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## CHARLOTTE HARBOR NEP 2014 WATERSHED SUMMIT PROCEEDINGS

Our vision in action



(Continued on the Back Cover)

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## Charlotte Harbor NEP 2014 Watershed Summit Proceedings Dedication & Acknowledgements

Guest Editor: Ms. Judy Ott.
Assistant Guest Editors: Mr. Michael Flannery, Dr. Penny Hall, Dr. Phil Stevens, Dr. Cassondra Thomas, and Dr. Dave Tomasko.

### Dedication

This 2014 Charlotte Harbor Watershed Summit Special Issue of the *Florida Scientist* is dedicated to one of our exceptional scientists, Dr. Ralph Montgomery, whom we recently lost and is dearly missed.

Dr. Montgomery was an outstanding, multi-disciplinary, environmental scientist who made valuable and long-lasting contributions to our understanding of the Charlotte Harbor ecosystems and their complexities. He was also known for his extraordinary personal character. Dr. Montgomery was a kind, respectful, and humorous gentleman who always interacted with citizens and colleagues in a patient and generous manner, willing to share complicated technical information about our local ecology in an understandable way.

Dr. Montgomery received his PhD from Florida State University in 1978, where he studied diatoms on coral reefs in the Florida Keys. He and his family moved to Charlotte County in 1981. During his tenure as a respected regional scientist, he helped create and direct an extensive hydrobiological monitoring and research program on the Lower Peace River and Upper Charlotte Harbor Estuary. Much of the data collection program he began in the early 1980s continues today. Dr. Montgomery's work on the hydrologic, physical, and chemical factors affecting phytoplankton production in our tidal river estuary is among the most comprehensive information of its kind in the state. He performed a large number of other projects for utilities and governmental agencies in Florida and will continue to be regarded as one of the leading authorities on the hydrology and water quality of the Peace River watershed. He also taught classes on a range of biological subjects at several educational institutions throughout the region. A tribute summary report of Dr. Montgomery's life and technical accomplishments will be available from the Charlotte Harbor National Estuary Program in the summer of 2016.

The Charlotte Harbor National Estuary Program and our partners sincerely appreciate Dr. Montgomery's dedication and exceptional contributions towards protecting the special resources of the Charlotte Harbor region. His scientific findings and creative problem solving will bring long-lasting benefits to the Harbor and its residents for many years to come. We wish him good fishing.

## Acknowledgements

The success of the 2014 CHNEP Watershed Summit at bringing together researchers, resource managers, elected officials and citizens from throughout the Charlotte Harbor region to share scientific information would not have been possible without the significant support of our partners and sponsors.

Special recognition is given to the esteemed Assistant Guest Editors who professionally and persistently coordinated the technical review of the manuscripts, including: Mr. Michael Flannery, Dr. Penny Hall, Dr. Phil Stevens, Dr. Cassondra Thomas, and Dr. Dave Tomasko.

Technical reviews of the manuscripts in this Special Issue were provided by many scientists who generously shared their expertise and time toward ensuring presentation of sound, well written scientific information. We genuinely thank these reviewers for their conscientious and thorough comments: Tom Ash (EPCHC), Patrick Biber (USM), Jaime Boswell (Jamie Boswell LLC), Melynda Brown (FDEP), Robert Burnes (Pinellas County), Jennifer Cannizzaro (USF), Marissa Carrozzo (Conservancy of SW FL), Bob Chamberlain (SJRWMD), Nora Demers (FGCU), Kellie Dixon (Mote), James Douglass (FGCU), Mike Duever (City of Napes), Don Duke (FGCU), James Evans (City of Sanibel), Michael Flannery (SWFWMD, retired), Lizanne Garcia (SWFWMD), Jennifer Gihring (SJRWMD), Jennifer Hecker (Conservancy of SW FL), Jeffrey Hill (UFL), Steve Krupa (SFWMD), Kimberly Lawrence (Osceola County), Eric Milbrandt (SCCF), Lori Morris (SJRWMD), John Ryan (Sarasota County), Ed Sherwood (TBEP), Shelley Thornton (Mosaic),

We also sincerely thank our many 2014 Charlotte Harbor Watershed Summit sponsors, including: Atkins, Benchmark, Jaime Boswell, CF Industries, Charlotte County, Charlotte Harbor Event and Conference Center, Friends of Charlotte Harbor Aquatic Preserves, Friends of Charlotte Harbor Estuary (CHNEP Friends), Earth Balance, Estero Bay Buddies, Florida Fish and Wildlife Conservation Commission, Jones Edmunds, Captain Joseph Kliment, Lemon Bay Conservancy, Mosaic, Mote Marine Laboratory, Peace River Manasota Regional Water Supply Authority, Science and Environment Council of Southwest Florida, Sierra Club Greater Charlotte Harbor Group, Southwest Florida Watershed Council, Stantec, Tom Winter and three anonymous donors.

In addition, this outstanding publication was enriched by the skills of the Florida Academy of Sciences Editors, including: Dr. James Austin, Dr. Jeremy Montague, Dr. Richard Turner, and Ms. Mary Vallianatos, as well as by Mr. Joseph Barriger from Allen Press. We greatly appreciate their expertise and patience with the many technical details involved with preparing this journal.

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 Special Issue of the Florida Scientists. 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, Pease 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.

And, most importantly, to our many exceptional partners - Thank You. The great diversity, magnitude and continuing nature of your contributions are essential to our collective successes at protecting and restoring the special natural resources of the Charlotte Harbor region.

# 2014 Watershed Summit: Our Vision in Action

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**Abstract** The theme of the 2014 Charlotte Harbor National Estuary Program Watershed Summit was "Our Vision in Action." The Charlotte Harbor National Estuary Program vision was developed through our partnerships and is featured in the adopted Comprehensive Conservation and Management Plan. The first Comprehensive Conservation and Management Plan was originally adopted in 2000. A vision section was added in 2008 and refined in 2013. Our overall vision is to restore altered hydrology, reverse water quality degradation, protect fish and wildlife habitat, and improve stewardship of natural resources. Our graphic vision was presented to the participants of the 2014 Watershed Summit. The participants used a technique called "Group Scoop" to identify the most significant accomplishments. The top four accomplishments in the Charlotte Harbor National Estuary Program's 18-year history represent each of the four vision topics. They were:

1. Urban Fertilizer Ordinances, a water quality accomplishment
2. K-12 Outreach and Education, a stewardship accomplishment
3. Peaceful Horse Ranch acquisition, a fish and wildlife habitat accomplishment
4. Minimum Flows and Levels, a hydrology accomplishment.

**Keywords** Accomplishments, national estuary program, vision, watershed management

## Background

National Estuary Programs (NEPs) were created under Section 320 of the Federal Clean Water Act. There are 28 in the United States and four in Florida. The Charlotte Harbor National Estuary Program (CHNEP) was designated in 1995 and is among the most recently designated. The Clean Water Act requires each NEP to convene a "Management Conference." CHNEP's Management Conference includes four committees, including one each for citizens, scientists/technicians, resource managers and elected/appointed officials. The Policy Committee consists of elected officials and top agency heads, and is the decision-making body of the CHNEP, which receives recommendations from each of the other three committees.

The Clean Water Act requires each NEP to adopt and implement a Comprehensive Conservation and Management Plan (CCMP). The CHNEP CCMP (CHNEP 2013) includes four priority problems, 15 quantifiable objectives and 64 priority actions. The priority problems include hydrologic alterations, water quality degradation, fish and wildlife habitat loss, and

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stewardship gaps. The CCMP is ambitious and requires CHNEP's broad partnerships to implement it. The 2014 Watershed Summit was used as an opportunity to identify the most significant achievements accomplished through CHNEP's partnership.

The CHNEP's 2014 Watershed Summit theme was "Our Vision in Action." The first session of the Summit was designed to highlight the conference theme. After the welcome, the audience was introduced to the "Our Vision in Action" game rules. The game was based on a game titled "Group Scoop" (Thiagarajan 2008). The summit participants were told that they were going to be shown the CCMP vision graphics. They were asked to think about the best action taken in the CHNEP area and write each idea on a five by eight inch index card. They were encouraged to generate as many ideas as possible. After the presentation of the CCMP vision graphics, audience members shared their ideas with individuals in neighboring seats. For each time neighbors shared their ideas, they were able to vote in favor of a single idea from among all of the ideas they discussed. They voted by signing the back of the index card with the action that they deemed the best.

As anticipated, audience members identified approximately 500 actions, many of the same actions individually. The final session of the Summit included a presentation of the "top ten" list of "Our Vision in Action." These were (in rank order):

1. Urban Fertilizer Ordinance
2. Kindergarten to 12th Grade High School (K-12) Outreach and Education
3. Peaceful Horse Ranch Land Acquisition
4. Minimum Flows and Levels (MFLs)
5. Stormwater Retrofits (Low Impact Development, sediment traps, etc.)
6. Expanding water quality monitoring partnerships
7. C-43 Reservoir on the Caloosahatchee River
8. Lake Okeechobee Regulation Schedule for flows of under 28,000 cfs
9. Lake Hancock Restoration Projects
10. Providing resources to Elected Officials and Policy makers.

The following sections describe the top four actions that have been accomplished to implement the CCMP. Each of the four actions identified at the Watershed Summit corresponds to a CCMP priority problem: Urban Fertilizer Ordinances (water quality), K-12 Outreach and Education (stewardship), Peaceful Horse Ranch Land Acquisition (fish and wildlife habitat), and Minimum Flows and Levels (hydrology).

**Urban fertilizer ordinances.** In 2007, the Southwest Florida Regional Planning Council (SWFRPC) adopted a series of resolutions designed to reduce urban water pollution. The first of these was #07-01 Fertilizer Resolution, adopted on March 15, 2007. It provided recommendations and guidelines to be considered by local government jurisdictions for the regulation and control of fertilizer application. The State of Florida followed this action with the adoption of a Model Ordinance (FDEP 2010). There are several significant differences between the State Model and the SWFRPC resolution including higher maximum nitrogen application rates and no restricted application in the rainy season by the State model. The coastal cities and counties within the CHNEP study area all adopted the SWFRPC resolution guidelines in preference over the State model, with effective dates ranging between March 2007 and June 2012. Most jurisdictions in the CHNEP area had adopted an urban fertilizer ordinance by December 2008. Coastal counties and cities north of CHNEP adopted similar stricter ordinances. The expansion of Urban Fertilizer limitations were largely in response to the November 2008 Tampa Bay Model Regional Fertilizer Ordinance, Sierra Club outreach, and SWFRPC assistance. The coastal area between Pinellas and Lee Counties has uniformly adopted measures recommended in the SWFRPC resolution, with some (often stricter) variation in ordinance language.

The effect of the Southwest Florida urban fertilizer ordinances was evaluated using the Charlotte Harbor Estuaries Volunteer Water Quality Monitoring Network (CHEVWQMN). Volunteers collect water quality data the first Tuesday of each month at dawn. There are 51 fixed stations throughout the CHNEP estuarine study area. The data are all collected under the direction of the Florida Department of Environmental Protection (FDEP) using consistent Standard Operating Procedures (SOPs) and laboratory methods. Though data collection was initiated November 1996, collection of all parameters appears to have been implemented beginning August 1998. Because each site is managed by a volunteer, not all sites are visited through the period of record. Ten of the 51 sites were suppressed from the analysis because of inconsistent data collection between 2000 and 2015. The suppressed sites include two in Charlotte Harbor, three in Gasparilla Sound, three in Pine Island Sound and two in Estero Bay. The data collection period for pre-ordinance assessment was April 2000 through March 2007. The period for post-ordinance assessment was from April 2012 through March 2015. March 2015 was the last month of data available at the time of this analysis. Data from April 2007 to March 2012 were suppressed for the purposes of the analysis.

Several parameters were analyzed. Variables included pre- and post-adoption of all ordinances (pre/post), salinity, total phosphorus (TP), total nitrogen (TN), total Kjeldahl nitrogen (TKN, a measure of ammonia, ammonium and organic nitrogen) and fecal Coliform (fcoli). The variables were tested for correlation (Table 1). Furthermore, the variables were tested under a partial correlation, controlling for salinity (Table 2).

The Kendall's Tau B rank correlation coefficient test was used because water quality data are often not normally distributed. Two periods were compared: 1) the period before the urban fertilizer ordinances were adopted including April 2000 through March 2007 and 2) the period after nearly all the urban fertilizer ordinances were adopted including April 2012 through March 2015. The pre-ordinance included 2250 samples or more. The post-ordinance

Table 1. Kendall's Tau B correlation coefficients (r, two-tailed) for CHEVWQMN data (N = sample size) April 2000 through March 2007 and April 2012 through March 2015.

| | | Pre- Post | Salinity ppt | TP ugl | TN ugl | TKN ugl |
|---|---|---|---|---|---|---|
| Salinity ppt | r | −0.009 | – | | | |
| | N | 4091 | | | | |
| TP ugl | r | −0.070** | −0.370** | – | | |
| | N | 3575 | 3504 | | | |
| TN ugl | r | −0.124** | −0.290** | 0.303** | – | |
| | N | 3449 | 3379 | 3441 | | |
| TKN ugl | r | −0.110** | −0.295** | 0.325** | 0.853** | – |
| | N | 3523 | 3452 | 3515 | 3428 | |
| Fcoli 100ml | r | 0.160** | −0.237** | 0.230** | 0.162** | 0.154** |
| | N | 3749 | 3677 | 3542 | 3417 | 3489 |

** Significant at the 0.001 level

period had 1199 samples. No statistically significant differences in salinity were detected.

Total phosphorus, total nitrogen and total Kjeldahl nitrogen all had a statistically significant decrease between pre- and post-adoption of all ordinances. Fecal coliform had a statistically significant increase between pre- and post-adoption of all ordinances, even though fecal coliform had a positive statistically significant relationship with all measures of nutrients (total phosphorus, total nitrogen and total Kjeldahl nitrogen) which decreased. Therefore, decreases in septic system pollution or animal waste cannot be suggested as a cause for nutrient reductions in the estuaries. The use of Mann-Whitney U test confirms that the distribution of total phosphorus, total nitrogen and total Kjeldahl nitrogen between pre- and post-ordinance periods were significantly different at the 0.001 level, whereas the distribution of salinity was not.

Controlling for salinity, variables including total phosphorus, total nitrogen and total Kjeldahl nitrogen all had a statistically significant decrease between pre- and post-adoption of all ordinances. Fecal coliform had a statistically significant increase. Missing cases were excluded pairwise.

Table 2. Partial correlation (r), controlling for salinity for CHEVWQMN data (N = sample size) for April 2000 through March 2007 and April 2012 through March 2015.

| | | Pre- Post | TP ugl | TN ugl | TKN ugl |
|---|---|---|---|---|---|
| TP ugl | r | −0.068** | – | | |
| | N | 3501 | | | |
| TN ugl | r | −0.141** | 0.204** | – | |
| | N | 3376 | 3376 | | |
| TKN ugl | r | −0.129** | 0.197** | 0.870** | – |
| | N | 3449 | 3449 | 3376 | |
| Fcoli 100ml | r | 0.137** | 0.031 | 0.011 | 0.001 |
| | N | 3674 | 3501 | 3376 | 3449 |

** Significant (2-tailed) at the 0.001 level

To verify that total phosphorous, total nitrogen and total Kjeldahl nitrogen were not declining prior to the adoption of urban fertilizer ordinances, trends within the pre-ordinance period were assessed using Kendall's Tau B rank correlation coefficient test. Over time, the period by year between April 2000 and March 2007, there was a statistically significant increase in total phosphorus, total nitrogen and total Kjeldahl nitrogen. Salinity decreased within the same period which would have explained the increase.

The adoption of Urban Fertilizer Ordinances by all eleven coastal city and county governments in the CHNEP study area represents tremendous cooperation. Florida House and Senate bills have been repeatedly introduced to preempt the local governments' ability to enforce ordinances more rigorous than the State model. Through the continued efforts of local elected officials and the local legislative delegations, municipal and county rights to control excess urban fertilizers remain in place so far. It appears that these actions have helped to decrease nutrient pollution in area estuaries.

**K-12 education.** Public information and education is a featured component of the CCMP, under the priority problem entitled "Stewardship Gaps." CHNEP provides a wide variety of programs through partnerships with the seven school districts, school programs such as Outdoor Classroom, non-profit organizations such as children's museums, individual teachers and individual students.

Partnerships with the seven school districts within the study area provides a cost effective way to reach the over 282,000 children enrolled in public school throughout the study area (FDOE 2013). The CHNEP works with each of its seven school districts to annually distribute their children's book, *Adventures in the Charlotte Harbor Watershed*, to every child at one grade level. For many of these children, this is the first book of their very own. Lee County School District created read-a-long videos with the author, a team of students, scientists and educators. Curriculum, coloring sheets, and a Spanish edition are a few of the additional resources CHNEP created. The program began in 2007 and is in its eighth year.

Each school year the CHNEP recognizes outstanding students whose work helps fulfill the program's CCMP at each of the five science fairs held in the study area. Two award-winning students presented their work at the 2014 Charlotte Harbor Watershed Summit poster session, along-side professional scientists.

School programs such as Outdoor Classroom and Environmental Education use CHNEP public outreach grants and micro-grants to augment their programs, providing students with creative and unique experiences that will improve their lifelong understanding of the natural world. For example, Harlem Heights Elementary Schools' Outdoor Classroom created an outdoor learning environment where fresh and salt waters mix in a mangrove forest. The Desoto County School District Outdoor Classroom used CHNEP grants to provide students the opportunity to travel to Charlotte Harbor to identify wildlife habitats, test the water quality and write essays about their experience.

Individual teachers and students have sought CHNEP micro-grants to help make their creative and unique ideas a reality. For example, CHNEP supported the first printing of student Zander Srodes' *Turtle Talk Activity Book* in 2004, the Spanish version in 2006, and other turtle talk efforts. Mr. Srodes has gone on the international stage to continue his effort to protect endangered sea turtles.

