coatings, in order to identify the most effective for boat hulls. This was done by applying the coatings to fiberglass panels and attaching the panels to a dock in a marina. When the panels were removed from the water, they were rinsed to remove any fouling that was not firmly attached. The data was analyzed qualitatively, on a scale of excellent, good or poor at preventing fouling and compared to cost per gallon (Nicholson, 2011).

In comparison, the present study quantitatively analyzed eight different anti-fouling coatings for their effectiveness at preventing fouling on water quality sensors at three different locations. To determine a more effective coating for the sensor systems, several parameters were analyzed (e.g. the types of fouling organisms, based on percent cover; the total individual percentages of fouling for each coating type over the course of four months; and the

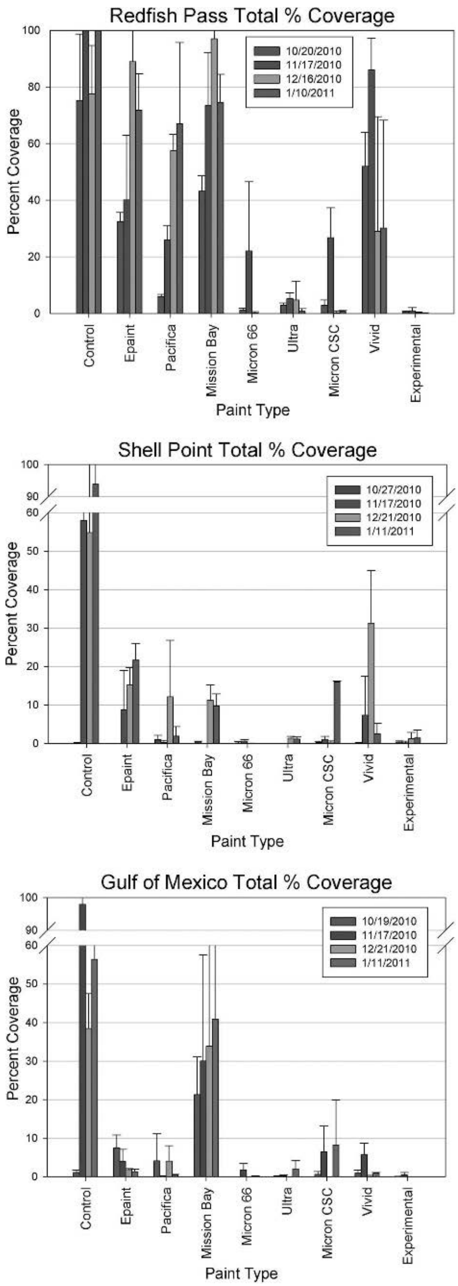

FIG. 6. Fouling as measured by percent coverage at the three sites, for each treatment and each sampling event. Percent coverage does not include biofilm.

TABLE 4. Tukey Pairwise Comparisons of the anti-fouling coatings at Redfish Pass. Percent cover data were square-root transformed. (*) indicates significance at $p<0.01$. The coatings were abbreviated as follows: E-Paint (E-Pa), Pacifica (Pac.), Mission (Miss.), Micron 66 (M-66), Ultra (Ult.), Micron CSC (M. CSC), Vivid (Viv.), Experimental (Exp.).

	Control	E-Pa.	Pac.	Miss.	M-66	Ult.	M.CSC	Viv	Exp.
Control	-	−3.21	−5.45*	−1.34	−10.98*	−10.75*	−10.42*	−4.53*	−12.08*
E-Pa.		-	−2.246	1.866	−7.773*	−7.541*	−7.215*	−1.322	−8.879*
Pac.			-	4.112*	−5.527*	−5.295*	−4.970	0.923	−6.633
Miss.				-	−9.64*	−9.41*	−9.08*	−3.19	−10.75*
M-66					-	0.233	0.558	6.451*	−1.106
Ult.						-	0.325	6.218*	−1.339
M.CSC							-	5.893*	−1.64
Viv								-	−7.557*
Exp.									-

amount of fouling expressed as a percentage of the coating currently being used). Permanent sensor systems are continuously in the water and are in high flow areas and therefore are prone to potentially high levels of fouling. It was found that Interlux Micron 66, Ultra and an Experimental paint, all copper-based paints, were the most effective at minimizing fouling based on total percent cover of fouling compared to the currently used e-Paint. This was especially true at Redfish Pass, a location with higher, less fluctuating salinities and greater potential for fouling. The differences between copper-based and non-copper-based coatings were less prominent at lower fouling locations.

Organism Surface Coverage—The most common category of fouling at all study sites was the biofilm. Once the biofilm had established itself, it likely provided a textured surface for attachment by other organisms, such as barnacles. Because biofilm does not cause damage to the sensors and is easily removed with light scrubbing, it was not included the statistical analysis to evaluate coatings and locations.

TABLE 5. Tukey Pairwise Comparisons of the anti-fouling coatings at Shell Point. Percent cover data were square-root transformed. (*) indicates significance at $p<0.01$. The coatings were abbreviated as follows: E-Paint (E-Pa), Pacifica (Pac.), Mission (Miss.), Micron 66 (M-66), Ultra (Ult.), Micron CSC (M. CSC), Vivid (Viv.), Experimental (Exp.).

	Control	E-Pa.	Pac.	Miss.	M-66	Ult.	M.CSC	Viv	Exp.
Control	-	−4.237*	−6.165*	−5.718*	−7.416*	−7.169*	−7.172*	−4.810*	−6.970*
E-Pa.		-	−1.929	−1.481	−3.179	−2.933	−2.935	−0.573	−2.733
Pac.			-	0.477	−1.250	−1.004	−1.006	1.355	−5.083
Miss.				-	−1.698	−1.452	−1.454	0.908	−1.252*
M-66					-	0.246	0.224	2.606	0.446
Ult.						-	−0.002	2.359	0.199
M.CSC							-	2.36	0.201
Viv								-	−2.160
Exp.									-

TABLE 6. Tukey Pairwise Comparisons of the anti-fouling coatings at Gulf of Mexico. Percent cover data were square-root transformed. (*) indicates significance at p<0.01. The coatings were abbreviated as follows: E-Paint (E-Pa), Pacifia (Pac.), Mission (Miss.), Micron 66 (M-66), Ultra (Ult.), Micron CSC (M. CSC), Vivid (Viv.), Experimental (Exp.).

	Control	E-Pa.	Pac.	Miss.	M-66	Ult.	M.CSC	Viv	Exp.
Control	-	−5.536*	−6.286*	−0.869	−7.369*	−7.135*	−6.223*	−6.311*	−7.653*
E-Pa.		-	−0.750	4.667*	−1.832	−1.599	−0.686	−0.774	−2.116
Pac.			-	5.417*	−1.083	−0.849	0.063	−0.024	−1.367
Miss.				-	−6.500*	−6.266*	−5.354*	−5.441*	−6.784*
M-66					-	0.234	1.146	1.058	−0.284
Ult.						-	0.912	0.824	−0.517
M.CSC							-	−0.088	−1.430
Viv								-	−1.342
Exp.									-

Salinity fluctuations at the three sites likely had an effect on the rate of fouling and thus the percent cover. At Redfish Pass, there were consistently higher levels of salinity with less variation and high rates of fouling, compared to the other two locations. The constant high salinity marine water is a more stable environment for the settlement of fouling organisms. An overall shift in the organism coverage was found on all coating types at Redfish Pass, particularly the non-copper based coatings. Initially amphipod tubes dominated but then changed to be dominated by barnacles by the end of the experiment. This was likely due to a shift in environmental conditions, such as lower salinities at the end of the study period. Barnacles can attach quickly to a surface, grow to an adult size and outcompete other organisms for space (Thiyagarajan et al., 2003). The high fouling rates at Redfish Pass could be partly explained by a constant flow of larvae from the Gulf of Mexico and Pine Island Sound because of the proximity to a tidal inlet along with optimal conditions for attachment and growth.

Shell Point and Gulf of Mexico had more variability and fluctuations in salinity than Redfish Pass (FIG. 4). The higher variability is caused by the proximity to the mouth of the Caloosahatchee River, a major river in lower Charlotte Harbor. It is likely that the salinity fluctuations led to lower fouling rates at these two sites. It is thought that fewer fouling species can tolerate lower salinities and may have been a less suitable environment for marine species. The primary fouling organisms as determined by percent cover at Shell Point were barnacles and macroalgae. Both of these organisms were among the highest percent coverages during the middle two sampling events of the study, when salinities were the most stable at Shell Point. The third location, Gulf of Mexico, was in between the other two locations in terms of salinity fluctuations but experienced the highest number of fouling organism categories throughout the study. The copper-free and copper containing coatings performed equally-well in the lower fouling locations while the copper containing coatings (with the exception of Vivid) performed significantly better at high fouling locations such as Redfish Pass.

Coating Type—The best performing coatings overall were Micron 66, Ultra, Micron CSC and the Experimental. Although Micron CSC was found to not have significantly different percent cover of fouling organisms than the other coatings mentioned. it had higher percent cover. Additional replication would likely provide statistical power needed to distinguish Micron CSC as a sub-optimal performer. These top performing coatings were copper based, which is the coating that was expected to outperform non-copper coatings. Micron 66 and Ultra had higher percentages of copper, 25–50% and 50–75% respectively, than the other coatings used, with the exception of Micron CSC (also 25–50%). The Experimental's coating's copper percentage is unknown as it is not yet commercially available (TABLE 1). Higher percentages of copper in the coatings appear to be related to the potential to reduce fouling. The four copper-based coatings were particularly effective at preventing fouling categories of barnacles and amphipod tubes. In comparison to biofilm growth, these fouling categories adhere strongly to the surface and removal can cause sensor damage or increase the roughness of the surface and likely higher fouling rates. The performance of these four coatings was significantly higher than e-Paint. These four coatings never had more than 55% of the fouling that e-Paint experienced.

Of the copper-free coatings, Pacifica performed the best at all study sites and performed significantly better than Vivid, a copper-based coating. Pacifica was an extremely soft ablative coating and this may have caused it to flake away more readily and contribute to its performance. Pettit Vivid, e-Paint, and SeaHawk Mission Bay performed poorly at all study sites. All three coatings were unsuccessful at preventing the attachment and growth of fouling organisms, particularly barnacles, after only one to two months. Vivid and Mission Bay also had the highest rates of fouling relative to e-Paint, with Mission Bay having over 100 percent of the fouling of e-Paint for eight out of the twelve sampling events. These results imply that these coatings would not be suitable alternative for underwater sensors-given that the sensors are stationary in the water continuously for up to six months. If these coatings were used to protect the bottom of boats, where the boat can move at higher velocities, the coatings would reduce fouling rates because of detachment or ablating.

CONCLUSIONS—The differences among locations in percent cover of fouling organisms were greater than expected. While the goal of this study may have been to determine the best coating for a variety of conditions throughout the RECON deployment area, the differences in fouling rates and performance of copper-free versus copper containing coatings warrants additional research. There may be efficiencies gained by using the same coating in a variety of locations but the performance of a coating varies depending on the location.

Throughout the deployment area copper-based coatings yielded better results (less percentage of fouling organisms) than copper-free coatings. The exception was Interlux Pacifica, which performed almost as well as the Interlux

Micron 66 and Ultra. Every coating tested performed better than e-Paint, the coating currently being used on RECON sensors, with the exception of SeaHawk Mission Bay. Using one of the better performing coatings will allow the RECON sensors to stay in the water longer in between servicing, thus reducing the annual maintenance cost for each unit. The results of this four month study can be used by SCCF and by other sensor users with deployed sensors in tropical, subtropical and warm temperate waters to help select an effective anti-fouling coating, thus reducing excessive fouling and related problems.

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GRASSED SWALE DRAINAGE PROVIDES SIGNIFICANT REDUCTIONS IN STORMWATER POLLUTANT LOADS

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Abstract: *To help refine pollutant loading estimates throughout Sarasota County, the County and the Southwest Florida Water Management District monitored multiple medium-density residential areas to quantify the difference in pollutant loads between areas served by swales and areas served by curb and gutter. Sites were selected to be as similar as possible (e.g., soils, age of development, lawn care intensity, imperviousness, etc.) so that differences in results could be attributed to use of curb and gutter versus swales. Conditions favorable to higher confidence flow monitoring and proximity of sites to each other were also considered in site selection. Monitoring results showed significant differences in flow volumes and pollutant concentrations between the two types of areas. Pollutant loads from the swaled areas were substantially lower (10-fold less for some pollutants) than those from curb and gutter areas. Additionally, the swaled areas demonstrated runoff response patterns characteristic of an area without directly connected impervious area. The findings from this study are highly supportive of low impact design and demonstrate the effectiveness of source control higher in the stormwater system, particularly for systems with disconnected impervious areas.*

Key Words: Stormwater, event mean concentration, EMC, low impact development, swale, best management practice, BMP, BMP performance

In the United States, pollution from stormwater runoff in urban areas is regulated under the National Pollutant Discharge Elimination System (NPDES) through Municipal Separate Storm Sewer Systems (MS4) permits. MS4 jurisdictions are required to obtain an NPDES permit and to develop a stormwater management program. To facilitate a better understanding of the sources and quantities of pollutant loading from stormwater runoff, local stormwater managers may develop pollutant loading models that, in turn, help to fulfill NPDES MS4 requirements. Pollutant loading models typically rely on land use and soils to predict pollutant loading from direct runoff for a given set of rainfall conditions. Land use types in the pollutant loading models must conform to how existing water quality monitoring data have been pooled in order to make use of the large volume of existing data. For example, all residential development within a certain density range is pooled into the single category of medium-density residential development, regardless of variations within the development that could potentially influence its pollutant loading characteristics. This study sought to determine if there was a significant difference in pollutant loading between medium-density residential areas

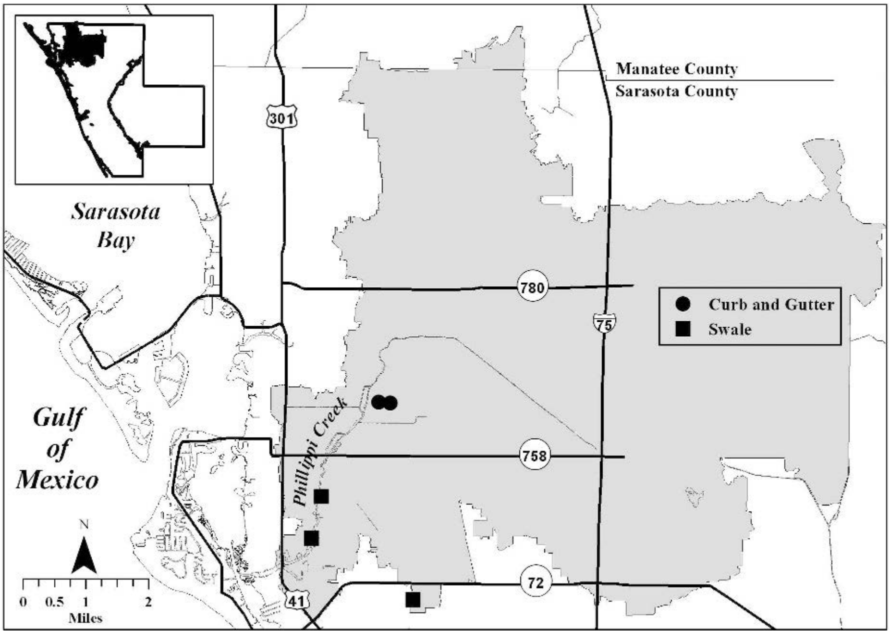

FIG. 1. Map of sampling site locations within Sarasota County's Phillippi Creek Watershed.

served by curb and gutter and those served by grassed conveyance swales/ ditches (not designed bioswales)—a common roadside feature in Sarasota County, Florida (the location of the study). Although conveyance swale drainage is not a recognized best management practice (BMP) under current State of Florida stormwater rules, compared to curb and gutter drainage, the runoff regime and aquifer recharge from swale drainage more closely resembles a natural system (Yang and Li, 2010). Additionally, the presence of curb and gutter or swales is rarely, if ever, distinguished when pooling monitoring data from medium-density residential areas.

METHODS—*Site selection*—In order to attribute differences in pollutant loading to the presence of roadside curb and gutter or roadside grassed conveyance swales, site selection was critical. Selected sites were all located within one portion of the County's Phillippi Creek watershed (Fig. 1), a developed watershed with primarily medium-density residential neighborhoods. Extensive effort was expended to ensure that the sites selected were as close to "all else equal" as possible (i.e., same age of development, soils, lawn care intensity, imperviousness, etc.) and that the physical characteristics of the sites were suitable for sampling (i.e., no standing water or obstructing canopy). Five sites were selected—three with grassed swale drainage and two with curb and gutter drainage. At the time of selection there was no standing water in any of the pipes, and after initial rain events there was no baseflow. Monitoring stations were located where swales discharged into concrete or corrugated pipes. Stormwater runoff quantity was measured at all five sites, and water quality was measured at all but one of the swale sites. Contributing areas at these sites ranged from 1.0 to 6.2 acres.

Sampling—Samplers (ISCO Avalanche) were programmed to begin collection of flow-weighted composite samples after a rainfall amount of 0.5 cm or greater within the time frame of one hour or less and to record rainfall and flow volume at 5-min intervals. Flow weighting was programmed on a site-specific basis and was adjusted over the course of the project as hydrologic conditions changed. There was no defined dry period interval between sampling events. The samplers composited the samples into a glass jar on a flow-weighted basis and cooled them to 3 °C. All samples were handled in accordance to relevant Florida Department of Environmental Protection standard operating procedures, and laboratory analyses were performed by Benchmark EnviroAnalytical—a NELAC-certified laboratory.

Data analysis—Water quality data were delivered by Benchmark EnviroAnalytical Laboratory in electronic format and stored in an Access database. Data management and quality control processes were performed using Microsoft Access© and SAS 9.2© software. Quality control involved qualifying data for known conditions that affected flow measurements (such as displaced sensors) and identification of data that were the result of staff maintenance on the equipment. Individual rain events were identified by a visual inspection of the data. Events were limited to coincident rainfall and discharge, thus rain events that did not generate discharge were not identified as events. At two of the swale sites there was intermittent but substantial baseflow. Baseflow was not separated from runoff in water quality samples; however, the rate of baseflow at the beginning and end of a discharge event was noted, and a linear interpolation was performed to fill the baseflow data between those points. Any flow in excess of the baseflow was considered to be direct runoff.

Data analyses were performed using SAS 9.2. Runoff coefficients (c) were calculated for each rain event at each site using the following equation:

$$C = Direct\ Runoff\ (cubic\ feet) \div [Rain\ (feet)\ *\ Drainage\ Area\ (square\ feet)]$$

Runoff coefficients were only calculated for rain events that generated runoff. Event-specific runoff coefficients were not normally distributed even after natural log transformation. Therefore, the Wilcoxon signed-rank test (Proc NPAR1WAY in SAS) was applied to the data. As there was some variability in percent impervious area between sites, the runoff coefficient for each event was divided by the site-specific percent impervious area to create a modified runoff coefficient (i.e., corrected for differences in impervious area).

Water quality results were analyzed by comparing the grouped swales sites versus the grouped curb and gutter sites. The raw data were not normally distributed; however, natural log transformation improved the data distribution to approximately normal for most analytes. T-tests were used in most cases (Satterthwaite method for unequal variances). In cases where the data were not normally distributed, the Wilcoxon signed-rank test was used. Where percent difference calculations were performed, a ratio of the values being compared was generated (with the smaller value being the numerator), and this ratio was subtracted from 1 (i.e., if comparing 0.25 with 0.5, $1-(0.25/0.5) = 0.5$, or 50%). The resulting value was considered the percent difference between the two values.

Estimated annual total pollutant loads (including baseflow) were generated using a rainfall factor that adjusted rainfall at each monitoring site to the 1.34-m annual mean for Sarasota County as determined from Southwest Florida Water Management District (SWFWMD) data. The rainfall factor was used to adjust the runoff observed at each site during the monitoring period to an average annual runoff. For example, if 1 m of rain fell at a site then the factor would be 1.34 (1.34/1). This factor was then applied to the direct runoff and baseflow volumes at each site to estimate the discharge that would have occurred during a year with average annual rainfall. The mean concentration of each analyte over the study period at each site was applied to the normalized discharge values to calculate the total load due to direct runoff and baseflow. With an absence of baseflow concentrations (i.e., they were not monitored due to funding limitations), direct runoff EMCs (Event Mean Concentration) were applied to baseflow volumes to compute average annual loads. These values likely overestimate baseflow loads and thus underestimate the difference between the two types of sites, as the curb and gutter sites have no baseflow. The total estimated load is the combined load due to direct runoff and baseflow.

TABLE 1. Summary of differences in water quality constituent concentrations at swale and at curb and gutter sites. T-test was used to test for differences in natural log transformed data. Percent difference was calculated as 1−((swale concentration)(curb and gutter concentration^{-1})). Standard deviations are presented in parentheses.

Analyte	Swale Mean Conc. (mg l^{-1})	Curb and Gutter Mean Conc. (mg l^{-1})	Percent Difference in Mean	p-value
Total Nitrogen	1.98 (0.75)	6.17 (4.41)	68%	0.0002
Total Kjeldahl Nitrogen	1.54 (0.61)	5.62 (4.18)	72%	<0.0001
Nitrate + Nitrite Nitrogen	0.43 (0.35)	0.55 (0.45)	22%	0.409
Total Phosphorus	0.77 (0.45)	0.99 (0.78)	22%	0.657
Ortho Phosphorus	0.44 (0.24)	0.52 (0.51)	15%	0.531
Total Suspended Solids	27.79 (28.1)	127.73 (129.37)	78%	0.0002
Biochemical Oxygen Demand	4.45 (2.75)	15.35 (10.52)	71%	0.0004

RESULTS—Rainfall during the study period (March–September 2010) was compared with the period of record in order to place the hydrologic conditions during the study within a historical context. Rainfall data were downloaded from the SWFWMD website and represent countywide composite monthly rainfall totals beginning in 1915.

The results of the comparison of study period rainfall to historic rainfall indicate that the rainfall during the study period was average in terms of the historical record in that it was near the long-term median value. During the study period, 0.99 m of rainfall was the composite total for Sarasota County, compared to the long-term March–September mean total of 1.03 m and the long-term median of 0.98 m. After pooling the data by site type (swale or curb and gutter), there were 19 water quality samples at swale locations and 23 water quality samples at curb and gutter locations. The sampling results for nitrogen (various forms), phosphorus (various forms), total suspended solids (TSS), and biochemical oxygen demand (BOD) are provided in TABLE 1.

Constituents that were present primarily in particulate form were found in significantly lower concentrations at swale sites than at curb and gutter sites ($p < 0.05$; TABLE 1). These included total nitrogen (TN), total Kjeldahl nitrogen (TKN), TSS and BOD. Total phosphorus (TP), ortho-phosphorus, and nitrate+nitrite nitrogen (NOx) are dissolved rather than particulate and did not differ significantly between site types. This suggests that the pollutant removal process at the swale sites is primarily driven by physical filtration of the runoff. Concentrations of particulate nitrogen were found to be exceptionally high during this study. Lab procedures were thoroughly reviewed and samples were run in duplicate to confirm these results. One possible explanation for these high values was an unusually active oak pollen season, as large amounts of oak pollen were observed in these systems.

There were a total of 121 runoff events from the two curb and gutter sites and 112 runoff events from the 3 swale sites. Modified runoff coefficients (corrected for impervious area) were significantly lower at the swale sites compared to the curb and gutter sites ($p < 0.0001$, test statistic = 7702), with

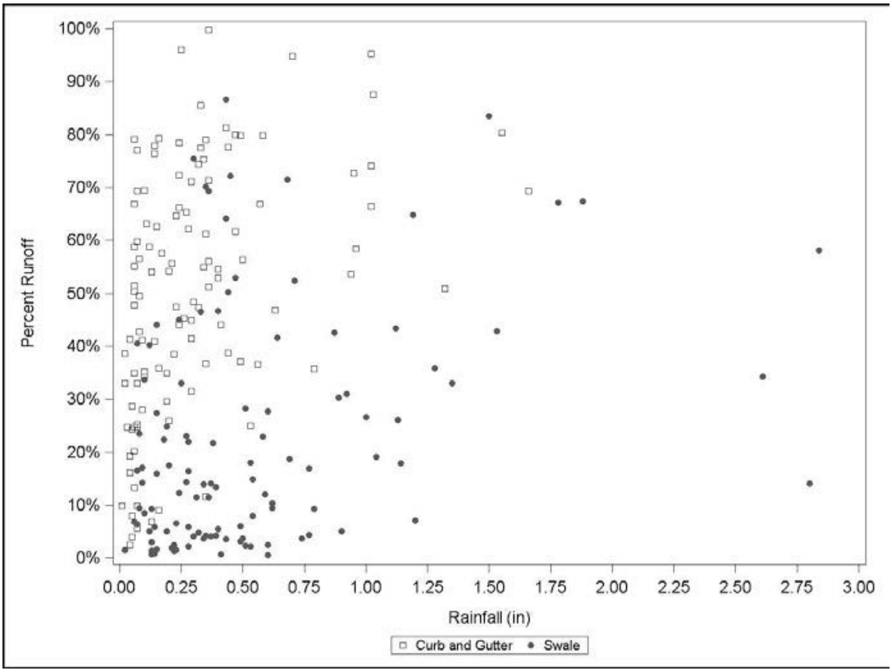

FIG. 2. Per event runoff (as a percentage of rainfall volume) versus event rainfall. Open squares represent curb and gutter sites; solid circles represent swale sites.

average runoff coefficient 58% lower at swales sites and median runoff coefficient 74% lower at swale sites than at curb and gutter sites. In addition to these reduced runoff coefficients, there was an average of 4.39 cm of rain that fell at the curb and gutter sites during the study period without generating runoff, compared to an average of 13.16 cm of rain that fell at the swale sites without generating runoff events. This indicates that small rain events are more frequently fully attenuated at swale sites than at curb and gutter sites. The direct runoff response (Fig. 2) was often greater at curb and gutter sites than at swale sites. Baseflow only occurred at swale sites with substantial slopes and only after large rain events, and was substantial relative to direct runoff when it occurred, often continuing for days following large events. Estimated annual loads due to direct runoff were lower at swale sites than curb and gutter sites for all analytes. The percent differences in estimated annual loads due to direct runoff (baseflow excluded) were 82% lower loads of total phosphorus, 93% lower loads of total nitrogen, 93% lower loads of BOD, and 95% lower loads of TSS (TABLE 2).

Estimated annual loading due to baseflow is combined with estimated annual loads from direct runoff to produce total estimated loads. These estimates are substantially different from the estimated loads due to direct runoff. The percent differences in estimated annual total loads (baseflow included) were 55% lower loads of total nitrogen, 54% lower loads of BOD, 80% lower loads of TSS, and 26% higher loads of total phosphorus; however,

TABLE 2. Estimated annual loading from swale and from curb and gutter sites. Estimates were generated by extrapolating the observed study period loads to the estimated annual loads based on long-term average rainfall (1.34 m). The estimated total load is the estimated load from direct runoff plus the estimated load from baseflow. Baseflow for the swale sites was based on an average of the area-weighted measured response from Swale 1 and Swale 2, as the baseflow values recorded at Swale 3 appear to be unrepresentative.

	Mean Estimated Load in Direct Runoff (kg km^{-2} y^{-1})		Mean Estimated Total Load (kg km^{-2} y^{-1})	
Parameter	Curb and Gutter	Swale	Curb and Gutter	Swale
Total Nitrogen	2401	172	2401	305
Total Kjeldahl Nitrogen	2202	133	2202	237
Nitrate + Nitrite as N	200	38	200	67
Total Phosphorus	382	67	382	117
Ortho-Phosphorus as P	193	38	193	67
Total Suspended Solids	51395	2322	51395	4229
Biochemical Oxygen Demand	5873	392	5873	690

we urge caution in the use of the baseflow data for two reasons. First, because baseflow-specific concentration data were not collected, direct runoff concentrations were used to estimate baseflow loading. Second, the area contributing to baseflow is unknown and is likely larger than the area contributing to direct runoff for at least one of the sites (Swale 3). These factors reduce our confidence in the accuracy of the baseflow load estimates.

DISCUSSION—The key finding in this study is that the estimated annual loads of monitored pollutants in direct runoff from medium-density residential areas served by grassed conveyances swales are less than 10% of those from similar areas served by curb and gutter for most constituents evaluated. This drastic difference is the result of significantly lower concentrations of particulate constituents and significantly lower flow volumes. To put this difference in perspective, total annual nitrogen loads from a medium-density residential area served by grassed conveyance swales and subject to no additional BMPs appear to be two to three times lower than those from a similar area served by curb and gutter with wet detention systems (assuming standard wet detention treatment)—the primary BMP used in the County.

The results of this study support the concept that pollutant removal by grassed swales is primarily a physical process (Deletic, 2005), with particulate analytes such as TSS and TKN having significantly lower concentrations in swale drainages and dissolved analytes such as NOx and ortho-phosphorous having similar concentrations in both swale and curb and gutter drainages. In this study, the majority of nitrogen was in the form of TKN rather than NOx. The significant reduction in concentration of total nitrogen by swale drainage is likely dependent upon nitrogen being in particulate form. The removal of TSS by grass filters has been studied previously, with various removal efficiencies reported, including 61–86% (Deletic and Fletcher, 2006) and 85% (Han et al., 2005). The estimated load reduction of TSS in this study is within the range observed in other studies (Barrett et al., 1998), providing further evidence in support of the findings of the present study.

After runoff coefficients were adjusted for site-specific variation in percent impervious area, runoff was significantly reduced at swale sites relative to curb and gutter sites. This is critically important because it indicates that even if swale, and curb and gutter sites discharged at the same concentrations, loads would still be significantly reduced at swale sites. Therefore, parameters such as NOx and TP, which did not have significantly different concentrations, still have significantly reduced loads. This reduction by infiltration in vegetated swales has been documented in a variety of studies (e.g., Barrett, 2008). It is also important to note that, for the calculation of runoff coefficients, only storm events generating runoff were considered. There were a number of rain events at swale sites for which the entire rainfall volume was fully attenuated on site. In fact, the water quantity results from the swale sites were indicative of areas with no directly connected impervious area.

The impact of swale drainage on loading is expected to vary on a site-specific basis, as there are a variety of features that will impact the efficacy of the swale drainage feature as a BMP. The use of these results should be governed by that understanding. However, these results are consistent with findings in other studies and likely represent a reasonable estimate of the overall impact of swale drainage in the region. Ultimately, it appears that swale drainage provides excellent stormwater treatment in Sarasota County and is more effective as a potential BMP than some other recognized BMPs such as wet detention ponds. The presence of low impact development infrastructure such as grassed swales should be accounted for when managers consider development or redevelopment and should be reflected in the models and permits.

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USING A COLLABORATIVE PARTNERSHIP TO MONITOR STORMWATER BEST MANAGEMENT PRACTICE EFFECTIVENESS: A PROCESS AND PROJECT SUMMARY

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Abstract: *A collaborative partnership of public agencies, private consultants, private developers, and the academic community is being successfully applied to monitoring the effectiveness of Best Management Practices (BMPs) used to treat stormwater in southwest Florida within the Charlotte Harbor National Estuary Program (CHNEP). The resulting publicly available data generated can be used to guide practical stormwater policy revisions by supplying region- and BMP-specific information to policy makers. The public-private-academic partnership provides a mechanism by which the strengths of each partner have combined into a common effort towards completing research projects efficiently, while generating quality data. This partnership has engaged in multiple BMP research projects, including a green roof study, a pervious/impervious pavement study, wet detention/aeration studies, a long term discharge study and a groundwater-surface water interaction study. The studies, some still ongoing, have produced notable results. Variations in stormwater volume and associated pollutant loading discharge are sensitive to site-specific conditions including amount of impervious area, depth to groundwater tables, rainfall distribution, and local geology. Additional monitoring is needed to better understand these relationships. Enhanced local information on BMP function and efficiency will facilitate the creation of treatment trains to best protect downstream receiving waters and meet the needs of natural systems.*

Key Words: Stormwater, BMPs, public/private partnerships, green roof, pervious pavement, groundwater, aeration, wet detention, stormwater monitoring

The purpose of this paper is to provide a summary of the process and methods used to implement five collaborative monitoring projects to evaluate the effectiveness of stormwater Best Management Practices (BMPs) in southwest Florida. The five monitoring projects were conducted in six residential communities within Lee and Charlotte Counties (see Fig. 1) by a public/private partnership. Four entities – a private developer (Bonita Bay Group), consulting firm (Johnson Engineering), state university (Florida Gulf Coast University) and state regulatory agency (Florida Department of Environmental Protection), conducted stormwater BMP monitoring from

FIG. 1. Locations of six residential communities where stormwater BMP monitoring studies were conducted.

2004 through present. The projects included studies of long term discharge behavior of stormwater management systems, groundwater and stormwater management system interactions, deep and shallow stormwater pond aeration, pervious and impervious pavement performance, and green roof utilization for stormwater treatment.

Conducting research studies in today's Florida watersheds can prove challenging given the state, local, and federal regulatory oversight overlaid with political and business interests, as well as those of private landowners. For example, when studying the effectiveness of a particular stormwater BMP, it may be necessary to gain permission to access public and private facilities, which can be difficult because of concerns and competing interests of the various stakeholders. A case in point involved a study to evaluate the effectiveness of wet detention systems for a large box store, (Johnson Engineering, Inc. 2009), which required access to the property. The business entity had legitimate concerns about granting access to the site, including questions about how the data would be used, and how the study activities might impact business, as well as liability issues. Based on the experience of the authors in this collaborative process, private and public entities generally support BMP studies and use of their sites for research. However, these apprehensions often override the willingness of potential partners to participate such stormwater BMP studies. In addition, monitoring the effectiveness of implemented stormwater BMPs is generally not required, contributing to the lack of data. The limited availability of technically sound studies of local BMP performance for southwest Florida makes such data collection imperative, since it can be used to improve system design, reduce pollutant loadings and facilitate regulatory processes.

For the five projects discussed in this paper, the collaborative partnership of regulatory, private developer, private consulting and academic entities provided a successful strategy for completing technically sound and cost-effective monitoring of stormwater BMP effectiveness. The partnership team approach was initiated in 2002, as the Florida Department of Environmental Protection (FDEP) began a long term effort to increase the effectiveness of Florida's stormwater treatment program with a focus on enhancing nutrient load reduction. As part of this effort, a team was assembled in 2002 to work on a series of Low Impact Development (LID) and Wet Detention effectiveness monitoring projects. A contract for this work was awarded to the Bonita Bay Group (BBG) in 2002 based on its long history of environmentally sensitive residential and mixed use master planned community projects that incorporate various LID practices and environmental restoration projects. BBG hired Johnson Engineering, Inc., to develop and implement a series of studies to this end because of the firm's experience with similar projects. During the design and implementation phases of the initial projects, it became apparent that a full time partner from academia would benefit the outcome of the studies, so staff from Florida Gulf Coast University (FGCU) was invited to join the collaboration team.

Each team member filled specific study needs. BBG contributed its experience with innovative LID practices and provides access to its communities. FDEP provided funding for the studies used to generate data that could be used to augment stormwater rulemaking. Johnson Engineering developed and implemented the monitoring projects conceived by the study team, and FGCU staff provided ongoing project design evaluation, data collection, and data analysis.

A multi-disciplinary team was able to respond efficiently to a variety of project challenges resulting in economical, scientifically sound data generation. Monitoring plan development, site selection, study techniques, identification of parameters of interest, timing of sampling and observations, and data interpretation are multi-disciplinary questions best managed by a team with a wide spectrum of skills. Private developers like BBG can best facilitate site access and community support. Private consulting staff from Johnson Engineering can carry out the day-to-day research activities, and react to routine study challenges such as extreme weather events, vandalism of equipment, and equipment maintenance issues. Representing the academic community, FGCU staff can review data sets and conduct data trend analyses, and expose students to water quality and BMP research projects involving operational stormwater facilities. As a regulatory and resource management agency, FDEP staff offer their experience with past BMP monitoring projects that helps to guide planning of studies and data collection techniques, which maximizes use of research study dollars. Additionally, because of a regulatory agency's close linkage to rulemaking, study findings can be incorporated into rules and policies leading to improved water quality in Florida watersheds. For example, results from studies performed by Johnson Engineering and others for the Florida Department of Transportation (FDOT) were used to update land use-related runoff values for roadways used in state stormwater ("Harper") calculations (Applied Technology and Management, 2010; FDOT memos April 25, 2011 and July 7, 2011).

Below are synopses of the five stormwater BMP projects conducted by the collaborative partnership since 2002, presented in chronological order of when the projects began. Three of the projects have been completed (Deep and Shallow Stormwater Pond Aeration, Pervious/Impervious Pavement Study and Green Roof) and two of the projects are currently still in the data collection stage (Long Term Discharge Study and the Groundwater Interaction Study).

METHODS—*Project locations*—The five collaborative stormwater BMP monitoring projects were conducted within six master planned residential communities in Lee and Charlotte Counties, developed by BBG. The residential communities include Sandoval, Shadow Wood Preserve, The Brooks, Mediterra and Twin Eagle (see FIG. 1). Three of the communities are located in Bonita Springs within the Estero Bay watershed, including Shadow Wood Preserve, The Brooks and Mediterra. Sandoval is located in Cape Coral within the Matlacha Pass watershed and Twin Eagles is located in Naples within the Golden Gate Canal System watershed. The stormwater sub-basins for the projects range from 163 to 783 hectares, including distinct basin boundaries and stormwater overflow structures (see TABLE 1). The typical habitats found within each of the projects include mixed coastal uplands and wetlands in three of the communities (Verandah, Sandoval and Shadow Wood Preserve) and inland flat prairie in three of the communities (The Brooks, Mediterra and Twin Eagles). The typical single-family home lot size is larger in Mediterra, Twin Eagles and Shadow Wood Preserve than in the other 3 communities. And, higher density, multi-family units are included in Verandah, Sandoval and The Brooks. Five of the residential communities, all but Sandoval, contain golf courses.

The five stormwater monitoring project purposes, sampling regimes, parameters, costs and results access and uses are summarized in TABLE 1 and described in more detail below, in chronological order.

Green Roof Study—The Green Roof Study was conducted from 2002 through 2005 in Shadow Wood Preserve (see FIG. 2). Prior to 2002, limited information existed on the design, construction, and planting of green roofs in south Florida. The four objectives of the study were to: 1) determine what types of vegetation are most suitable for roof-top stormwater retention in southwest Florida semi-tropical climate, 2) demonstrate the feasibility and effectiveness of un-irrigated green roof systems by evaluating differences in vegetation health and survivability using different substrates, 3) evaluate the potential for this Low Impact Development (LID) BMP to reduce the quantity of rain runoff, and 4) determine if green roof stormwater treatment credits might be utilized in state regulatory processes. The study was originally implemented as a collaborative effort of Roof Systems, Inc., Bonita Bay Group, FDEP, and Johnson Engineering (Roofscapes, Inc., 2003).

To accomplish the objectives, a demonstration green roof system was installed during July 2003 on the roof of a non-air conditioned metal golf maintenance storage building in Shadow Wood Preserve (see FIG. 2c). The green roof included three test plots approximately 74 square meters each, separated buffers of 0.5 m wide. A consistent type of growth media, lightweight shale aggregate with a volumetric maximum moisture content of 35%, of variable depths, 8 cm, 10 cm and 14 cm, was used within the three test plots. Initial vegetation species included those in the genera *Sedum* (stonecrop), *Delosperma* (aizoaceae), *Euphorbia* (crown-of-thorns), and *Portulaca*, with a small trial group of plants from the genus *Zephyranthes* (rain lily), *Aloe*, *Spartina*, and *Agapanthus*. Plants were installed in groups of 10–15 distributed evenly over each of the three zones, using approximately 1300 individual plants.

The green roof was monitored 2003 to 2006 on a casual basis at approximately monthly intervals., By 2006, most of the original planted species had died off slowly, with the exception of a few aloes, rain lilies, and crown-of-thorns, most likely due to extended dry periods experienced typically between November and June, and high temperatures typically experienced during the July–September periods. In March 2007, following the design and installation of a cistern/irrigation system, the roof was replanted with a group of species that were more tolerant of these conditions, approximately 1200 total plants. Those were common aloe, crown-of-thorns, and rain lily, from the original planting list,, as well as blue flag iris, purple lovegrass, beach sunflower beach purslane and muhly grass. Monthly observations were then made which included plant counts and overall percent coverage (see TABLE 2). Vegetation, physical and water quality conditions were monitored (TABLE 1). Vegetation was monitored by trained ecologists on staff with Johnson Engineering for percent cover, and survivability, Plant observations were made in accordance with guidelines set out in South Florida Water Management District (SFWMD) Environmental Monitoring Report Guidelines. Physical conditions were monitored for temperature, heat flow, moisture content, rainfall and flow, using sensors and recorders generally accepted in the instrumentation industry. Rainfall and flow measurements were used to calculate rainfall runoff volumes. Roof thermal characteristics were recorded from readings produced by a set of sensors installed on and in the roof media and stored in a standard Campbell Scientific data logger for later analysis. Water quality parameters monitored included: cadmium, chromium, copper, zinc, ortho-phosphate, total phosphorus, Kjeldahl nitrogen, nitrate+nitrite, total nitrogen, total ammonia and total suspended solids. Laboratory analyses were conducted by Benchmark Labs, using standard FDEP field procedures and Standard Methods or EPA approved laboratory analysis techniques.

Data was compiled into a series of Excel files and PDF files containing the lab results, including blanks and duplicate testing to ensure lab data integrity

The total estimated budget for the Green Roof Study was approximately $198,000, with about 35% in-kind services provided by the collaborative partners (TABLE 1).

Pervious/Impervious Pavement Study—The Pervious/Impervious Pavement Study was conducted from 2003 through 2005 in the Shadow Wood Preserve (see FIG. 1). The objective of the project was to evaluate the effectiveness of porous concrete (pervious pavement) as a stormwater BMP by comparing the water quality loadings in the stormwater runoff from porous concrete and standard asphalt (impervious pavement).

To accomplish the objective, a temporary clubhouse parking lot in Shadow Wood Preserve was constructed with pervious pavement on one side (0.25 hectares) and standard asphalt pavement on the other side (0.45 hectares), separated by a gradient that prevented surface water runoff from moving between the two sides. Runoff from each pavement type flowed to a storm grate located near the center of each of the two distinct parking areas.

The water flow and quality monitoring was conducted from December 2003 through January 2005 during ten rainfall events that produced discharge.

The storm grates for each parking area were outfitted with modified inlet grate boxes that forced the runoff through v-notch weirs in order to measure flows using level sensing bubbler modules. Additionally, a water table monitor well was installed between the two parking areas and outfitted with a pressure transducer and datalogger to measure groundwater levels (see FIG. 3). Automated samplers were used to collect flow composited water quality samples of stormwater runoff from each pavement type. The samplers were also equipped with dataloggers and additional monitoring equipment to collect rainfall and flow data. Groundwater levels were also monitored. Flow composited samples were collected using automated samplers.

TABLE 1. Summary of Collaborative Stormwater BMP Monitoring Projects.

Project	Green Roof	Pervious Impervious Pavement	Deep & Shallow Stormwater Pond Aeration	Long Term Discharge	Groundwater Interaction
Purpose	Measure plant viability & stormwater quantity & quality from Green Roof.	Measure water quality/ quantity from porous pavement and asphalt.	Measure effects of aeration on wet detention pond water quality.	Measure rainfall & wet detention pond discharges.	Measure surface & groundwater interactions in wet detention ponds.
Project Period	2002–2008	2003–2005	2004 & 2006	2006–2012	2010–2012
Partners					
Lead	Bonita Bay Group	Bonita Bay Group	Bonita Bay Group	Bonita Bay Group	Bonita Bay Group
Design	Roofscapes, Inc.	Bonita Bay Group	FGCU/JEI	FGCU/JEI	Johnson Engineering
Location	Lee County	Lee County	Lee County	Lee/Collier County	Lee County
Field Collection	Johnson Engineering	Johnson Engineering	Johnson Engineering	Johnson Engineering	Johnson Engineering
Lab Analyses	Benchmark Labs	US Biosystems	Benchmark Labs	Benchmark Labs	Benchmark Labs
Data Analyses	Johnson Engineering	FGCU/JEI	FGCU/JEI	FGCU/JEI	Johnson Engineering
Reporting	Johnson Engineering	Johnson Engineering	Johnson Engineering	Johnson Engineering	Johnson Engineering
Community Locations	Shadow Wood Preserve	Shadow Wood Preserve	The Brooks	Verandah, The Brooks, Mediterra, Sandoval, Shadow Wood Preserve, Twin Eagles	Verandah and Shadow Wood Preserve
Drainage Basin Size	Roof Area 2400 SF	1.0 Ac	4 Lakes Totaling 8.9 Ac	6 Basins Totaling 3490 Ac	444 Ac/242 Ac
No. of Sampling Sites	3	2	4	11	8
Frequency of Sampling	Event Based (23 total)	Event Based (10 total)	Bi-Weekly	Event Based (178 total)	varies
Field Parameters	plant species, temperature, heat flow, moisture content, rainfall, runoff flow, …	rainfall, runoff flow, groundwater levels, …	dissolved oxygen, ORP, pH, temperature, turbidity, specific conductance	water levels, runoff flows, rainfall, pH, temperature, specific conductance	water levels

TABLE 1. Continued.

Project	Green Roof	Pervious Impervious Pavement	Deep & Shallow Stormwater Pond Aeration	Long Term Discharge	Groundwater Interaction
Laboratory Parameters	ammonia, cadmium, chromium, copper, Kjeldahl nitrogen, nitrate+nitrite, ortho-phosphate, nitrogen, phosphorus, suspended solids	ammonia, cadmium, chromium, copper, Kjeldahl nitrogen, nitrate/nitrite, ortho-phosphate, nitrogen, phosphorus, suspended solids	turbidity, ammonia nitrogen, Kjeldahl nitrogen, nitrogen, nitrate+nitrite, ortho-phosphorus, phosphorus, chlorophyll a, specific conductance	ammonia, Kjeldahl nitrogen, nitrate/nitrite, ortho-phosphate, nitrogen, phosphorus, suspended solids, copper, iron, lead	none
Estimated Total Project Cost	$198,000	$40,000	$70,000	$640,000	$45,000
Estimated % In-Kind by Partners	35 BBG	100 BBG	10 BBG	10 BBG	20 BBG
Funding Sources	FDEP	FDEP	FDEP	FDEP	FDEP
Data Available from	mlohr@johnsoneng.com	mlohr@johnsoneng.com	mlohr@johnsoneng.com	mlohr@johnsoneng.com	mlohr@johnsoneng.com
Report Available from	mlohr@johnsoneng.com	mlohr@johnsoneng.com	mlohr@johnsoneng.com	mlohr@johnsoneng.com	mlohr@johnsoneng.com
Uses of Results	BMP Improvement/ Rulemaking	BMP Improvement/ Rulemaking	BMP Improvement/ Rulemaking	Future Pollution Loading Assessments/Rulemaking	Future Pollution Loading Assessments/Rulemaking

FGCU - Florida Gulf Coast University;
JEI - Johnson Engineering, Inc.
BBG - Bonita Bay Group.
FDEP - Florida Department of Environmental Protection.
BMP - Best Management Practice.

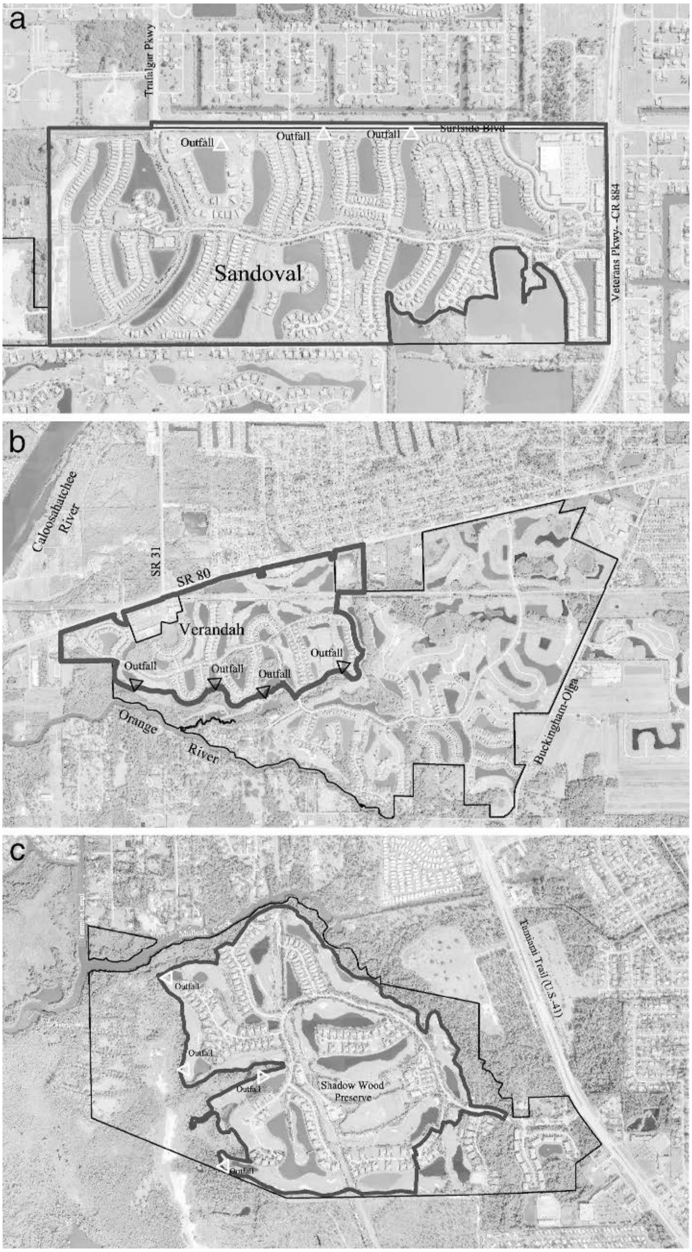

FIG. 2. Locations of stormwater BMP monitoring sites within study communities. a) Sandoval Long Term Discharge Study monitoring site locations. b) Verandah Long Term Discharge Study monitoring site locations. c) Shadow Wood Preserve Green Roof, Pervious/ Impervious Pavement, Long Term Discharge and Groundwater Interaction Study monitoring sites.

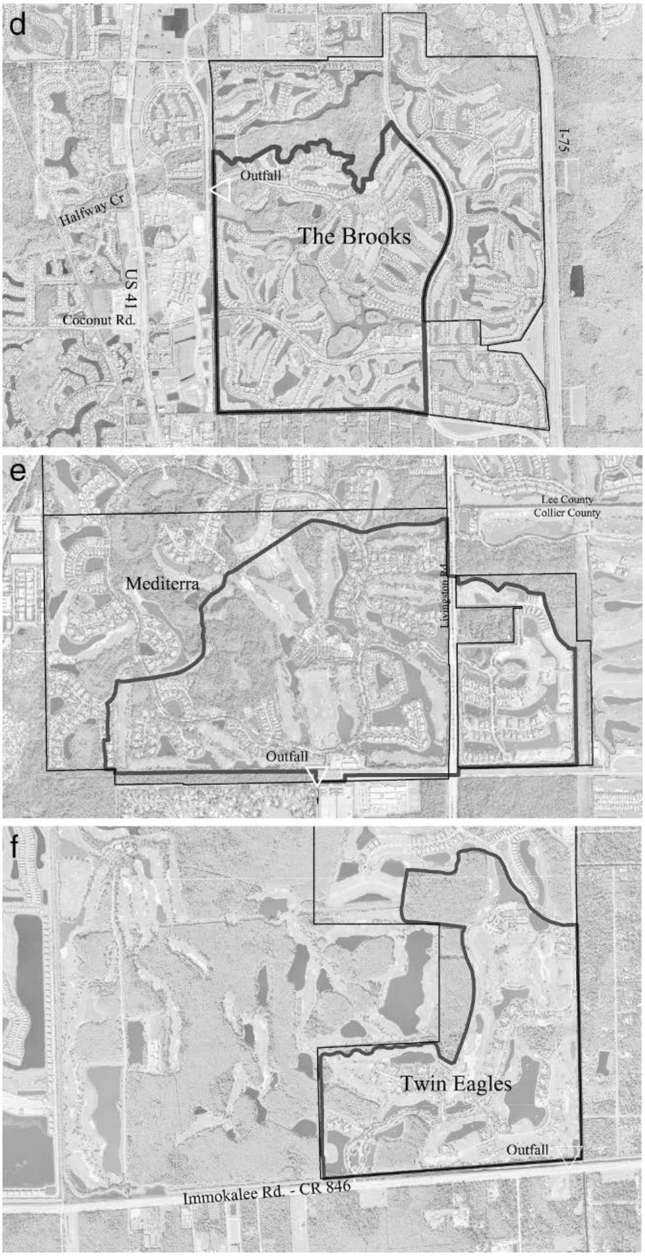

FIG. 2. Continued. d) The Brooks Deep and Shallow Stormwater Pond Aeration, Long Term Discharge and Groundwater Interaction Study monitoring sites. e) Mediterra Long Term Discharge and Groundwater Interaction Study monitoring sites. f) Twin Eagles Long Term Discharge and Groundwater Interaction Study monitoring sites.

TABLE 2. Summary of Green Roof Study vegetation plantings and survival from April 2007–April 2008 at Shadow Wood Preserve.