CHNEP works with nonprofit organizations such as Lakes Education/Action Drive, Imaginarium, Explorations V Children's Museum, and others to develop exhibits, events, and games to teach about a wide variety of issues important to CHNEP and its CCMP. CHNEP supported WGCU public media through its *Curious Kids* programming.

This multi-prong approach has reached or will reach nearly every child in the public schools system in one or more methods.

**Peaceful Horse Ranch land acquisition.** The CHNEP CCMP calls for doubling the amount of conservation lands in the study area from 1998 acreages by 2025. Within the last 16 years, over 90 percent of this objective has been achieved. Though the 4,414-acre Peaceful Horse Ranch is not the largest of the acquisitions, it was cited as one of the greatest CCMP implementation accomplishments. Its position at the confluence of the Peace River and Horse Creek (along with buffers for 7.6 miles of the Peace River and 5.6 miles of Horse Creek) render it an important acquisition to protect Charlotte Harbor (FDEP 2014).

In addition, the acquisition has a remarkable history that intersects with long-standing concerns by CHNEP regarding phosphate mining permitting practices. In 2010, the Board of Trustees of the Internal Improvement Trust Fund added Peaceful Horse Ranch to the Florida Forever priority list in the newly designated Climate Change Lands category. A little over one year later, the Mosaic Company purchased Peaceful Horse Ranch at a bankruptcy auction to offer as part of a legal settlement with Sierra Club, the People for the Protection of the Peace River and Manasota 88 over the South Fort Meade phosphate mining permit. However, FDEP refused to accept the property for management (Pittman 2013). The terms of the agreement said that if FDEP did not accept it, the property would go to the Conservancy of Southwest Florida. The Conservancy of Southwest Florida successfully facilitated transfer of the property to the Board of Trustees of the Internal Improvement Trust Fund with the Florida Department of Agriculture and Consumer Services (FDACS) Florida Forest Service as the management agency. Florida Commissioner of Agriculture Adam H. Putnam described the donation as important to protect "Florida's beautiful landscape and wildlife corridor, while creating new opportunities for recreational activities and agricultural operations" (FDACS 2014).

CHNEP (2011) stated that almost 14 percent of the CHNEP's landmass is under conservation management, but the Peace River basin has the least percentage under conservation management at 7 percent. The CHNEP CCMP

Vision shows acquisition of a buffer for the Peace River and its tributaries as one of the largest unserved conservation needs. The transfer of the Peaceful Horse Ranch is an important step in the assembly of the buffer. The phosphate industry's role toward this significant step is notable. The Peaceful Horse Ranch land acquisition represents the first off-site acquisition and restoration project associated with a phosphate mining permit application within the Peace River basin. Subsequent off-site protection and restoration opportunities have been incorporated into the first Peace River Basin phosphate mining permit applications as of 2014.

**Minimum Flows and Levels.** As the CHNEP was developing its first CCMP, the water management districts were developing their first adopted Minimum Flows and Levels (MFLs) to limit withdrawals that "would be significantly harmful to the water resources or ecology of the area" (State of Florida 2015). Consequently, MFL development by our water management district partners with review by the CHNEP partnership was featured within the hydrologic alteration problem area of the CCMP. Within the CHNEP watershed, MFLs have been adopted for the Peace River (Upper, Middle, and Lower), Myakka River (Upper and Lower), and the Caloosahatchee River and Estuary. In addition, MFLs have been proposed for Dona Bay/ Shakett Creek below Cow Pen Slough and adopted for lower west coast aquifers.

MFL adoption has been valuable because 1) quantitative standards provide the public with a simple tool to determine if flows are adequate or if there is a problem; 2) MFLs have been developed with the best scientific information available, peer review and public comment; and 3) failure to meet adopted MFLs has prompted projects and policies which are improving flows.

Based on the statute language, the first technical issue is the definition of "significant harm." Throughout Southwest Florida Water Management District (SWFWMD) documentation, a 15 percent change in habitat availability compared to estuarine baseline conditions has been the designated threshold for "significant harm."

South Florida Water Management District (SFWMD) documentation states that significant harm is "harm that requires multiple years for the water resource to recover" (SFWMD 2000). For the Caloosahatchee River and Estuary, analysis of valued ecosystem components and other key species requirements formed a picture of general flow needs. SFWMD then proposed MFL criteria for the Caloosahatchee River and Estuary based on maintaining the wild celery/tape grass (*Vallisneria americana*) using a hydrologic model, a salinity model, and a *Vallisneria* growth model. The MFL is exceeded when the 30-day average salinity concentration exceeds 10 parts per thousand or the daily average salinity exceeds a concentration of 20 parts per thousand at the Ft. Myers salinity station. This corresponds roughly to a mean monthly flow of 300 cfs at S-79, otherwise known as the Franklin Locks. Douglass (2014) confirms 450 cfs flow at S-79 is the lower limit for maintaining *Vallisneria* habitat and 2800 cfs is the upper limit for maintaining *Halodule wrightii* habitat in the Caloosahatchee estuary.

The Franklin Lock (S-79) was constructed on the Caloosahatchee River in 1966. The year 2014 is the first year since construction that average monthly flows fell within the envelope of 450 to 2800 cfs. (Appendix) presents average monthly flow collected at S-79 from USGS station 2292900 beginning in 1966. In 2014, the lowest mean monthly flow was 613 in May and the highest mean monthly flow was 2730 cfs in September. The low and high average monthly flows are within the salinity targets defined by SFWMD based on valued ecosystem components and other key species requirements. The dispersed water storage projects in the Kissimmee River Basin created 51,400 acre-feet of retention and storage on public and private land by the end of fiscal year 2014 contributing to this success. In addition, the Lake Okeechobee Regulation Schedule adopted in 2008 (USACOE 2008) provides policy tools to help meet MFLs.

MFLs have prompted actions by the SWFWMD. The Lake Hancock Lake Level Modification Project (construction completed in 2015) is expected to provide approximately 50 percent of the minimum flow requirements for a 20-mile portion of river, protect thousands of acres of floodplain and reduce nitrogen levels by 27 percent (SWFWMD 2007).

Adoption of MFLs has prompted projects and policies to ensure natural systems receive adequate water. They provide clear criteria that communicate to citizens, scientists and elected officials the state of hydrologic flow to the rivers and estuaries.

## Summary and Conclusions

The participants at the 2014 Watershed Summit identified 10 significant achievements of the Charlotte Harbor National Estuary Program (CHNEP) and its partners toward implementing the Comprehensive Conservation and Management Plan (CCMP.) The top four selected address one of each of the four priority problems areas of the CCMP, including water quality degradation, hydrologic alteration, fish and wildlife habitat loss, and stewardship gaps. It appears that each selected accomplishment incorporates a wide variety of related actions. The adoption of Minimum Flows and Levels (MFLs) provides citizens and resource managers with tools to evaluate hydrologic flows and select resource management policies and projects to meet the MFL targets. Urban Fertilizer Ordinances involved cooperation of all eleven coastal local governments as well as neighboring governments to apply a consistent improved treatment of residential and commercial fertilizer application. A key land acquisition project demonstrated a way to achieve CCMP goals in concert with phosphate mine permitting. Environmental education geared to K-12 not only reaches that age bracket but also their families and friends.

Environmental restoration of a seven-county CHNEP watershed requires the unified but independent actions of citizens and government. These efforts represent our "Vision in Action."

**Acknowledgements** To all the participants in the Charlotte Harbor National Estuary Program and at the 2014 Watershed Summit, thank you.

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Appendix. Mean monthly flows at S-79.

| | Month | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Year | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Oct | Nov | Dec |
| 1966 | | | | | 2,241 | 4,853 | 5,636 | 6,350 | 4,371 | 2,954 | 742 | 117 |
| 1967 | 121 | 198 | 88 | 10 | 10 | 1,300 | 2,145 | 1,714 | 798 | 1,411 | 109 | 10 |
| 1968 | 38 | 59 | 42 | 10 | 547 | 5,622 | 7,169 | 4,631 | 1,073 | 1,346 | 901 | 479 |
| 1969 | 211 | 158 | 4,558 | 2,106 | 2,634 | 5,008 | 2,145 | 4,153 | 1,465 | 6,772 | 6,869 | 4,789 |
| 1970 | 7,486 | 5,245 | 8,829 | 7,970 | 1,574 | 5,371 | 5,050 | 1,857 | 624 | 459 | 98 | 10 |
| 1971 | 25 | 61 | 56 | 10 | 113 | 274 | 993 | 1,084 | 2,411 | 1,234 | 357 | 86 |
| 1972 | 31 | 109 | 33 | 119 | 57 | 1,458 | 546 | 228 | 370 | 85 | 369 | 195 |
| 1973 | 223 | 592 | 395 | 134 | 10 | 548 | 1,645 | 2,537 | 2,603 | 762 | 29 | 104 |
| 1974 | 20 | 21 | 10 | 18 | 154 | 1,940 | 7,376 | 10,751 | 5,248 | 311 | 64 | 212 |
| 1975 | 25 | 10 | 201 | 175 | 318 | 855 | 1,745 | 705 | 1,917 | 942 | 62 | 21 |
| 1976 | 56 | 460 | 236 | 192 | 342 | 627 | 688 | 1,637 | 1,035 | 188 | 135 | 39 |
| 1977 | 527 | 133 | 92 | 135 | 782 | 773 | 767 | 1,173 | 1,550 | 105 | 90 | 747 |
| 1978 | 308 | 204 | 715 | 62 | 485 | 817 | 1,855 | 4,063 | 1,758 | 643 | 235 | 455 |
| 1979 | 4,088 | 4,172 | 3,033 | 84 | 915 | 192 | 363 | 340 | 4,408 | 5,937 | 1,217 | 1,171 |
| 1980 | 2,506 | 4,561 | 3,106 | 3,297 | 1,700 | 613 | 529 | 1,178 | 1,823 | 177 | 248 | 46 |
| 1981 | 30 | 267 | 144 | 81 | 106 | 253 | 81 | 901 | 1,238 | 4,892 | 33 | 4 |
| 1982 | 3 | 28 | 67 | 194 | 686 | 6,053 | 3,510 | 3,158 | 1,992 | 4,892 | 176 | 240 |
| 1983 | 2,060 | 10,079 | 10,321 | 8,198 | 2,473 | 2,923 | 1,331 | 2,455 | 2,925 | 2,657 | 741 | 831 |
| 1984 | 941 | 1,889 | 5,536 | 5,830 | 1,537 | 3,336 | 6,264 | 4,079 | 1,526 | 528 | 777 | 259 |
| 1985 | 323 | 48 | 294 | 458 | 480 | 983 | 1,985 | 2,376 | 3,687 | 922 | 156 | 130 |
| 1986 | 343 | 84 | 1,228 | 15 | 181 | 2,917 | 2,528 | 4,209 | 2,286 | 723 | 863 | 514 |
| 1987 | 1,659 | 937 | 1,921 | 1,909 | 1,012 | 600 | 1,412 | 1,229 | 1,442 | 2,792 | 4,488 | 980 |
| 1988 | 576 | 1,269 | 2,223 | 804 | 136 | 362 | 1,648 | 2,895 | 1,113 | 111 | 680 | 187 |
| 1989 | 199 | 351 | 531 | 722 | 46 | 700 | 1,397 | 1,491 | 1,538 | 987 | 30 | 197 |
| 1990 | 68 | 479 | 6 | 396 | 91 | 439 | 1,453 | 2,567 | 799 | 663 | 26 | 10 |
| 1991 | 1,010 | 185 | 57 | 371 | 1,426 | 1,732 | 3,989 | 3,114 | 1,653 | 1,287 | 233 | 89 |
| 1992 | 98 | 486 | 428 | 474 | 159 | 3,657 | 3,084 | 3,676 | 2,628 | 654 | 182 | 74 |
| 1993 | 2,400 | 2,891 | 1,434 | 3,268 | 178 | 946 | 763 | 1,489 | 3,778 | 2,698 | 618 | 108 |
| 1994 | 626 | 855 | 394 | 447 | 207 | 1,945 | 1,549 | 1,853 | 4,869 | 4,835 | 3,910 | 5,519 |

Appendix Continued

| Year | Month | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Oct | Nov | Dec |
| 1995 | 5,406 | 3,819 | 2,681 | 1,274 | 124 | 1,731 | 3,394 | 8,287 | 9,357 | 10,391 | 6,785 | 2,708 |
| 1996 | 2,348 | 331 | 267 | 1,017 | 696 | 4,304 | 3,813 | 1,012 | 389 | 1,037 | 24 | 272 |
| 1997 | 68 | 472 | 250 | 458 | 357 | 832 | 1,401 | 2,500 | 2,009 | 884 | 394 | 2,952 |
| 1998 | 5,632 | 8,296 | 10,156 | 6,291 | 2,095 | 477 | 821 | 3,195 | 2,759 | 1,105 | 2,578 | 296 |
| 1999 | 665 | 98 | 7 | 780 | 301 | 3,601 | 3,185 | 2,739 | 3,961 | 4,853 | 4,170 | 1,779 |
| 2000 | 809 | 17 | 342 | 1,351 | 2,914 | 494 | 992 | 486 | 1,816 | 798 | 148 | 0 |
| 2001 | 154 | 0 | 30 | 32 | 126 | 474 | 2,115 | 2,999 | 5,599 | 1,657 | 447 | 311 |
| 2002 | 490 | 454 | 693 | 245 | 431 | 3,753 | 5,441 | 2,795 | 5,024 | 1,709 | 766 | 2,816 |
| 2003 | 3,870 | 1,887 | 738 | 714 | 1,958 | 5,904 | 3,591 | 7,469 | 8,962 | 4,692 | 1,369 | 1,695 |
| 2004 | 1,651 | 1,902 | 902 | 642 | 267 | 700 | 582 | 4,040 | 5,518 | 9,356 | 4,435 | 963 |
| 2005 | 1,401 | 1,183 | 2,820 | 2,683 | 3,410 | 9,096 | 10,928 | 8,847 | 4,983 | 4,087 | 9,187 | 5,903 |
| 2006 | 769 | 1,558 | 1,146 | 1,028 | 625 | 625 | 1,039 | 2,816 | 5,495 | 71 | 17 | 265 |
| 2007 | 238 | 253 | 0 | 0 | 0 | 218 | 255 | 171 | 312 | 84 | 7 | 43 |
| 2008 | 59 | 43 | 36 | 124 | 0 | 330 | 1,516 | 5,780 | 4,744 | 1,709 | 416 | 535 |
| 2009 | 635 | 468 | 150 | 271 | 735 | 1,937 | 3,551 | 2,766 | 2,572 | 205 | 137 | 569 |
| 2010 | 462 | 542 | 2,137 | 2,824 | 4,758 | 3,352 | 4,872 | 2,503 | 1,366 | 292 | 504 | 277 |
| 2011 | 211 | 318 | 256 | 49 | 29 | 234 | 1,056 | 1,829 | 1,681 | 2,153 | 1,027 | 318 |
| 2012 | 459 | 483 | 369 | 284 | 497 | 663 | 961 | 2,546 | 3,948 | 5,047 | 1,070 | 729 |
| 2013 | 691 | 1,162 | 759 | 764 | 1,328 | 3,777 | 10,145 | 10,654 | 7,711 | 2,961 | 778 | 715 |
| 2014 | 679 | 1,099 | 1,131 | 762 | 613 | 854 | 1,567 | 2,171 | 2,730 | 1,499 | 1,086 | 1,342 |

Biological Science

# A spectral optical model and updated water clarity reporting tool for Charlotte Harbor seagrasses

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**Abstract** Seagrass communities are a critical response endpoint used by the Charlotte Harbor National Estuary Program (CHNEP) to manage estuarine water quality in southwest Florida. This paper describes advances made to quantify how primary light attenuation parameters affect the amount and quality of light reaching seagrass target depths through the development of a spectrally explicit optical model, based on partitioned absorption and scattering, and parameterized as a function of color, chlorophyll, and turbidity. The model was developed and validated using empirical data collected throughout CHNEP estuarine management areas from Dona and Roberts Bays to Estero Bay and produces estimates of both attenuation coefficients ($K_d$) and % Photosynthetically Active Radiation (%PAR) at specified depths. The calibrated model was remarkably well suited to predict water clarity throughout the estuaries and allows for the prediction of $K_d$ without reliance on field light estimates which have been shown to include a great deal of uncertainty in these shallow water estuaries. The model is suitable to replace routine field-based monitoring of light attenuation and model output was easily incorporated into existing statistical tools designed to report on annual water clarity conditions relative to a baseline period (2003-2007) when seagrasses were stable in this estuary.

**Keywords** Charlotte Harbor, light requirements, optical model, PAR, seagrass, water clarity

## Introduction and Background

The Charlotte Harbor estuary, located in southwest Florida, is shallow, subtropical, and connects to the Gulf of Mexico through several passes between barrier islands (McPherson et al. 1996). The 90,650 ha estuary has been divided into 14 segments (Figure 1) of relatively homogeneous seagrass and water quality conditions (JEI 2009); Dona and Roberts Bays (DRB), Upper Lemon Bay (ULB), Lower Lemon Bay (LLB), Tidal Myakka River (TMR), Tidal Peace River (TPR), East Wall (EW), West Wall (WW), Bokeelia (BOK), Cape Haze (CHZ), Pine Island Sound (PIS), Matlacha Pass (MP), Tidal Caloosahatchee River (TCR), San Carlos Bay (SCB) and Estero Bay (EB). Compared to others in SW Florida, the estuary has high levels of colored dissolved organic matter (CDOM) in the upper reaches due to freshwater inflows from the Myakka, Peace and Caloosahatchee Rivers. The lower portions, generally experience higher salinity and higher clarity waters resulting from more direct exchange with the Gulf of Mexico, though some segments (e.g. Lemon Bay and Estero Bay) are more lagoonal.