Common Name	Apr 07	May 07	Jul 07	Sep 07	Nov 07	Jan 08	Apr 08	No. Lost
Plot 1 (South Panel)								
Percent cover	UNK	UNK	UNK	50%	25%	65%	80%	N/A
Undesirable plant species coverage	UNK	UNK	UNK	15%	0%	0%	8%	N/A
Common aloe	189	180	180	102	95	94	92	97
Crown-of-thorns	79	71	65	60	54	53	49	30
Blue flag iris	NP	NP	56	3	0	1	1	55
Marsh elder	1	1	1	1	1	1	1	0
Beach sunflower	1	4	12	6	7	7	20	−19
Rain lily	UNK	UNK	15	13	0	0	15	0
Beach purslane	6	0	UNK	7%	0%	48	28%	N/A
Muhly grass	4	1	80	30	31	28	26	−22
Plot 1 Totals	±280	±257	±408	±215	±188	±232	±217	
Plot 2 (Center Panel)								
Percent cover	UNK	UNK	UNK	65%	40%	55%	65%	N/A
Undesirable plant species coverage	UNK	UNK	UNK	10%	0%	1%	12%	N/A
Common aloe	173	173	173	149	155	160	154	19
Crown-of-thorns	96	92	47	34	34	28	27	69
Blue flag iris	NP	NP	24	0	0	0	0	24
Purple lovegrass	NP	NP	47	47	51	47	47	0
Rain lily	UNK	UNK	8	3	0	0	0	8
Beach sunflower	1	4	14	13	10	12	36	−35
Beach purslane	12	0	UNK	12%	0%	16	13%	N/A
Muhly grass	3	12	10	1	1	1	2	1
Plot 2 Totals	±285	±281	±323	±247	±251	±264	±264	
Plot 3 (North Panel)								
Percent cover	UNK	UNK	UNK	55%	35%	55%	82%	N/A
Undesirable plant species coverage	UNK	UNK	UNK	5%	0%	1%	3%	N/A
Common aloe	84	84	85	167	180	193	222	−138
Crown-of-thorns	71	71	63	57	54	55	56	15
Purple lovegrass	NP	NP	28	28	15	26	22	6
Beach sunflower	10	0	2	2	2	2	3	7
Beach purslane	8	4	UNK	25%	0%	36	25%	N/A
Rain lily	UNK	UNK	9	23	15	13	8	1
Muhly grass	20	0	7	1	1	1	1	19
Plot 3 Totals	±193	±159	±194	±278	±267	±326	±304	

Water flow parameters were determined using standard V-notch hydraulic calculations based on water levels. Water quality samples were analyzed for: cadmium, chromium, copper, zinc, ammonia nitrogen, nitrate+nitrite, total Kjeldahl nitrogen, total nitrogen, orthophosphate, total phosphorus, and total suspended solids. Laboratory analyses were conducted by US Biosystems using standard FDEP field procedures and Standard Methods or EPA approved laboratory analysis techniques. Data was compiled into a series of Excel spreadsheets representing the water levels and laboratory test results.

FIG 3. Pervious Pavement Study at Shadow Wood Preserve showing the inlet and sampler.

The total estimated budget for the Pervious/Impervious Study was approximately $40,000, with about 100% in-kind services provided by the collaborative partners (TABLE 1).

Deep and Shallow Stormwater Pond Aeration Study—The Deep and Shallow Stormwater Pond Aeration Study was conducted in 2004 and 2006 in the Brooks Preserve (see FIG. 1 and FIG. 4). The objectives of the study were to: 1) evaluate differences in water quality in wet detention ponds of various depths under aerated and non-aerated conditions, and 2) determine if the presence or absence of aeration in the wet detention ponds impacts stratification of the ponds.

To accomplish the objectives, dissolved oxygen levels and other water quality parameters were monitored in four wet detention ponds of varying depths, from 3 to 5.5 m (see FIG. 2) in under aerated and non-aerated conditions.

Monitoring was conducted in two 15-day phases in 2004 from September to December and in 2006 from September through October, (see TABLE 1). The 2004 testing was done under static or no-flow conditions in the lake system, and the 2006 phase was done during a period when the lake system was experiencing flows though the system and eventually out the final discharge structure. During each phase, aerators in two of the four ponds were turned off and the other two were left on. A variety of water quality parameters were collected for both the 2004 and 2006 phases of the study including: turbidity, pH, ammonia nitrogen, total nitrogen, nitrate+nitrite, ortho-phosphorus, total phosphorus, chlorophyll-a and specific conductance. Field monitoring methods used submersible datasondes (see FIG. 5), portable multi-parameter meters and traditional grab sampling for laboratory analysis. All field and laboratory testing done using FDEP Standard Operating Procedures and Standard Methods or EPA approved methods for wet lab analysis. The monitoring was repeated to test the effects of aeration under static (no discharge) and flowing (discharge) conditions over the two-year period. Multivariate analysis was applied to this water quality data set to examine similarities and differences among the detention ponds through time

FIG. 4. Deep and Shallow Stormwater Pond Aeration Study at the Brooks showing the collaborative process in June 2007.

and under the different flow conditions. Data was compiled into a series of Excel files and PDF summaries to contain the field data, laboratory data and water level data. The study generally indicated mixed results as far as levels of tested water quality parameters were concerned, but indicated a clear stabilization of dissolved oxygen levels and a general lack of stratification in the aerated lakes.

The total estimated budget for the Deep and Shallow Stormwater Pond Aeration Study was approximately \$70,000 with about 10% in-kind services provided by the collaborative partners (TABLE 1).

Long Term Discharge Study—The Long Term Discharge Study (LTDS) was implemented in 2007, is on-going and is being conducted in all six of the project residential communities (see FIG. 1). The objective of the study is enhance understanding of the behavior of stormwater discharges from wet detention systems in southwest Florida under a variety of landforms and development types, by monitoring water stage and rainfall conditions.

To accomplish the objective, rainfall and runoff quantity and quality are being measured in six wet detention stormwater pond systems, located in the various project residential communities (see FIG 2a–2f). The stormwater drainage sub-basin areas are provided in TABLE 1.

Beginning in the spring of 2007, rainfall, flow and water quality began being monitored. Rainfall and water levels (flows) were recorded at hourly intervals and water quality samples taken based on discharge events generated by rainfall at the six locations (TABLE 1). Physical conditions being monitored include continuous rainfall and water levels upstream and downstream of the stormwater pond discharge structures (see FIG. 6). Discharges are monitored using datalogging pressure transducers installed at the outfall structures (Johnson Engineering, 2007), using industry standard pressure transducers from Infinities USA and In-Situ. The installation and set up was done under the direction of registered professional engineers licensed to practice in the state of

FIG. 5. Deep and Shallow Stormwater Pond Aeration Study Aeration Study at The Brooks Showing the submersible datasondes used in the study.

Florida and the survel level loops to the water level gages overseen by a Florida registered Surveyor and Mapper. Detailed information on the Infinities USA instruments can be found at http://infinitiesusa.com/Site/Pressure_Water_Level_Data_Loggers.html, and on the In-Situ Level Troll 500 instruments at http://www.in-situ.com/products/Water-Level/Level-TROLL-Family. The pressure transducers are housed in PVC pipes and suspended just above the pond bottoms at locations that should remain constantly submerged throughout the year. The cables suspending the pressure transducers link them to dataloggers mounted on top of open-ended 2-inch PVC pipes, for the Infinities equipment. In the case of the In-Situ equipment, the pressure transducer and datalogger are both located on a probe suspended by a cable from the top of the PVC pipe. The PVC pipe is strapped to a steel pipe driven into the pond bottom, or fixed to the outfall structure in a rigid fashion.

Starting in 2007, water quality monitoring was added to the study (Johnson Engineering, 2008). The parameters are consistent for all six project sites and include standard nutrients and some metals (see TABLE 1). Again, water quality samples are collected according to FDEP Standard Operating Procedures (SOP) for field and laboratory work and analyzed by Benchmark Laboratories, a state certified analysis laboratory, using Standard methods or EPA approved methodologies From 2008 through 2012, 82 surface water sample sets were collected during discharge conditions, 82 surface water sample sets collected during static or no discharge conditions, as well as 38 sample sets from adjacent groundwater wells, for a total of 202 water quality data sets. A summary of mean values for the tested surface water parameters for all project locations is attached as FIG. 9. Discharge volumes are calculated from measured water levels and outfall structure geometry utilizing established and accepted hydraulic formulas (Brater and King, 1976; Merit, 1983; Seelye, 1996). The water levels and formulas are entered into Microsoft Excel spreadsheets and hourly and annual discharge volumes are calculated and the results are displayed graphically. Water quality data are compiled into a series of annual Excel files and reviewed for

FIG. 6. Long Term Discharge Study at the Brooks showing the south outfall structure.

data irregularities and inconsistencies prior to chart preparation. A typical stage-rainfall hydrograph is depicted in FIG. 8.

Currently data collection is planned through November of 2012 with a final report in January of 2013. See *Long Term Surface Water Discharge Study Report for Calendar Year 2009* (Johnson Engineering, 2010) for additional details.

The total estimated budget for the LTDS for operations from 2006–2012 is approximately \$595,000, with about 10% in-kind services provided by the collaborative partners (TABLE 1).

Groundwater Interaction Study—The Groundwater Interaction Study was initiated in 2008 as a part of the overall LTDS; it is on-going and is being conducted at two of LTDS project locations, the Verandah and Shadow Wood Preserve (FIG. 1). The objective of the study is to enhance understanding of surface and ground water interactions in wet detention systems in southwest Florida, specifically as they relate to annual discharge volumes and pollutant loadings.

To accomplish the objective, groundwater level and quality data are being collected, to augment surface water level, rainfall, surface water quality and discharge data from the LTDS.

Surface discharge behaviors at the six LTDS sites monitored to date suggest that surface water-groundwater interactions may exert influence over how stormwater exits some of the stormwater systems. For example, the infrequency of discharge events at Twin Eagles (2% of runoff during 2008 and 2009) indicates that a significant quantity of stormwater may exit the wet detention ponds via the highly transmissive Surficial aquifer (7,700 m^2/d; Missimer International, 1998). Major managed canals proximate to Twin Eagles along the southern and eastern site boundaries may control water table gradients in the area, which in turn could influence discharge from the wet detention systems. On the other end of the spectrum, fine sand with silt and clay characterized by low transmissivity (25 m^2/d or less; CDM, 2005) composes the Surficial aquifer at the Verandah project site. Consequently, water within the Verandah stormwater management

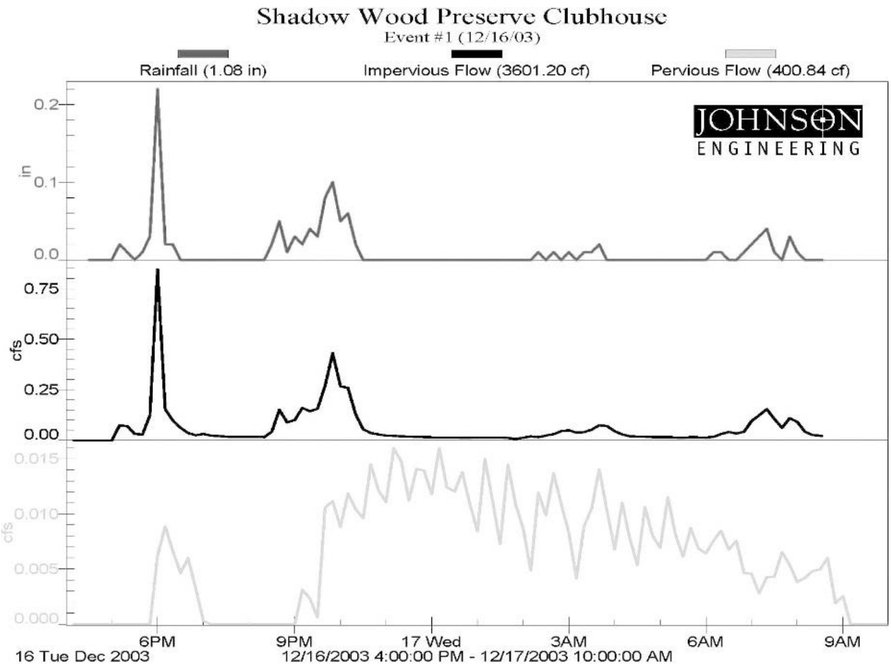

FIG. 7. Pervious/Impervious Pavement Study example event discharge chart.

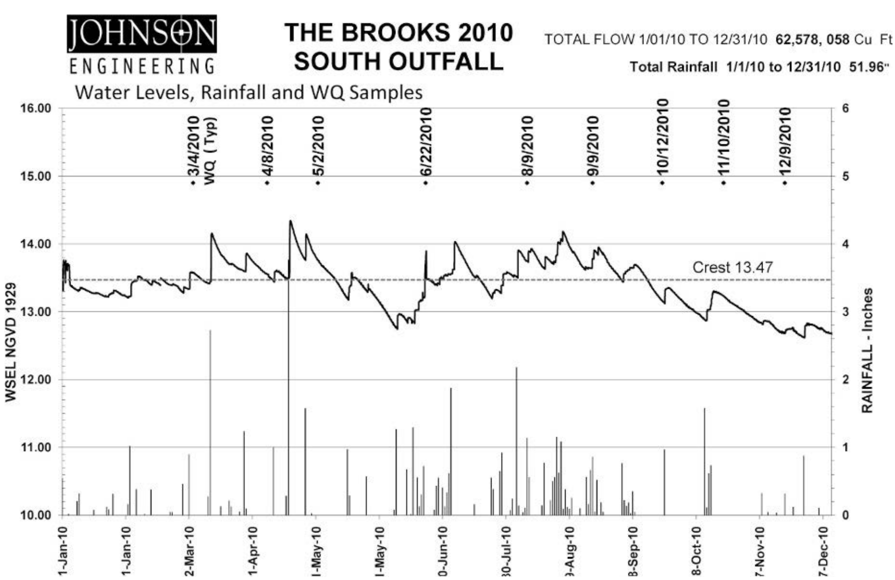

FIG. 8. Long Term Discharge Study example stage and rainfall hydrograph with water quality sampling dates for 2010.

LTDS Surface Water Quality Summary Table 2007-2012

Annual Average Surface Water Samples

JOHNSON ENGINEERING

	Verandah						Sandoval						Shadow Wood Preserve					
	2007	2008	2009	2010	2011	2012	2007	2008	2009	2010	2011	2012	2007	2008	2009	2010	2011	2012
Number of Sample Sets	**1**	**5**	**5**	**8**	**4**	**8**	**1**	**5**	**5**	**10**	**4**	**7**	**0**	**5**	**3**	**9**	**3**	**8**
Nitrogen NH_3 mg/l	0.05	0.08	0.13	0.08	0.07	0.10	0.09	0.23	0.42	0.38	0.07	0.20	NS	0.17	0.07	0.05	0.06	0.05
Nitrogen Kjeldahl mg/l	0.91	1.11	0.94	1.18	0.99	1.00	1.01	1.72	1.44	1.52	1.49	1.32	NS	1.17	1.17	1.22	1.56	1.15
Nitrate+Nitrite mg/l	0.01	0.01	0.07	0.08	0.09	0.14	0.01	0.03	0.05	0.18	0.05	0.24	NS	0.05	0.02	0.03	0.06	0.03
Nitrogen Total mg/l	0.91	1.11	1.01	1.26	1.08	1.13	1.01	1.75	1.49	1.71	1.54	1.55	NS	1.21	1.19	1.22	1.60	1.19
Ortho Phosphorus mg/l	0.03	0.03	0.09	0.02	0.01	0.03	0.01	0.28	0.16	0.20	0.12	0.34	NS	0.03	0.02	0.01	0.01	0.01
Total Phosphorus mg/l	0.12	0.11	0.12	0.10	0.06	0.06	0.04	0.37	0.23	0.32	0.24	0.46	NS	0.08	0.03	0.05	0.04	0.04
Chlorophyll a mg/M^3	27.5	47.0	13.6	45.5	2.5	5.5	25.7	44.3	31.5	17.8	43.0	15.1	NS	20.4	15.3	14.2	12.4	13.7
Chloride mg/l	NS	156	131	139	200	141	NS	105	154	137	240	374	NS	265	214	201	288	167
Total Suspended Solids mg/l	4.6	4.8	3.6	6.9	8.9	2.2	5.6	9.6	5.3	7.9	12.8	7.6	NS	5.2	4.1	8.0	16.7	7.0
Total Hardnhess mg/l	NS	307	220	295	194	175	NS	206	188	204	211	247	NS	309	301	295	287	244
Copper ug/l	NS	1.17	1.14	1.74	5.05	2.96	NS	9.96	4.80	10.69	12.54	7.38	NS	3.40	2.75	13.71	8.18	5.17
Iron ug/l	NS	248	339	243	69	91	NS	86	75	95	316	128	NS	146	100	112	133	126
Lead ug/l	NS	0.67	0.67	0.83	0.77	0.67	NS	0.67	0.67	1.00	0.67	0.67	NS	0.67	0.67	0.75	0.67	0.76

	The Brooks						Mediterra						Twin Eagles					
	2007	2008	2009	2010	2011	2012	2007	2008	2009	2010	2011	2012	2007	2008	2009	2010	2011	2012
Number of Sample Sets	**0**	**5**	**3**	**9**	**3**	**7**	**0**	**4**	**1**	**10**	**3**	**6**	**0**	**3**	**1**	**10**	**3**	**5**
Nitrogen NH_3 mg/l	NS	0.06	0.10	0.06	0.03	0.03	NS	0.12	0.05	0.04	0.03	0.03	NS	0.09	0.07	0.05	0.05	0.11
Nitrogen Kjeldahl mg/l	NS	0.84	0.85	0.96	1.04	0.95	NS	0.70	0.91	0.80	0.80	0.99	NS	1.11	1.28	1.25	1.34	1.35
Nitrate-Nitrite mg/l	NS	0.01	0.02	0.02	0.03	0.01	NS	0.01	0.01	0.03	0.02	0.04	NS	0.03	0.07	0.05	0.04	0.09
Nitrogen Total mg/l	NS	0.85	0.87	0.97	1.07	0.95	NS	0.71	0.92	0.83	0.82	1.03	NS	1.14	1.35	1.29	1.38	1.44
Ortho Phosphorus mg/l	NS	0.03	0.07	0.01	0.01	0.02	NS	0.02	0.02	0.01	0.01	0.02	NS	0.04	0.01	0.01	0.01	0.01
Total Phosphorus mg/l	NS	0.07	0.08	0.05	0.04	0.05	NS	0.06	0.04	0.04	0.03	0.05	NS	0.10	0.06	0.06	0.04	0.03
Chlorophyll a mg/M^3	NS	24.0	16.4	12.8	9.5	11.5	NS	25.3	23.9	18.2	8.3	36.4	NS	44.5	21.6	22.1	13.3	22.5
Chloride mg/l	NS	74	91	103	132	132	NS	23	29	26	29	27	NS	10	18	18	20	20
Total Suspended Solids mg/l	NS	2.8	2.5	6.5	6.3	2.3	NS	5.6	5.3	4.2	5.7	7.3	NS	9.7	11.4	7.7	16.1	6.2
Total Hardnhess mg/l	NS	178	166	193	140	170	NS	268	198	280	245	242	NS	206	314	282	267	259
Copper ug/l	NS	3.95	2.37	12.79	8.80	5.72	NS	3.32	3.03	3.41	2.49	5.16	NS	3.17	2.50	3.18	2.11	2.20
Iron ug/l	NS	140	46	75	69	64	NS	378	84	101	63	187	NS	61	161	48	74	72
Lead ug/l	NS	0.67	0.67	0.73	0.77	0.67	NS	0.93	0.67	0.69	0.71	0.81	NS	0.67	0.67	0.67	0.67	0.67

FIG. 9. Long Term Discharge Study water quality sampling summary 2007–2012.
1. Averages include discharge and static samples.
2. The following projects contain multiple sample sites in the combined averages: Verandah, Mediterra and Twin Eagles.
3. NS = not sampled.
4. ND = not detected at minimum detectable level (MDL).
5. Groundwater samples are not included in this table.

system cannot enter the surrounding aquifer as readily. This site discharges a greater volume of the overall runoff (nearly 50% of runoff); consequently, more pollutant loading is discharged annually despite the study basin's proximity to the Orange River along the southwest site boundary, which has typical water levels several feet below control elevations of the wet detention system (3.5 to 4.0 feet NGVD for the study basins vs. approximately 0 feet NGVD for the Orange River).

The two groundwater interaction monitoring sites are intended to represent the two extremes of discharge behavior:, in the Verandah representing a high-discharge site with low Surficial aquifer transmissivity, and Shadow Wood Preserve, representing a low-discharge site with high Surficial aquifer transmissivity (see FIG. 2). Both sites were selected because of the existing water table monitoring networks and well defined boundary conditions. Both monitoring sites have several existing Surficial aquifer wells that have been incorporated into the monitoring network.

In October 2010, Surficial and Sandstone aquifer monitor wells were either installed or were existing at Verandah and Shadow Wood Preserve (see FIG. 2.) and equipped with data logging pressure transducers identical to those used to monitor water levels in the LTDS primary program. Instruments at the Mullock Creek and the Orange River sites monitor water levels at hourly intervals surface water level elevations of selected ponds within the study basins are measured also at hourly intervals and compared with data from the groundwater monitoring wells. Just prior to the aquifer performance testing, the groundwater level recorders are reprogrammed to record at closer intervals, some at 3 minute intervals and some at 1 second intervals to properly monitor the drawdown effects of the pumping operation on the aquifer being tested. In addition, evapotranspiration (ET) gauges were installed at both sites in March 2011, in accordance with manufacturer's instructions. These ET gages were electronic recording devices sold by Novalynx and capable of evaporating a reservoir equivalent to 12" total volume, but did not deliver consistent long term reliable data. Beginning in March 2011, quarterly water quality measurements commenced at the Surficial and Sandstone aquifer monitoring wells . Wells were sampled for the same water quality parameters measured in the stormwater ponds (see TABLE 1). Water quality samples are being collected and analyzed according to the same FDEP protocols as the surface water sampling in the general LTDS program. Prior to sample collection, field measurements for temperature, pH, specific conductance, dissolved oxygen and turbidity will be allowed to stabilize within the levels specified in the FDEP SOPs for groundwater sampling, as described in Sections 3.1 or 3.5 of *FS 2200 Groundwater Sampling*.

Following the groundwater level data collection starting in October of 2011,the hydraulic properties of the Surficial and Sandstone aquifers were assessed through on-site testing programs at both sites. Data generated from constant rate pumping tests was fitted to type curves using the Aquifer Win32 software in order to determine the hydraulic properties of the aquifer (Fetter, 1994; Sanders, 1998). Groundwater flow modeling (with Winflow or MODFLOW software) was also used to simulate the test conditions and calibrated to match observed water levels during the test.

In 2013, water budgets will be developed to quantify in-flows and out-flows for the Surficial aquifer within the selected study basins,, including rainfall, ET, groundwater flow into and out of the study basins, groundwater flow to or from underlying aquifers, and irrigation water withdrawals and contributions. Data will be compiled into a series of standard Excel files and reviewed for data irregularities and inconsistencies and water budgets will be calculated using industry standard good engineering practices. Based on data collected during the monitoring program, flow nets will also be developed to illustrate groundwater flow conditions for selected time periods. See *Groundwater Monitoring Plan Long Term Discharge Project* (Johnson Engineering, 2011) for additional detail.

Water quality data are being compiled into a standard Excel database and general statistical tests, including maximum, minimum and mean values will be conducted using standard Excel functions.

The total estimated budget for the Groundwater Interaction Study is approximately $45,000, with about 20% in-kind services provided by the collaborative partners (TABLE 1).

RESULTS—*Green Roof Study*—A summary of the vegetation results of the study is provided in TABLE 2. The results of the initial plantings done in 2003 were that with the lack of additional irrigation through the seasonal dry periods from November to June and the high summer temperatures prevented the initial plant species from thriving, as all species showed significant decreases in percent cover and in plant numbers in all 3 test plots. Based on the short term study results, the *Sedum* varieties did not tolerate the hot humid conditions as well as the other species planted. With the exception of *Sedum,* the planted species grew steadily through the initial 2003 summer and fall wet season and the *Portulaca* expanded across the entire roof. By the following late spring of 2004, all species were showing signs of low moisture stress, with some moderate loss of plants. The less drought tolerant species were replaced with plants in the genera *Aloe*, *Euphorbia*, *Helianthus*, *Portulaca*, and *Spartina* prior to the summer of 2005. By 2006, all planted species grew well during the wet seasons, but had difficulty maintaining healthy populations through the cooler, extended dry seasons. In 2007, a cistern and irrigation system were developed and installed to capture and store roof runoff from periods of excess rainfall for later application to the plants via a standard drip tube irrigation system. In 2007, after the cistern and irrigation system were fully operational, plants in the genera *Aloe*, *Euphorbia*, *Eragrostis*, *Iris*, *Helianthus*, *Portulaca, Muhlenbergia*, *Spartina*, and *Zephyranthes* were again installed on the green roof. The *Iris* and *Muhlenbergia* did not acclimate well to the green roof conditions, but the remaining species did fairly well through the summer, fall, and early winter season. Aloe and Euphorbia have persisted across the entire roof to date. Casual observations indicate that shallow rooted xeric species may be a more viable plant option for green roofs in south Florida.

Results of the rainfall, flow and water quality measurements are summarized in TABLE 3. Over the two year study, rainfall events ranged from 6.3 cm to 0.8 cm and runoff flow were calculated to provide a mechanism for doing flow composited sampling. Total phosphorus ranged from 0.78 mg/l to 0.03 mg/l and total nitrogen ranged from 10.94 mg/l to 0.92 mg/l.

More detailed study results and data are shown in TABLE 2.

Pervious/Impervious Pavement Study—Results of the physical, water quality, rainfall and flow measurements are summarized in TABLE 4. Over the two year study, 10 monitored rainfall events ranged from 0.53 cm to 3.81 cm and the average runoff per hectare was 40.8 m3 for the impervious area and 18.7 m3 for the pervious area. Total phosphorus ranged from 0.013 mg/l to 0.100 mg/l and total nitrogen ranged from 0.62 mg/l to 1.92 mg/l in the impervious area samples. In the pervious area, samples showed total phosphorus ranged from 0.049 mg/l to 0.430 mg/l and total nitrogen ranged from 1.01 mg/l to 1.86 mg/l. No significant differences were found in concentrations of total nitrogen in the samples collected from the pervious vs. impervious pavement areas. Total phosphorus levels were higher in the pervious samples. However, the runoff volumes measured from the pervious

TABLE 3. Summary of Green Roof Study rainfall and water quality monitoring results from 2003–2008 at Shadow Wood Preserve.

Rainfall 2003–2008	
Total Events	589
Total Rainfall	678 cm
Ave. Event Rainfall	1.1 cm
Max. Daily Event Rainfall	12.7 cm
Monitored Events	23
Avg. Monitored Event	2.7 cm
Max Monitored Event	6.3 cm
Min Monitored Event	0.8 cm

Water Quality 2007–2008 15 Sampling Events		
	TN	TP
Site GRI		
Avg.	3.09 mg/l	0.26 mg/l
Max.	7.52 mg/l	0.78 mg/l
Min.	1.11 mg/l	0.02 mg/l
Site GR2		
Avg.	2.67 mg/l	0.20 mg/l
Max.	9.24 mg/l	0.63 mg/l
Min.	0.92 mg/l	0.03 mg/l
Site GR3		
Avg.	3.27 mg/l	0.29 mg/l
Max.	10.94 mg/l	0.77 mg/l
Min.	1.04 mg/l	0.04 mg/l

parking area were significantly less than those from the impervious pavement side resulting in significantly higher load reductions. Pervious pavement runoff volumes for each monitored event ranged from 5%–65% of the calculated total potential runoff volume and generally did not occur until 20–30 minutes after each rain event began, as opposed to the nearly immediate runoff response from the impervious pavement. An example study discharge event chart depicting reduced runoff is shown in Fig. 7.