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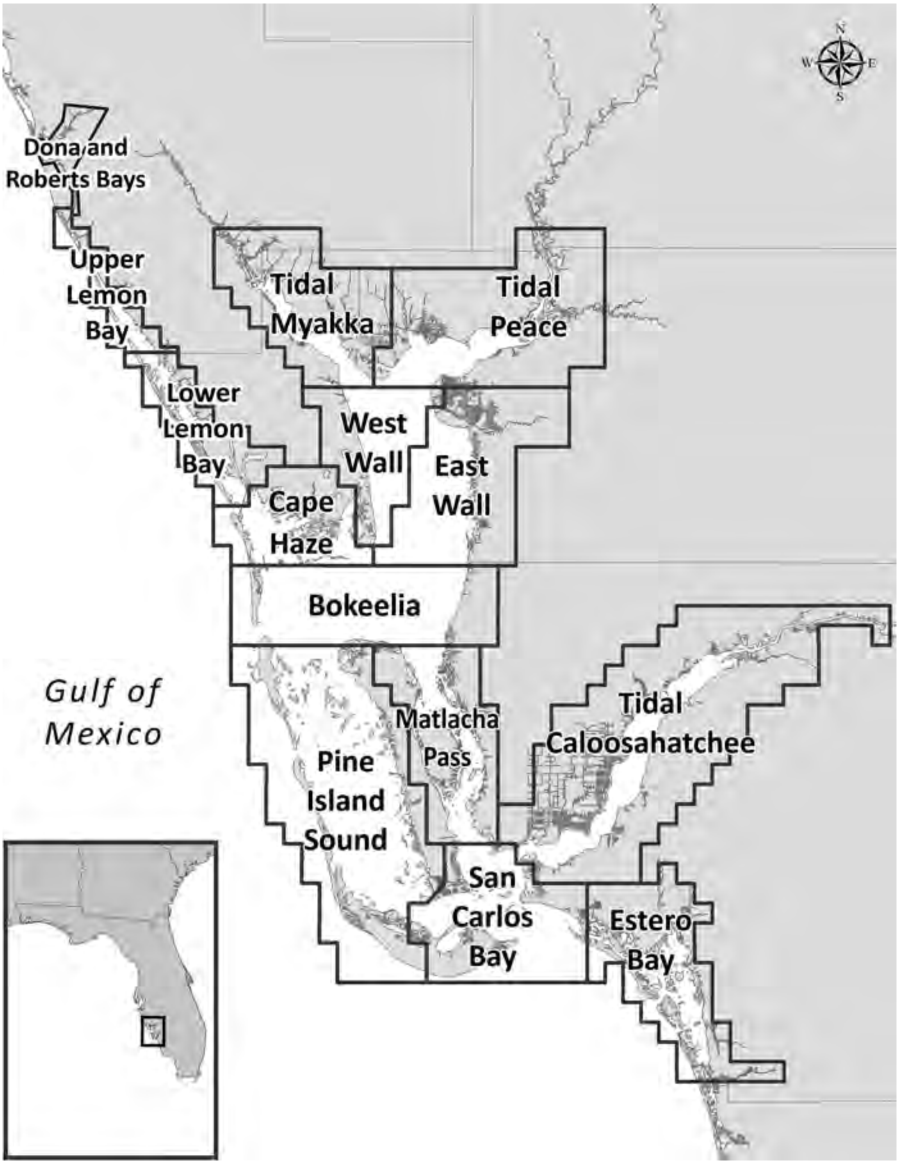


Figure 1. Charlotte Harbor study area and seagrass segments.

The Charlotte Harbor National Estuary Program (CHNEP) manages this region through the implementation of the Comprehensive Conservation and Management Plan (CHNEP 2008) using science-based decision making and management tools. Protection and restoration of seagrass acreage in this productive ecosystem represents a cornerstone of much of the management effort. While mechanistic relationships between water quality, other conditions, and the living resource requirements of seagrass are not fully understood, the preponderance of evidence suggests that water clarity is a limiting factor in determining the depth distribution, and therefore areal extent, of seagrass.

Management efforts for seagrasses established target seagrass acreages by segment (Janicki et al. 2009) as the greater of either the 1950's baseline acreage (adjusted for non-restorable areas) or the mean of all recent seagrass surveys from 1988, 1994, 1999, 2001, 2004, and 2006. Segments were designated for "Restoration" when recent acreages were less than adjusted baseline and for "Protection" when recent acreages were greater. Seagrass management targets for the highly colored tidal river segments where interpreting aerial photography is difficult (DRB, TMR, TPR, and TCR) were assigned a "Protection" or "Restoration" designation based on regulatory status and local expertise (JEI 2011).

Target depths to achieve desired seagrass cover had been previously established from observed depth distributions in either 1995-2005 or in 2003 (Corbett 2006, Corbett and Hale 2006) and ranged from 0.9-2.4 m. Photosynthetically active radiation (PAR) needed to maintain seagrasses was initially estimated at 25% of immediately subsurface values based on regional literature (Tomasko and Hall 1999, Dixon 2000, Greening and Janicki 2006). Combining target depths and 25% PAR resulted in estimated water clarity targets, measured as vertical diffuse attenuation coefficients of PAR ($K_{dPAR}$), of 0.6-1.4 $m^{-1}$. Comparing field measured $K_{dPAR}$ with $K_{dPAR}$ modeled from linear combinations of regression-derived partial attenuation coefficients (McPherson and Miller 1994, Corbett 2006), however, indicated a lack of predictive capacity in the regression relationship (Wessel and Corbett 2009). There were also notable artifacts in observed $K_{dPAR}$ time series attributed to field technique changes.

The most recent seagrass management efforts (JEI 2010, 2011) proposed water clarity targets for light-limited seagrass, on the premise that improving water clarity would result in an increase in the areal extent of seagrasses. By identifying water clarity during a reference period as a target, it was not necessary to explicitly quantify the light requirements of seagrass or the impacts of other potentially limiting factors such as salinity. Segment-specific water clarity targets were ultimately established using selected percentiles of cumulative frequency distribution curves of measured $K_{dPAR}$ during the reference period (2003 – 2007) when seagrass cover was generally acceptable and water clarity data were available. The recently established CHNEP Water Clarity Reporting Tool (JEI 2010, 2011, Wessel et al. 2013) employed a binomial test to evaluate annual distributions of measured $K_{dPAR}$ with respect to that of the reference period for progress towards clarity goals within individual segments.

An apparent optical property, $K_{dPAR}$ is a function not only of the inherent optical properties of absorption and scattering, but is also affected (both positively and negatively) by the angular distribution of light (sky and cloud conditions; sun angle). In addition, field measurements of KdPAR in shallow systems can be "contaminated" by wave climate, bottom reflectance, and other issues beyond the samplers' control.

Comparison of field $K_{dPAR}$ with modeled $K_{dPAR}$ (Dixon et al. 2010) indicated that an existing, spectrally explicit optical model had substantially improved coefficients of determination relative to field observations than did the partial attenuation coefficient approach. In addition, the spectral model approach was spatially consistent over multiple estuarine segments. Given the shallow depths, "bright" sand bottoms in much of the estuary, and difficulties in the field measurement of attenuation coefficients, Dixon suggested that spectrally explicit model estimates may more consistently represent water column attenuators of light compared to actual field measurements of light attenuation. Based on this information, the CHNEP funded the creation of an optical model in which $K_{dPAR}$ could be reproducibly computed from the monitoring parameters of color, chlorophyll *a,* and turbidity. The model was to be calibrated against only the highest quality field measurements of $K_{dPAR}$ made in multiple segments of the CHNEP study area. The calibrated model was then to be applied to the entire body of water quality monitoring data in order to 1) estimate water clarity conditions for each of the estuary segments, 2) evaluate water clarity with respect to a designated reference period judged optimal for seagrasses, and 3) assess changes in water clarity over time.

## Materials and Methods

**Field and water quality data.** Water quality data were available from 1998 through December 2011 for stratified random and fixed stations sampled by the Coastal Charlotte Harbor Monitoring Network and for DRB and ULB sampled by Sarasota County. There were 13,232 samples with complete parametric coverage of color, chlorophyll, and turbidity data. Chlorophyll data were predominantly available as chlorophyll *a*, corrected or adjusted for pheophytin. Turbidity was determined by nephelometry and methodologies were uniform across all data providers. Color values were determined either through visual determination of unfiltered samples as apparent color, or by spectrophotometric determination of absorption on filtered samples (0.45 μm) at 465 nm and reported in platinum cobalt units (PCU). Maximum water quality values recorded were 420 $\mu g\ L^{-1}$ Chlorophyll *a*, corrected for pheophytin, 68 NTU of turbidity, 375 PCU of spectrophotometric true color ($Color_{PCU\text{-}465}$), and 507 PCU of visually determined apparent color ($Color_{PCU\text{-}Vis}$).

In association with the water quality monitoring, measurements of PAR irradiance at two or more depths per station were made with quantum sensors exhibiting a spectrally flat response to photons between 400-700 nm. Techniques to collect PAR data have varied over the years. In addition to both single and paired sensors, both flat sensors ($2\pi$) measuring irradiance and spherical ($4\pi$) sensors measuring scalar irradiance have been used. Experience in SW Florida estuaries (Dixon and Kirkpatrick 1995, Dixon et al. 2010), and literature (Kirk 1994a) indicate that $K_{dPAR}$ determined with $2\pi$ sensors is approximately equal to attenuation of scalar irradiance and so no distinction between sensor type was made. When in-air readings were available, multiple readings made with a single sensor were ratio corrected for varying incident irradiance ("air correction"). Values of $K_{dPAR}$ were computed as the negative slope of *ln* (irradiance) as a function of depth.

**Model calibration data.** Field data for optical model calibration were limited to $K_{dPAR}$ with coefficients of determination ($r^2$) greater than 0.95 and less than 1.0 (i.e. more than 2 data points), for the $r^2$ associated with air-corrected $K_{dPAR}$ to be greater than or equal to the $r^2$ of uncorrected $K_{dPAR}$, and for the $K_{dPAR}$ of an ambient station to be greater than or equal to 0.20 $m^{-1}$, the approximate $K_{dPAR}$ of pure water alone. Restricting calibration data to stations with an overall measurement depth greater than 1.5 m improved agreement between modeled and observed $K_{dPAR}$

by identifying data more likely to have been collected under optically deep conditions (consistent with model assumptions). Calibration data were available in all but two segments and emphasized the estimated $K_{dPAR}$ range of less than 1.6 $m^{-1}$ needed for support of seagrasses. The two segments without data in the calibration were DRB and ULB, where measurement techniques only employed two depths and all $r^2$ were 1.0. Maximum water quality data associated with the selected field observations (n=445) were absorption coefficient at 440 nm ($a_{g440}$) of 11.8 $m^{-1}$, $Color_{PCU\text{-}Vis}$ of 150 PCU, $Color_{PCU\text{-}465}$ of 130 PCU, Chlorophyll *a* of 88.0 μg $L^{-1}$, and turbidity of 11.1 NTU and a field $K_{dPAR}$ of 2.7 $m^{-1}$.

**Optical model.** The empirical optical model to be used originated with equations presented in Kirk (1981, 1984, 1991, 1994b) in which the vertical diffuse attenuation coefficient for a given wavelength ($K_{d\lambda}$) is a function of the cosine ($\mu_0$) of the solar zenith angle, and wavelength-specific total absorption ($a_{t\lambda}$) and scattering ($b_\lambda$) coefficients.

$$K_{d\lambda} = (1/\mu_0) * [a_{t\lambda}^2 + (g_1 * \mu_0 - g_2) * a_{t\lambda} * b_\lambda]$$

The empirical model approach has been repeatedly validated and used under a wide range of water quality conditions (Kirk 1981, Kirk 1984, Gallegos et al. 1990, Kirk 1991, Gallegos 1994, Kirk 1994b, Dixon and Kirkpatrick 1999, Gallegos 2001, Gallegos 2005, Gallegos et al. 2006, Johansson 2007, Biber et al. 2008, Johansson et al. 2009, Dixon et al. 2010, Johansson 2012, Dixon 2014). Models based on the solution of radiative transfer equations (Mobley 1994) have also been used to validate the spectral empirical models (Gallegos 2001), resulting in very high and unbiased correlations between the two approaches in optically deep areas. The agreement between empirical and radiative transfer models results further implies that much of the source of scatter between observed and empirically model $K_{dPAR}$ values should be attributed to making observations under conditions not modeled (optically shallow systems, significant amounts of skylight) or to the difficulty of making $K_{dPAR}$ measurements, rather than to bias in the modeling approach. Model outputs retain the assumptions under which they were formulated, simulating $K_{d\lambda}$ under optically deep conditions (i.e. no measurable bottom reflectance) and under direct sunlight only (i.e. direct radiance from the solar disk in the “black” sky, and not including irradiance due to scattered light or skylight from the remaining portion of the hemisphere) (Kirk 1984, Gallegos 1994).

For individual water quality and $K_{dPAR}$ observations, the solar zenith angle was derived from the day of the year, solar declination, station latitude, and recorded time, and was adjusted for the time constant, the offset in minutes between the longitude of the station and the longitude of the eastern boundary of the start of the local time zone (Kirk 1994a). Air mass, the path through the atmosphere that sunlight travels for a given solar elevation relative to when the sun is directly overhead was computed from solar elevation, using an empirical adjustment for a curved rather than a plane-parallel atmosphere (Kasten and Young 1989). Similar computations were used to calculate the zenith angles for maximum solar elevation (solar noon) on individual sampling days. From the solar zenith angle in air, the in-water zenith angle and cosine ($\mu_0$) were computed with Snell’s Law and the relative index of refraction between air and water.

To obtain representative spectra of sunlight incident on the water column, the spectral distribution of extraterrestrial radiation (ASTM 2003) was adjusted for the Earth-Sun distance for the day of the year (Kirk 1994a). “Global Tilt” values (spectral radiation from solar disk plus sky diffuse and diffuse reflected from ground on south facing surface tilted 37° from horizontal, under 1.5 SA) (ASTM 2003) were geometrically adjusted to normal values, and using Beer’s Law, approximate atmospheric extinction coefficients were computed between adjusted global tilt and solar spectrum at top of atmosphere at the mean Earth-Sun distance (ASTM 2003).

Individual solar elevations, resulting air mass, and derived atmospheric extinction coefficients were used to compute the approximate spectral incident irradiance just below the water surface ($I_{0\lambda}$) for each observation. Resulting PAR (integration of $I_{0\lambda}$ between 400 and 700 nm) resulted in a seasonal range of $I_{0PAR}$ of 1800-2000 μmol $m^{-2}$ $s^{-1}$, consistent with field observations of in-air irradiance. Reductions in irradiance due to reflectance at the air-water surface, although a function of solar elevation (Austin 1974, as shown in Kirk 1994b) was considered spectrally flat and neglected.

The optical model was formulated to obtain $a_{t\lambda}$ and $b_{\lambda}$ as empirical spectral functions of the water quality monitoring parameters that represent the characteristic particulate and dissolved attenuating substances (color, chlorophyll, and turbidity). Total absorption ($a_{t\lambda}$) was partitioned into that attributable to water ($a_{w\lambda}$), chlorophyll or phytoplankton pigments ($a_{ph\lambda}$), non-chlorophyllous particulates or detritus ($a_{d\lambda}$), and dissolved color or *gelbstoff* ($a_{g\lambda}$) with a non-linear wavelength dependence of each term.

$$a_{t\lambda} = a_{w\lambda} + a_{g\lambda} + a_{ph\lambda} + a_{d\lambda}$$

Water absorption spectra were drawn from literature values (Pope and Fry 1997). Values, supplied every 2.5 nm, were splined (Matlab 6.R12) to 2 nm increments. Remaining partitioned absorption coefficients were measured on specialized samples and then quantified as a function of associated water quality values to permit the application of the optical model to existing water quality data. The model is therefore regionally specific, representing the typical CDOM, phytoplankton, and particulates present.

**Model formulations.** Spectral scans of $a_g$ were performed on samples collected from stations in TMR, TPR, EW, WW, CHZ, and BOK between April 1997 and March 1998 and from additional stations in MP, PIS, and SCB between June and November 1997 (n=137) (Dixon and Kirkpatrick 1999). Samples were filtered through 0.2 micron filters (Sterivex$^{TM}$). Absorption scans were zeroed at 700 nm where thermal artifacts are minimal and CDOM absorption expected to be absent. A negative exponential function of wavelength (Bricaud et al. 1981), $a_g$ is generally computed using both a reference value and a spectral slope determined at low wavelengths where absorption is maximized. The exponential fit to spectral $a_g$ determined in one wavelength region, however, is imperfect in other spectral regions. As a result of the typically high concentrations of CDOM in Charlotte Harbor waters, little light penetrates the water column at low wavelengths and absorption due to water alone limits penetration for longer wavelengths. Minimum $K_{d\lambda}$ values and maximum resulting %PAR penetrating the water column occurs near mid spectra. Accordingly, the wavelength of the reference $a_g$ (550 nm) and the spectral range used to derive spectral slopes ($S_g$, 500-600 nm) was selected to optimize model agreement in the region of minimum $K_{dPAR}$ and maximum PAR. Information on $a_{g440}$ was retained for comparison with other literature.

Monitoring programs differed in the technique of color analysis, resulting in two methods of computing reference $a_{g440}$ and $a_{g550}$ from reported color values. Spectrophotometric color (based on absorption at 465 nm) was back transformed from PCU to the $a_{g465}$ of platinum cobalt standards (Mote Marine Laboratory unpublished data) as:

$$a_{g465} = \mathrm{Color}_{PCU\text{-}465} / 16.02895$$

The resulting $a_{g465}$ of samples was converted to $a_{g440}$ based on the nearly linear relationship of $a_{g465}$ and $a_{g440}$ in the 1997-1998 ambient samples of:

$$a_{g440} = 1.58595463 * a_{g465}{}^{0.95895333}$$

Apparent color determined visually was converted to $a_{g440}$ based on concurrent scans with measured $a_{g440}$ and visual color determinations of the 1997-1998 samples.

$$a_{g440} = 0.02842795 * \mathrm{Color}_{PCU\text{-}Vis}{}^{1.18483956}$$

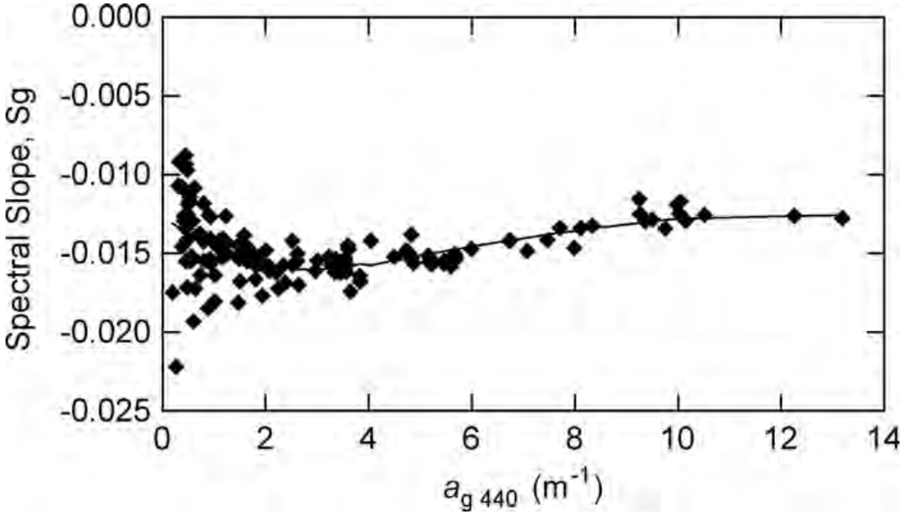


Figure 2. Spectral slope, $S_g$, 500-600 nm, as a function of observed $a_{g440}$ for 1997-1998 data at selected stations.

Subsequently, $a_{g550}$ was computed from either type of derived $a_{g440}$ using $a_{g440}$ and $a_{g550}$ relationships of the 1997-1998 samples and limiting results to 0.00 m$^{-1}$ and greater:

$$a_{g550} = \mathrm{MAX}(0{,}0.00771348 * (a_{g440})^2 + 0.16458477 * a_{g440} - 0.01053129)$$

Spectral slopes, $S_g$, of dissolved color were determined on *ln*-transformed $a_g$ data between 500-600 nm from the 1997-1998 samples and also modeled as a function of $a_{g440}$. The function was complex (Figure 2) and was fit as a three part equation to mimic observed slopes and to reasonably account for slopes at color values higher than observed in the calibration data.

$$\begin{aligned} S_g = {} & \text{If } a_{g440} <= 4.0, \\ & \quad 0.00040224 * a_{g440}{}^2 - 0.00232357 * a_{g440} - 0.01267477 \\ & \text{If } a_{g440} > 4.0 \text{ and } a_{g500} <= 13.5, \\ & \quad -0.00002682 * a_{g440}{}^2 + 0.00087068 * a_{g440} - 0.01882298 \\ & \text{If } a_{g440} > 13.5 \\ & \quad -0.01177 \end{aligned}$$

Relationships between $a_{g440}$ and $S_g$ were determined from samples with a maximum $a_{g440}$ of 13.1 m$^{-1}$ (400 PCU for Color$_{\text{PCU-Vis}}$, and 260 PCU for Color$_{\text{PCU-465}}$), while samples to be modeled for status evaluations were as high as 33.7 m$^{-1}$ (507 PCU for Color$_{\text{PCU-Vis}}$, and 375 PCU for Color$_{\text{PCU-465}}$). Slopes for samples with color values outside the available calibration range were limited to that observed for the highest calibration samples. Slopes were somewhat smaller than other literature references due to the longer wavelengths used for slope determination but comparable to those noted for similar highly colored waters (Gallegos 2005). The range in slopes at low $a_{g440}$ values was likely indicative of CDOM from multiple sources. Other numeric models of $a_g$ were also explored (Twardowski et al. 2004) but offered no improvements in fit across larger wavelength ranges. Absorption due to color, $a_g$, was then computed as:

$$a_{g\lambda} = a_{g550} * e^{[Sg * (\lambda - 550)]}$$

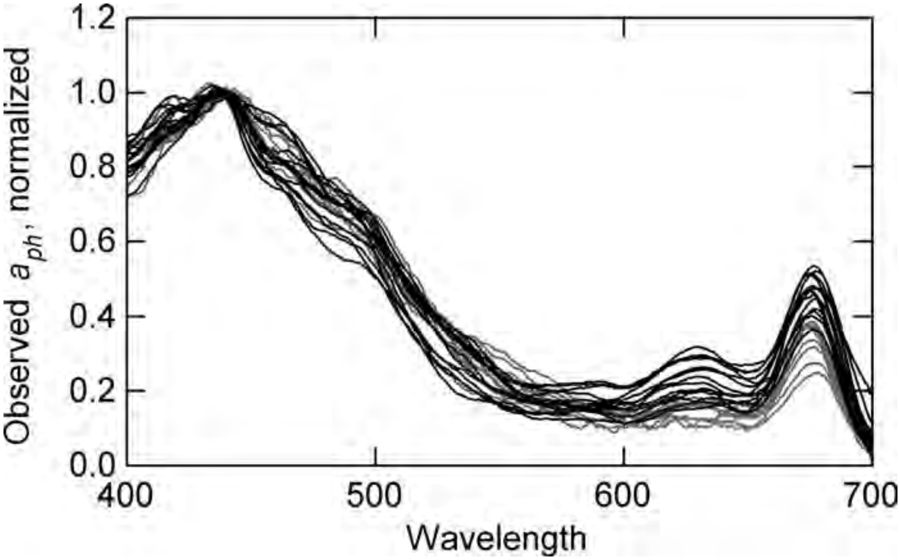


Figure 3. Chlorophyll absorption spectra, $a_{ph}$, normalized to $a_{ph440}$ indicating remaining seasonal variation; May-grey, August-black.

Absorption coefficients due to chlorophyll pigments and detritus were determined on the 1997-1998 samples by the method of Kishino (1985), Butler (1962), and Cleveland and Weidemann (1993). Samples ($n=129$) were filtered through glass fiber filters (GF/F, 0.7 μm), scanned for absorbance relative to a clean filter, extracted with methanol to remove pigments, and rescanned. Absorption coefficients were computed from optical density by multiplying by 2.303 and dividing by path length (based on filter clearance area and volume filtered). Path amplification (*Beta* factors; representing the additional absorption that occurs due to the increased path length of light and opportunities for absorption as light transits the filter) were set at 1.0 until model calibration. Chlorophyll absorption was computed as the difference between the pre- and post-solvent extraction scans, while detrital absorption was that of the depigmented scan.

Typically, available spectral absorption profiles of chlorophyll are normalized to absorption at a given wavelength (such as 440 nm), and a single normalized absorption spectra computed. The mean spectra is then scaled by the chlorophyll-specific absorption at 440 nm ($a^{*}_{ph440}$), itself computed via regression of $a_{ph440}$ as a function of chlorophyll content determined either spectrophotometrically or fluorometrically (Arar and Collins 1997, APHA 2005). Collectively, however, normalized spectra displayed considerable seasonal variation (Figure 3) related to chlorophyll concentration, likely attributable to varying phytoplankton species. Accordingly, chlorophyll absorption for each wavelength was individually modeled as a series of power relationships of chlorophyll (Figure 4). Coefficients of determination, $r^2$, were generally 0.9 or above, with the form as:

$$a_{ph} = C * Chl^{B}$$

The modeled chlorophyll absorption, although less variable than that of the original data (Figure 5), did maintain some portion of the seasonal variation in normalized absorption profiles with fits substantially improved in the low absorption region of >500 nm. The formulae for $a_{ph}$ were developed with chlorophyll values as high as 143 μg $L^{-1}$, relative to a maximum of 420 μg $L^{-1}$ in the data to be modeled. Power relationships prevented sample chlorophyll values greater than calibration data from resulting in lower computed absorption. Similar to $a_w$ values, C and B coefficients were held as tabular data within the model.

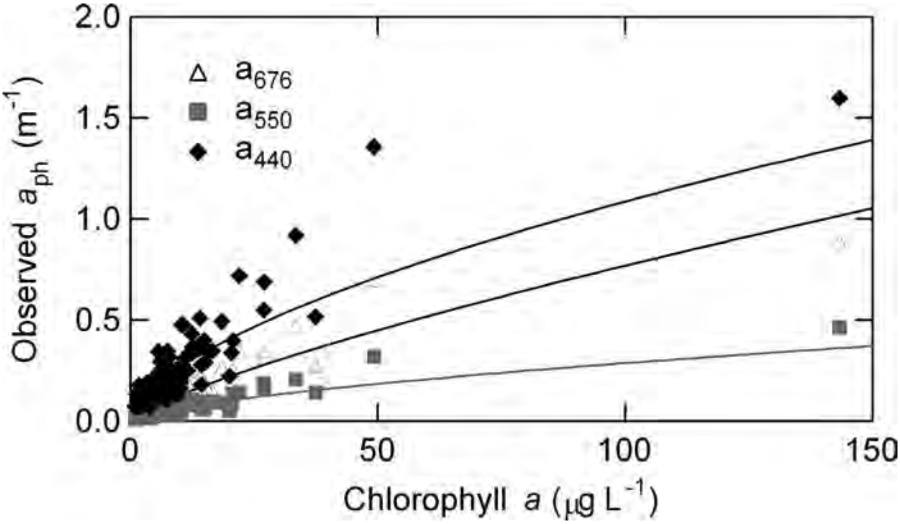


Figure 4. Selected wavelengths of the direct model of chlorophyll absorption, $a_{ph}$; 440, 550, and 676 nm.

Spectral absorption coefficients due to detritus ($a_d$) were obtained from the post-solvent extraction scans of particulates described above. Detrital absorption includes not only that due to suspended mineral and organic detrital particles, but also that due to the remaining structural components of de-pigmented phytoplankton. Detrital absorption is well represented by a negative exponential similar to that of dissolved color (Bricaud et al. 1981). Continuing to maximize model

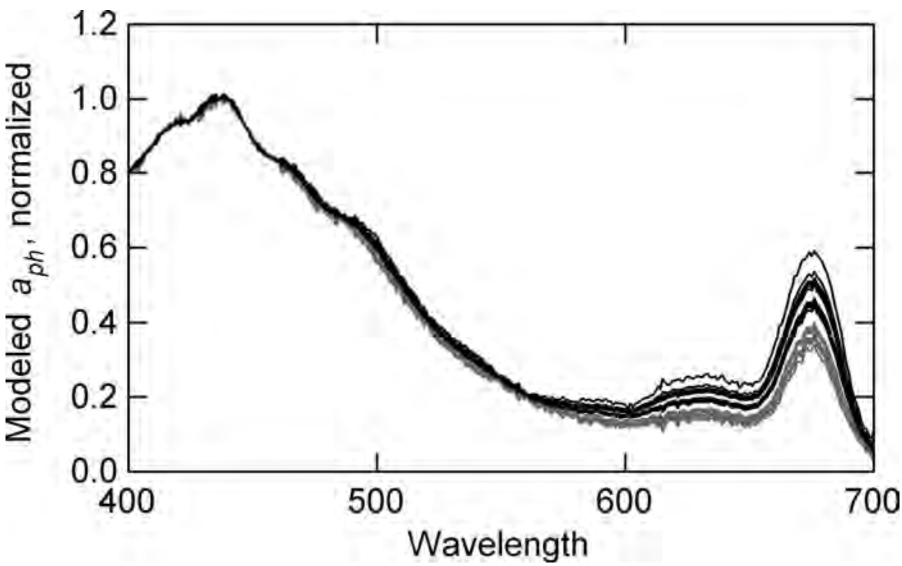


Figure 5. Modeled chlorophyll absorption, $a_{ph}$, normalized to $a_{ph440}$; May-grey, August-black.

accuracy in the region of low overall attenuation, spectral slopes were determined from 500-600 nm, using a reference wavelength of 550 nm:

$$a_d = a_{d550} * e^{[Sd * (\lambda - 550)]}$$

Reference $a_{d550}$ and spectral slopes were computed as the optimal function of available water quality parameters and were significant functions of both turbidity and dissolved color, $a_{g440}$, again limiting $a_{d550}$ to 0.0 m$^{-1}$ or greater.

$$a_{d550} = \mathrm{MAX}(0,\text{-}0.037345 + 0.026239 * \mathrm{Turbidity} + 0.012644 * a_{g440})$$

$$S_d = \text{-}0.00000419 * (a_{g440})^2 + 0.00024174 * a_{g440}\ \text{-}0.01120509$$

Spectral slopes, $S_d$, exhibited a shallow slope with $a_{g440}$ but were not a significant function of turbidity. Maximum turbidity data and $a_{g440}$ data used to develop the empirical formula of $a_d$ were 16.9 NTU and 10.5 m$^{-1}$, respectively, relative to 68 NTU and 33.7 m$^{-1}$ in the data set to be modeled. The color dependence of both $a_{d550}$ and detrital spectral slope, $S_d$, indicated that precipitated humic and fulvic acids (components of dissolved color) likely form the basis of much of the turbidity and particulates in the Charlotte Harbor system.

Particulate scattering is reported to be best described as a direct function of turbidity (Morel and Gentili 1991, Gallegos et al. 2008) with an inverse wavelength dependent term and scattering at a reference wavelength approximately equal to 0.92 to 1.1 times turbidity (Kirk 1994a). Exponents to the normalized wavelength term (Morel 1973, DiToro 1978, Gallegos et al. 2009) range between 0.8 and 1.1, with an occasional dependence on color. Scattering due to water alone (Buiteveld et al. 1994) was 300-7000 times smaller than that estimated from particulates and was subsequently ignored. Results from the Charlotte Harbor model were not strongly dependent on the precise scattering formula, but gave marginally better agreements using values of:

$$b_\lambda = 1.1 * \mathrm{Turbidity} * (555/\lambda)^{0.8}$$

For model calibration, $a_t$ and $b$ were computed from the water quality data as described above, $\mu_0$ was computed for the time of sampling, and wavelength-specific $K_{d\lambda}$ computed using the optical model formulation. Spectral incident irradiance ($I_{0\lambda}$) at the time of collection was then attenuated at each wavelength, using the appropriate $K_{d\lambda}$, to compute $I_{z\lambda}$, at both the minimum ($z_1$) and the maximum ($z_2$) depths at which PAR measurements were made.

$$I_{z\lambda} = I_{0\lambda} * e^{(\text{-}Kd\lambda * z)}$$

The resulting modeled irradiance at $z_1$ and $z_2$ was then integrated from 400-700 nm to simulate the PAR readings obtained by field instrumentation. With strong spectral structure in $K_{d\lambda}$, declines of 10-20% in $K_{dPAR}$ can be expected with increasing depth due to the progressive removal of the more strongly attenuated wavelengths. Modeling $K_{dPAR}$ between the depths actually used for field observations was expected to provide a more appropriate comparison and also emphasizes that modeled $K_{dPAR}$ is strictly applicable only to the depths between which it was derived. Modeled $K_{dPAR}$ was computed as

$$\text{Modeled } K_{dPAR} = \text{-}\,ln(\Sigma\,[I_{z2\lambda}] / \Sigma\,[I_{z1\lambda}]) / (z_2 - z_1)$$

Optical model calibration consisted of iteratively varying model parameters and maximizing the agreement between modeled and field-observed $K_{dPAR}$. Comparisons were conducted using a Line of Organic Correlation (LOC) (Helsel and Hirsch 2002) in which potential errors are assumed in both the *x*- and *y*-dimensions. A desired slope for the relationship would be between 0.8-1.2, with an intercept within $\pm 0.20$ m$^{-1}$, the approximate theoretical $K_{dPAR}$ of pure water alone between 0.0 and 0.5 m. In addition, residuals were examined to minimize distribution with respect to water quality variables.

Modeled results proved relatively insensitive to the range of literature values for coefficients $g_1$ and $g_2$ and so values of 0.473 and 0.218, empirically determined for the midpoint of the euphotic zone (Kirk 1994a), were used. Other model parameters to be adjusted during calibration were path amplification factors applied to the absorption due to chlorophyll and detritus. The $a_d$ and $a_{ph}$ quantities measured by the filter pad method are considered to be biased high relative to the same materials in suspension as result of increased scatter and additional opportunities for absorption as light transits the filter pad (Bricaud and Stramski 1990, Nelson and Robertson 1993). Factors can be either a fixed or absorption-dependent quantity, can vary by particle size or phytoplankton species, and can range by a factor of 6 or more (Mueller et al. 2003), but are generally used to reduce computed partial absorptions for materials in suspension.

As no direct measurements of suspensions or path amplification were available for Charlotte Harbor particulates, factors were adjusted to optimize modeled $K_{dPAR}$ results with respect to field observations and to remove parameter-dependent distributions in residuals. Initial residuals displayed strong positive correlations (model underestimation) with both turbidity and chlorophyll concentrations. Rather than reducing computed $a_d$ and $a_{ph}$ by a *Beta* factor, enhancement factors of 1.6 and 1.8 for $a_d$ and $a_{ph}$, respectively, were required to minimize residuals across the range of available water quality data. Absorption dependent factors (Mueller et al. 2003) did not substantively improve model fit relative to fixed factors. Absorption due to dissolved color is not typically accorded a *Beta* adjustment or enhancement as $a_g$ is measured directly on filtered samples. Model residuals with respect to $a_{g440}$ computed from color analyses, however, were similarly distributed until an enhancement factor of 1.5 was applied to $a_{g440}$.

Although the actual times of field measurements and resulting sun angles were used for calibration, model application for clarity trend assessment was conducted at local noon on the date of sample collection and $K_{dPAR}$ was modeled between the surface and the segment-specific seagrass target depths. The use of the standardized conditions and local noon produced attenuation coefficients that were independent of station order, could be considered a daily maxima for the given water quality parameters and direct sunlight assumptions, and were unaffected by time of day, cloud cover, segment or measurement depth, or bottom type, responding only to water column attenuators.