Deep and Shallow Stormwater Pond Aeration—Results of the rainfall, flow and water quality measurements are summarized in TABLE 5. Rainfall events

TABLE 4. Summary of Pervious/Impervious Pavement Study rainfall and water quality monitoring results from 2003–2005 at Shadow Wood Preserve.

Rainfall 2003–2005	
Total Monitored Events	10
Max. Daily Event Rainfall	3.8 cm
Min. Daily Event Rainfall	0.5 cm
Avg. Runoff from Pervious	18.75 m^3/ha
Avg. Runoff from Impervious	40.84 m^3/ha

Water Quality 2003–2005		
	TN	TP
Impervious		
Avg.	1.07 mg/l	0.04 mg/l
Max.	1.92 mg/l	0.10 mg/l
Min.	0.62 mg/l	0.01 mg/l
Pervious		
Avg.	1.51 mg/l	0.20 mg/l
Max.	1.86 mg/l	0.43 mg/l
Min.	1.01 mg/l	0.05 mg/l

TABLE 5. Summary of Deep and Shallow Lake Aeration Study rainfall and water quality monitoring results from 2004 (September–December) and 2006 (August–October) at The Brooks.

Rainfall	
Phase 1 (Static): Sept.–Dec. 2004	
Total Events	28
Total Rainfall	10.2 cm
Avg. Event Rainfall	0.4 cm
Max. Daily Event Rainfall	1.5 cm
Phase 2 (Dynamic): Aug.–Oct. 2006	
Total Rainfall	35.6 cm
Monitored Events	47
Avg. Monitored Event	0.8 cm
Max. Monitored Event	1.6 cm

Water Quality		
Phase 1 (Static): Sept.–Dec. 2004, 12 Sampling Events		
	TN	TP
Avg.	0.73 mg/l	0.08 mg/l
Max.	2.35 mg/l	0.78 mg/l
Min.	0.20 mg/l	0.02 mg/l
Phase 2 (Dynamic): Aug.–Oct. 2006, 16 Sampling Events		
	TN	TP
Avg.	0.91 mg/l	0.17 mg/l
Max.	1.71 mg/l	0.37 mg/l
Min.	0.65 mg/l	0.05 mg/l

for the October–December 2004 study period totaled 10.2 cm from 28 events and no runoff discharged. Total phosphorus for this period ranged from 0.02 mg/l to 0.78 mg/l and total nitrogen ranged from 0.20 mg/l to 2.35 mg/l. For the August–October 2006 study period there were 47 rainfall events totaling 35.6 cm and ranging from 0.02 cm to 1.6 cm with total phosphorus ranging from 0.05 mg/l to 0.37 mg/l and total nitrogen ranging from 0.65 mg/l to 1.71 mg/l. In addition to these laboratory measured parameters, in place YSI EDS6600 data sondes were suspended in the lakes during the test periods and collected readings at 15 minute intervals for pH, temperature, conductivity, turbidity, chlorophyll-a and dissolved oxygen (D.O.), providing a very discrete dataset to use in conjunction with the wet laboratory results. Results showed consistently higher and more stable levels of D.O. in the aerated lakes both during the 2004 and the 2006 phases.

The results of the study indicate that aeration helped stabilize fluctuations in water quality and limited stratification that can sometimes result in anoxic conditions in deeper ponds. In addition, multivariate analysis of the data showed clear trends of seasonal differences over-riding patterns driven by depth or aeration, and that throughout the study each pond appeared to retain its own characteristics that separated it from the other ponds. Additional study details and discussion are available from FGCU staff and co-authors, Ceilley and Everham.

Long Term Discharge Study—Results of the physical, water quality, rainfall and flow measurements to date are summarized in TABLE 6. An example water level and rainfall chart depicting periods of runoff and rainfall for 2010 at The Brooks is provided in FIG. 8.

Preliminary results from 2007–2012, found that rainfall events ranged from 0.20 cm to 19.4 cm at each site, compared to an average of approximately 1.15 cm for all events. Discharge runoff flow ranged from 14 m^3/hectare to

TABLE 6. Summary of Long Term Discharge Study rainfall and water quality monitoring results from 2007 at 6 study sites: Verandah (180 ha), Sandoval (161 ha), Shadow Wood Preserve (98 ha), The Brooks (475 ha), Mediterra, (311 ha) and Twin Eagles (186 ha).

Rainfall 2007–2012	Verandah	Sandoval	Shadow Wood	The Brooks	Mediterra	Twin Eagles
Total Monitored Events	657	611	641	720	651	765
Total Rainfall	788.4 cm	855.4 cm	705.1 cm	720.0 cm	781.2 cm	841.5 cm
Avg. Rainfall Event	1.2 cm	1.4 cm	1.1 cm	1.0 cm	1.2 cm	1.1 cm
Max. Daily Rainfall Event	10.8 cm	11.5 cm	12.7 cm	15.9 cm	11.3 cm	19.4 cm
Avg. Discharge	1123 m^3/ha	1288 m^3/ha	216 m^3/ha	549 m^3/ha	42 m^3/ha	14 m^3/ha

Water Quality 2007–2012
Surface Water: 164 Static and Discharge Sampling Events

	Verandah		Sandoval		Shadow Wood		The Brooks		Mediterra		Twin Eagles	
	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP
Avg. mg/l	1.13	0.09	1.61	0.32	1.26	0.05	0.94	0.05	0.86	0.04	1.34	0.05
Max. mg/l	1.79	0.20	2.27	0.68	1.79	0.19	1.27	0.16	1.29	0.06	1.66	0.17
Min. mg/l	0.75	0.03	1.01	0.04	1.03	0.01	0.54	0.02	0.55	0.02	1.12	0.01

Ground Water: 38 Sampling Events

	Verandah		Sandoval		Shadow Wood		The Brooks		Mediterra		Twin Eagles	
	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP	TN	TP
Avg. mg/l	2.99	0.12	n/a	n/a	4.35	0.13	n/a	n/a	n/a	n/a	n/a	n/a
Max. mg/l	3.61	0.41	n/a	n/a	26.60	0.50	n/a	n/a	n/a	n/a	n/a	n/a
Min. mg/l	0.82	0.15	n/a	n/a	0.75	0.02	n/a	n/a	n/a	n/a	n/a	n/a

1,288 m^3/hectare. Total phosphorus in surface waters ranged from 0.01 mg/l to 0.68 mg/l and total nitrogen ranged from 0.54 mg/l to 2.27 mg/l.

Among the six study sites, certain sites consistently discharge high percentages of total basin inputs every year, as illustrated by Sandoval, while for the same period other sites, such as Twin Eagles, do not discharge at all for successive years, despite receiving similar annual rainfall totals. Data show that the volume of runoff that is discharged is not directly related to the volume of rainfall. Rather, a variety of site specific factors will ultimately determine a BMP's actual discharge frequency and volumes. Water resource professionals have long suspected this to be the case but had limited data to support observations. Water quality findings to this point are, as expected, somewhat variable. The water quality of the vast majority of samples collected since 2007 is equal to or better than the State of Florida standards, verifying the proper performance of the permitted wet detention systems (Johnson Engineering, 2010).

Groundwater Interaction Study—Results of the rainfall and flow measurements are included with the general results of the LTDS and are not independently collected for the groundwater study. Preliminary results from

October 2010 to December 2012, found that 265 rainfall events (276 cm) occurred greater than 0.02 cm up to 10.8 cm at Verandah and 365 rainfall events (384 cm) occurred greater than 0.02 cm up to 9.4 cm at Shadow Wood Preserve. Total phosphorus in the groundwater samples in the 38 sample sets ranged from 0.02 mg/l to 0.50 mg/l and total nitrogen ranged from 0.75 mg/l to 26.6 mg/l. Water level and water quality data collection has been completed. commenced, and approximately 25 months of data has been collected to date. Aquifer performance testing to calculate hydraulic parameters of transmissivity, storage and leakance for both the Surficial and Sandstone aquifers have been completed. The study is ongoing and future analyses and modeling are pending.

DISCUSSION—*Green Roof Study*—This project demonstrated that green roofs can be incorporated into low impact development strategies in southwest Florida, and they can, with proper maintenance, irrigation, and plant selection, survive and thrive in a semi-tropical climate. Our ability to fully understand the correlation between roof design, media selection, and plant selection is limited by the size and relatively short duration of the project, but casual observation indicates the green roof profile design is secondary to the correct choice of plants. Based on our observations, green roof design in Florida performs best with a cistern to help reduce stormwater volume by recycling the runoff. Consideration of a media mixture that is lighter in color and holds less heat than the expanded shale mixture used should be explored. Several of these observations and recommendations have been incorporated into the recently completed green roof studies at the University of Central Florida (UCF). The knowledge gained from the Shadow Wood Preserve green roof and the subsequent UCF green roof is being used in the new Statewide Stormwater Treatment Rule currently in the rulemaking process with the FDEP. The collaborative team approach proved quite valuable at the end of the initial planting phase in 2006 when it became apparent that plant survivability was questionable with dry season irrigation. The team coordinated a variety of funding, design and installation of the cistern/irrigation system and facilitated the successful completion of the study.

Pervious/Impervious Pavement Study—Estimated percent runoff volumes from the impervious pavement area actually exceeded 100% of the calculated total. Additional runoff from the clubhouse entrance drive, which was adjacent to the impervious pavement area, may have contributed to the excess runoff. Field observations also underscored the importance of the subsurface material in allowing the runoff to be filtered and absorbed effectively. Immediately following one intense rain event, the flow across the pervious pavement surface dried up relatively quickly, but water continued to percolate out of the ground along the edge of the pervious pavement, directly in front of the stormwater drain. Overall, pervious pavement was shown to be an effective BMP in reducing stormwater pollutant loadings due to the substantial reduction in runoff volume and flow rate as compared to runoff from standard asphalt. The collaborative team approach in integrating the funding, construction of pervious concrete parking lots and implementation of the monitoring activities during normal clubhouse events throughout the study, turned out to be very successful.

Deep and Shallow Stormwater Pond Aeration—A major issue associated with the design of wet detention systems has been restrictions on how deep they can be built. It has been hypothesized that deeper ponds go anaerobic and thereby release pollutants stored in the bottom sediments back into the water column. Current regulations limit the depth of wet detention systems to prevent anaerobic conditions, or require aeration in ponds exceeding a certain depth. Our ability to fully understand the pond water quality dynamics was limited by a lack of quantification of the impacts of groundwater flow, aquatic flora and fauna, and management activities both within the ponds and adjacent watersheds. These factors should be carefully considered in future studies. The successful coordination of the local regulatory agencies, community residents, and golf club staff, as well as the monitoring guidance and detailed analysis provided by FGCU staff, worked very well. The combination of the land developer (BBG) coordinating site access and approvals, and the day to day monitoring activities provided by the private consultant (JEI) was quite effective in keeping the project on schedule and within budget.

Long Term Discharge Study—With one more year of data collection remaining, clear trends have emerged. The amount of stormwater runoff discharged from a typical wet detention system in southwest Florida is variable and may not be predictable without detailed knowledge of hydraulic characteristics of the Surficial aquifer and vertical distance to a stable water table, particularly in tidally influence areas. This study also demonstrates that with relatively simple instrumentation it is possible to continuously monitor stormwater discharges from communities with defined basins and discharge structures, and produce reliable discharge calculations.

In practice, wet detention systems may function like a retention system for a part of a year and only discharge for a few months during and immediately following the peak of the wet season, as shown by the discharge data collected. Stormwater runoff must ultimately exit all the sites in order to maintain a mass balance, but under certain conditions infiltration and groundwater transport may be significant factors in determining how much stormwater is discharged to surface waters. As a result, standard methodologies for assessing the effectiveness of wet detention systems in reducing pollutant loadings may not be applicable under all circumstances. The groundwater interaction study is designed to quantitatively assess this hypothesis.

This research project and its findings have important implications with respect to the State of Florida's water regulatory programs, especially its stormwater/environmental resource permitting rules, but also with respect to the implementation of Total Maximum Daily Loads and meeting load reductions established as part of the wasteload allocation (WLA) for regulated points or the load allocation (LA) for nonpoint sources. The treatment effectiveness of wet detention systems is related primarily to the treatment volume and the residence time of the system. However, ancillary factors affecting treatment efficiency include the amount of infiltration that may occur, the seasonal high water table fluctuation, and the relationship between the wet detention system's control elevation and the seasonally high water table. The results of this research dispute some common assumptions about the functioning and effectiveness of wet detention systems. Generally, it is assumed that wet detention systems discharge after most or all storm events, especially during the wet season when recovery of the treatment and flood control volume may not occur between storm events. As seen from the results of this project, this is not always true, especially when the soils and underlying geology at a site allow for substantial infiltration. These results suggest that much more careful consideration must be given to site specific conditions in assessing the discharge behaviors of wet detention systems, especially in evaluating soils, geology, and seasonal high water table conditions. This is especially true for systems that potential discharge to impaired waters where the level of treatment is "net improvement" whereby the post-development loading can't exceed the predevelopment loading from the project site. Instead of making wet detention systems larger, some areas may have soil, geology, and water table conditions that allow wet detention systems to get much higher levels of treatment because they do not discharge as often as assumed. The Long Term Discharge Study has shown significant variations in discharge behavior exist among six well maintained, typical communities in the southwest Florida region. These variations within a region can be dramatic, as evidenced in this study by a large community like Twin Eagles in Collier County with no discharge for an entire year and during the same year Sandoval, for example, in Lee County, has significant discharges and similar annual rainfall. It is not known with certainty, how these variations might apply to other parts of Florida, and additional work is needed to better understand the stormwater discharge behaviors in a wider area.

As with the previous studies in this paper, the collaborative team approach has solved problems and overcome obstacles throughout the life of the project that could not be easily addressed by any of the individual team members. When the water quality or groundwater components were added to the program, for example, FDEP would streamline the funding acquisition. The private developer (BBG) would intervene where needed to provide site access, the private consultant would develop and implement the monitoring plan with the guidance of FGCU staff, and react to tropical storm events, for example.

Groundwater Interaction Study—Based on trends observed during previous years of monitoring under the LTDS, Verandah is expected to discharge for several months each year, with groundwater flowing from northeast to southwest across the study basin from the topographic high near SR-80 towards the Orange and Caloosahatchee Rivers. Shadow Wood Preserve did not discharge water through any control structure in 2007 and discharged approximately 200,000 cubic meters during both 2008 and 2009, despite 46.7 more centimeters of rainfall in 2008 than 2009. Groundwater may flow radially from the center of the project site towards Mullock Creek and tidally influenced lands to the west, following the existing topography. The site also may receive groundwater inflow from the east near US-41, where land surface is significantly higher than on-site.

Upon completion of data collection, water level data and aquifer parameters determined from on-site testing will be used to quantify the volume of groundwater flowing through the study basins and the influence of groundwater hydraulics on stormwater management system discharge.

Collaborative Team Approach—As with most studies, additional questions and research needs became apparent as each of the projects summarized above progressed. Certain observations either supported prior assumptions about the systems or highlighted the need for auxiliary research to help explain the behaviors of the BMPs evaluated. The flexibility of the collaborative approach allowed additional components to be added to existing studies on relatively short notice, such as the addition of the water quality and groundwater interaction studies to the LTDS.

The BMP studies conducted by the team to-date continue to generate observations and questions that may guide the development of additional studies investigating specific phenomena. For example, the Green Roof Study showed that this type of low impact development technique could be successful in the sub-tropical climates given the proper design and maintenance. However, additional studies are needed to assess thermal profiles for various host structures and growth media, irrigation system designs, and plant selection, as well as water quality of the roof run-off. Similarly, Pervious/Impervious Pavement Study noted that the technology for manufacturing and placing the pervious pavements needed improvements to overcome raveling, which was objectionable to the residents in the community. Observations from this study are one of the reasons FDEP staff have been working with the Florida Concretes Product Association to implement a training and certification program for pervious concrete contractors.

The Long Term Discharge Study, while still ongoing, is providing important information on discharge behavior for typical wet detention systems in residential communities permitted under current SFWMD stormwater rules. The variation in the annual stormwater volume and the associated pollutant loading discharged from the study sites is a particularly interesting observation. The monitoring results indicate that wet detention systems may act as retention systems under certain rainfall and runoff conditions, which may occur for a substantial part of the year. This indicates that the wet detention systems are more effective in reducing stormwater pollutant loads than previously thought.

However, these are site specific conclusions based on a long term data set and additional monitoring is needed to better understand the relationships between wet pond design, especially setting the control elevation, and surficial aquifer characteristics. Furthermore, when estimating stormwater loadings discharged to downstream receiving waters, one must consider the variations that we have seen based on site-specific groundwater conditions and geology. Additional information on how groundwater interacts with the surface water in the study communities will be generated over the next year with data from that program.

Data generated by these studies can be valuable in advancing the body of knowledge related to the effectiveness of stormwater BMPs in southwest Florida. Enhanced local information on the function, applicability and efficiency of a variety of BMPs will aid in the creation of treatment trains that can best protect downstream receiving waters and better meet the needs of natural systems. The FDEP wanted data on the effectiveness of Low Impact Design and wet detention BMPs, as part of its efforts to improve nutrient removal as it developed the Statewide Stormwater Treatment Rule (Chapter 62-347). The involvement of the FDEP in developing and implementing these studies can enable an accelerated integration of real-world data into the rulemaking process, as occurred with incorporation of revised roadway run-off values used in the Harper calculations for FDOT projects. Upon completion of the ongoing studies, specifically the LTDS and associated groundwater interaction study, the data and results will be made publicly available through the FDEP and be available for fine tuning wet detention design criteria to optimize their nutrient removal. In conclusion, the results of these stormwater monitoring projects demonstrate that the collaborative team approach is an effective and efficient approach to evaluating and improving BMPs in southwest Florida.

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Biological Sciences

A WATERSHED ANALYSIS OF PERMITTED COASTAL WETLAND IMPACTS AND MITIGATION ASSESSMENT METHODS WITHIN THE CHARLOTTE HARBOR NATIONAL ESTUARY PROGRAM

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Abstract: *From 2004 to 2008, 1,834 Environmental Resource Permits were issued for development projects in coastal wetlands within the Charlotte Harbor National Estuary Program. We evaluated 118 sites utilizing three wetland functional assessment methods (WFAMs): Hydrogeomorphic Method (HGM), Wetland Rapid Assessment Procedure (WRAP), and Uniform Mitigation Assessment Method (UMAM). All functioned as designed and produced similar assessments of wetlands but yielded different mitigation results. The study results showed that HGM was most effective in identifying and quantifying wetland functions of coastal wetlands. UMAM and WRAP are useful but delivered mitigation ratios less than one in both function and area, resulting in net wetland losses. The wetland area lost over the study period was small relative to total wetland area, but the permitted wetland elimination is gradually reducing the extent of wetlands. The process relocates wetland functions out of impacted watersheds toward off-site mitigation areas. The WFAM site evaluations indicated an equal or greater balance of ecosystem functions within the total service area when utilizing off-site mitigation. However, there is loss of wetland area and function in the donor watershed and an increase in function, but not area, of wetlands in the receiving watershed.*

Key Words: Charlotte Harbor, coastal wetlands, mangroves, mitigation, salt marsh, wetland functional assessment, HGM, UMAM, WRAP

This study: 1) identifies regional effects of current wetland permitting processes and programs of compensatory wetland mitigation; 2) evaluates the success of state and local mitigation strategies implemented in the Charlotte Harbor National Estuary Program (CHNEP) study area, focusing on coastal marine and estuarine habitats; and 3) provides suggestions for improving mitigation for wetland loss. Management criteria and implementation success are assessed for both public and private mitigation lands.

Restoration and maintenance of high environmental quality should sustain the coastal economic base for tourism, fishing, recreation and the quality of life for area residents (CHNEP CCMP 2010). It is essential that the wetland regulatory process maintain and protect these resources. The pace of changes in the coastal wetlands of the CHNEP is not distinguishable with large scale mapping tools and requires close examination of the wetland regulatory and mitigation processes to be observed and measured.

TABLE 1. Comparison of the variables evaluated for the three wetland functional assessment methods.

HGM 14 Variables	UMAM 3 Variables	WRAP 6 Variables
Degree of Marsh Dissection	Location & Landscape Support	Wildlife Utilization
Proportion of Tidally Connected Edge to Total Edge	Water Environment	Wetland Canopy
Total Effective Patch Size	Community Structure (vegetation and/or benthic)	Wetland Ground Cover
Hydrologic Regime		Habitat Support/Buffer
% Cover by Typical Plant Species		Field Hydrology
Nekton Habitat Complexity (# different habitat types)		Water Quality Input & Treatment
Wildlife Habitat Complexity (# different habitat types)		
Surface roughness (Manning's; 3 sub-components)		
Mean Total % Vegetative Cover		
Mean Vegetative Structure Index		
Mean Width of Marsh		
Relative Exposure Index (fetch)		
Distance to Navigation Channel or 2 m Depth		
Soil texture		

Wetland Functional Assessment Methods (WFAM) are procedures that are designed to evaluate current wetland functions and predict potential changes to a wetland's functions that may result from proposed activities. In this study we evaluated the apparent success of assigned mitigation actions by the use of the current prevalent wetland mitigation functional assessment methods (WFAMs). These are the federal Hydrogeomorphic Methodology (HGM) (Shafer et al., 2002), the State of Florida's Uniform Mitigation Assessment Method (UMAM) (Florida Department of Environmental Protection, 2004), and the South Florida Water Management District's Wetland Rapid Assessment Procedure (WRAP) (Miller and Gunsalus, 1997). WRAP was also employed by the US Army Corps of Engineers (USACOE) for a time period ranging from January 1, 2002 to December 31, 2008. A summary of the variables collected in each WFAMs provided in TABLE 1.

Prior to beginning the study, Environmental Resource Permits (ERPs) within the CHNEP were inventoried. ERPs available as electronic files from the Florida Department of Environmental Protection (FDEP), the South Florida Water Management District (SFWMD), and the Southwest Florida Water Management District (SWFWMD) for the study period of January 1, 2004 to December 31, 2008 were reviewed. The review revealed that 10,186 ERP permitting actions occurred within the study area. The watersheds with the most permitting activity were the Caloosahatchee River in the SFWMD (28% of the total) and the Peace River in the SWFWMD (25%). Permitting activities occurring in costal wetland habitats or at the coastline account for

18%. of the permits. The coastal watersheds with the most active permitting activities were the Caloosahatchee River (27% of coastal actions) and the Pine Island Sound/ Matlacha Pass watershed (23%). The least active coastal permitting occurred in the Myakka River (4%) and Dona and Roberts Bays (5%) watersheds.

Materials and Methods—*Site selection*—The Wetland Functional Assessment Method (WFAM) assessment process began with the selection of an Environmental Resource Permit (ERP) based on the proximity of the project to tidally influenced wetlands, and on whether or not wetland impacts were anticipated. Aerial photos for each study site, dating as far back as 1998, were downloaded from county websites and saved as part of the site study file. Physical access to the site was then determined and sites were accessed via road, trail or charter boat. In three cases the property owner could not be determined or contacted so the sites were assessed from the nearest public access (road, right-of-way, or waterway). A total of 118 sites were selected for the WFAM assessment, as shown in Fig. 1. For this study, a site identification number was given to each site based on the date of assessment and the number of sites assessed that day. The distribution of projects assessed was determined to some degree by the facility of finding pertinent permit information.

Equipment—Equipment used in the field included: a Trimble GPS unit with ArcGIS ArcPad software, digital camera, functional assessment field data sheets, functional assessment "reference sheets", field guides, binoculars, aerial photos, and YSI water quality sensor. The water quality sensor was incorporated into the project in 2008, as it became available from another study.

Study Personnel—WFAM field assessment, office evaluations and data analysis were conducted by the authors, James Beever as Principal Planner and the Whitney Gray as Environmental Scientist.

WFAM scoring—Each site was determined to be pre-construction or post-construction of the permitted project. Data sheets for each assessment method were filled out by study scientists and the same information was entered into the GPS unit. Flora and fauna observed at the site were recorded. Photographs were taken at each, with emphasis on the project area, wetland vegetation, alterations of vegetation, and wildlife observed. Surrounding conditions were recorded for spatial context.

If the site being assessed was in the pre-construction state, data sheets were also completed for the post-construction state as predicted by the conditions of the permit, as defined in the protocols for each WFAM. If the site was assessed post-construction, data sheets were also completed for a pre-construction condition based on historical aerial photos and staff reports from the permit file. Local knowledge was sometimes helpful in this process as well.

The total time at each site doing all three functional assessment methods as a team averaged approximately one hour.

Water quality—Measurements of water temperature, dissolved oxygen, pH, and salinity were taken at each site starting in 2008 when the water quality meter was acquired.

Data Management and Analysis—Following the field visit, all collected data, photographs and measurements were downloaded and stored in appropriate Access, Excel and Geo- databases. The data sheets were completed in the office because some information in the functional assessments was more efficiently determined from the desktop. Scores from the functional assessments were then entered into databases that summarized wetland impacts and mitigation. Finally, a narrative was written for each site summarizing the conditions at the time of assessment; the nature of the project being permitted; the wildlife, wetland canopy, and wetland groundcover observed at the

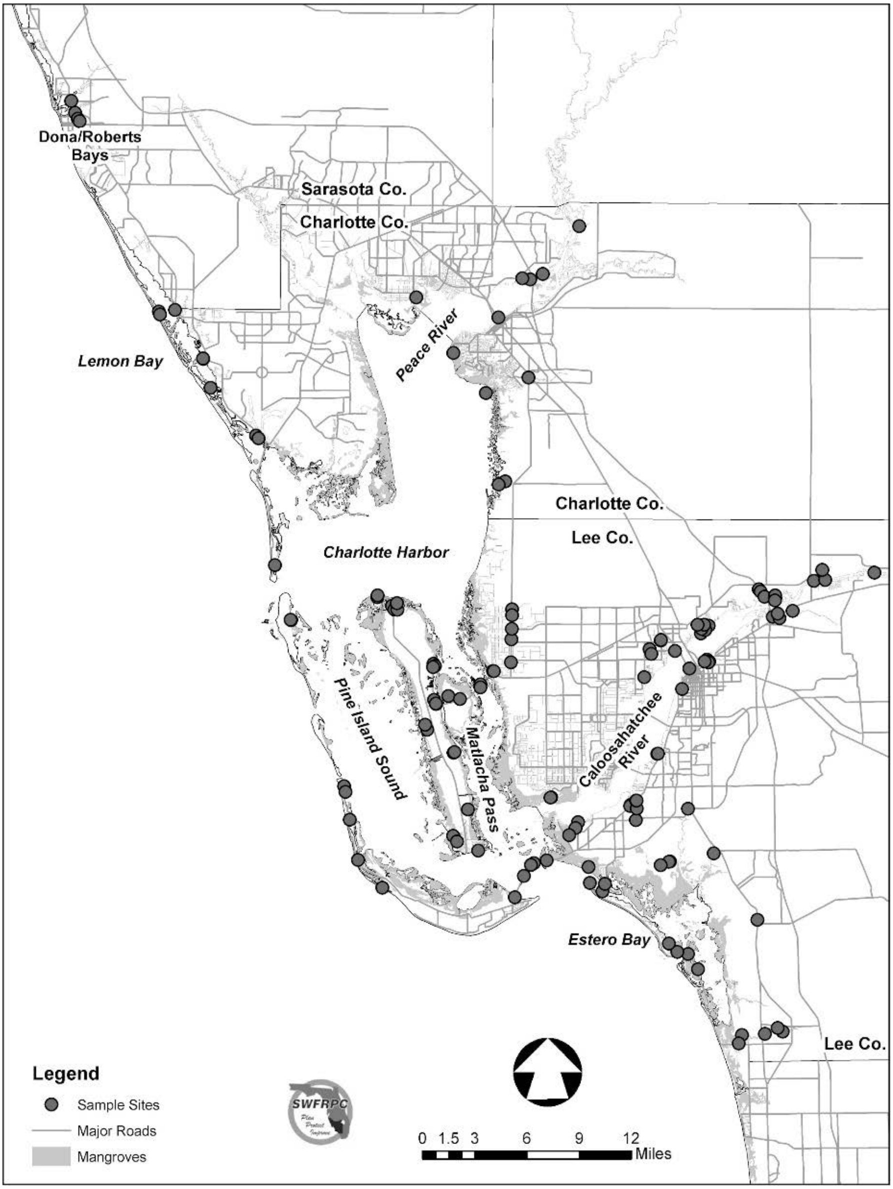

FIG. 1. Location of Project Sample Sites.

site; the habitat support around the site; and the hydrology of the site. The conditions predicted for either post- or pre-construction state, were recorded.