**Model application.** Following optical model calibration, modeled $K_{dPAR}$ was computed from all available water quality data, using segment-specific seagrass target depths as $z_2$, and with $z_1$ set to 0.0 m. The solar elevation at local noon was used for each date, minimizing $K_{dPAR}$ for the given date and water quality, and removing time biases in sampling. Model results provided $K_{dPAR}$ values independent of weather (wind, clouds), overall water depth, and bottom type.

**Water Clarity Reporting Tool.** The application of the Water Clarity Reporting Tool (Wessel et al. 2013) was identical in approach to that earlier applied to *field* measurements of $K_{dPAR}$, but in this updated instance was applied to *modeled* $K_{dPAR}$ values. Data were limited to $K_{dPAR}$ computed from randomized stations, sampled as part of the Coastal Charlotte Harbor Monitoring Network, and the randomized stations of DRB and ULB. The density of data was reviewed to avoid bias from a changing number of stations or missing sampling periods. The TCR segment changed from five monthly stations to three during 2006, but the three samples continued to extend over the entire segment. There were very few data from MP in 2003 and so the reference period for this segment was 2004-2007. Appropriate application of the Reporting Tool in the future depends on maintaining a similar density, frequency, and seasonal timing of sampling efforts.

The distribution of $K_{dPAR}$ during the reference period was considered to be most representative of the target light requirements of seagrasses in each segment. Selected percentiles ($30^{th}$, $70^{th}$) from the reference distribution were chosen as benchmark points. As described in Wessel et al. (2013), the

Table 1. Water clarity scores, categories, and colors: Declining (D), Caution (C), Stable (S) and Improving (I) categories with color codes of dark grey (red), light grey (yellow), and white (green).

| Score | Protection | Restoration |
|---|---|---|
| -2 | C, D* | D |
| -1 | C | C |
| 0 | S | C |
| 1 | S | C |
| 2 | S | I |

binomial test (Wackerly et al. 1996) was used to establish a scoring method to evaluate each year's water clarity data relative to the benchmark points. If more than 30% of the $K_{dPAR}$ measurements were significantly below the 30% benchmark (alpha=0.05), then the water clarity was considered to be improving and was assigned a value of +1. If less than 30% of the values were significantly below the 30% benchmark, then the water clarity was considered to be degrading and was assigned a value of -1. Non-significant differences were assigned a 0. Similar scores were assigned based on the comparison of the 70$^{th}$ percentiles and the scores summed for each year-segment combination. Combined scores could range from -2 to 2. Scores were computed provided data were available in approximately nine of twelve months. The score from an initial monitoring year of a segment may be biased by an incomplete year of water quality data.

Using the computed scores, categories were assigned to call attention to changing water clarity. Emphasis varied based on whether a segment had been designated as a "Protection" or a "Restoration" segment, recognizing that protection of water clarity required a "hold the line" strategy to maintain ambient conditions, while restoration required an improvement in water clarity. Stability in scores relative to the benchmark period is considered sufficient for the protection targets but not for the restoration targets. The grading system employed (Table 1) assigned both category and color codes.

## Results

Overall model calibration of data from 12 segments, evaluated as a LOC between modeled and the highest quality, field-observed $K_{dPAR}$, was excellent overall, exhibiting a slope of 0.982 and an intercept of -0.002 m$^{-1}$ (Figure 6). The median residual was 0.015 m$^{-1}$, and the median root mean square error was 0.156 m$^{-1}$. Over 60% of residuals fell within 0.00±0.20 m$^{-1}$. The optical model was mechanistically satisfying in that similar attenuation results from similar water quality values in all segments. Not all segments contained sufficient data for a segment-specific assessment of model performance as *n* by segment ranged between 10 and 79, but the median residuals by segment all ranged within 0.000 ± 0.221 m$^{-1}$, with most between 0.000 ± 0.100 m$^{-1}$. The two largest median

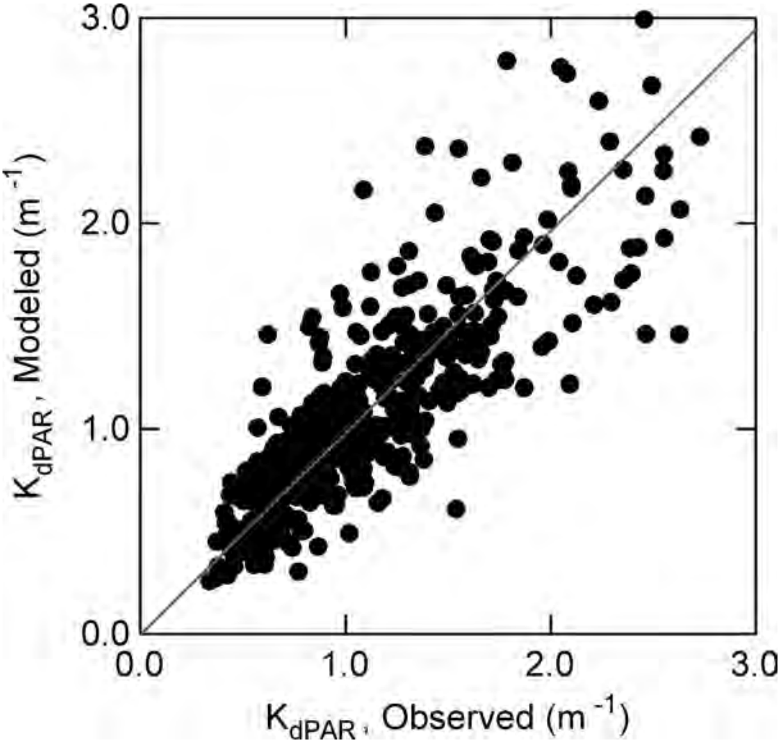


Figure 6. Calibration data of modeled $K_{dPAR}$ as a function of observed (LOC in grey). All segments with the exception of DRB and ULB.

residuals were for the segments with the highest (BOK) and one of the lowest (TPR) water clarities implying that any biases were not due to overall model performance or to the methods of computing partial attenuation coefficients from water quality variables. An independent test of model performance using data from two segments (DRB and ULB) that had not been included in the calibration demonstrated that the optical model was a robust tool with broad regional applicability. The model provides a method of computing water clarity from analyses performed in a controlled laboratory environment, and removes the many uncertainties of field $K_{dPAR}$ measurements, as well as eliminating biases due to the measurement depth dependence of $K_{dPAR}$, sampling time, varying cloud cover, and bottom type.

For DRB and ULB segments, $r^2$ criteria were not available to identify the highest quality field $K_{dPAR}$ but maximum observation depths of 0.7 m or greater were selected. As a result, distribution of data for DRB and ULB was larger than that used for model calibration (Figures 7 and 8) and some observations may have been censored if more than two depth observations been available. The LOC of the DRB and ULB data ($n$=1250), which were not used in model calibration, resulted in a slope of 1.006, an intercept of 0.217 m$^{-1}$, and median residual of -0.233 m$^{-1}$, indicating a robustness and broad regional

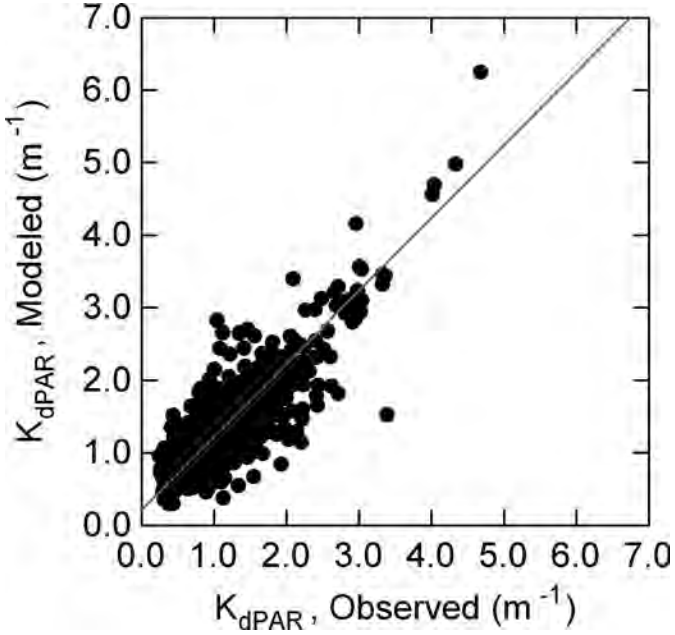


Figure 7. Modeled $K_{dPAR}$ as a function of observed (LOC in grey) for DRB and ULB.

applicability of the empirical optical model. The overestimation of $K_{dPAR}$ in DRB and ULB was primarily associated with higher turbidity levels. These segments are near inlets from the Gulf of Mexico and have periodic, light colored, mineral turbidity associated with coastal wave energy (personal observation). The instances of dominantly mineral turbidity are poorly represented in the data used to derive $a_d$ and would result in a computed $a_d$

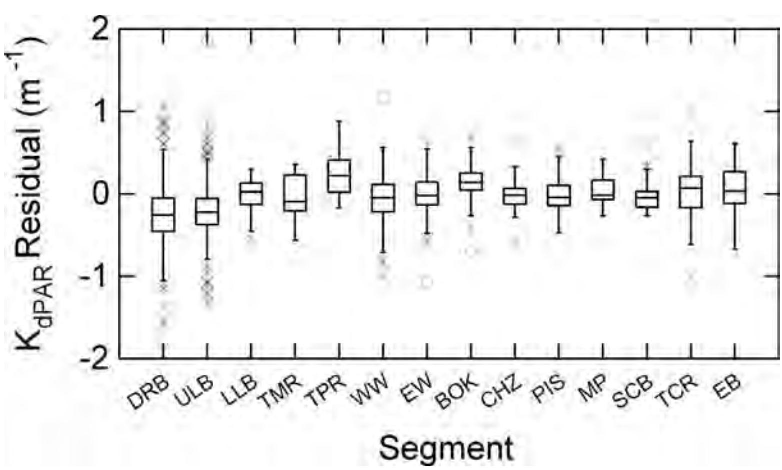


Figure 8. Residuals of $K_{dPAR}$ by segment.

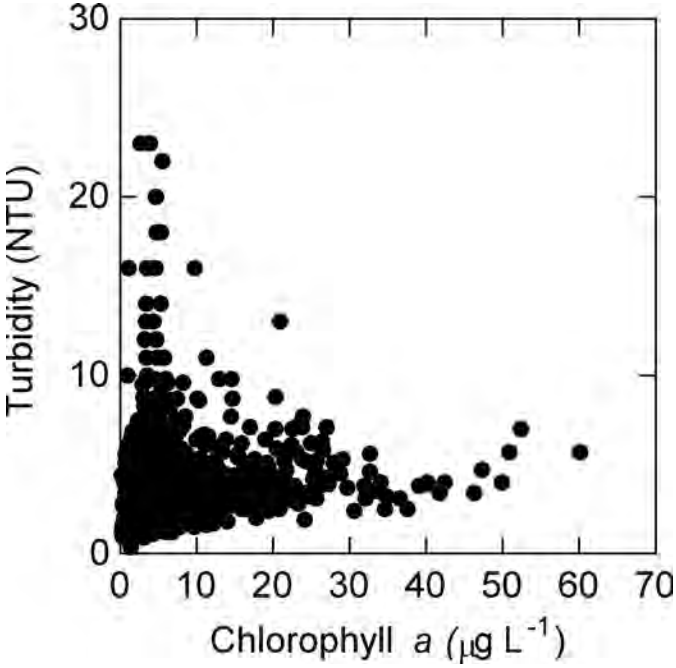


Figure 9. Distribution of turbidity with respect to chlorophyll for DRB and ULB.

biased high (and a model overestimate) relative to the more absorptive, organic particulates of the remaining segments against which model parameters were calibrated. This explanation is further supported by the distribution of elevated turbidity with respect to low chlorophyll concentrations (Figure 9).

Modeled $K_{dPAR}$ values were compiled for the reference period by segment. The benchmark 30$^{th}$ and 70$^{th}$ percentiles were computed (Table 2) and illustrated as a function of target depths (Figure 10). The values of $K_{dPAR}$ as a function of target depth indicated a coherent progression of declining water clarity with declining target depths for all segments of Charlotte Harbor for which high confidence target depths were available. The more riverine segments with poorly defined target depths due to low water clarity exhibited a much larger variation in $K_{dPAR}$ percentiles. Seagrass in these segments typically appears in the more saline regions within the segment, and so the overall segment water clarity, determined from stratified random stations, is likely lower than for the comparatively small region of remaining seagrass.

Annual distributions and percentiles for all years and segments with data were compared to the reference percentiles and scored for significant differences. Annual scores of "Protection" and "Restoration" segments appear in Tables 3 and 4. In addition to segment-specific performance, a number of region-wide trends were evident in the time series of scores. The years of 2001-2003 were often relatively decreased in water clarity, while 2007 was "Stable" or "Improving" in all segments. More recently, clarity declined in many segments in 2010. In 2011,

Table 2. Selected percentiles of the frequency distribution of modeled $K_{dPAR}$ from 2003-2007, target depths, and median %PAR, by seagrass segment.

| Segment | Protection (P) Restoration (R) | $30^{th}$ %-ile ($m^{-1}$) | 70th %-ile ($m^{-1}$) | Target Depth (m) | Median %PAR |
|---|---|---|---|---|---|
| DRB | R | 0.90 | 1.35 | 1.0 | 36 |
| ULB | P | 0.85 | 1.17 | 2.0 | 14 |
| LLB | R | 0.75 | 1.13 | 2.0 | 16 |
| TMR | P | 1.47 | 2.46 | 0.9 | 18 |
| TPR | R | 1.39 | 2.41 | 1.0 | 15 |
| WW | R | 0.87 | 1.40 | 1.4 | 21 |
| EW | R | 0.71 | 1.17 | 1.4 | 28 |
| CHZ | P | 0.74 | 1.06 | 1.9 | 19 |
| BOK | P | 0.56 | 0.88 | 2.4 | 19 |
| PIS | R | 0.71 | 0.98 | 2.2 | 16 |
| MP | R | 0.62 | 0.92 | 2.0 | 20 |
| SCB | P | 0.57 | 0.91 | 2.2 | 22 |
| TCR | R | 1.68 | 2.92 | 1.0 | 9 |
| EB | R | 0.96 | 1.39 | 1.6 | 18 |

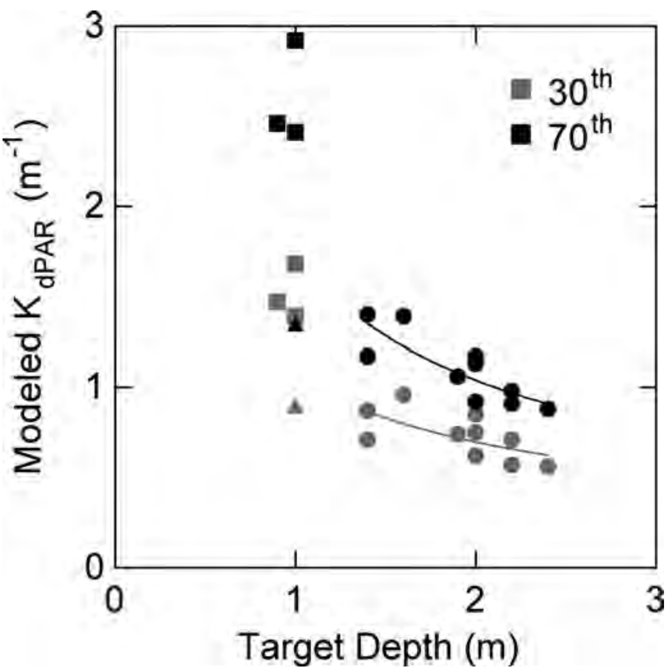


Figure 10. Distribution of $30^{th}$ (grey) and $70^{th}$ (black) percentiles of modeled $K_{dPAR}$ with a smooth applied to non-riverine segments (circles) and riverine segments (DRB – triangles; TMR, TPR, TCR – squares).

Table 3. Annual scores for segments with "Protection" designation.

| | PROTECTION | | | | | |
|---|---|---|---|---|---|---|
| Year | ULB | TMR | BOK | CHZ | PIS | SCB |
| 1998 | -2 | | | | | |
| 1999 | -2 | | | | | |
| 2000 | -1 | | | | | |
| 2001 | 0 | -2 | | -2 | | |
| 2002 | 0 | 0 | -2 | -2 | | 2 |
| 2003 | -1 | -2 | -2 | -2 | | 1 |
| 2004 | 0 | 0 | 0 | 0 | 0 | 2 |
| 2005 | 0 | -1 | -1 | 0 | -1 | -1 |
| 2006 | 1 | 1 | 0 | 0 | -1 | -2 |
| 2007 | 1 | 2 | 2 | 1 | 2 | 2 |
| 2008 | 0 | 0 | 2 | -1 | 2 | 0 |
| 2009 | 1 | 0 | 1 | -1 | 0 | -1 |
| 2010 | -1 | 0 | 0 | -1 | 0 | -1 |
| 2011 | -1 | -1 | 1 | -2 | 0 | -1 |

Table 4. Annual scores for segments with "Restoration" designation.

| | RESTORATION | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Year | DRB | LLB | TPR | WW | EW | MP | TCR | EB |
| 1998 | | | | | | | | |
| 1999 | | | | | | | | |
| 2000 | | | | | | | | |
| 2001 | | -2 | -2 | -2 | -2 | | | |
| 2002 | | -1 | -1 | -1 | -2 | | -2 | |
| 2003 | -2 | -1 | -2 | -2 | -2 | | -2 | |
| 2004 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | -1 |
| 2005 | 0 | 0 | -1 | -2 | -2 | -2 | -2 | 0 |
| 2006 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 1 |
| 2007 | 2 | 1 | 2 | 2 | 2 | 2 | 2 | 2 |
| 2008 | 1 | -1 | 0 | 1 | 1 | 0 | 1 | -1 |
| 2009 | 0 | | -1 | -1 | -1 | 0 | 0 | 0 |
| 2010 | -2 | | -1 | 1 | 1 | -2 | 1 | -2 |
| 2011 | -1 | | 0 | -1 | 0 | -2 | 1 | -1 |

there were some improvements over 2010. The segments of ULB, TMR, BOK, and PIS were the segments which recorded the most stable or improving categories over the period of record, consistent with the large areas of seagrasses present in these regions.

The model also provides estimates of %PAR at target depths (Table 2, above) but %PAR values are highly dependent on targets depths (some of which have known uncertainties), and should be interpreted and used with caution. Model assumptions of an optically deep system are implicit and seagrass at target depths undoubtedly receive additional %PAR from bottom reflectance. Due to the exponential relationship between %PAR and $K_{dPAR}$, transformations between summary $K_{dPAR}$ statistics and %PAR are inappropriate and can be 70-80% of summary statistics based on individual %PAR data. In addition, due the strong spectral distribution of $K_{d\lambda}$, values of $K_{dPAR}$ only pertain to the depths for which they were modeled; in this case seagrass target depths. For the same water quality, $K_{dPAR}$ can vary by 10-20% when modeled over the range of target depths. Fortunately, model application is straightforward, and target depths readily modified for investigations of potential management actions.

## Discussion and Conclusions

Modeled and measured $K_{dPAR}$ measurements often show extensive scatter in their relationship and a discussion of the contributing causes will enhance the appreciation of model calibration results. Most importantly, model formulation is based on simulating optically deep conditions illuminated by direct sunlight only, such that results should be a function of water column attenuators and sun angle alone. However, when PAR measurements at depth are enhanced by bottom reflection (especially noted for scalar PAR sensors), observed $K_{dPAR}$ can be decreased relative to water column attenuation alone (negative residuals). As an apparent optical property, $K_{dPAR}$ is also influenced by the angular distribution of light in addition to the inherent optical properties of absorption and scattering. In shallow, clear waters, bottom reflectance can substantially increase the angular distribution, potentially increasing measured $K_{dPAR}$ (positive residuals). For both low and high solar elevation, if skylight is the dominant contribution to irradiance (early or late day, or overcast), modeled $\mu_0$ based on solar elevation can fail to capture the ambient subsurface angular distribution of light, resulting in either positive or negative residuals. The scatter in individual data points in model calibration represents the potential range of these collective influences.

Much of the Charlotte Harbor estuary is quite shallow, and while model calibration data were censored to obtain field data that were minimally impacted by bottom reflectance, some influence undoubtedly remained. Field data also obviously contain the combined effects of wave climate, skylight, and cloud cover. By calibrating to field data, the optical model implicitly incorporates the mean effects of these factors.

This paper advances previous efforts in deriving empirically based model estimates through the inclusion of techniques for computing $a_{ph\lambda}$ as a function of chlorophyll concentration, using 550 nm as reference wavelengths to compute $a_{g\lambda}$ and $a_{d\lambda}$, and the application of enhancement factors to $a_{ph}$, $a_g$ and $a_d$ in order to remove the dependence of $K_{dPAR}$ residuals on chlorophyll, color, and turbidity concentrations. Computing $a_{ph\lambda}$ as a series of wavelength-specific power relationships permitted the capture of seasonal variations in accessory pigments that were associated with overall chlorophyll concentrations and possibly with seasonally dominant phytoplankton species. The conventional approach using $a^*_{ph440}$ and a single spectra did not adequately simulate these variations. The use of a mid-spectral reference wavelength for $a_{g\lambda}$ and $a_{d\lambda}$ as used in this work was important for highly absorbing waters with a strong spectral dependence and minimum overall $K_{d\lambda}$ in the mid-spectral region.

Improved precision in the partial absorption coefficients in the mid-spectral region resulted in the largest increase in precision of $I_{z\lambda}$, and consequently the largest increase in precision of $K_{dPAR}$.

The application of enhancement factors to $a_{ph}$, $a_g$ and $a_d$ clearly improved model performance across the range of relevant water clarity and water quality parameters. A similar optical model calibrated against field $K_{dPAR}$ (Biber et al. 2008) did not apply enhancements and resulted in model underestimation of 23% for $K_{dPAR}$ values greater than 1.5 $m^{-1}$. The Charlotte Harbor model, however, was calibrated in the range of 0-3 $m^{-1}$, useful in this highly colored system, with similar overall accuracies comparable to those described by Biber et al. (2008).

Work in the St. Johns River, another highly colored system (Gallegos 2005), quantified inherent optical properties of absorption and scattering, and partial absorption coefficients as functions of water quality variables, chlorophyll, color, and total suspended solids (TSS). Gallegos' work identified that fine particulates (0.22 – 0.7 μm), roughly proportional to CDOM and likely organic in nature, contributed substantial absorption and scattering but were not included in the direct measurements of $a_g$ or $a_d$ or the resulting relationships with water quality variables. Failing to model the absorptive properties of these fine particulates would result in model underestimates of $K_{dPAR}$ relative to field measured quantities. The Charlotte Harbor system undoubtedly has similar fine organic particulates and the enhancement factors applied to $a_d$ and $a_g$ in this work could be considered to account for these absorptive properties in some measure. The effects of fine particulates were included in the estimates of scattering in the present work (determined as a function of turbidity on unfiltered samples), although variations were observed that were attributed to particulate type (mineral versus organic). There were no independent measures of scattering with which to additionally refine scattering formulations but the model results in this absorption dominated system were relatively insensitive to a variation in scattering of $\pm$15%.

The relationship of $a_{ph\lambda}$ with corrected chlorophyll concentrations, however, should adequately represent both the contributions of typical accessory pigments, as well as the packaging and absorption cross sections associated with typical phytoplankton species without an additional enhancement factor. Review of $a_{ph}$ derivations, indicated modeled $a_{ph}$ less than observed in much of the 20-140 μg $L^{-1}$ chlorophyll range. A repeat of model calibration with no $a_{ph}$ enhancement, indicated an only slightly lower quality model (slope=1.044, intercept of -0.010 $m^{-1}$) for the 0-88 μg $L^{-1}$ calibration range. From this result, it is apparent that the enhancement factor for chlorophyll affects $K_{dPAR}$ only minimally at low chlorophyll in this high color system, but did improve model agreements at high chlorophyll. The needed enhancement may be attributed to smaller bloom-forming species with higher effective absorption cross sections relative to bulk chlorophyll content.

Although provisions were made to reasonably accommodate water quality data outside of the range used for either determination of partial absorption coefficients or calibration, the model should be considered an approximation above $K_{dPAR}$ of ~3 $m^{-1}$. Fortunately, earlier estimates of seagrass requirements as well as the percentiles derived from the modeled $K_{dPAR}$ of this work fall within this range. The lack of a full range of water quality data in the calibration data set was not as problematic as may initially have been perceived.

The use of $K_{dPAR}$, rather than %PAR, to assess water clarity reduced the sensitivity to selected and possibly uncertain target depths, and the Water Clarity Reporting Tool remains a valid assessment of changes in water clarity over time, even if target depths are not optimum. The optical model and the Water Clarity Reporting Tool combined provide an assessment of changes in water clarity resulting from changes in water column attenuators alone, and a mechanism for the convenient display of results to the general public. Comparisons of future scores to the reference period remain valid only as long as the design frequency and spatial density of the monitoring program remains essentially the same as during which the reference values were developed.

The use of the optical model is not limited to the periodic assessment of water clarity. A versatile tool, it can be applied to other sampling efforts within the Charlotte Harbor region. A highly relevant investigation would be the site-specific modeling with water quality data at the depths of seagrass transects to correlate modeled %PAR and the associated seagrass performance at multi-annual time scales. Application of a similar scoring technique to salinity may help determine the extent to which changes in annual scores are correlated with climatic variations. The Optical Model and the Water Clarity Reporting Tool enhance the ability of the CHNEP to collect and disseminate information relevant to seagrass protection and restoration and to concisely report estuarine conditions to CHNEP constituents.

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# Variation of light attenuation and the relative contribution of water quality constituents in the Caloosahatchee River Estuary

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**Abstract** Submerged aquatic vegetation (SAV) is an important ecosystem resource in the Caloosahatchee River Estuary (CRE). Light attenuation is a primary factor controlling SAV distribution and abundance. Multiple data sets collected over decades throughout the CRE were compiled to develop a multiple regression model and quantify the relationship between water quality constituents (color, turbidity, and chlorophyll a (Chl *a*)) and light attenuation. The model was then used to assess the relative contribution of each constituent as a function of location in the estuary and magnitude of freshwater discharge at the Franklin Lock and Dam (S79). On average, both color and turbidity accounted for about 40% of variation of light attenuation, while Chl *a* accounted for 10%. Relative contributions varied with freshwater inflow and location. Color dominated in the upper estuary and/or during higher flow conditions. Turbidity was a major factor in the lower estuary and during lower flow conditions. Chl *a* dominated in the upper estuary, only rarely, during very low flow discharge. The observed relative contributions of color, turbidity, and Chl *a* suggest that while nutrient load reduction might improve water clarity, the assessment of the extent of improvement must account for the contributions of color and turbidity.

**Keywords** Caloosahatchee River Estuary, Charlotte Harbor National Estuary Program, chlorophyll *a*, color, light attenuation, turbidity

## Introduction

It is well established that light is a fundamentally important factor determining the composition, abundance, productivity and distribution of estuarine flora in estuarine systems (Alpine and Cloern 1988, McPherson et al. 1990, Kemp et al. 2004, Murrell et al. 2009, Buzzelli et al. 2014). Consequently, loss of light transparency in the water column is one of the major threats to environmental quality in many estuaries (Cloern 2001). For example, reduced light availability has often been implicated in declines of submerged aquatic vegetation (SAV) distribution (Kemp et al. 2004, Greening and Janicki 2006). SAV are sensitive to changes in light conditions because they generally require a high percentage (~20%) of surface irradiance for net primary production. It is critically important, therefore, to maintain adequate water clarity to preserve or restore the health of estuarine environments (Steward et al. 2005, Corbett and Hale 2006, Crean et al. 2007, Greening et al. 2014).

Multiple optically important constituents affect light transmission in a water column including colored dissolved organic matter (CDOM) or commonly

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called color, suspended sediments, phytoplankton (Chlorophyll *a* or Chl *a*) and water itself (Kirk 1994). The concentrations of suspended sediments are measured in several different ways, each with its own method dependent terminology. These include total suspended solids (TSS), tripton, suspended particular matter (SPM) and turbidity. For the purpose of this paper, we refer to turbidity to indicate the contribution of suspended sediments to light attenuation. The relative contribution of these constituents to total light attenuation varies from region to region and/or season to season (McPherson and Miller 1994, Christian and Sheng 2003, Kelble et al. 2005, Le et al. 2013, Chen et al. 2014). In the main stem of Charlotte Harbor, total light attenuation was partitioned between color (~13%), Chl *a* (~27%) and turbidity (~54%) (McPherson and Miller 1994). Similarly, color, Chl *a* and turbidity accounted for 22%, 16% and 54%, respectively, of total light attenuation in Tampa Bay with the relative contribution of each constituent varying seasonally (McPherson and Miller 1994, Le et al., 2013). In the lower Caloosahatchee River Estuary (CRE), turbidity was often a major factor controlling light attenuation during the dry season, while color was a major factor during high freshwater inflow (Buzzelli et al. 2014, Chen et al. 2014). By contrast, in Florida Bay, turbidity was constantly the dominant component accounting for >80% of total light attenuation with negligible contributions from chlorophyll *a* and CDOM (both <5%) (Kelble et al. 2005, McPherson et al., 2011).

These assessments used multiple linear regression models to partition light attenuation among optical constituents. While this empirical approach can be relatively simple to derive from limited field measurements and is readily applicable for management purposes (e.g. Corbett and Hale 2006), it has inherent limitations as compared with more robust mechanistic models (Gallegos 2001). First, the effects of optical constituents are additive in multiple regression models. In nature, it is more likely that they are non-linearly related (Kirk 1994). Second, the derived regression coefficients are valid only over the ranges of observed concentrations. Any extrapolation of estimated coefficients beyond observed ranges likely introduces uncertainties (Gallegos 2001, Wessel and Corbett, 2009). Therefore, recent studies focused on developing bio-optical models to estimate light attenuation from water quality constituents (Gallegos 2001, 2005, Biber et al. 2008, Dixon et al., 2010). However, development of such a model requires extensive measurements of region-specific inherent optical properties (e.g. spectral scattering and absorption coefficients), which are not readily available to most estuarine resource managers. Thus, it is still desirable to develop an accurate and practically applicable empirical model linking light attenuation to routinely measured water quality constituents, provided sufficient field measurements are available (Steward and Green 2007).

The relationship between water quality constituents and light attenuation and the relative contribution of water quality constituents to light attenuation have been studied in the CRE and other regional estuaries. However, previous studies were based on data from limited spatial and temporal coverages

(McPherson and Miller 1994, Doering et al. 2006, Chen et al. 2014). Given the inherent limitations of a linear regression model and its application for establishing water quality targets (Corbett and Hale 2006), it is preferable to evaluate the potential uncertainty associated with these models. The objectives of this study were (1) to establish a linear relationship between light attenuation and water quality constituents using data collected over decades and across the whole CRE; and (2) to examine how the relative contributions of water quality constituents to light attenuation varies with freshwater inflow and location. The spatial and temporal variations in relative contribution of water quality to light attenuation provide an improved understanding of how water quality constituents interact to control light conditions in the CRE, and insight for establishing nutrient load reduction targets based on meeting seagrass light requirements through a nutrient-dependent reduction in Chl *a* concentration.

## Materials and Methods

**Study area.** The CRE is located on the southwest coast of Florida. It extends from the head of the estuary at the Franklin Lock and Dam (S79) approximately 48 km downstream to San Carlos Bay, where it empties into the Gulf of Mexico (Figure 1). S79 separates the freshwater river from the estuary and acts, in part, as a salinity barrier. Freshwater inflow to the CRE has three different sources: Lake Okeechobee, runoff from the upstream watershed, and inputs from the tidal basin downstream of S79. Based on the salinity gradient and the distribution of SAV, the CRE is commonly divided into four regions: the Upper, Middle, Lower estuary and San Carlos Bay (Figure 1).

**Water quality and light attenuation.** Water quality and light attenuation data in the CRE collected from three monitoring programs (Figure 1 and Table 1) conducted between 1994 and 2010 were analyzed for this study. These include (1) monthly samples collected at 12 stations from 1994-1999 under the Caloosahatchee Estuary (CAL) program; (2) wet season and dry season samples at 12 stations from 2000-2002 under the Environmental Research and Design (ERD) program, and (3) monthly samples at 6 stations from 2005-2010 under the Center for Environmental Studies (CES) program. All data were collected by the South Florida Water Management District (SFWMD) or contactors. Data for the CAL and CES programs were downloaded from DBHYDRO (http://my.sfwmd.gov/dbhydroplsql/show_dbkey_info.main_menu). Data for the ERD are available upon request. Table 1 shows the distribution of sampling stations within regions.

Water quality constituents including salinity, turbidity, color, and Chl *a* were sampled at 0.5 m below the surface. All water quality constituents were sampled and processed in the laboratory following the methods described in the South Florida Water Management District's Laboratory Standard Operating Procedures (Lab SOPs) (SFWMD 2005, SFWMD 2013). Water samples were filtered through GF/G glass fiber filters which were subsequently subjected to extraction using an acetone solution. Chl *a* concentrations were determined on extracts using a Turner AU-10 fluorometer with a narrowed bandwidth of 436 nm (excitation filter) and 680 nm (emission filter). The color of the samples was determined by spectroscopic comparison of filtered samples to a platinum-cobalt (PCU) color solution at 465 nm in a one centimeter path-length cell. The detection range is from 1 to 500 PCU. Turbidity was measured with YSI 6600 water quality instruments that were calibrated using a two point calibration (0 Nephelometric Turbidity Unit (NTU), deionized water) and 123 NTU standards (Formazin supplied by YSI) as recommended by the manufacturer. The detection limits of turbidity sensors are from 0 to 1000 NTU with a precision of 0.1 NTU and an accuracy of 0.3 NTU or 3% of readings (whichever is greater). For CAL and CES data,

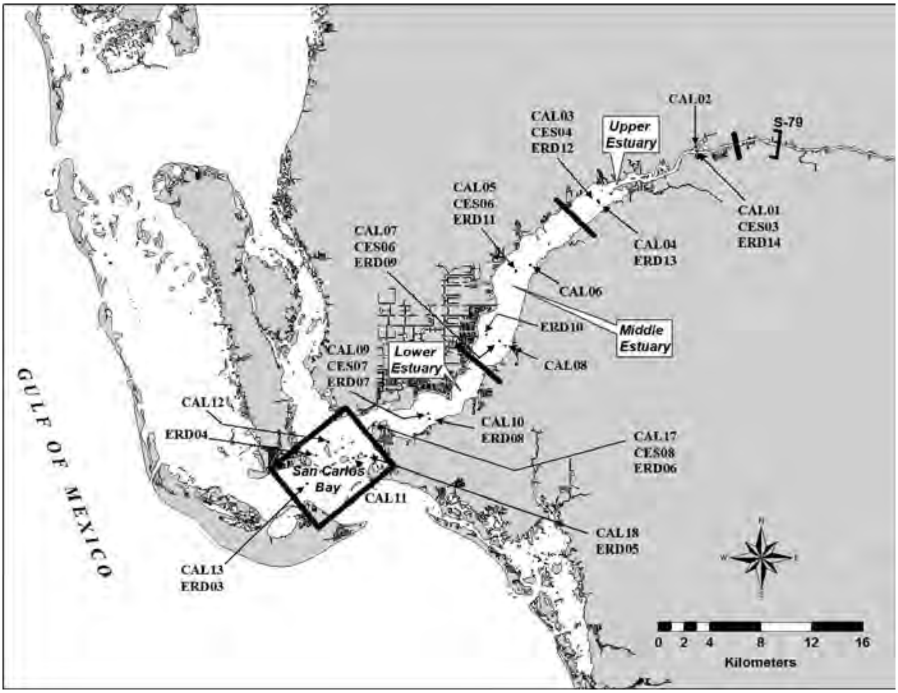


Figure 1. Map of the Caloosahatchee River Estuary, showing regions and water quality monitoring stations from three different programs, namely the Caloosahatchee Estuary (CAL) program; the Environment Research and Design (ERD) Program, and the Center for Environmental Studies (CES) program.

turbidity values were measured in the laboratory following the methods described by the Laboratory Standard Operating Procedures (SOPs, SFWMD, 2014), specifically using a two point calibration (8 and 32 NTU). The details of sampling and water quality analysis also appeared in Doering et al. (2006) and Chen et al. (2014).