RESULTS—*Sites evaluated*—The locations of the site visits are displayed on FIG. 1. The number and size of study sites are summarized by watershed, county and construction status in TABLE 2. The sites were distributed in 9 of the

TABLE 2. Summary of the number and size of WFAM assessment sites by watershed and county within the study area.

	No. of Projects				Wetland Size			
	Pre-Project	Post-Project	Total	%	Total (ha)	Min. (ha)	Max (ha)	Ave. (ha)
Watershed								
Dona/Roberts Bays	0	4	4	3%	1.0	0.073	0.45	0.25
Lemon Bay	0	8	8	7%	0.5	0.004	0.26	0.06
Charlotte Harbor/ Gasparilla Sound	5	5	10	8%	2.3	0.004	1.86	0.23
Peace River	4	1	5	4%	29.7	0.069	18.62	5.93
Pine Island Sound	8	3	11	9%	36.9	0.012	29.98	3.35
Matlacha Pass	16	1	17	14%	44.3	0.004	21.45	2.61
Caloosahatchee R.	33	5	38	32%	64.3	0.000	22.14	1.69
San Carlos Bay	3	4	7	6%	1.9	0.012	0.96	0.28
Estero Bay	14	4	18	15%	18.2	0.004	6.94	1.01
Total	83	35	118	100%	199.1	<0.000	29.98	1.69
County								
Sarasota	0	4	4	3%	1.0	0.073	0.45	0.25
Charlotte	6	13	19	16%	32.3	0.004	18.62	1.70
Lee	77	18	95	81%	165.8	<0.000	29.98	1.74
Total	83	35	118	100%	199.1	<0.000	29.98	1.69

CHNEP watersheds, with 33 of the 118 site located in the Caloosahatchee River watershed and 15% were in Estero Bay (TABLE 4). Of 118 sites examined, 81% were in Lee County, 16% were in Charlotte County, and 3% were in Sarasota County. Seventy percent of projects were assessed in the pre-construction condition, and 30% in the post-construction condition. A total of 199 hectares of wetlands were evaluated at the study sites, ranging in size from less than 0.004 hectares to 30 hectares.

It is important to note that if a project site contained two or more distinct wetland types, each was assessed separately. Also, some projects contained on-site mitigation which was assessed separately.

Project types—The types of projects assessed are provided in TABLE 4. The largest category of projects was residential, comprising almost 17% of the assessment locations. However, when all dock, dock-related and marina projects are added to residential projects, this total increased to 25% of all projects. Roadway projects made up almost 16% and shoreline hardening projects and recreation projects each made up 8% of all projects. Preservation projects comprise the same proportion of projects (4%) as riprap additions.

Habitat types—Habitat types at each location of a functional assessment were assigned according to Florida Land Use Cover Classification System (FLUCCS) number (Florida Department of Transportation, 1999), as shown in TABLE 3. Mangroves (FLUCCS 612) made up the largest category of habitat

TABLE 3. Summary of the number and size of WFAM assessment sites by wetland type and Florida Land Use Cover Codes (FLUCC) within the study area.

		No. of Projects			Wetland Size			
Primary Habitat	FLUCC	Subtotal	Total	%	Total (ha)	Min. (ha)	Max. (ha)	Ave. (ha)
Exotic Wetland Hardwoods	6100		1	1%	2.1	2.076	2.08	2.08
Littoral Shelf	6510		1	1%	0.1	0.134	0.13	0.13
Mangroves			75	64%	91.9	0.004	29.98	1.22
Mangrove Forest	6120	3						
Mangrove Fringe	6120	55						
Mangrove Swamp	6120	13						
Mangroves	6120	2						
Mangrove Overwash Island	6120	1						
Basin Mangrove Forest	6120	1						
Saltmarsh, Tidal Flats and Saltern								
Marsh	6420	3	24	20%	99.2	<0.000	22.14	4.13
Salt Marsh	6420	12						
Tidal Marsh	6420	2						
Low Marsh	6421	1						
High Marsh	6434	4						
Tidal Flat	6510	1						
Saltern	7203	1						
Shoreline	7100		2	2%	0.1	0.012	0.12	0.07
Submerged	6450		15	13%	5.7	0.004	1.86	0.38
Total			118	100%	199	<0.000	29.98	1.69

assessed at 63%. Within that category, mangrove fringe made up 46%. Other types of mangrove habitat encountered included basin mangrove forest, mangrove overwash island, mangrove swamp and mangrove forest. The salt marsh category included saltern, high marsh, low marsh and tidal flats, and comprised 20% of all habitats assessed.

Area of wetlands reviewed—Of the 118 sites visited, a total of 199 hectares of coastal wetlands were subject to review for potential impacts. The largest area of coastal wetlands on a project site was 30 hectares. A total of 21.5 hectares of on-site coastal wetland loss was permitted in the 5 year timeframe. This is a 11% of loss between the pre-construction condition and post-project condition.

The largest on-site loss for a single project site was 4.5 hectares. On average a permit included 0.2 hectares, or roughly one half acre, of coastal wetland loss. This fits a general pattern of many small impacts of less than 0.2 hectares each summing to a larger total area of 21.5 hectares.

Plant and animal species observed—One hundred ninety four plant species were observed in the 118 project sites reviewed. This includes 73% native plants, 22% non-native plants, and 5% of debated origin.

TABLE 4. Summary of the number and size of WFAM assessment sites by project type within the study area.

Project Type	No. of Projects		Wetland Size			
	Total	Percent	Total (ha)	Min. (ha)	Max. (ha)	Ave. (ha)
Boardwalk	2	2%	2.8	0.061	2.70	1.38
Bridge	1	1%	0.1	0.093	0.09	0.09
Commercial	4	3%	3.6	0.024	0.00	0.90
Dock	16	14%	1.4	0.004	0.45	0.09
Dock/seawall	3	3%	0.3	0.036	0.12	0.09
Filling	1	1%	0.1	0.134	0.13	0.13
Flood Control	3	3%	0.1	0.004	0.03	0.02
Mangrove Alteration	1	1%	0.1	0.065	0.06	0.06
Marina Impact	11	9%	7.5	0.008	2.65	0.68
Mitigation	5	4%	27.0	0.077	21.45	5.40
Navigation	12	10%	30.6	0.004	18.62	2.55
Parking Lot	1	1%	0.3	0.336	0.34	0.34
Preservation	4	3%	45.2	0.911	22.14	11.31
Recreational	9	8%	0.9	0.004	0.27	0.10
Residential	20	17%	69.9	0.008	29.98	3.50
Riprap	4	3%	0.1	0.008	0.02	0.01
Roadway	18	15%	9.0	<0.000	5.50	0.50
Seawall	2	2%	0.1	0.004	0.08	0.04
Stormwater Treatment	1	1%	0.2	0.202	0.20	0.20
Total	118	100%	199	<0.000	29.98	1.69

A wide variety of wildlife (184 species) was observed, directly or indirectly, during project site visits, including 10 mammal species, 55 bird species, 8 reptile species, 3 amphibian species, 36 fish species, 44 terrestrial invertebrate species, 28 marine/aquatic invertebrate species. The invertebrates include 15 butterfly, 2 moth, 7 dragonfly, 4 damselfly, 7 other insect, nine crab, 13 crustacean species, and 17 mollusks species (TABLE 5).

WFAM scores—The variables assessed in the three WFAM are listed on TABLE 1. All calculated WFAM scores are tabulated on TABLES 6 through 8. TABLE 6 shows the scores for pre-construction conditions, followed by TABLE 7

TABLE 5. Animal species totals observed at study sites.

Species Group	No. of Species	No. of Occurrences
Total Terrestrial Invertebrates	44	110
Total Marine/Aquatic Invertebrates	28	87
Total Fish	36	170
Total Amphibians	3	3
Total Reptiles	8	29
Total Birds	55	221
Total Mammals	10	38
Total	184	658

TABLE 6. Summary of WFAM pre-project scores by wetland type within the study area.

Habitat Category	No. of Projects	Wetlands (ha)	Mean Pre-Project Scores		
			HGM	UMAM	WRAP
Exotic Wetland Hardwood	1	2.1	0.54	0.43	0.63
Std Dev Exotic Wetland			n/a	n/a	n/a
Min. Exotic Wetland			0.54	0.43	0.63
Max. Exotic Wetland			0.54	0.43	0.63
Littoral Shelf	1	0.13	0.31	0.27	0.42
Std Dev Littoral			n/a	n/a	n/a
Min. Littoral			0.31	0.27	0.42
Max. Littoral			0.31	0.27	0.42
Mangrove	75	91.9	0.81	0.72	0.72
Std Dev Mangrove			0.12	0.15	0.16
Min. Mangrove			0.23	0.15	0.09
Max. Mangrove			0.98	0.97	0.99
Saltmarsh	24	99	0.69	0.58	0.66
Std Dev. Saltmarsh			0.17	0.21	0.17
Min. Saltmarsh			0.38	0.17	0.36
Max. Saltmarsh			0.95	0.93	0.97
Shoreline	2	0.13	0.73	0.82	0.72
Std Dev. Shoreline			0.18	0.12	0.30
Min. Shoreline			0.60	0.73	0.50
Max. Shoreline			0.86	0.90	0.93
Submerged	15	5.7	0.42	0.56	0.57
Std. Dev Submerged			0.17	0.17	0.19
Min. Submerged			0.15	0.30	0.29
Max. Submerged			0.82	0.83	1.00
All Projects	118	199	0.73	0.67	0.69
St. Dev All Projects			0.19	0.18	0.17
Min. All Projects			0.15	0.15	0.09
Max. All Projects			0.98	0.97	1.00

with the post-construction conditions. The change between pre-construction and post-construction scores are provided in TABLE 8.

HGM, UMAM, and WRAP scores range from 0 to 1.0. Using HGM, the mean wetland functional assessment score for pre-construction site evaluation was 0.72, with a standard deviation of 0.19 and a range of 0.98 to 0.15. In most cases HGM scored the pre-construction wetlands as having a higher functional score than the other two methods. The mean UMAM pre-construction functional assessment score was 0.66 with a standard deviation of 0.18 and a range of 0.97 to 0.15. Generally, UMAM scored the pre-construction wetlands as having a lower functional score than the other two methods. Using WRAP, the mean pre-construction score was 0.69 with a standard deviation of 0.18 and a range of 1.0 to 0.09. During the course of this study, all visited site were determined to be jurisdictional wetlands by both the authors and permitting

TABLE 7. Summary of WFAM post-project scores by wetland type within the study area.

Habitat Category	No. of Projects	Wetlands (ha)	Mean Post-Project Scores		
			HGM	UMAM	WRAP
Exotic Wetland Hardwood	1	2.1	0.48	0.22	0.47
Std Dev Exotic Wetland			n/a	n/a	n/a
Min. Exotic Wetland			0.48	0.22	0.47
Max. Exotic Wetland			0.48	0.22	0.47
Littoral Shelf	1	0.13	0.30	0.22	0.39
Std Dev Littoral			n/a	n/a	n/a
Min. Littoral			0.30	0.22	0.39
Max. Littoral			0.30	0.22	0.39
Mangrove	75	91.9	0.72	0.60	0.63
Std Dev Mangrove			0.21	0.19	0.18
Min. Mangrove			0.00	0.00	0.00
Max. Mangrove			0.96	0.93	1.00
Saltmarsh	24	99	0.65	0.47	0.58
Std Dev. Saltmarsh			0.25	0.28	0.23
Min. Saltmarsh			0.00	0.00	0.00
Max. Saltmarsh			0.95	0.92	1.00
Shoreline	2	0.13	0.55	0.53	0.59
Std Dev. Shoreline			0.47	0.42	0.30
Min. Shoreline			0.22	0.23	0.37
Max. Shoreline			0.88	0.83	0.80
Submerged	15	5.7	0.41	0.50	0.54
Std. Dev Submerged			0.12	0.12	0.15
Min. Submerged			0.15	0.32	0.32
Max. Submerged			0.62	0.67	0.81
All Projects	118	199	0.65	0.55	0.60
St. Dev All Projects			0.23	0.21	0.19
Min. All Projects			0.00	0.00	0.00
Max. All Projects			0.96	0.93	1.00

agency staff. One reviewer of the study suggested that WRAP scores <0.25 generally indicate that an area is not a jurisdictional wetland (J. Beever, Personal Communication, 2012).

Utilizing SPSS, the pre-construction scores from all three methods were significantly correlated with each other using 0.01 level two-tailed tests for Pearson's Correlation Coefficient, Kendall' s tau-b, and Spearman's rho. HGM scores were significantly different from WRAP (sig. 0.003) and from UMAM (sig. <0.001) scores. In contrast, UMAM and WRAP pre-construction scores were not determined to be statistically significantly different (sig. 0.018) among all projects.

As shown on TABLE 7, the mean post-construction score using HGM was 0.65, with a standard deviation of 0.23 and a range of 0.96 to 0. Generally, HGM gave the post-construction mitigation and on-site enhanced and/or

TABLE 8. Summary of WFAM pre-project scores minus post-project scores by wetland type within the study area.

Habitat Category	No. of Projects	Wetlands (ha)	Mean Pre- Minus Post Project Scores		
			HGM	UMAM	WRAP
Exotic Wetland Hardwood	1	2.1	0.06	0.21	0.16
Std Dev Exotic Wetland			n/a	n/a	n/a
Min. Exotic Wetland			0.06	0.21	0.16
Max. Exotic Wetland			0.06	0.21	0.16
Littoral Shelf	1	0.13	0.01	0.05	0.03
Std Dev Littoral			n/a	n/a	n/a
Min. Littoral			0.01	0.05	0.03
Max. Littoral			0.01	0.05	0.03
Mangrove	75	91.9	0.09	0.12	0.10
Std Dev Mangrove			0.20	0.17	0.15
Min. Mangrove			−0.51	−0.25	−0.43
Max. Mangrove			0.72	0.79	0.63
Saltmarsh	24	99	0.05	0.11	0.08
Std Dev. Saltmarsh			0.19	0.22	0.19
Min. Saltmarsh			−0.24	−0.27	−0.19
Max. Saltmarsh			0.56	0.47	0.76
Shoreline	2	0.13	0.18	0.29	0.13
Std Dev. Shoreline			0.28	0.30	0.00
Min. Shoreline			−0.02	0.07	0.13
Max. Shoreline			0.38	0.50	0.13
Submerged	15	5.7	0.00	0.06	0.03
Std. Dev Submerged			0.15	0.11	0.07
Min. Submerged			−0.36	−0.13	−0.10
Max. Submerged			0.31	0.24	0.19
All Projects	118	199	0.07	0.11	0.09
St. Dev All Projects			0.20	0.18	0.15
Min. All Projects			−0.51	−0.27	−0.43
Max. All Projects			0.72	0.79	0.76

preserved wetlands a higher functional score than the other two methods. Using UMAM functional wetland analysis, the mean post-construction wetland functional assessment score for all projects was 0.55, with a standard deviation of 0.21, and with a range of 0.92 to 0. The mean post-construction score using WRAP was 0.60, with a standard deviation of 0.19 and a range of 1.0 to 0. All three methods were significantly correlated with each other using 0.01 level two-tailed tests for Pearson's Correlation Coefficient, Kendall' s tau-b, and Spearman's rho. HGM scores were significantly different from WRAP (sig. <0.001) and UMAM scores (sig. <0.001). The UMAM and WRAP post-construction scores were statistically significantly different (sig. <0.001).

The difference between pre- and post-construction WFAM scores reflect the difference in wetland functions that occur as the result of the activities

TABLE 9. Summary of WFAM scores for mitigation for those projects that proposed mitigation.

Mitigation Category	No. of Projects	Wetlands (ha)	HGM	UMAM	WRAP
On-Site Mitigation	24	135.0	0.81	0.75	0.79
Std Dev			0.2	0.36	0.4
Min			0.31	1.0	1.0
Max.			1.00	0.21	0.16
Off-Site Mitigation	17	12.85	0.93	0.92	0.92
Std Dev			0.11	0.13	0.13
Min.			0.71	0.68	0.66
Max.			1.0	1.0	1.0
All Projects	30*	147.85			

* Eleven projects had both on-site and off-site mitigation.

(filling, dredging, wetland removal) of the project. The mean difference between pre- and post-construction UMAM functional assessment scores for all projects was 0.12 with a standard deviation of 0.18, with a range of 0.79 to −0.27. The mean difference between pre- and post-construction WRAP functional assessment scores for all projects was 0.09 with a standard deviation of 0.15, and a range of 0.76 to −0.43. The mean difference between pre- and post-construction HGM functional assessment scores for all projects was 0.07 with a standard deviation of 0.20, and a range of 0.67 to 0. Seventy percent of the reviewed sites were in pre-construction condition and the future condition was projected per the WFAM method utilized. Thirty percent of the reviewed projects had been constructed and the pre-construction condition was obtained from the permit information provided by the agency or in the case of incomplete permitting files obtained from mapping and comparison to immediately adjacent wetlands.

HGM was not significantly different than WRAP (sig. 0.12), but was significantly different than UMAM (sig. 0.001). UMAM and WRAP methods were statistically significantly different in their results (sig. 0.006) for all projects.

Mitigation—Of the 118 projects, a total of 30 proposed some form of mitigation, as shown TABLE 9. This included 13 projects with on-site mitigation, six projects with off-site mitigation, and 11 projects with both on-site and off-site mitigation. The total area of all on-site mitigation was 135 hectares. Off-site mitigation totaled 12.85 hectares, principally at the Little Pine Island Wetland Mitigation Bank.

The Little Pine Island Wetland Mitigation Bank has a service area that includes most of the CHNEP coastal area. Two sites utilized as mitigation credits for permits on Little Pine Island Wetland Mitigation Bank were assessed for the study. Mean assessment scores there were: HGM 0.90 ± 0.04; UMAM 0.89 ± 0.02; and WRAP 0.93 ± 0.04.

To calculate the functional units of mitigation that need to be balanced in the permitting process, the total wetland area assessed was multiplied by its functional assessment score. We expected that the post-construction functional units would be equal to or greater than the number of functional units lost (i.e. no net loss). There are two ways that functional units could be lost: if no mitigation was implemented or required; or if the functional units lost by completion of the project were greater than the functional units generated by mitigation.

A mitigation ratio is the comparison of the difference in wetland functional capacity or wetland area provided by the wetland prior to the project and the wetland functional capacity or wetland area remaining after the project including the mitigation provided as part of the project. The mitigation ratio for all UMAM scores was 1.5 with a standard deviation of 3.26. This was skewed by four projects with high or very high mitigation ratios generated by large on-site wetland preserves on Pine Island. The mitigation ratio for all HGM scores was 1.61 with a standard deviation of 3.45. The mitigation ratio for all WRAP scores was 1.61 with a standard deviation of 3.5. If the four unusual projects with high or very high mitigation ratios generated by large on-site wetland preserves on Pine Island are removed from the analysis, then the mitigation ratios for each method would be: for HGM equal to 1.1, with a standard deviation of 0.88; for UMAM equal to 1.02, with a standard deviation of 0.91; and for WRAP equal to 1.08, with a standard deviation of 0.93. Utilizing t-tests, the mitigation ratios generated for all projects were not statistically significantly different between HGM, UMAM, and WRAP.

Of the twelve projects that utilized an off-site mitigation area, often a mitigation bank, 54% were located in a different watershed (as defined by the Water Management District), than the bank and 46% were within the same watershed as the bank.

Eighty-three percent of off-site mitigation was located at Little Pine Island Mitigation Bank in the Matlacha Pass watershed, 8% at the Island Park Mitigation Bank in the Estero Bay watershed, and 9% in the Dinkins Bayous area in Pine Island Sound.

DISCUSSION—The study results indicate that all three wetland functional assessment methods function as designed, and produce results that are similar in their assessment of coastal wetlands, but yield somewhat different mitigation results (TABLE 6).

The actual measured rate of permitted/authorized wetland loss in this study from the 118 projects reviewed by on-site visits was 4.3 hectares. This is 0.01 percent of the 32,028 hectares of coastal wetlands in the CHNEP (Beever et al., 2011). If the average rate of real wetland area loss of 0.19 hectares per project is applied to the total 1,834 coastal ERP Permit Actions over the five year study period, this would hypothetically project a wetland loss rate of approximately 68 hectares per year. However, the wetland functional assessment balance would indicate no loss of wetland functions, since enhancements and preservations were occurring in other already extant wetlands at on-site and off-site mitigation areas.

Based on the study results, the projected hypothetical loss rate is 0.2% of the total current wetlands habitat in the CHNEP (Beever et al., 2011). This is below the margin of error in aerial photography mapping of these habitat resources (Kautz et al., 2007; SFWMD, 2006; SWFWMD, 2011). Because this loss is primarily occurring in urban landscapes, such as the Caloosahatchee River, Peace River, and Captiva Island, the relatively small wetland loss may already be overlooked in the land use/land cover mapping method utilized to map these resources.

While the loss of wetland area and functional appears relatively small over the five-year period examined compared to the total extent of wetlands that continue to exist, the permitted wetland elimination is gradually reducing the total extent of coastal wetlands in watershed of the CHNEP. This is in contrast to the perception of the public and regulatory entities that no wetland functional loss occurring as a result of the balancing process used in the functional assessment tools. Additionally, wetland functions are being relocated out of impacted watersheds and into the watershed that is able to provide the approved off-site mitigation in the category of coastal wetland habitats that are being impacted. While the functional assessment evaluation shows a mathematical balance for the total service area that is equal or greater for a project that utilizes a mitigation bank, there is a measured loss of wetland area and function in the donor watershed, with the potential for increase in function but not area of wetlands created in the receiving watershed. HGM was effective in quantifying functions of coastal wetland ecosystems (mangroves, salt marsh, intertidal and subtidal) within the CHNEP study area as a result of its measured objective variables and calibration to the wetland type being reviewed. UMAM and WRAP provided to be of utility but generally delivered a mitigation ratio in both functions and area that is less than one, which allowed for systematic net loss of both wetland function and area.

The use of any functional assessment method with off-site mitigation, including mitigation banks, can result in a balance of wetland functions being retained if the actual performance of the mitigation and the time lag to achieve the final mitigation state are accounted for. However, it can also result in a net loss of wetlands area and/or a net loss of wetland function while appearing on the ledger to have been an equivalent trade of mitigation for loss of function from the permitted impacts. This can occur in six different but potentially co-occurring ways including:

1. Relocation of the wetland functions out of the watershed.
2. The loss of area and functions to conservation easement mitigation credits that do not increase function or area of wetlands.
3. The presumption that the final wetland functional assessment score for the mitigation bank will be 1.0.
4. Creation of an inverse mitigation ratio. Wetlands to be impacted are assessed as having a low functional score, while the promised mitigation

wetland is granted a 1.0 perfect score. As a result, for example, three hectares of impacted wetland may be offset by one hectare of mitigation wetland.

5. Insufficiency of mitigation credit purchase tracking.
6. Existence of unidentified wetlands that sustain impacts that are never mitigated.

Conclusions—Based on the study results, the authors provide the following ten suggestions for improving mitigation for coastal wetland impacts in the CHNEP study area. The suggestions are not presented in order of importance, as utilization of any of the suggestions will improve the wetland mitigation process.

1. A handheld GPS device with GIS capability and digital functional assessment worksheets should be used when conducting functional assessments of wetlands to improve the speed and accuracy of data collection and allow field data to be digitally linked to site latitude/longitude. Additionally, use of a spreadsheet designed to compile and calculate functional assessment scores in the office for pre-site visit review and post-site visit processing can reduce time needed to complete functional assessments, improve accuracy and generate permanent documentation of the results. Utilizing the GPS, GIS and spreadsheet methods allows the HGM to be conducted within the time range of a rapid assessment method.
2. HGM is the most appropriate method for conducting wetland functional assessments for regulatory purposes because of the thorough scientific review involved in developing the method. HGM has been documented to be the most objective, complete, replicable, and accurate of the three assessed wetland functional assessment methods (Brinson, 1993; Brinson et al., 1997). However, it is the authors' opinion that in Florida and the CHNEP study area, it is unlikely that HGM will replace UMAM in the near future, principally due to non-scientific, legal and legislative reasons.
3. UMAM should include a real mitigation success weighting factor based on empirically measured success rates of the types of mitigation used as offsets. A major limitation of functional assessment imbalance is the assumption that completed mitigation will ultimately perform as well as natural un-impacted wetlands of the same type. It is very difficult to create wetland mitigation that will achieve the same level of functions as an area that has never been disturbed (Kusler and Kentula, 1990). WFAM scores utilized in calculating future mitigated wetland conditions need to be based on empirical evaluation of actual completed wetland mitigation areas. Calculating more accurate scores is now possible because many mitigation wetlands exist for both on-site and off-site efforts, including mitigation banks (Reiss et al., 2007). These completed mitigation sites could be used to determined a measure by which mitigation is balanced in

the permit review process. For example: if a proposed mitigation area is performing at a 0.9 UMAM functional assessment level, in order to offset the impacts to 10 hectares of a pre-construction wetland having a UMAM score of 0.62, a total of 6.9, not 6.2, functional assessment units would be needed to offset the functional losses. This is needed because the mitigation wetlands would only be performing 90% of the functional capacity of a natural wetland of the same type.