Vertical profiles of photosynthetically active radiation (PAR) were generally obtained at depth intervals of 0.25-0.50 m with a LI-COR, LI-193 spherical quantum sensor, and a LI-1400 or LI-1000 data logger. Above water surface radiation was also simultaneously measured using a deck sensor to correct for potential errors introduced by clouds. The diffuse attenuation

Table 1. Sampling stations from three monitoring and research programs in the four regions of the Caloosahatchee River Estuary. ERD: the Environmental Research and Design (ERD) program, CAL: Caloosahatchee Estuary (CAL) program, and CES: the Center for Environmental Studies.

| | Programs | | | |
|---|---|---|---|---|
| Regions of Estuary | ERD | CAL | CES | Km from S79 |
| Upper Estuary | 12,13,14 | 01,02,03,04 | 03,04 | 6-14 |
| Middle Estuary | 09,10,11 | 05,07,08 | 05,06 | 14-28 |
| Lower Estuary | 06,07,08 | 09,10,17 | 07,08 | 28-41 |
| San Carlos Bay | 03,04,05 | 11,12 | na | 41-49 |

coefficient ($K_d$, $m^{-1}$) of PAR was calculated from those profiles as the linear regression coefficient of ln (Iz) with respect to depth based on the following equation

$$I_z = I_0 \times e^{-K_d \times Z} \tag{1}$$

where $I_o$ (μmol $sec^{-1}$ $m^{-2}$) is PAR just below the water surface, and $I_z$ (μmol $sec^{-1}$ $m^{-2}$) is PAR at depth Z (m). To achieve comparability between stations of different depths, only data from the top 2 m were used in the calculation.

**Data analysis.** In order to account for spatial variation in water quality and light attenuation, data were grouped in the four regions (Figure 1). In each region data were sorted by year and month and then averaged across stations. This produced a set of observations in each region for which descriptive statistics such as minima, medians, maxima and percentiles were calculated.

The relationship between $K_d$ and color, turbidity, and Chl *a* was examined using multiple linear regression analysis (SAS v9.3). Stepwise regressions relating variation of light attenuation to water quality constituents were first applied to specify the significant independent variables ($p<0.05$). Then a multiple regression model with a forced intercept of 0.15 was developed (equation 2).

$$K_d = K_{water} + (k^*_{color} \times \text{color}) + (k^*_{turbidity} \times \text{turbidity}) + (k^*_{Chl\,a} \times \text{Chl}\,a) \tag{2}$$

Where $K_{water}$=0.15 $m^{-1}$ is the contribution of water to total light attenuation at a depth of ~ 2 meters (Dixon 2014, personal communication). $k^*_{color}$ ($PCU^{-1}$ $m^{-1}$), $k^*_{turbidity}$ ($NTU^{-1}$ $m^{-1}$), and $k^*_{Chl\,a}$ ($m^2$ $mg^{-1}$) are regression coefficients for color, turbidity and Chl *a*, respectively. The relative contributions of water constituents were calculated by dividing products of water quality concentrations and their respective linear regression coefficients by the estimated $K_d$.

To assess the appropriateness of the linear regression model, we evaluated residuals of the model (Kachigan 1991) and their normality with the Shapiro-Wilk test (SAS v9.3). Model performance was examined across the range of $K_{d,}$ with three quantities. The first was the ratio of predicted $K_d$ (P_Kd) using equation 2 to measured $K_d$ (M_Kd) from equation 1. This ratio (P_$K_d$:M_$K_d$) provides a measure of an overall bias of predicted values. If the model accurately estimates the observed light attenuation, the ratio would be centered on one. The second quantity was the absolute percentage difference (APD) between predicted and measured $K_d$ (equation 3).

$$APD = \frac{|M\_K_d - P\_K_d|}{M\_K_d} \times 100 \tag{3}$$

It provides a measure of uncertainties of prediction, because a large APD indicates the modeled values deviate far from the measured values. The final quantity was root mean square error (RMSE), which is a statistic indicating how accurately the model predicts $K_d$ (equation 4)

$$RMSE = \sqrt[2]{\frac{(M\_K_d - P\_K_d)^2}{n}} \tag{4}$$

Where n is the number of samples. Similar statistics have been used to compare performance of different models (e.g. Wessel and Corbett 2009).

**Freshwater inflow.** Average freshwater inflow (cubic feet per second (CFS), or 1 CFS ~=0.538 $m^3$ $s^{-1}$) at S79 over the 30 days prior to sampling was calculated from daily flow rate. The 30 day average flow was then assigned to four flow categories (<450, 450-2800, 2800-4500 and >4500, CFS) to examine variations of $K_d$ and contributions of water constituents as a function of freshwater flow. The 30 day averaging period was selected based on previous studies, which suggested the average flushing time of the CRE was about 1 month ranging from a few days to more than 60 days, depending on freshwater inflow and location in the estuary (Doering et al. 2006,

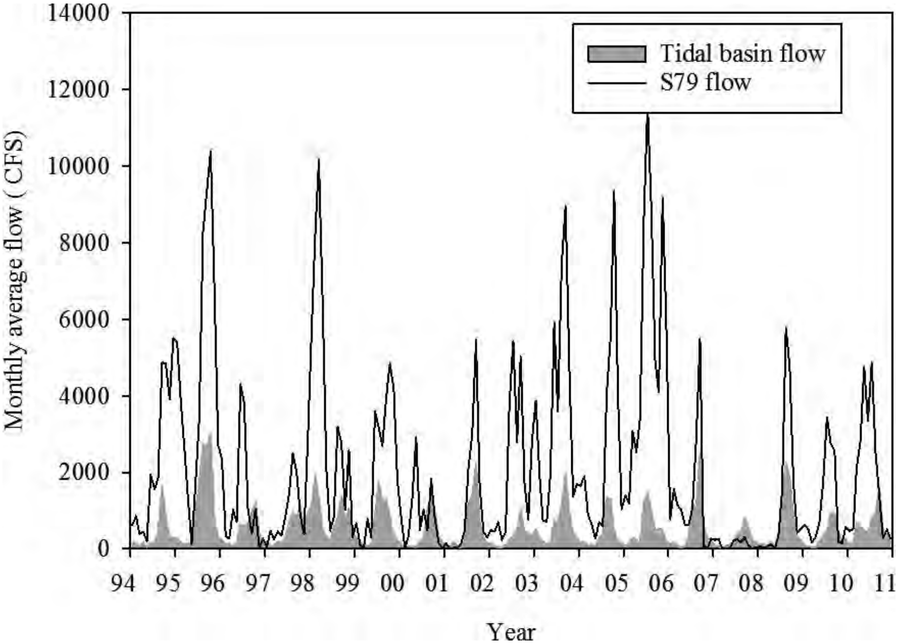


Figure 2. Monthly average freshwater flows (CFS) from at the Franklin Lock and Dam (S79) and tidal basin.

Wan et al. 2013). The four categories were selected based on performance measures used in the Restoration Coordination and Verification program (RECOVER) (Buzzelli et al. 2014). Similar flow categories were used by Doering et al. (2006) to study the variations of water quality with freshwater inflow. Flow from the tidal basin was modelled using a linear watershed model developed by Wan and Konyha (2015).

## Results

**Freshwater inflows from S79 and tidal basins.** Freshwater inflow from the S79 to the CRE showed large variations with monthly average freshwater flow of 2145 CFS ranging from 0 to >11590 CFS during the study period (1994-2010) (Figure 2). The relatively high flows occurred in the years of 1995, 1998 and 2003-2006, while relatively low flows occurred in the 2007-2009 period. Tidal basin flows had a similar pattern as S79 flow with a monthly average of 530 CFS and a range of from 0 to 3000 CFS. On average, tidal basin flow accounted for ~ 20% of total flows into the CRE. However the tidal basin flow can be larger than S79 flow in a drought year.

**Variations of light attenuation and water quality constituents.** Both light attenuation and water quality constituents showed large variation across the CRE (Table 2). The light attenuation coefficient ranged from

0.3 - 6.5 $m^{-1}$ with an average and standard deviation of 1.9 ± 1.0 $m^{-1}$. Turbidity varied from 0.2 - 23.0 NTU with average and standard deviation of 4.7 ± 3.0 NTU. Relative to ranges of light attenuation and turbidity, both color and Chl *a* displayed more pronounced fluctuations with larger ranges and variances. Indeed, both Chl *a* and color varied over 3 orders of magnitude from 0.1 - 76.0 mg $m^{-3}$ and 1 - 331 PCU, respectively. Light attenuation and water quality constituents, except turbidity, all displayed a distinctive spatial gradient with higher values in the upper estuary. Turbidity varied little across the CRE with relatively higher values in the lower estuary than those in other regions.

**Linear regression of light attenuation and relative contribution of water quality constituents.** Results from stepwise linear regression indicated that all three water quality parameters were significantly related to light attenuation. The three water quality constituents were used to estimate the regression coefficients (equation 5). The predicted $K_d$ values compared well with observed $K_d$ (Figure 3). In addition, the residual plot shows a random pattern over predicted Kd and the Shapiro-Wilk test revealed that residuals were normally distributed (Figure 4). Therefore the linear model is appropriate.

Together color, turbidity and Chl *a* explained 64% of the total variation in $K_d$ and the regression coefficients were all significantly greater than 0 ($p < 0.001$). The median ratio ($P\_K_d$:$M\_K_d$) between the predicted and measured $K_d$ was 0.95 (mean ratio was 1.00), indicating that the overall bias of the model was very small. The median APD was 20%, while the mean APD was 22%. The RMSE was 0.67 $m^{-1}$ over the range of observed light attenuation coefficients (0.3 - 6.5 $m^{-1}$). The $K_d$ values predicted by the model were also comparable to those calculated using a previous model (McPherson and Miller 1994), hereafter called M&M model, equation 6. The M&M model explained about 60% of the variability in $K_d$ in the CRE with a similar model performance: the median $P\_K_d$:$M\_K_d$, the median APD and the RMSE were 0.92, 17% and 0.73 $m^{-1}$, respectively.

$$K_d = 0.15 + (0.014 \times \text{color}) + (0.15 \times \text{turbidity}) + (0.022 \times \text{Chl}\ a)\ (n = 512,\ p < 0.001) \quad (5)$$

$$K_d = 0.30 + (0.014 \times \text{color}) + (0.062 \times \text{turbidity}) + (0.049 \times \text{Chl}\ a) \quad (6)$$

Using the regression coefficients from equation 5 and concentrations of color, turbidity and Chl *a* observed in the field, the relative contributions of color, turbidity and Chl *a* to the predicted $K_d$ were estimated. The contribution of each constituent was further examined as a function of location and the magnitude of freshwater inflow (Table 3 and Figure 5). On average, both color and turbidity accounted for about 40% of light attenuation, while Chl *a* contributed about 10%. However, the relative contributions varied with freshwater inflow and region of

Table 2. Statistical summary of water quality (color (PCU), turbidity (NTU), chlorophyll *a* (mg m$^{-3}$) and light attenuation (m$^{-1}$) in different regions of the CRE. The sample sizes for Upper, Middle, Lower estuaries and San Carlos Bay are 174, 133, 116 and 89, respectively.

| Parameter | Regions | Minimum | 25 percentile | Median | 75 Percentile | Maximum | Average | Standard Deviation | Sample Size |
|---|---|---|---|---|---|---|---|---|---|
| Color | Upper Estuary | 8 | 38 | 60 | 120 | 331 | 85 | 66 | 174 |
| | Middle Estuary | 6 | 21 | 38 | 80 | 268 | 62 | 58 | 133 |
| | Lower Estuary | 1 | 9 | 19 | 54 | 311 | 39 | 49 | 116 |
| | San Carlos Bay | 1 | 4 | 11 | 24 | 102 | 20 | 24 | 89 |
| | CRE | 1 | 17 | 38 | 80 | 331 | 58 | 60 | 512 |
| Turbidity | Upper Estuary | 0.2 | 2.4 | 3.5 | 5.6 | 16.7 | 4.4 | 3.2 | 174 |
| | Middle Estuary | 0.4 | 2.4 | 3.5 | 4.7 | 15.4 | 4.1 | 2.6 | 133 |
| | Lower Estuary | 0.5 | 2.5 | 4.2 | 6.2 | 23.0 | 4.7 | 3.2 | 116 |
| | San Carlos Bay | 1.3 | 3.8 | 5.6 | 6.9 | 16.7 | 5.6 | 2.5 | 89 |
| | CRE | 0.2 | 2.6 | 3.8 | 6.0 | 23.0 | 4.7 | 3.0 | 512 |
| Chl *a* | Upper Estuary | 0.1 | 4.0 | 8.0 | 16.6 | 76.4 | 12.2 | 13.0 | 174 |
| | Middle Estuary | 0.3 | 3.0 | 5.9 | 10.8 | 76.6 | 9.9 | 12.0 | 133 |
| | Lower Estuary | 0.2 | 2.0 | 3.6 | 7.2 | 59.2 | 7.0 | 10.4 | 116 |
| | San Carlos Bay | 0.2 | 2.7 | 4.1 | 9.4 | 39.9 | 7.4 | 8.1 | 89 |
| | CRE | 0.1 | 3.0 | 5.5 | 11.1 | 76.6 | 9.7 | 11.7 | 512 |
| Kd | Upper Estuary | 0.3 | 1.9 | 2.5 | 2.3 | 6.5 | 2.6 | 0.9 | 174 |
| | Middle Estuary | 0.3 | 1.2 | 1.8 | 2.8 | 5.6 | 2.1 | 1.1 | 133 |
| | Lower Estuary | 0.3 | 0.9 | 1.3 | 1.9 | 4.8 | 1.5 | 0.9 | 116 |
| | San Carlos Bay | 0.4 | 0.9 | 1.2 | 1.5 | 3.0 | 1.3 | 0.5 | 89 |
| | CRE | 0.3 | 1.2 | 1.8 | 2.7 | 6.5 | 1.9 | 1.0 | 512 |

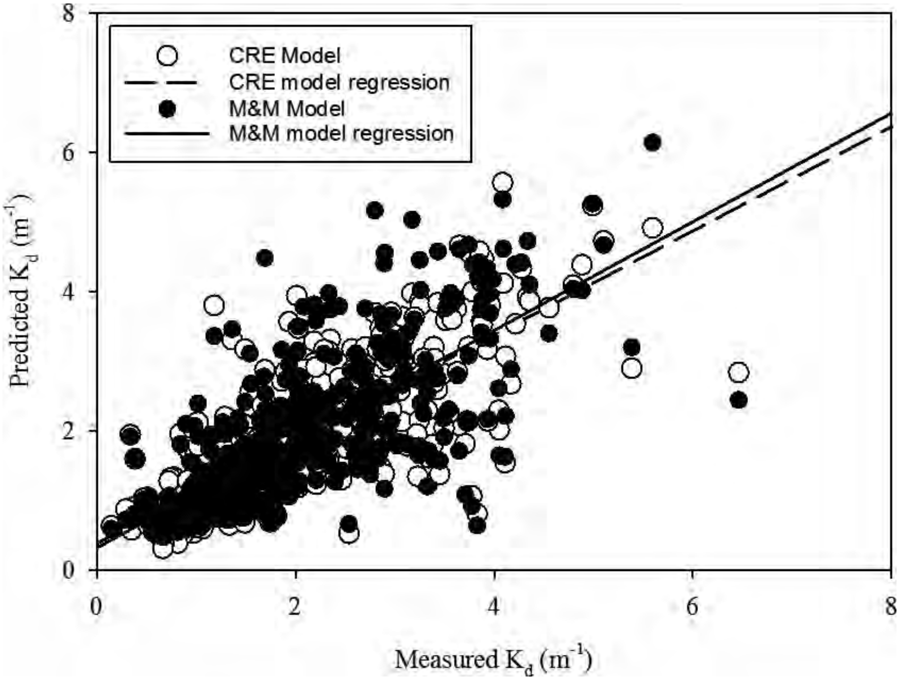


Figure 3. Relationships between measured $K_d$ and predicted $K_d$ from linear regression model in this study (CRE model, open circles) and the McPherson and Miller model (1994, M&M model, filled circles). The dashed line and solid line are linear regressions.

the estuary. Color was the main factor impacting light attenuation in the upper estuary with its contribution increasing with freshwater inflow in all regions. Turbidity was the second-most important factor with its contribution to light attenuation increasing with the distance from S79. As freshwater inflow increased, the contribution of turbidity decreased while the contributions from color increased. However, Chl *a* contribution varied with the regions and flow conditions. In the upper estuary, the Chl *a* contribution did decrease with increasing freshwater inflow. In the middle and lower estuary, increasing flows initially increased Chl *a* contribution of light attenuation, and when freshwater inflow exceeded 4500 CFS, Chl *a* contribution decreased. In all regions, Chl *a* contributed the least to total light attenuation with the maximum percentage less than 17%, and average of less than 12% regardless of region and freshwater inflow. To further examine the contribution of Chl *a* to total light attenuation, we examined the frequencies of different ranges of percent contribution to total light attenuation through a histogram. Results show that Chl *a* contributed <10% to total light attenuation in 304 (60%) of the total samples (512). In contrast, only 2 of 512 observations had a Chl *a* contribution >50%, which occurred in the upper estuary at low freshwater inflow (30 day average flows are 688 and 0 CFS, respstively) and very high Chl *a* (>70 mg $m^{-3}$) (Figure 6). Similarly in only 6 cases did Chl *a* contribution exceed >40% (Figure 6).