4. Conservation easements and preservations, alone, should not receive wetland functional mitigation credits. This study and others (Brown and Lant, 1999; Brumbaugh and Reppert, 1994; Dahl, 2005; Environmental Law Institute, 2004; Reiss et al., 2007) have shown that the practice of granting mitigation credit solely for filing a conservation easement encourages net wetland loss in both function and area. Conservation easement mitigation credits do not increase wetland functions, area, or offset the permanent loss. Conservation of mitigation, designed to offset wetland impacts in perpetuity, is not a separate component of physical mitigation activities, whether the goal is enhancement, restoration or creation (Federal Register, 1995). The value of permanent conservation should be incorporated into the mitigation activity and not treated an accomplishment that provides real physical functions (Brown and Lant, 1999; Dennison and Schmid, 1996; Eggers, 1992; Kusler and Kentula, 1990; Reiss et al., 2007). This might increase the costs of mitigation, but would minimize net loss of wetland functions associated with both on-site and off-site mitigations that rely solely on preservation as the mitigation method.
5. All mitigation options within the local watershed should be required to be examined before seeking options outside the watershed, to reduce in-watershed loss of wetland area and functions (USACOE and USEPA 2008, Federal Register (1995). Keeping the critical hydrologic, water quality, biological, social, and environmental justice issues associated with mitigation within the same watershed as the impacts it is essential for maintaining wetland functions. Sequencing "in-watershed first" also enhances avoidance and minimization of impacts during project design, because less abundant sources of wetlands for credits would be available within a given watershed to offset large scale wetland impacts.
6. A full tracking system of mitigation credits should be developed and implemented, to ensure that permitted mitigation is actually performed for both on-site and off-site mitigation plans. The importance of mitigation tracking is emphasized by the relatively low percent of mitigation plans implemented for current and long-term projects, particularly when projects are approved by one agency but reviewed and regulated by another agency. The results of this study indicate that, currently, there is not adequate linkage between mitigation needs, calculated credits or mitigation bank credits purchased to satisfy that need. The need for mitigation tracking has also been identified by USACOE, US Fish and

Wildlife Service (USFWS), university and legal experts in previous studies (Brown and Lant, 1999; Brumbaugh and Reppert; 1994; Dahl, 2005; Environmental Law Institute, 2004; Reiss et al., 2007).

7. All three functional assessment methods, HGM, UMAM and WRAP, should be adjusted to increase, not decrease, the total number of hectares of mitigation required, if the goal is to assure no net loss of wetlands and restoration of past wetland losses. The results of this study indicated that a balance sheet of pre- and post- project functional assessment scores does not equate to a balance of wetland mitigation or achieve no net loss of wetland area or functions. The ratios used in wetland mitigation permitting prior to the development of functional assessment tools were not arbitrary. They were based on time lag, probability of performance, distance from impacted wetlands, and the recognition that wetlands play a vital role in a landscape context. In the past, the greater-than-one-to-one ratios achieved a no-net-loss of wetlands for the projects reviewed. Significant areas of wetlands were protected, restored, and put into management using ratio methods. In contrast, the current inverse ratios generated by the functional assessment methods result in net losses of wetland area and function. This is particularly apparent when exotic plant cover effects in the pre-construction wetlands are over-stated in "best professional judgment" weighted assessments.
8. Photographic evidence of the absence of wetlands should be required for all permit applications involving shoreline alterations. The photographs should include aerial, ground-level and from the water view. This study showed that most permit applicants and reviewers correctly identify the presence of wetlands and potential impacts for proposed activities. However, 6% of the projects reviewed contained wetlands that were not indicated as present, did not consider those wetlands in the permitting process, and were not require to mitigate for those wetland losses.
9. Mangrove trimming and other activities that damage wetland functions should cease in conservation easement mitigation areas, to maintain benefits to water quality, fish and wildlife. This study and others found that a major reason for failure of long-term mangrove mitigation was due to trimming for aesthetic views (Beever and Loflin 1989; Beever et al., 2011). If regulatory agencies are unable or unwilling to enforce this functional degradation, alternate mitigation should be required.
10. Rip-rap as an alternative shoreline habitat should be examined scientifically, to compare its function to natural and other types of shorelines, including living shorelines containing vegetation. Historically, the shoreline of the CHNEP was not naturally hard lime rock and the native invertebrate communities of nearshore bottoms are adapted to soft sand and mud sediments, seagrass beds, algae beds and oyster bars. There has been a regulatory presumption that rip-rap provides valuable hard surface habitat for coastal benthic organisms, the fish and wildlife that feed upon them, and water quality benefits from filter feeding. However, through the course of this study, the predicted enhanced of benthic communities by rip-rap was not

observed. A variety of negative habitat effects associated with rip-rap were observed, including habitat for non-native invertebrates, inadequate rooting areas for emergent vegetation, stunted growth in mangroves, and habitat for drift and filamentous algae representative of high nutrient conditions.

In conclusion, wetland functional assessment methods were developed to improve wetland regulatory and mitigation processes, above ratio/area methods which mandated multiple hectares of mitigation in return for a single hectare of wetland loss. The goal was to ensure no net loss of wetland area and functions. While this study showed that functional assessment methods do work, the overall effects of using the WFAMs is to cause a gradual loss of wetland area and functions in donor watersheds, with a slow increases in wetland functions, but not area, in receiving watersheds. Implementing the suggestions provided above could enhance achievement of the original "no net loss" intent of wetland functional assessment methods, and contribute to future protection, and perhaps restoration, of coastal wetlands within the CHNEP.

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RETROSPECTIVE ANALYSIS AND SEA LEVEL RISE MODELING OF COASTAL HABITAT CHANGE IN CHARLOTTE HARBOR TO IDENTIFY RESTORATION AND ADAPTATION PRIORITIES

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Abstract: *Charlotte Harbor, Florida, is a series of interconnected shallow estuaries surrounded by low-lying uplands and a population of approximately 780,000 people. Direct and indirect impacts of human development as well as an accelerating rate of sea level rise have had and will continue to have dramatic effects on the distribution of this system's coastal habitats. The long term sustainability of this estuarine system and surrounding human communities depends on understanding past and predicted future coastal scenarios, allowing effective adaptation, restoration and management decisions. To understand historical changes, we compared recent coastal habitat distribution information to that reported in an earlier study (Harris et al., 1983) using geospatial analysis. To understand likely future conditions, we applied the Sea Level Affecting Marshes Model (SLAMM) over a 100 year period using slower, moderate and faster sea level rise (SLR) scenarios of 0.7 m, 1.0 m and 2.0 m, respectively. Our analyses show that while some coastal wetland habitats increased over the sixty year period from 1945 to present, modeling results through 2100 predicted net losses of tidal flat, coastal forest and inland freshwater marsh under all three SLR scenarios. Mangrove swamp and saltmarsh decreased under the fastest rate of SLR modeled.*

Key Words: Adaptation, coastal systems, marsh, oyster reef, restoration, seagrass, Sea Level Affecting Marshes Model (SLAMM), sea level rise (SLR)

The purpose of this retrospective and prospective study was to spatially characterize and quantify both past and future changes in coastal habitats throughout the Charlotte Harbor system to support effective resource management, restoration and climate change adaptation decisions. Charlotte Harbor is located in southwest Florida (Fig. 1) and consists of a series of interconnected estuaries surrounded by low-lying wetlands and uplands, making the region sensitive to sea level rise (SLR). Since the 1940's, coastal habitats in the Charlotte Harbor system have been substantially altered by human development (Beever et al., 2009; Harris et al., 1983) and these alterations are expected to continue into the future as development and sea level rise progress (http://www.esterofl.org/EsteroLife/growth/taxbase.htm, 3/27/12; Beever et al., 2009). In southwest Florida, alteration of coastal systems will continue whether sea level rises 18–59 cm by 2100, as predicted by the Intergovernmental Panel on Climate Change (IPPC, 2007), or at the higher rates predicted by models that include the melting of polar ice caps and other

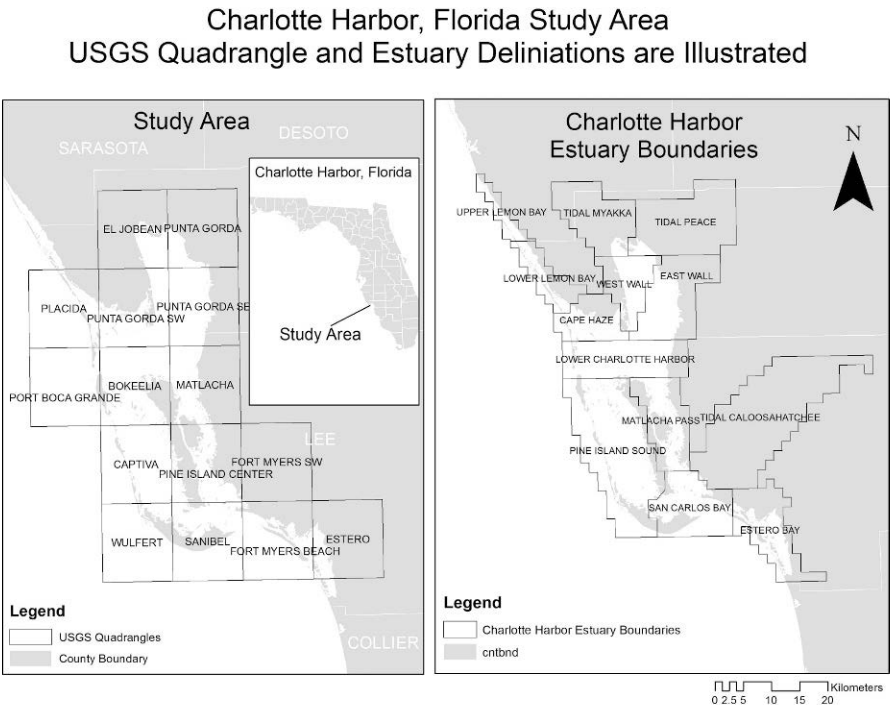

FIG. 1. Study area showing USGS quadrangle names on the left and Charlotte Harbor National Estuary Program (CHNEP) estuaries on the right.

factors (e.g., CCSP, 2008; Mitrovica et al., 2009; Overpeck et al., 2006; Rahmstorf et al., 2007).

Estuaries are especially vulnerable to environmental changes because their productivity varies with the qualities and quantities of water exchanged between the ocean and the adjacent uplands and watersheds (Nicholls et al., 2007). As with many Gulf of Mexico estuaries, the Charlotte Harbor estuaries are bounded in places by urbanized lands (CHNEP, 2008). The human altered landscapes limit the opportunities for intertidal and coastal wetland systems to migrate to higher elevations in response to SLR (Harris and Cropper, 1992). SLR impacts will be observable over decadal time scales (CCSP, 2008; IPPC, 2007; Mitrovica et al., 2009; Overpeck et al., 2006; Rahmstorf et al., 2007), which is within the planning horizons for coastal development. Coastal communities would benefit economically, socially and environmentally by implementing climate change adaptation strategies such as redirecting development away from those natural areas that will be impacted by SLR and maintaining sediment transport to marsh systems (Titus et al., 2009; USEPA, 2009). Understanding how SLR is likely to affect the distribution of coastal natural habitats provides an opportunity to assess the vulnerability to both natural and anthropogenic environments. Better decisions now will result in healthier and safer natural and human communities into the future.

Among the useful tools available to enhance understanding of the effects of SLR on coastal wetland systems is the Sea Level Affecting Marshes Model (SLAMM). SLAMM was developed by the U.S. Environmental Protection Agency (USEPA) in the mid-1980s (Park et al., 1986). SLAMM v6.1 beta dated March 2011 (http://warrenpinnacle.com/prof/SLAMM/index.html), was used for these analyses. SLAMM employs a decision tree that integrates geometric and qualitative relationships between elevation–submergence and wave action–erosion to simulate the dominant processes involved in wetlands change and shoreline modifications during SLR. The five primary processes used to predict wetland changes with SLR are inundation, erosion, overwash, saturation, and accretion.

SLAMM has been applied in different locations around the United States (Glick and Clough, 2006), but several early applications used relatively low resolution elevation (1.5 m contours) based on National Elevation Data (NED). The coarse resolution elevation data requires SLAMM to extrapolate elevations based on other factors, such as land cover data provided by the National Wetlands Inventory (NWI). Comparison of SLAMM results using inferred elevation information versus the recently available high resolution Light Detection and Ranging (LiDAR) elevation data revealed differences in predicted habitat distributions between the two methods of over 170% depending on the habitat type (Geselbracht et al., unpublished data). However, a hindcast of SLAMM using high resolution LiDAR elevation data in the Waccasassa Bay area of the Florida Gulf Coast found that SLAMM predicted the same patterns of coastal forest loss as that observed in 30 years of field plot data (Geselbracht et al., 2010).

Consistent with other areas of Florida's Gulf of Mexico coast, the Charlotte Harbor estuaries have extremely low relief geomorphology. The 1.0 m elevation contour extends inland from the shore as far as 3 to 10 km. These low elevations make the southwest Florida coast particularly vulnerable to SLR (Titus and Richman, 2000) and emphasize the need for accurate elevation data for SLAMM analyses in the Charlotte Harbor region.

To characterize past coastal system changes, we conducted a retrospective (1945 – most recently available) geospatial analysis of habitat trends over a 258,500 ha area of the greater Charlotte Harbor estuaries (Fig. 1). The geospatial analysis compared data derived from aerial photo-interpretation of historic and contemporary conditions. We conducted the analysis to determine long term trends in coastal habitat extent which can be utilized in coastal restoration, conservation and management decision-making.

We also conducted a prospective (2008–2100) SLAMM analysis to predict future coastal habitat conditions for the same areas examined in the historical analysis and compared results to spatially characterize and quantify changes. For the prospective SLAMM analysis, we used recently available high resolution LiDAR-derived elevation data. We evaluated the impacts of 0.7 m (IPCC A1B Maximum scenario), 1.0 m, and 2.0 m SLR by the year 2100. Both the retrospective (1945 – most recently available) and prospective (2000–2100)

TABLE 1. USGS quadrangle maps and associated estuaries in the study area.

USGS Quadrangle	Associated Estuary	Estuary Size (hectares)
El Jobean	Tidal Myakka River	16,804
Punta Gorda	Tidal Peace River	25,056
Placida	Lower Lemon Bay	8,318
Punta Gorda SW, El Jobean	Charlotte Harbor West Wall	11,011
Punta Gorda SE	Charlotte Harbor East Wall	19,273
Placida, Punta Gorda SW	Cape Haze	10,405
Port Boca Grande, Bokeelia, Matlacha	Lower Charlotte Harbor	18,372
Wulfert, Bokeelia, Captiva	Pine Island Sound	39,095
Pine Island Center, Matlacha	Matlacha Pass	15,018
Fort Myers SW	Tidal Caloosahatchee	42,907
Ft. Myers Beach, Sanibel	San Carlos Bay	17,033
Estero, Fort Myers Beach	Estero Bay	16,578

analyses are provided to assist with development of restoration, management and sea level rise adaptation strategies that can be employed to sustain and improve coastal habitat productivity and resiliency while better protecting human communities and economic opportunities. The most recently data available was used, ranging from 1999 to 2007, depending on location and habitat type, as described below.

METHODS—*Study area*—Charlotte Harbor is located in southwest Florida (26°44′58.14″N, 82°07′32.89″W) and is a large, subtropical, estuarine complex approximately 56 kilometers in length comprised of interconnected estuaries, coastal bays and tidal rivers, as shown in FIG. 1 and TABLE 1 Harris et al., 1983; Stevens et al., 2007). This large estuarine system contains at least 71,680 ha of open water and 320 km of shoreline not including the numerous mangrove islands. Water depth of the system averages approximately 1.8 m. Several state Aquatic Preserves are designated in Charlotte Harbor, which is considered one of the most pristine and productive estuarine systems in Florida (FDEP, 2009; Pierce et al., 2003). Three large rivers flow into Charlotte Harbor, the Caloosahatchee, the Peace and the Myakka together draining approximately 78,800 ha. The thirteen estuaries and coastal bays that comprise the Charlotte Harbor system are diverse and productive and are designated as the Charlotte Harbor National Estuary Program (CHNEP). Each of the estuaries has unique biological, geomorphological, water quality and watershed conditions, and associated resource management priorities. Throughout the estuaries, there currently exist tens of thousands of hectares of freshwater and salt water marshes, mangrove swamps, coastal forests, tidal flats, cypress, tidal swamps, beaches and oyster reefs (Pierce et al., 2003). For purposes of our analyses, we used the same study area as in Harris et al. (1983), which encompasses a 258,500 ha portion of the Charlotte Harbor area (somewhat less than the CHNEP area) and is identified by the USGS quadrangle maps illustrated in FIG. 1. We conducted both retrospective and prospective analyses of coastal system change to better understand long term changes in the system. Having a better understanding of past and likely future changes is valuable information that can be used to guide restoration and coastal resilience decisions.

Retrospective analysis (1945–2007)*—We conducted a quantitative, comparative geospatial analysis of coastal habitat change in the 258,500 ha study area over the period 1945 to the most recently available data, which ranged from 1999 to 2007 depending on the location and habitat type. We compared distribution of saltmarsh, mangrove swamp, tidal flat, seagrass and oyster reef habitat from the period 1945 to 1982 as reported in Harris et al. (1983) to the most recent distribution information for these coastal systems. USGS quadrangle maps were used as comparison units to be consistent with the unit of measurement in Harris et al. (1983). In 1983, Harris et al., quantitatively assessed change in coastal system extent through interpretation of 1982 aerial photographs taken at a 1:24,000 scale and compared them to data collected in 1945 using photographic and photo-interpretation techniques. To determine the most recent spatial extent of saltmarsh, mangrove swamp and tidal flat habitat, we used the Florida Natural Areas Inventory (FNAI) Cooperative Land Cover 1.1 (CLC) map which uses the Florida Land Cover Classification System (FLCS) categories to describe various wetland and upland land cover types including developed dry land (http://myfwc.com/research/gis/data-maps/terrestrial/fl-land-cover-classification/). The CLC map pieces together all the latest land cover datasets available statewide. The northern part of the study area is within the Southwest Florida Water Management District (SWFWMD) and the FNAI data were generally collected in 2008. The southern portion of the study area is within the South Florida Water Management District (SFWMD) and these data were generally collected in 2004. To determine recent seagrass distribution, we utilized geospatial data from the SWFWMD and SFWMD collected in 2008 (SWFWMD, 2008). For the most recent oyster reef distribution, we used the geospatial oyster reef coverage available from the Florida Fish and Wildlife Conservation Commission's Fish and Wildlife Research Institute (FWC-FWRI, 2008), which was collected in the Charlotte Harbor area in 1999.

Prospective SLAMM analysis—The prospective coastal system analysis was conducted by modeling the impacts of sea level rise (SLR) using the Sea Level Affecting Marshes Model (SLAMM). The modeling was conducted for a 100 year period covering the years 2000 through 2100 using three SLR scenarios: the Intergovernmental Panel on Climate Change A1B Maximum, which is equivalent to 0.7 m, 1.0 m and 2.0 m. As in previous versions, SLAMM 6.1 beta (http://warrenpinnacle.com/prof/SLAMM6/ SLAMM6_Technical_Documentation.pdf) requires a set of raster input files, a table of site parameters specific to the study area, and a set of model parameters that are entered when a simulation is run. In this study, the raster inputs were a digital elevation model (DEM), a slope layer, and a land cover layer which includes natural communities. The non-spatial, site specific input parameters required to run SLAMM include the photo date of the land cover layer, the date the DEM was created, direction offshore that the subsite faces, historic trend in sea level rise, several tidal elevation parameters (NAVD88 correction, salt elevation, and great diurnal tide range) and the rates of erosion, sedimentation, and accretion for certain wetland types. To accommodate the varying tidal elevations within the Charlotte Harbor system, we created four subsites (i.e., defined polygons over which site parameters are held constant) within the overall study area (see FIG. 2). The "global site" is the remainder of the study area, i.e. the area outside of the subsites. The four subsites include: Peace and Myakka River Estuaries, Estero Bay, Caloosahatchee Estuary and Cape Haze. Site specific information on all SLAMM input parameters is provided below and summarized in TABLE 2. While tidal elevations varied among the site/subsites, the DEM and slope were held constant.

Prospective SLAMM analysis input, land cover raster—The FNAI Cooperative Land Cover (CLC) map (www.fnai.org) was used to classify natural communities and land cover types in the SLAMM analysis. The source data used to create this map are the same as the vegetation data used for the above described retrospective analysis and represents the most recently available land cover conditions. FNAI uses the Florida Land Cover Classification System (FLCS) categories to describe various wetland and upland land cover types including developed dry land (http://myfwc.com/research/gis/data-maps/terrestrial/fl-land-cover-classification/). We examined the FLCS categories and assigned the SLAMM category that most closely matched the vegetation or land use description, as summarized TABLE 3 and shown in FIG. 3. The National Wetlands Inventory (NWI) wetland types were also compared and assigned to SLAMM categories (Clough et al., 2010). Areas identified by the NWI as tidal flats replaced the CLC classification if they overlaid water. Small areas of coastal forest and marsh in the Peace River were coded by the NWI as tidal categories and were designated as tidal categories in our vegetation cover. In addition, the beach distribution data were inconsistent between the SWFWMD and SFWMD data. The SWFWMD dataset did not

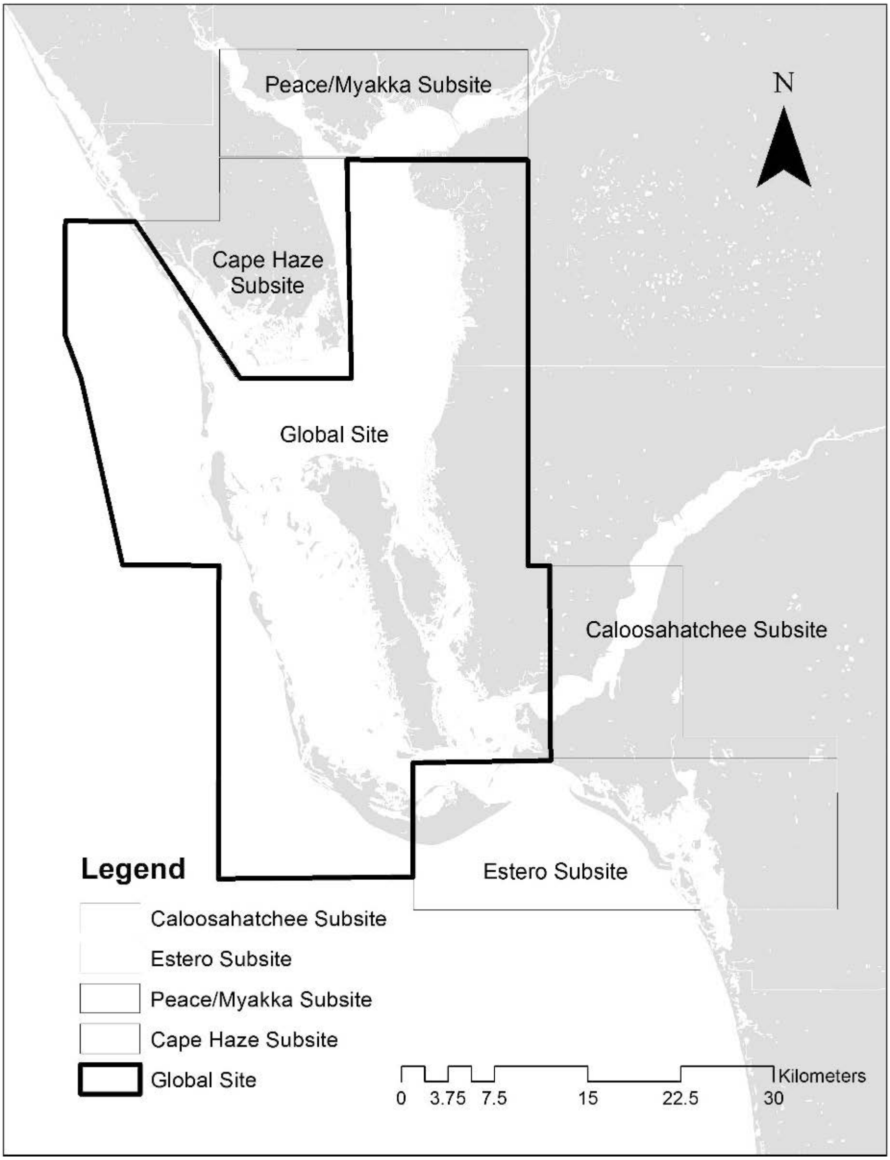

FIG. 2. The study area used in the SLAMM analyses is made up of a global site and subsites. Boundaries are illustrated. Subsites were created to accommodate variations in tidal elevations in the study area. The global site is the portion of the study area outside of subsites.

show beach distribution data along the barrier island in the northern portion of the study area. To correct this, we eliminated the beach information from the vegetation raster and added in the FWC-FWRI beaches layer, which originated from the FWC-FWRI's 2003 Florida Vegetation and Land Cover dataset (http://www.fnai.org/gisdata.cfm).

TABLE 2. SLAMM input parameters for the global site and subsites. The global site is the area outside of the subsites within the study area.

Parameters	Global Site	Subsites			
	Charlotte Harbor	Peace/Myakka River	Estero Bay	Caloosahatchee River	Cape Haze
Land cover photo date	2004	2007	2004	2004	2008
DEM Date	2007	2005	2007	2007	2007
Direction Offshore	West	South	West	West	West
Historic trend in sea level rise (mm/yr)	2.4	2.4	2.4	2.4	2.4
NAVD correction [MTL - NAVD88 (m)]	−0.178	−0.142	−0.185	−0.127	−0.171
Great diurnal tide range (m)	0.529	0.624	0.75	0.401	0.461
Salt elevation (m above MTL)	0.434	0.511	0.615	0.527	0.378
Marsh erosion rate (horizontal m/yr)	0.23	0.23	0.23	0.23	0.23
Coastal forest erosion rate (horizontal m/yr)	0.23	0.23	0.23	0.23	0.23
Tidal flat erosion rate (horizontal m/yr)	0.7	0.13	0.7	0.13	0.13
Saltmarsh accretion rate (mm/yr)	7.2	7.2	7.2	7.2	7.2
Brackish marsh accretion rate (mm/yr)	7.2	7.2	7.2	7.2	7.2
Tidal FW marsh accretion rate (mm/yr)	7.2	7.2	7.2	7.2	7.2
Beach sedimentation rate (mm/yr)	0.3	0.3	0.3	0.3	0.3
Frequency of overwash (years)	31	31	31	31	31
Used elevation pre-processor [True, False]	FALSE	FALSE	FALSE	FALSE	FALSE

TABLE 3. FNAI Cooperative Land Cover (CLC) codes and land cover types assigned for SLAMM analyses throughout the study area. Where NWI and CLC tidal features corresponded, the SLAMM vegetation assignment was changed. Tidal swamp was assigned to the entire CLC polygon with which it overlapped.

CLC Land Cover Code	SLAMM Category
1110–1150, 1210–249, 1311–1330, 1400–1410, 1500, 1610–1660, 1710–1740, 1811, 1831, 1880, 2114, 2221, 2410, 7000–300, 18311–18323, 22311, 183111–183252, 222111, 1832121–1832151	Undeveloped Dry Land
1670	Ocean Beach
1821–822, 1832–1877, 3240–260, 18211–18225, 18324, 182111–182136	Developed Dry Land
2100–2113, 2120–2141, 2300	Inland Freshwater Marsh
2210–2214, 22131–22132, 221311–221312	Cypress Swamp
2215–2220, 2222–2242, 2420–2450, 7400, 22211–22212, 22312–22332	Coastal forest
3100–3115, 3117–3118, 3200–230, 4100–140, 4200–210, 8000	Inland Open Water
3116, 4160, 5000	Estuarine Water
4170	Inland Shore
5200–5220, 9100	Tidal Flat
5230, 52111	Rocky Intertidal
5240	Regularly Flooded Marsh
5250	Mangrove
5251, 21112–21212	Inland Freshwater Marsh
6000	Open Ocean
21231	Tidal Fresh Marsh
22151	Tidal Swamp

Prospective SLAMM analysis input, elevation raster—Digital elevation models (DEMs) derived from high resolution LiDAR data collected by the SWFWMD the Florida Division of Emergency Management (FDEM) Coastal LiDAR Project were downloaded from the NOAA Coastal Services Center's Digital Coast website. The elevation data were downloaded as a DEM in the State Plane Coordinate System (Florida West 1983), with a vertical datum of NAVD88 by averaging ground points within a 5 m cell. The floating-point DEM data were converted to an ArcGIS grid format (Esri ArcGIS 9.3), re-sampled to 30 m cell size, and clipped to the study area. The LiDAR from which the DEM was derived meets or exceeds a 1.2 m horizontal accuracy and 0.20 m fundamental vertical accuracy at the 95% confidence level. Metadata with links to the technical reports for the LiDAR data collection are available at (http://www.csc.noaa.gov/digitalcoast/data/coastallidar/index.html).

Areas of open ocean and tidal creeks that contained "no data" values were set equal to 0. A slope raster in degrees from the DEM was then defined. In all of the model runs, LiDAR-derived elevation data were used; therefore, use of the SLAMM preprocessor was not required.

Prospective SLAMM analysis inputs, site specific parameters—As discussed above, two sets of land cover data were used for this study and photo date varied among the "global site" and subsites (see TABLE 2). The Charlotte Harbor, Estero Bay and Caloosahatchee Estuary global site and subsites fall within the SFWMD boundaries and the land cover data for these site/subsites were collected in 2004. The Peace/Myakka River Estuaries and Cape Haze subsites fall within the SWFWMD boundaries and the land cover data for these subsites were collected in approximately 2007/2008.

The LiDAR data used to create the DEM used for our analysis was collected in 2007, except for the Peace subsite which was collected in 2005. The land cover and DEM dates allow SLAMM to calibrate initial land cover condition to the latest land cover photo date. The direction offshore parameter is used by the SLAMM decision tree to determine the context of a particular cell in relation to offshore areas and informs the direction of habitat conversion. For all but the Peace/

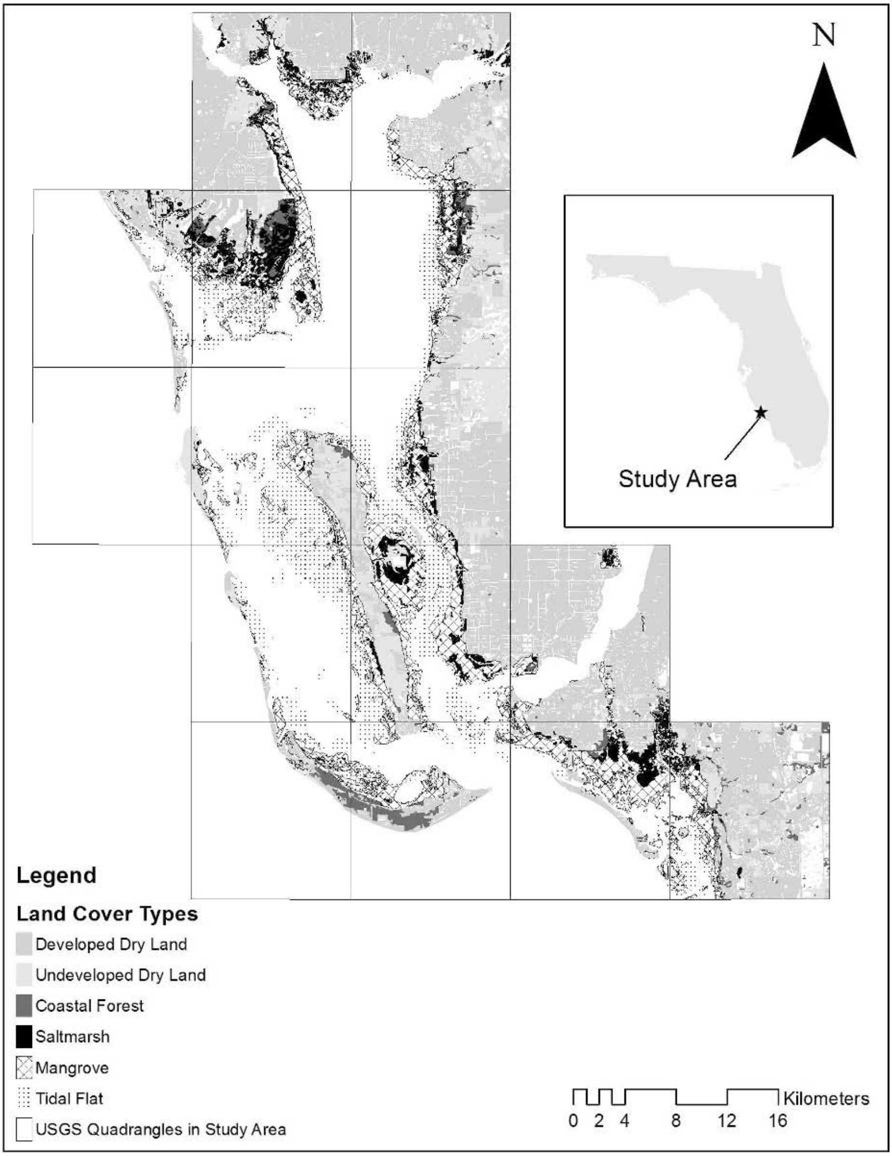

FIG. 3. Land Cover Types in the Charlotte Harbor Study Area. Source is the Cooperative Land Cover, Florida Natural Areas Inventory (FNAI, 2010).

Myakka River Estuaries subsite this direction is west. For the Peace/Myakka River subsite, direction offshore is predominantly south. Historic trend in sea level rise provides SLAMM with the information required to calibrate both the land cover and DEM rasters to the initial condition (year 2007).

SLAMM requires parameters for: converting elevation values to the MTL datum (NAVD88 correction); for the maximum daily tide range (great diurnal tide); and for the elevation at which freshwater wetlands and dry land begin (salt elevation). The SLR rate along with the NAVD88 correction, great diurnal tide range and salt elevation are either published on the NOAA Tides and Currents website (http://tidesandcurrents.noaa.gov/), or are calculated from data found on this website. The NAVD88 correction (MTL-NAVD) was calculated for NOAA stations that had published values for these datums; otherwise, the correction was calculated using NOAA's vdatum program (vdatum.noaa.gov).. Great diurnal tide range was from published station values. These values were averaged within subsites to produce the final SLAMM input parameters.

Salt elevation was also derived from data published on the NOAA tides and currents website. Salt elevation is the elevation boundary between salt water, saltmarsh or brackish marsh and freshwater marsh or dry land. It was calculated by examining 3 years of tide data for the Naples tide station (NOAA tides and currents website). We used the Naples station because the Fort Myers tide station, while within the study area, is substantially influenced by the Caloosahatchee River, limiting its applicability for estimation in the remainder of the study area. From the 3 years of tide data, we calculated a frequency distribution to identify the elevation at which the high tide occurred no more than once a month (i.e., the salt elevation). For the Naples tide station, we then calculated the ratio of 'salt elevation' to mean high water (MHW) and applied this ratio to short-term tide stations in the study area to estimate salt elevation at these stations. The salt elevations for all tide stations within the global site and each of the subsites were then averaged to derive the salt elevation for the study area and each subsite.

For erosion rates, the most proximate data to our study area in the literature was from the Crystal Bay area approximately 230 km north of Charlotte Harbor. In that area, Hine and Belknap (1986) found that saltmarsh eroded at a rate of 0.2 horizontal m/year for coastal embayments. In our study area, coastal forest is set back from the coast except for a small stand found in northern Estero Bay. Because we were unable to identify erosion rate data for coastal forest in the study area or region, the same saltmarsh erosion rate (0.2 horizontal m/year) was also used for coastal forests.

To calculate tidal flat/beach erosion rate, we downloaded long-term coastal erosion rate data from the National Assessment of Shoreline Change (Morton et al., 2004). These data are only applicable to the beaches and tidal flats facing the Gulf of Mexico, not tidal flats located within Charlotte Harbor and its associated estuaries. Using the National Assessment of Shoreline change dataset, we were able to calculate the average erosion rate of 0.7 horizontal m/yr. for the Charlotte Harbor global site and Estero Bay subsite which have Gulf-facing beaches and tidal flats. Because no erosion data for the interior tidal flats and beaches could be located, we applied the same erosion rate calculated above to the Peace/Myakka River Estuaries, Caloosahatchee River Estuary and Cape Haze subsites.

SLAMM is sensitive to the accretion rate parameter and so this rate is a significant source of uncertainty in SLAMM applications (Chu-Agor M.L et. al 2010). In addition, little published information is available on accretion rates along Florida's Gulf Coast. The closest marsh accretion rate data was from Cedar Creek near Crystal River, Florida approximately 230 km north of the study area. In this area, Leonard et al. (1995) measured an accretion rate of 7.2 mm/yr. in saltmarsh, the predominant type of marsh in our study area. Accretion rate was not specifically available for brackish and tidal freshwater marsh, so the rate for saltmarsh was used for all marsh types.

For beach sedimentation rate, we used data from the most proximate source, the Southwest Florida Everglades, estimated to be 0.3 mm/year (Scholl et al., 1969). Frequency of overwash was calculated from historic hurricane information available from NOAA's Coastal Service Center website (http://www.csc.noaa.gov.hurricanes). We conducted a query of all recorded tropical storms passing through the Charlotte Harbor "bay" area. The first recorded tropical storm passing through this area was recorded in 1888. Since that time, 3 tropical storms and 8 hurricanes have passed through this area. We assumed hurricanes rated as a category 2 or higher resulted in overwash of the barrier island system. Since 1888, 4 category 2 or higher hurricanes have passed through the area, so on average approximately 1 overwash event has occurred every 31 years over the 123 year period of record (1888 to 2011).

SLAMM model runs—Because LiDAR-derived elevation data were used, use of the SLAMM preprocessor was not required. In addition, the soil saturation algorithm was turned off, the connectivity algorithm was enabled, and the SLAMM default elevations for coastal wetland system types were utilized (see SLAMM technical documentation for additional information; Clough et al., 2010). All scenarios were run with developed dry land set to "protected". This setting assumes developed areas are surrounded by a dike that will protect them from sea level rise. SLAMM can also be run with developed dry land not protected. We utilized the "protect developed dry land" setting as we assumed efforts will be made to protect existing developed dry land from inundation. We ran SLAMM using the scenarios 0.7 m, 1.0 m, and 2.0 m SLR by the year 2100. The 0.7 m SLR scenario was chosen because it is an IPCC (2007) scenario (A1B maximum). The other two SLR scenarios were selected based on recent projections of the magnitude of SLR to the year 2100 (CCSP, 2008; Mitrovica et al., 2009; Overpeck et al., 2006; Rahmstorf et al., 2007).

SLAMM model output—SLAMM provides output in both tabular and graphic formats. For each output SLAMM predicts the spatial distribution of each wetland system, open water area and upland land type (developed and undeveloped dry land) so change in spatial distribution over time can be calculated by comparing output to the initial condition. The graphic output provides a spatial depiction of where habitat changes are simulated to occur. As with the tabular output, change over time can be described quantitatively and qualitatively by comparing initial condition with output years.

Comparison of historic and prospective coastal habitat analyses—Although we used a global site and subsites to best approximate existing conditions in the study area as input into SLAMM, SLAMM does not provide model results by global site/subsite. Therefore, we used geospatial analysis to compare historic changes in coastal wetland distributions to SLAMM simulated future changes and presented the results for each USGS quadrangle map in the study area to allow comparison to the earlier study by Harris et al (983).

RESULTS—*Retrospective analysis (1945 – most recently available)*—The results of the retrospective analysis of habitat changes between 1945 and the most recently available habitat data are shown in TABLE 4 by USGS quadrangle map. To reference the estuaries associated with each quadrangle map, please see TABLE 1. Results are presented for five habitat types (saltmarsh, mangroves, tidal flats, seagrass and oysters) and three period comparisons (1945–1982, 1982–most recent and 1945– most recent). For this study, we defined substantial changes as those greater than 10% or 100 ha. Where habitat increased from "none present", we reported the results in hectares rather than percent change. The results indicate that throughout the study area, from 1945–1999/2004/2007, saltmarsh and tidal flat habitat increased substantially (+123% and +927%, respectively), while seagrass, mangrove swamp and oyster reef habitat decreased substantially (−25%, −25% and −86%, respectively). It should be noted that the large observed increase in tidal flat extent is more likely due to differences methods and conditions between years than a real gain in tidal flat habitat. Possible differences in methods include tide stage during image collection, photographic methods and/or photo interpretation methods. Because the tide stage and/or time of day for the 1945 and 1982 aerial photography is not available, a comparison of the historic and recent photography is not possible and would be required to better estimate change in extent of tidal flat habitat. For the period 1982–1999/2004/2007, throughout the study area, seagrass habitat remained relatively stable (+6%), saltmarsh increased substantially (+356%), and mangrove swamp and oyster reef declined (−32% and −75%, respectively). Again, caution is advised for interpreting the tidal flat data due to methodological differences.

Over the period 1945 to the most recent year habitat data were available, change in extent of coastal habitats by sub-area did not always follow the pattern of change observed for the study area as a whole. While saltmarsh increased by 123% throughout the study area, a substantial decrease (−68%) was observed in the Fort Myers SW USGS quadrangle (Tidal Caloosahatchee River). Saltmarsh extent remains relatively unchanged (−6%) in the Punta Gorda (Tidal Peace River) quadrangle area. The Port Boca Grande (Lower Charlotte Harbor) and Captiva (Pine Island Sound) quadrangle areas continue to have very little salt marsh habitat. Over the same period (1945–2004/2007), saltmarsh increased substantially in the following quadrangle areas: Bokeelia/Pine Island Sound (+385%), El Jobean/Tidal Myakka River (+13%), Estero/Estero Bay (+17%); Fort Myers Beach/Estero Bay (+183%), Matlacha/Matlacha Pass (+301%), Pine Island Center/Matlacha Pass (+231%), Placida/Cape Haze (+433%), Punta Gorda SE/Charlotte Harbor East Wall (88%), Punta Gorda SW/Charlotte Harbor West Wall/Cape Haze (+878%) and Sanibel/San Carlos Bay (+162%).

Over the study period (1945–2004/2007), while mangrove swamp decreased by 25% in the study area as a whole, mangrove swamp increased substantially in the following USGS quadrangle areas: Estero/Estero Bay (+26%), Fort Myers SW/Tidal Caloosahatchee River (+37%), Matlacha/Matlacha Pass (+25%) and Port Boca Grande (+44%). During the same period, substantial decreases in mangrove swamp were seen in these quadrangle maps: Bokeelia (−56%); Captiva (−13%;), Fort Myers Beach (−32%), Pine Island Center (−36%), Placida (−55%;), Punta Gorda (−41%), Punta Gorda SE (−51%;), Punta Gorda SW (−47%;) and Sanibel (−30%). And, mangrove swamp remained fairly stable in the El Jobean (−3%) and Wulfert (−2%) quadrangle areas.

Regarding seagrass and oyster reef habitat, all quadrangle areas experienced a substantial loss with the exception of Placida (Cape Haze and Lower Lemon Bay) and Punta Gorda SW (West Wall and Cape Haze; +4% and +7%, respectively) for seagrass habitat and Bokeelia (Pine Island Center), Fort Myers Beach (Estero and San Carlos Bays), Fort Myers Southwest (Tidal Caloosahatchee) and Sanibel (San Carlos Bay; +11 ha, +1 ha, +1 ha, and +6%, respectively) for areas that had oyster reef habitat in 1945. Focused seagrass restoration and conservation efforts may explain why this habitat increased by 6% during the period 1982 to 2008. Because of the large uncertainty surrounding how the earlier tidal flat data were collected, we will only say that the only quadrangle area where tidal flat distribution did not increase substantially over the study period 1945 to 2004/2007 is Fort Myers Southwest (−13%; i.e., Tidal Caloosahatchee River).

TABLE 4. Change in coastal habitat distribution over time for (a) saltmarsh, (b) mangrove, (d) tidal flat, (e) seagrass, and (f) oyster habitats. Data sources include: (1) Harris et al. (1983) for 1945 and 1982 all habitats, (2) FNAI Cooperative Land Cover for 2004 and 2007 saltmarsh, mangrove, coastal forest and tidal flat habitats, (3) SWFWMD and SFWMD for 2008 for seagrasses, and FWC for 1999 for oyster habitats.

	1945 (hectares)	1982 (hectares)	1945–1982% Change (ha)	2004/2007 (hectares)	1982–2007% Change (ha)	1945–2007% Change (ha)
(a) SALTMARSH (USGS Quadrangle)						
Bokeelia	12	10	−17%	57	486%	385%
Captiva	0	3		13	349%	(+10)
El Jobean	713	619	−13%	803	30%	13%
Estero	222	160	−28%	261	64%	17%
Fort Myers Beach	311	302	−3%	879	191%	183%
Fort Myers SW	560	138	−75%	181	31%	−68%
Matlacha	187	0	−100%	751	(+751)	301%
Pine Island Center	287	80	−72%	951	1092%	231%
Placida	64	0	−100%	339	(+339)	433%
Port Boca Grand	0	0	0%	2	(+2)	(+2)
Punta Gorda	223	57	−75%	208	268%	−6%
Punta Gorda SE	172	0	−100%	322	(+322)	88%
Punta Gorda SW	177	68	−61%	1726	2423%	878%
Sanibel	9	0	−100%	23	(+23)	162%
Wulfert	0	0	0%	24	(+24)	(+24)
Total Study Area	2936	1436	−51%	6542	356%	123%
(b) MANGROVES (USGS Quadrangle)						
Bokeelia	3544	3731	5%	1572	−58%	−56%
Captiva	1033	1121	9%	898	−20%	−13%
El Jobean	3433	4321	26%	3324	−23%	−3%
Estero	2769	3280	18%	3481	6%	26%
Fort Myers Beach	6032	5955	−1%	4127	−31%	−32%
Fort Myers SW	1936	1190	−39%	2649	123%	37%
Matlacha	4243	5821	37%	5290	−9%	25%
Pine Island Center	8937	11291	26%	5760	−49%	−36%
Placida	1083	968	−11%	483	−50%	−55%
Port Boca Grand	39	32	−18%	56	75%	44%
Punta Gorda	4310	2799	−35%	2532	−10%	−41%
Punta Gorda SE	2821	3502	24%	1377	−61%	−51%
Punta Gorda SW	6885	8251	20%	3645	−56%	−47%
Sanibel	3067	2943	−4%	2137	−27%	−30%
Wulfert	1392	1426	2%	1371	−4%	−2%
Total Study Area	51524	56631	10%	38701	−32%	−25%
(c) TIDAL FLATS (USGS Quadrangle)						
Bokeelia	21	13	−40%	3755	29817%	17735%
Captiva	23	0	−100%	6769		29233%
El Jobean	306	51	−83%	433	748%	41%
Estero	126	68	−46%	980	1341%	678%
Fort Myers Beach	314	147	−53%	786	436%	150%
Fort Myers SW	153	21	−86%	133	519%	−13%
Matlacha	513	21	−96%	4371	21067%	751%
Pine Island Center	941	145	−85%	9711	6600%	932%
Placida	108	57	−47%	1962	3312%	1715%

TABLE 4. Continued.

	1945 (hectares)	1982 (hectares)	1945–1982% Change (ha)	2004/2007 (hectares)	1982–2007% Change (ha)	1945–2007% Change (ha)
Port Boca Grand	0	0		40		
Punta Gorda	347	38	−89%	1577	4001%	354%
Punta Gorda SE	438	103	−76%	2536	2356%	479%
Punta Gorda SW	1186	437	−63%	2407	451%	103%
Sanibel	60	1	−98%	10229	842072%	16971%
Wulfert	0	0		920		
Total Study Area	4537	1102	−76%	46607	4128%	927%
(d) SEAGRASS (USGS Quadrangle)						
Bokeelia	4921	4602	−6%	4442	−3%	−10%
Captiva	8060	4114	−49%	5434	32%	−33%
El Jobean	661	362	−45%	383	6%	−42%
Estero	1586	523	−67%	979	87%	−38%
Fort Myers Beach	1452	1063	−27%	463	−56%	−68%
Fort Myers SW	593	77	−87%	0	−100%	−100%
Matlacha	2340	2000	−15%	1949	−3%	−17%
Pine Island Center	4640	3921	−16%	3566	−9%	−23%
Placida	1057	634	−40%	1100	73%	4%
Port Boca Grand	155	27	−83%	0	−100%	−100%
Punta Gorda	361	313	−13%	183	−41%	−49%
Punta Gorda SE	1719	1442	−16%	1028	−29%	−40%
Punta Gorda SW	2786	2332	−16%	2975	28%	7%
Sanibel	2144	1595	−26%	1915	20%	−11%
Wulfert	1113	678	−39%	634	−6%	−43%
Total Study Area	33587	23682	−29%	25051	6%	−25%
(e) OYSTER REEF (USGS Quadrangle)						
Bokeelia	0	15	(+15)	11	−26%	(+11)
Captiva	23	0	−100%	5	(+5)	−79%
El Jobean	0	2	(+2)	0	−100%	0%
Estero	20	13	−37%	14	14%	−28%
Fort Myers Beach	1	1	50%	2	85%	178%
Fort Myers SW	0	0	0%	1	(+1)	(+1)
Matlacha	0	3	(+3)	0	−95%	0%
Pine Island Center	209	123	−41%	12	−90%	−94%
Placida	22	23	2%	0	−100%	−100%
Port Boca Grand	0	0	0%	0	0%	0%
Punta Gorda	2	2	25%	0	−100%	−100%
Punta Gorda SE	0	0	0%	0	0%	0%
Punta Gorda SW	70	11	−84%	0	−100%	−100%
Sanibel	3	4	25%	3	−15%	6%
Wulfert	0	0	0%	0	0%	0%
Total Study Area	349	197	−44%	49	−75%	−86%

Prospective analysis (2008–2100)—The results of the SLAMM analyses from 2008–2100 are shown in TABLES 5 and 6 and FIG. 4 through FIG. 6. The results indicate substantial changes in coastal wetland systems under all three SLR scenarios modeled. Under the modest SLR scenario of 0.7 m by 2100

TABLE 5. SLAMM results under 0.7 m (IPCC A1B maximum) and 1.0 m SLR scenarios through 2100. Although SLAMM can only use one date as an input parameter, the Initial Condition represents the most recently available data, 2004 or 2007, depending on the location within the study area.

SLR Scenario	Initial	0.7 m	0.7 m	0.7 m	1.0 m	1.0 m	1.0 m
Date	2004/2007	2100	Change	Change	2100	Change	Change
	(hectares)	(hectares)	(hectares)	%	(hectares)	(hectares)	%
Developed Dry Land	50,531	50,507	−24	0%	50,483	−48	0%
Undeveloped Dry Land	17,672	14,964	−2,708	−15%	12,704	−4,968	−28%
Open Ocean	65,946	65,975	29	0%	66,052	106	0%
Estuarine Open Water	63,124	82,635	19,511	31%	86,574	23,450	37%
Mangrove Swamp	22,535	27,072	4,537	20%	27,846	5,311	24%
Tidal Flat	20,490	1,343	−19,147	−93%	647	−19,843	−97%
Saltmarsh	5,770	5,751	−19	0%	5,473	−297	−5%
Inland Open Water	5,455	4,882	−573	−11%	4,584	−871	−16%
Coastal Forest	3,498	1,344	−2,154	−62%	835	−2,663	−76%
Inland-Fresh Marsh	1,695	1,634	−61	−4%	1,379	−316	−19%
Cypress Swamp	1,300	1,298	−2	0%	1,297	−3	0%

TABLE 6. SLAMM results under 2.0 m SLR scenarios through 2100. Although SLAMM can only use one date as an input parameter, the Initial Condition represents the most recently available data, 2004 or 2007, depending on the location within the study area.

SLR Scenario	Initial	2.0 m	2.0 m	2.0 m
Date	2004/2007	2100	Change	Change
	(hectares)	(hectares)	(hectares)	%
Developed Dry Land	50,531	50,283	−248	0%
Undeveloped Dry Land	17,672	7,941	−9,731	−55%
Open Ocean	65,946	66,775	829	1%
Estuarine Open Water	63,124	121,115	57,991	92%
Mangrove Swamp	22,535	3,809	−18,726	−83%
Tidal Flat	20,490	1,488	−19,002	−93%
Saltmarsh	5,770	101	−5,669	−98%
Inland Open Water	5,455	3,949	−1,506	−28%
Coastal Forest	3,498	589	−2,909	−83%
Inland-Fresh Marsh	1,695	750	−945	−56%
Cypress Swamp	1,300	1,294	−6	0%

(IPCC A1B maximum) (TABLE 5; FIG. 4) the model predicted substantial change (>10% and >100 ha) in three coastal wetland systems through 2100: mangrove swamp increased (+ 20%), tidal flat nearly disappeared (−93%) and coastal forest decreased (−62%). Other substantial changes in land and water cover types include a 15% loss of undeveloped dry land, a 31% increase in estuarine open water areas and an 11% decrease in inland open water areas.

Under the moderate SLR scenario of 1.0 m by 2100 (TABLE 5; FIG. 5), the predicted changes were for the same coastal wetland systems and open waters, but the magnitude of change was greater. Under this scenario, mangrove swamp increased (+24%), tidal flat all but disappeared (−97%) and coastal forest decreased by 76%. In addition, inland freshwater marsh decreased under this scenario by 19%. Undeveloped dry land, estuarine open water and inland open water areas also changed substantially (−28%, +37% and −16%, respectively).

Under the higher rate of SLR modeled of 2.0 m by 2100 (TABLE 6; FIG. 6), all wetland systems with at least 1,000 ha in the study area currently decreased substantially including mangrove swamp, tidal flat, saltmarsh and inland freshwater marsh (−83%, −93%, −98%, −83% and −56%, respectively). Cypress swamp experienced little change (−6 ha). In addition, undeveloped dry land in the study area decreased by 55%, estuarine open water increased by 92% and inland open water decreased by 28%.

The most significant predicted land type changes by 2100 under the moderate 1.0 m SLR scenario are shown in TABLE 7 and FIG. 7. Reviewing potential changes under this scenario shows that the largest system transition is from tidal flat to estuarine open water (20,041 ha; 7.8% of study area). Other transitions representing at least 500 ha include undeveloped dry land to mangrove swamp (4,092 ha; 1.6% of study area), coastal forest to mangrove

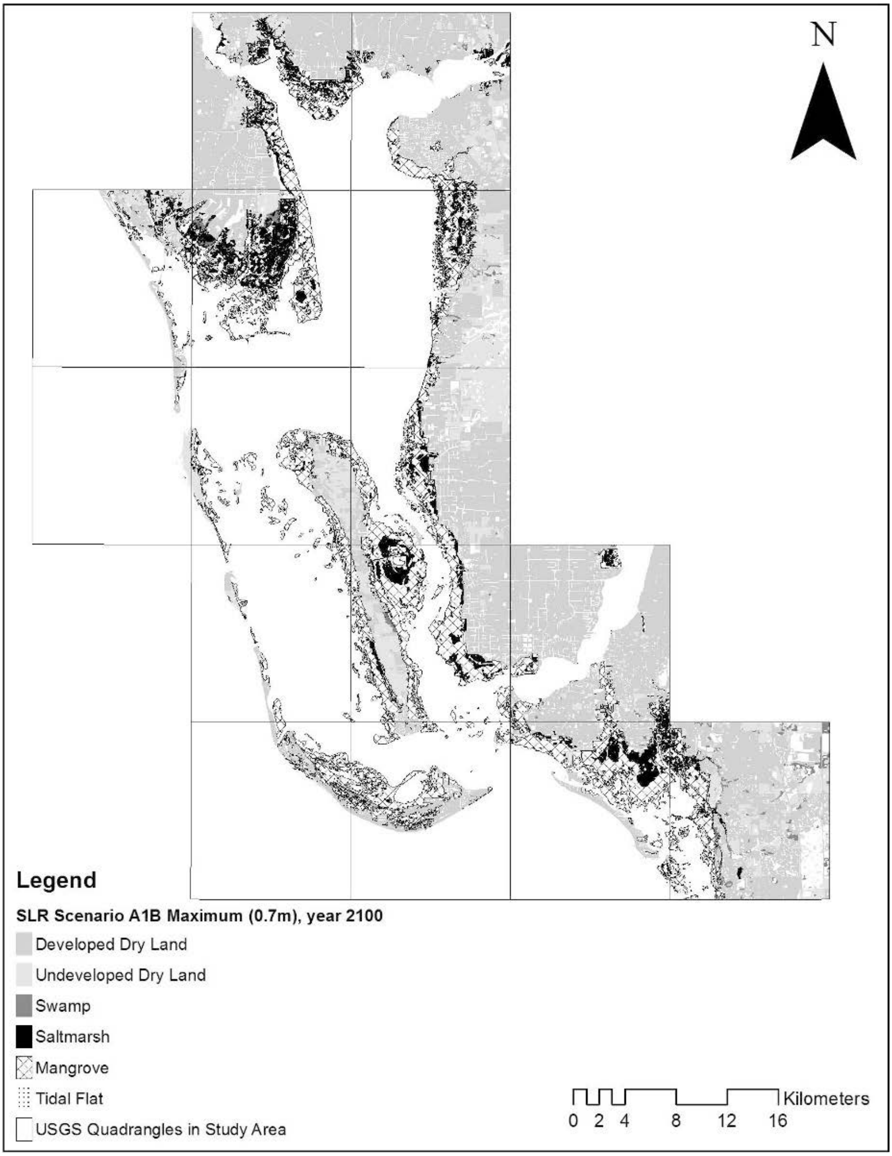

FIG. 4. SLAMM results under 0.7 m SLR scenario through 2100 (IPCC A1B maximum). In this scenario, developed dry land was treated as protected in the model run, assuming there would be no loss of this land type.

swamp (1,921 ha; 0.7% of study area), mangrove swamp to estuarine open water (962 ha; 0.4% of study area) and coastal forest to estuarine open water (704 ha; 0.3% of study area. The quadrangle areas with the greatest transition of tidal flat to estuarine water include Captiva, Pine Island Center, Bokeelia, Matlacha, and Punta Gorda SW and SE (i.e., Pine Island Sound, Matlacha

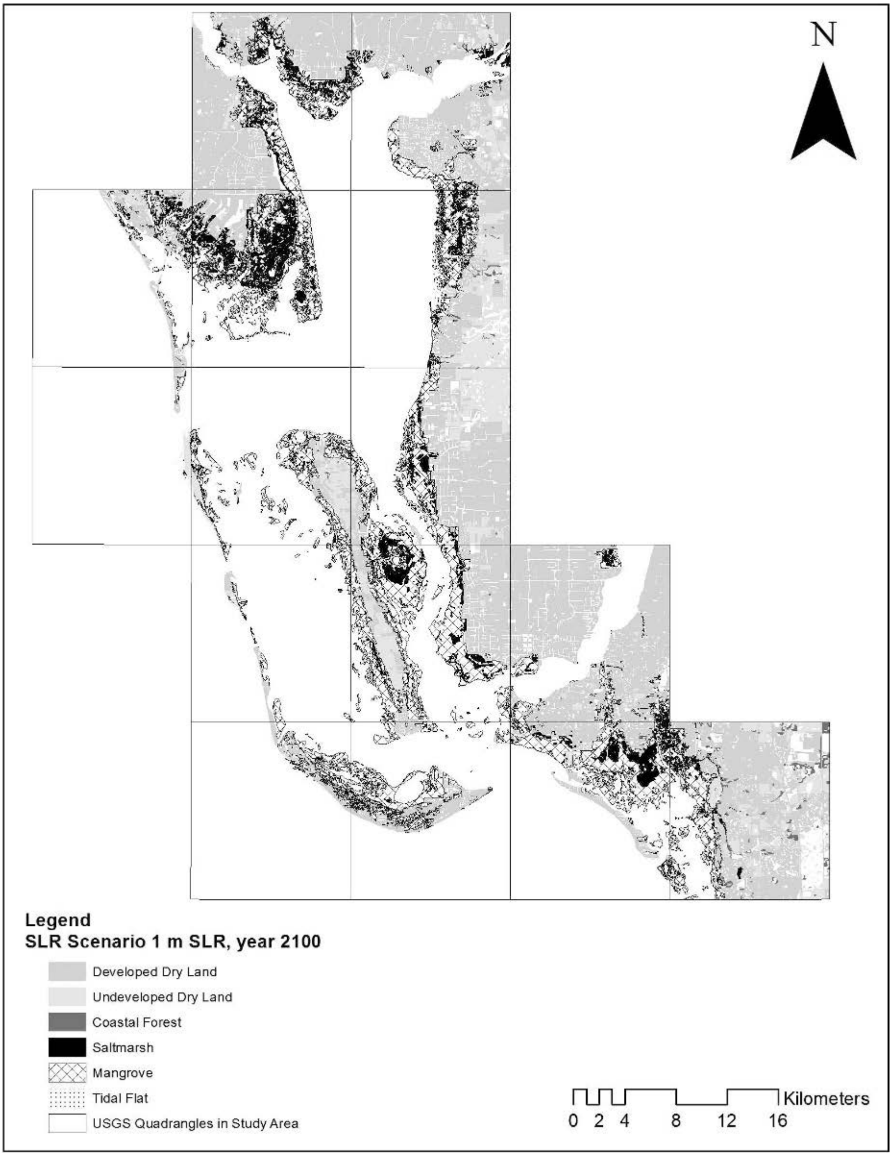

FIG. 5. SLAMM results under 1.0 m SLR scenario through 2100. In this scenario, developed dry land was protected in the model run, assuming there would be no loss of this land type.

Pass, Cape Haze, West Wall and East Wall). The quadrangle areas with the greatest transition of undeveloped dry land to mangrove swamp include Punta Gorda SW and SE and El Jobean (i.e., Cape Haze, West Wall, East Wall and Tidal Myakka River). The transition of coastal forest to mangrove swamp

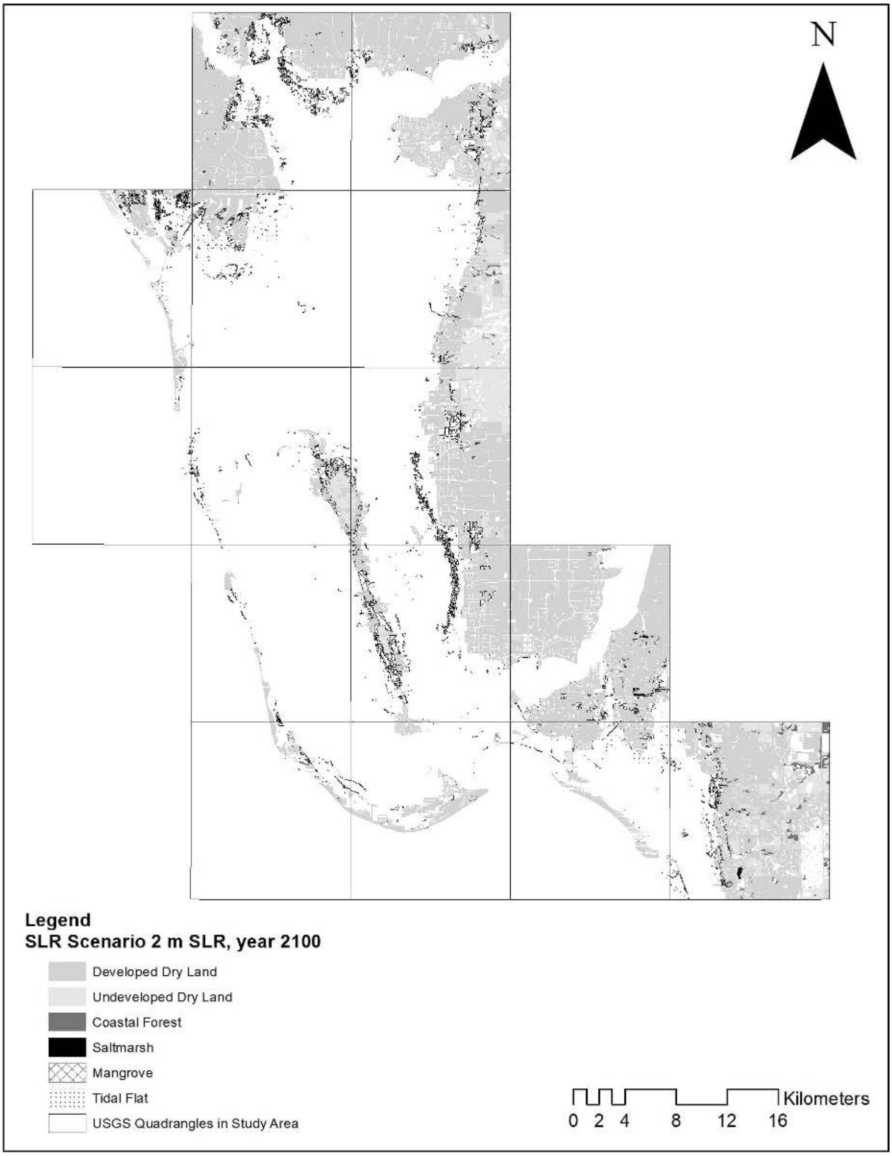

FIG. 6. SLAMM results under 2.0 m SLR scenario through 2100. In this scenario, developed dry land was protected in the model run, assuming there would be no loss of this land type.

primarily occurs in the Sanibel, Punta Gorda SE and SW and Wulfert quadrangle areas (i.e., San Carlos Bay, Cape Haze, West Wall, East Wall and southern Pine Island Sound).

DISCUSSION—From 1945 to 1982, Harris et al. (1983) found a substantial change in some of the coastal habitats examined in the study area, including

TABLE 7. Predicted transition of coastal wetland systems using SLAMM under a 1.0 m SLR scenario by 2100. Only transitions > 500 ha are included.

Transition From/To	Hectares	Percent of Study Area
Tidal Flat to Estuarine Open Water	20,041	7.8%
Undeveloped Dry Land to Mangrove Swamp	4,092	1.6%
Coastal forest to Mangrove Swamp	1,921	0.7%
Mangrove Swamp to Estuarine Open Water	962	0.4%
Coastal forest to Estuarine Open Water	704	0.3%

saltmarsh (−51%), mangrove swamp (+10%), tidal flat (−76%), seagrass (−29%); and oyster reef (−44%; TABLE 4). Harris et al (1983) attributed most of the coastal habitat losses directly or indirectly to coastal development and assumed mangrove swamp expansion was primarily a result of sea level rise and warming temperatures (longer periods between hard freezes). Our analysis shows that since the Harris et al. (1983) analysis, the loss trends for oyster reef habitat have continued (−75% from 1982 to 1999), seagrass extent has slightly improved (+6% from 1982 to 2004/2006), saltmarsh extent has expanded considerably (+302% from 1982 to 2004/2007), and mangrove swamp has declined (−32% from 1982 to 2004/2007; TABLE 4). In the context of this project, it was not possible to assess the change in extent of tidal flat.

Saltmarsh expansion may be at least partially a result of saltmarsh moving into areas that were previously freshwater marsh based on a visual comparison of salt and freshwater marsh distributions in earlier and later land cover maps (i.e., NWI 1999 versus CLC 2008). The retrospective analysis results suggest that given the substantial increase in study area saltmarsh habitat, perhaps at the expense of freshwater wetlands, hydrologic restoration and/or modification where possible may be required to re-establish the diversity of wetland systems that were present prior to extensive urban and suburban development.

Suggested priority restoration areas for coastal wetland systems based on the results of the retrospective and prospective are summarized in TABLE 8 for each estuary (refer to TABLE 1 for associated USGS quadrangle maps). Priority areas for hydrologic restoration and/or modification include the: Punta Gorda SW, Pine Island Center, Fort Myers Beach, Matlacha, Placida and Punta Gorda SE USGS quadrangles. Saltmarsh areas at or adjacent to Cape Haze, West Wall, East Wall, Matlacha Pass and Estero Bay are included within these priority areas.

Despite management practices beginning in the late 1960's that created a buffer system around many of the Charlotte Harbor estuaries and protected mangrove swamps from coastal development, our results show that 25% of mangrove swamp habitat was lost in the study area from 1945 to 2004/2007 (TABLE 4). The quadrangle areas most affected were: Pine Island Center, Punta Gorda SW, Fort Myers Beach and Punta Gorda. These areas include Matlacha

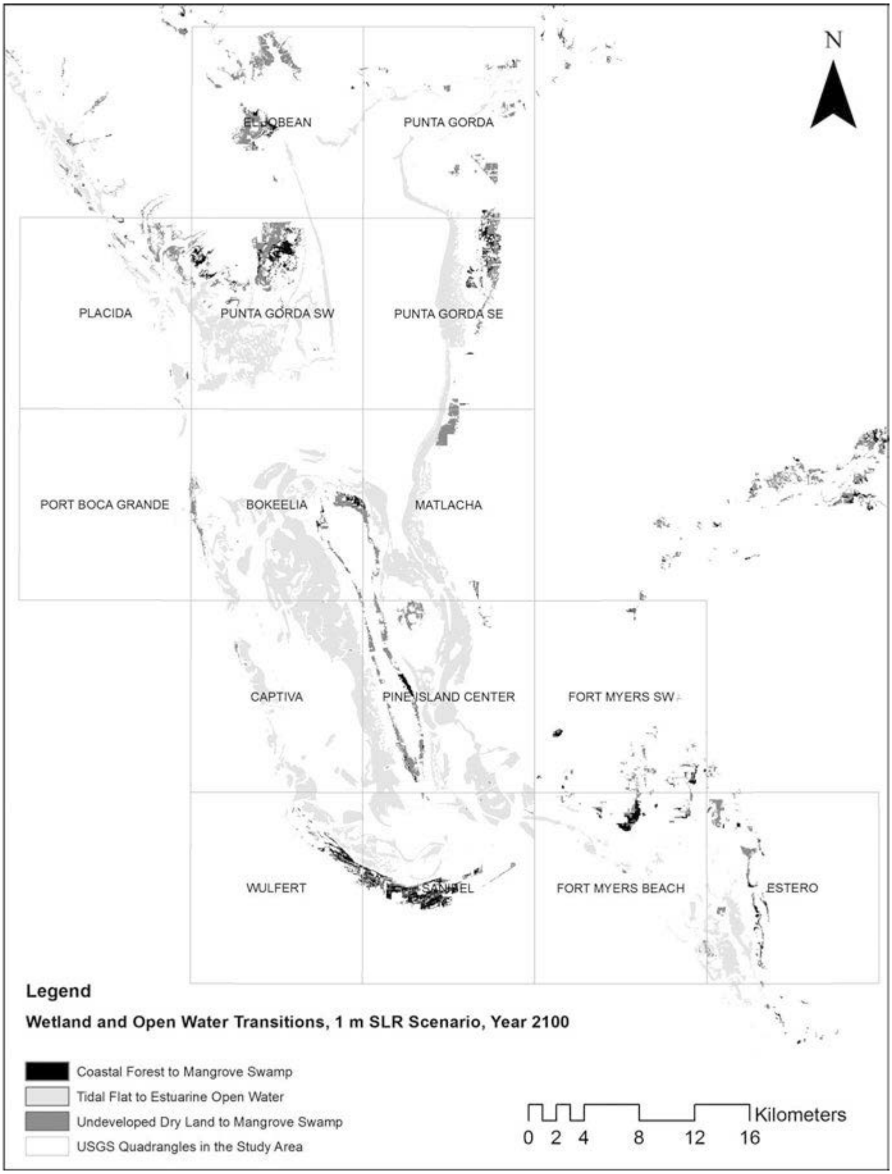

FIG. 7. Greatest land type transitions by 2100 under the moderate 1.0 m SLR scenario. Only changes representing 0.5% or more of the study area are illustrated.

Pass, Cape Haze, West Wall, Estero and San Carlos Bays, East Wall (see TABLE 8). Some of this loss has been attributed to development which required extensive dredge-and-fill activities (Harris et al., 1983). Given the protective nature of mangrove swamps in coastal storms and their relatively high habitat

TABLE 8. Priority restoration areas for coastal wetland systems based on retrospective and prospective analyses of past and predicted future habitat change in the study area.

Estuary	Salt-marsh	Mangrove Swamp	Seagrass	Oyster Reef	Tidal Flat (Facilitate Mangrove Growth)	Coastal Forest	Inland Fresh-water Marsh
Tidal Myakka River		X					
Tidal Peace River							
Lower Lemon Bay							
Charlotte Harbor West	X	X		X		X	
Charlotte Harbor East	X	X			X	X	
Cape Haze	X	X		X	X	X	X
Lower Charlotte Harbor							
Pine Island Sound				X	X	X	
Matlacha Pass	X	X		X	X		
Tidal Caloosahatchee			X				
San Carlos Bay		X	X		X	X	
Estero Bay	X	X	X		X		

value (Das and Vincent, 2009; Odum et al., 1982), opportunities for restoring mangrove swamps in these areas should be explored.

Our results also show that seagrass habitat in the study area has diminished since 1945 in most areas, with the exception of the Placida and Punta Gorda SW quadrangle areas, including Cape Haze and Charlotte Harbor West Wall. However, since 1982, several locations have shown increases in seagrass distribution including: Captiva, Estero, Placida, Punta Gorda SW and Sanibel quadrangle areas, corresponding to Pine Island Sound, Estero Bay, Cape Haze, West Wall and San Carlos Bay. These increases are likely due to the implementation of numerous water quality improvement projects over the last few decades (CHNEP, 2008; SFWMD, 2008; SWFWMD, 2000). Submerged habitats throughout the study area will continue to benefit from implementation of the ongoing water quality improvement efforts and efforts to protect and restore seagrass. Two priority areas for reversing seagrass loss based on past losses are the Fort Myers SW and the Fort Myers Beach quadrangle areas, including the Tidal Caloosahatchee River, San Carlos Bay and Estero Bay (see TABLE 8).

Our analysis indicates that the areas of greatest loss of oyster reefs include Pine Island Center, Punta Gorda SW, Placida and Captiva (i.e., Matlacha Pass, Cape Haze, West Wall and Pine Island Sound; Table 8). Consequently, these areas are candidates for oyster reef restoration. However, current and anticipated future conditions (salinity, substrate and hydrologic regime, etc.) will need further examination to clarify where restoration is likely to be successful. Some of the oyster reef losses may have resulted from direct impacts such as coastal development, filling, dredging and/or harvesting. In other cases, losses may have been a result of indirect impacts such as degraded water quality or modified hydrologic regime (Harris et al, 1983). Ongoing hydrological restoration activities may need to be completed in some areas before oyster reef restoration efforts can be successful.

Looking to the future, the SLAMM results predicted that the Charlotte Harbor system within our study area will lose substantial areas of some coastal systems by the year 2100 under the moderate 1.0 m SLR scenario (FIG. 5 and TABLE 5). Coastal wetland systems that are predicted to lose more than 25% of current area are tidal flat and coastal forest (−97% and −76%, respectively). While it would be difficult if not impossible to slow the transition of tidal flat areas to shallow open water areas as sea level rises, it may be possible to preserve this area as coastal wetlands if mangrove swamp colonization is allowed and/or encouraged. Areas that are predicted to lose the most tidal flat include Cape Haze, East Wall, Pine Island Sound, Matlacha Pass, San Carlos Bay and Estero Bay (TABLE 8). Loss of coastal forest systems with rising sea level could perhaps be slowed by influencing freshwater flows, sedimentation and nutrient loading (Lewis, 1992; Saha et al., 2011; USEPA, 2009; Williams et al., 1999). Areas most vulnerable to coastal forest loss include those adjacent to Cape Haze, West Wall, East Wall, Pine Island Sound and San Carlos Bay where under the moderate 1.0 m SLR by 2100 coastal forest transitions to mangrove swamp and shallow open water areas.

Under the higher SLR scenario (2.0 m by the year 2100), three additional coastal wetland systems suffer substantial area losses. These include saltmarsh (−98%), mangrove swamp (−83%) and inland freshwater marsh (−56%; Table 6). Under this SLR scenario, saltmarsh transitions to shallow subtidal open water and tidal flat habitat. Some of the same approaches noted above to slow the rate of loss of coastal forest (influencing freshwater flows, sedimentation and nutrient loading) could also be used to slow the loss of saltmarsh. In addition, natural habitat restoration and/or creation techniques that reduce shoreline erosion may help slow the rate of saltmarsh loss from some areas. One such technique is the creation and/or restoration of oyster reefs/reef structures along the offshore edge of saltmarshes transitioning to tidal flat or open water. These oyster reef restoration and creation projects have been shown to enhance accretion of sediments and the expansion of marsh vegetation (Dumesnil, 2011). The portions of the study area most vulnerable to saltmarsh loss are those at or adjacent to Cape Haze, Matlacha Pass (Little Pine Island) and northern Estero Bay (Table 8).

While mangrove swamp is able to expand under the 1.0 m SLR scenario by 2100, it is largely replaced by open water under the more rapid 2.0 m SLR by 2100 SLAMM scenario. If SLR occurs at this higher rate, the extensive loss of mangrove systems would increase the vulnerability of both human and natural communities in this area (Das and Vincent, 2009). Mangrove swamps are known to mitigate the effects of storm surge and in the tropics they serve as the base of the estuarine food web (Spalding et al., 2010). In the face of higher rates of SLR, strategies should be implemented to enhance the survival of the protective mangrove systems to the greatest extent possible. One strategy would be to encourage mangroves to migrate onto undeveloped dry land as it becomes increasingly inundated by eliminating as many physical obstacles as possible. Areas where mangrove swamp is most likely to colonize adjacent undeveloped dry land include Lower Lemon Bay, East Wall, Pine Island, Lower Charlotte Harbor, Matlacha Pass and Estero Bay.

Under the 2.0 m SLR scenario, inland freshwater marsh primarily transitions to mangrove swamp, particularly in the Cape Haze area (Table 8). The same techniques noted above for slowing the transition of coastal forest could be applied to slowing the transition of inland freshwater marsh, namely restoring and/or enhancing freshwater flows and facilitating enhanced sedimentation of the system.

The prospective analysis did not address seagrass or oyster reef as SLAMM does not address these habitat types. Seagrass may be able to expand substantially as sea level rises and there are some early indications of migration in Charlotte Harbor (Ott, 2010). Oyster reef may have similar opportunities, but is less likely to expand without human intervention (e.g., additional management measures, oyster reef restoration and/or re-establishment of more natural water flow regimes) as reefs have not naturally rebounded over the last several decades (Geiger, 2009).

The habitat changes noted in the Charlotte Harbor area are not unique to this region. Large areas of coastal wetlands across the Gulf of Mexico are likely to be lost as sea level rises unless adjacent inland habitats are protected from development and hydrologic modification (Geselbracht et al., 2010). The Charlotte Harbor region has an advantage compared to many other regions of the state as large areas of coastal wetlands were protected from development beginning in the late 1960s (USEPA, 1992). These buffer lands will facilitate the upslope migration of some low-lying coastal habitats, but will likely be insufficient under higher rates of SLR. In the developed portions of the study area, where coastal wetlands were largely eliminated, human communities are most vulnerable to SLR impacts including the risks from coastal storms. The vulnerability of these human communities will increase as sea level rises (Shepard et al., 2011) because the coastal wetlands that remain will be unable to migrate to higher elevations where blocked by structures, roads and other development. In such areas, efforts should be made to accommodate upslope migration of coastal habitats such as mangrove swamp as a means of not only preserving ecological values, but as a way of improving the protection of human property and welfare. Where coastal wetlands remain connected to undeveloped lands at a higher elevation, efforts to avert development are advisable. In some areas, coastal wetlands may persist longer than in other areas if mangrove swamp colonization is fostered and oyster reefs are restored in lieu of hardened shorelines where stabilization is required. Mangrove swamps and oyster reef communities can help stabilize sediments, protect shorelines from wave-generated erosion, and mitigate vulnerability of coastal communities to natural hazards and SLR (Das et al., 2009; Meyer et al., 1997; Spalding et al., 2010).

Protecting healthy coastal wetland systems in the face of SLR is of added importance in the context of the Gulf of Mexico Deepwater Horizon BP oil spill in April 2010. The Gulf's healthy coastal wetlands will serve as critical refugia for numerous species following this and potential future spills. Many approaches for mitigating the loss of highly productive coastal wetlands are being suggested including living shorelines and oyster reef restoration to stabilize shorelines and maintaining sediment loads and freshwater flows to maintain marshes and other wetlands (EPA, 2009; IPCC, 1990). Regardless of the approaches adopted, mitigation and adaptation strategies need to be flexible so as to increase the probability that these coastal wetland systems and the services they provide will be conserved into the future. The quantitative and spatial data developed in this study provides a synopsis of the coastal wetland changes that have taken place in the Charlotte Harbor system over the last 60 plus years and the changes that are likely to occur as a result of SLR in the future. Future predictions of habitat distribution changes using SLAMM could be improved if some of the uncertainty regarding marsh accretion in the study area could be addressed through data collection.

The information provided by our retrospective and prospective analyses can be used to identify where specific types of coastal wetland restoration are most needed in the Charlotte Harbor study area and support the climate change adaptation planning and implementation underway in the Charlotte Harbor region (Beever et al., 2009a; Beever et al., 2009b). This work provides the framework for more detailed restoration siting studies that will incorporate such considerations as land use, land ownership, water quality and hydrologic conditions. Taking action now to protect the region's coastal wetland systems will not only result in maintaining a healthy coastal ecosystem, but will maintain the natural system's ability to protect the region's human communities.

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