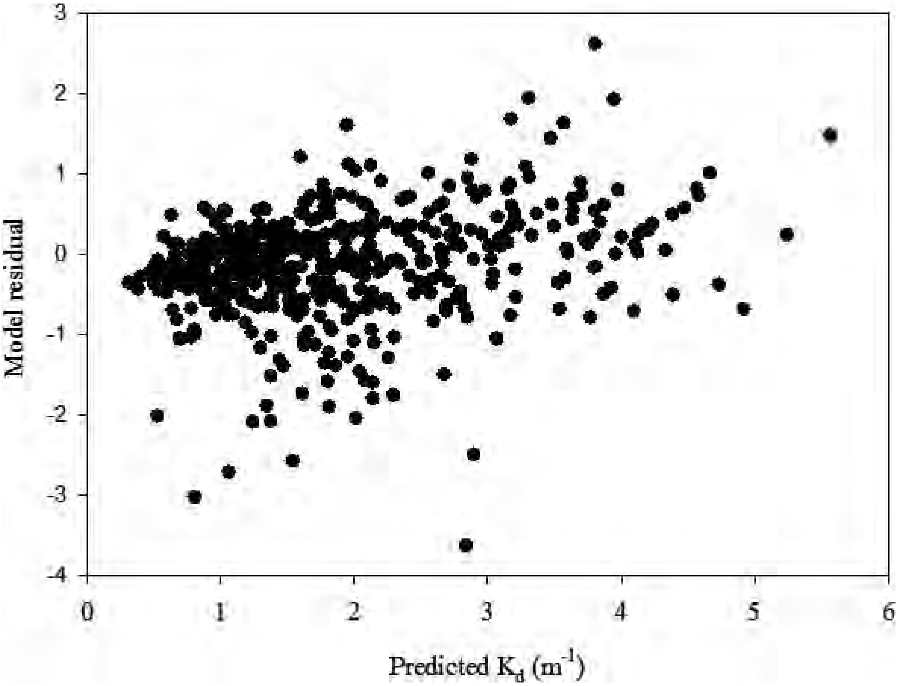


Figure 4. The distribution of model residuals over predicted light attenuation showing a random and normal distribution pattern of the model residuals.

## Discussion

A new empirical regression model was developed which links light attenuation with water quality constituents in the CRE. Although similar to other models developed for the CRE and other estuarine systems (McPherson and Miller 1994, Doering et al. 2006, Chen et al. 2014), the newly developed model may represent a statistically more rigorous relationship for the CRE. This is because the data were collected from broader spatial and temporal coverages encompassing larger variability in both light attenuation and water quality concentrations. Given the potential limitations of linear models to partition light attenuation among water quality constituents, it is critically important that a data set used for the development of a model cover sufficient spatial and temporal variability for a system (Gallegos 2001, Steward and Green 2007). This model predicted $K_d$ very well as compared with measured values ($r^2 = 0.64$) and performed well in terms of accuracy (mean ratio between $P\_K_d$: $M\_K_d$ of 1.0) and precision (mean APD of 22%). In addition, this model has a similar prediction performance as the M&M light model (McPherson and Miller 1994) with both explaining > 60% of the variation in measured light attenuation and comparable $P\_K_d$:$M\_K_d$ and APD.

While the newly developed model is comparable to the M&M model in terms of model performance, some regression coefficients are quite different.

Table 3. Contribution (%) of color, Chl *a* and turbidity to light attenuation with regions and freshwater inflows (CFS).

| Water quality constitents | Flow < 450 | | | | Averages over regions |
|---|---|---|---|---|---|
| | Upper Estuary | Middle Estuary | Lower Estuary | San Carlos Bay | |
| Sample size | 72 | 50 | 34 | 26 | 182 |
| Color | 37.8 | 34.9 | 22.8 | 10.6 | 30.3 |
| Chl *a* | 16.3 | 10.1 | 6.8 | 7.3 | 11.5 |
| Turbidity | 37.0 | 39.5 | 51.3 | 67.1 | 44.6 |
| | 450 <= Flows < 2800 | | | | |
| Sample size | 67 | 55 | 51 | 39 | 212 |
| Color | 49.4 | 38.0 | 24.4 | 12.1 | 33.6 |
| Chl *a* | 12.8 | 13.9 | 8.1 | 9.7 | 11.4 |
| Turbidity | 30.4 | 38.5 | 53.4 | 64.2 | 44.2 |
| | 2800 <= flows < 4500 | | | | |
| Sample size | 16 | 16 | 17 | 11 | 60 |
| Color | 71.0 | 58.8 | 47.3 | 26.7 | 52.9 |
| Chl *a* | 6.4 | 10.4 | 11.9 | 17.9 | 11.1 |
| Turbidity | 17.5 | 25.1 | 33.2 | 47.8 | 29.5 |
| | Flows >= 4500 | | | | |
| Sample size | 19 | 12 | 14 | 13 | 58 |
| Color | 70.8 | 64.1 | 44.2 | 35.3 | 55.1 |
| Chl *a* | 3.0 | 6.1 | 10.8 | 11.9 | 7.5 |
| Turbidity | 20.5 | 24.2 | 37.5 | 44.5 | 30.7 |
| | Average over all regions and all flows | | | | |
| Color | | | 37.1 | | |
| Chl *a* | | | 11.0 | | |
| Turbidity | | | 41.1 | | |

The differences may be, in part, due to different approaches applied. We used a forced intercept in the multiple regression analysis, while McPherson and Miller (1994) let the model automatically determine the intercept. However, using an unforced intercept approach in linear regression often leads to a higher intercept (it would be 0.60 in our case). Any intercept greater than 0.3 would be an overestimate of the water contribution to total light attenuation in the study area based on the minimum $K_d$ (0.3 $m^{-1}$) observed in the measurements. On average, the latter approach would lead to color accounting for 31% of total light attenuation with turbidity and Chl *a* contributing 23% and 9%, respectively. The second potential reason for the differences may be related to different data sets with different ranges and mean conditions of water quality constituents employed to estimate the regression coefficients (Doering et al. 2006). Differences notwithstanding, the overall conclusion that CDOM and turbidity dominate light attenuation in the CRE is valid regardless of the model applied. However, it is worth noting that the models are empirically based. Future efforts should include further validation of this new model with more field measurements and development of a spectral bio-optical model for the CRE (Biber et al. 2008, Dixon et al. 2010).

Our study found light attenuation was primarily controlled by color in the upper region of the CRE and by turbidity in the more marine San Carlos Bay.

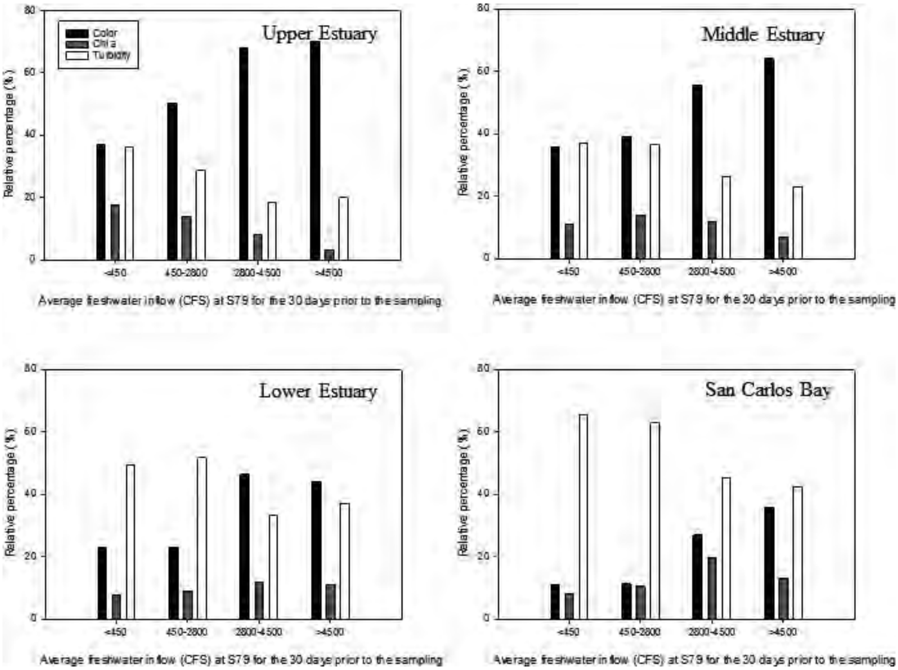


Figure 5. Variations of contributions (%) of color, Chl *a* and turbidity to light attenuation in different freshwater inflows and regions.

In all regions, Chl *a* contributed the least to light attenuation relative to color or turbidity. This low contribution of Chl *a* to the total light attenuation is consistent with previous results in the CRE (McPherson and Mille, 1987, McPherson and Miller 1994, Dixon and Kirkpatrick 1999, Doering et al. 2006) and other subtropical estuaries (Christian and Sheng 2003, Kelble et al. 2005, Cannizzaro et al. 2013) (Table 4). Our results also show that Chl *a* might become a major factor influencing light attenuation in the upper estuary at the very low freshwater when increased light availability and residence time promote phytoplankton bloom in the upper estuary (Doering et al. 2006, Buzzelli et al. 2014). However, the dominance of Chl *a* in light attenuation occurred under very rare conditions (only 2 cases among 512 total samples). Furthermore, the relatively low contribution of Chl *a* to light attenuation in the CRE is different from observations in other large estuaries (e.g. Chesapeake Bay) and even other south Florida estuaries (e.g., Tampa Bay), where $K_d$ was dominated by the variability of phytoplankton and sediment (Gallegos 2001, Le et al. 2013). The differences are mostly due to the higher color concentration in the CRE.

The relatively small contribution of Chl *a* to light attenuation as compared to color and turbidity in the CRE has important management implications. Strategies to manage external nutrient loads are often designed around improving water clarity and light penetration by lowering Chl *a* concentrations

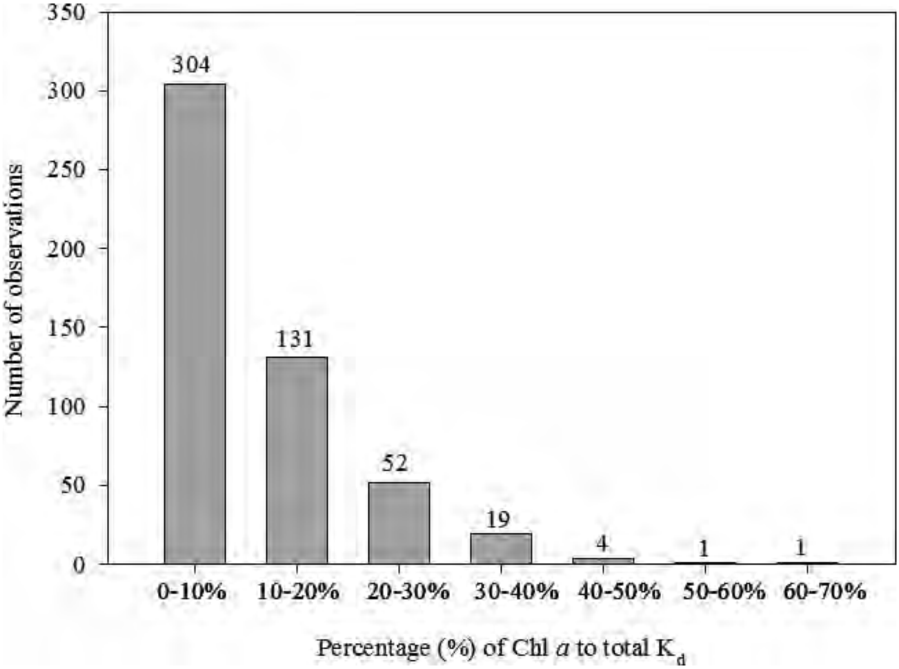


Figure 6. The number of observations at different percentage of Chl *a* to total light attenuation, showing it is rare (2 sample among total samples of 512) that Chl *a* was a dominant factor controlling light attention.

to meet seagrass light requirements (Steward and Green 2007, Buzzelli et al. 2014, Greening et al. 2014). Such strategies have been very successful in systems like Tampa Bay and Chesapeake Bay where Chl *a* is one of major attenuators of light and water column phytoplankton depend on external nutrient loads (Orth et al. 2010, Le et al. 2013, Greening et al. 2014). However, in systems like the CRE, where color and turbidity dominate light attenuation, reducing the nutrient load may not be as effective in improving water clarity.

Table 4. Comparison of percent of components of $K_d$ for the different estuaries or the same estuary from different studies. The listed values represent mean percentages over the study area. If no average values are available, the reported ranges are used here.

| Estuary | Color | Chl *a* | Turbidity/Tripton/TSS | Refences |
|---|---|---|---|---|
| Charlotte Harbor | 21% | 4% | 73% | McPherson and Miller (1987) |
| Charlotte Harbor | 22% | 16% | 55% | McPherson and Miller (1994) |
| Tampa Bay | 13% | 27% | 54% | McPherson and Miller (1994) |
| Tampa Bay | 22% | 25% | 52% | Le et al. (2013) |
| Indian River Lagoon | 5% | 16% | 78% | Christian and Sheng (2003) |
| Florida Bay | 1% | 2% | 89% | Kelble et al. (2005) |
| Albemarle-Pamlico Sound | 15-42% | 19-21% | 39-45% | Biber et al. (2008) |
| the CRE | 40% | 11% | 38% | this study |

Indeed, McPherson et al. (2011) found that nutrient control alone may not be sufficient to permit seagrass recolonization in Florida Bay because the light attenuation in the system was dominated by suspended sediment. Recently, seagrass modeling results demonstrated that in the Southern Indian River Lagoon, a system similar to the CRE where light was controlled primarily by color and turbidity, seagrass coverage was not very sensitive to changes in Chl *a* (Buzzelli et al. 2012). More interestingly, the latest results from an integrated model in the CRE suggested that nutrient load reduction through decreasing fresh water inflow would be more effective in improving water clarity and increasing seagrass coverage than nutrient load reduction by reducing incoming nutrient concentrations (Buzzelli et al. 2014). This is, in part, because decreased freshwater inflow significantly reduced color concentration and color accounts for higher percentage of light variation than Chl *a* in the CRE. The relatively lower contribution of Chl *a* to the total light attenuation and modeling results in the CRE suggest that while nutrient load reduction will likely improve water clarity, the assessment of effectiveness of improvement should account for the significant contributions of color and turbidity to light attenuation in the system.

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# Evaluating light attenuation and low salinity in the lower Caloosahatchee Estuary with the River, Estuary, and Coastal Observing Network (RECON)

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**Abstract** The southern portion of the Charlotte Harbor region, which includes Pine Island Sound, San Carlos Bay, and the lower Caloosahatchee Estuary, has over 11,700 ha of submerged aquatic vegetation (SAV). The SAV species in the region have been used as environmental indicators because they are affected by nutrient loading, algae blooms, and freshwater discharges. Management approaches to reduce nutrient loading, phytoplankton concentrations, and high freshwater discharges in the region have also been applied to meet water clarity targets (light attenuation). In an effort to understand the duration and effect of low salinity periods in the lower estuary on water clarity, salinity data at several River, Estuary, and Coastal Observing Network (RECON) sites were analyzed. Optical parameters associated with increased light attenuation (fluorescent dissolved organic matter, chlorophyll, turbidity) were significantly higher during lower salinity periods (less than 25). In addition, discrete light attenuation coefficients, collected as part of RECON monthly maintenance, were analyzed. A synthesis and evaluation of the conditions in the lower Caloosahatchee during the study period (2008-2014) suggest that flow and load reductions would result in increased water clarity.

**Keywords** FDOM, freshwater discharge, *in situ* chlorophyll, seagrass extent, water clarity, RECON

## Introduction

The management of freshwater flows to an estuary can have profound effects on conditions and ecosystem function (Alber 2002, Kimmel 2002). Increasing worldwide demands for freshwater and the loss of critical habitats in coastal areas (Lotze et al. 2006) are contributing to the need for prioritization, allocation, and delivery of freshwater to coastal systems (Montagna et al. 2002). One approach using optimal flow criteria provides a mechanism by which freshwater is allocated to preserve estuarine characteristics (e.g., estuarine gradients, turbidity maximum) and to protect sessile oligohaline (e.g., *Vallisneria americana*), euryhaline (e.g., *Crassostrea virginica, Callinectes sapidus*), and euryhaline species (e.g. *Cynoscion nebulosus, Thalassia testudinum*) that are found in bays and estuaries. However, oscillations in climatic conditions, such as increased hurricanes, can also lead to variable flows,

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nutrient loading (Doering and Chamberlain 1999), and periods of hypoxia (Vaquer-Sunyer and Duarte 2008). Similarly, droughts and floods can lead to the absence of critical spring phytoplankton blooms (Nichols et al. 1986) or reductions in nursery habitats, such as seagrasses or oysters. Consequently, there is a growing need for monitoring of inshore, nearshore, and oceanic conditions over relatively large distances (e.g. 100–200 km) and in "real-time" (Johnson and Needoba 2008) to understand both natural and anthropogenic sources of environmental variability. Estuarine systems have many gradients that change rapidly over short time-scales (hours to days). This requires frequent and simultaneous measurements at multiple locations (Malone 2003) to understand how complex systems are organized and to provide recommendations for how to improve the management of natural resources and maximize ecosystem services (Costanza et al. 1997).

The Caloosahatchee is a channelized and highly managed estuary where freshwater flows are determined by a series of dams and locks which are connected to Lake Okeechobee. At any time the freshwater flows can be any combination water from the Caloosahatchee watershed and from Lake Okeechobee. The Caloosahatchee River is used as a conveyance to release water when levels in the lake are too high. During drought periods, the Caloosahatchee Estuary often does not have sufficient freshwater from the tidal watershed and the oligohaline zone is highly compressed. This problem is exacerbated by the S79 lock and dam, which presents a barrier to upstream movement of the oligohaline zone. Furthermore, the rapid urbanization of the southwest Florida coast has replaced wetlands and riparian areas that once stored and treated stormwater within the Caloosahatchee watershed. The result is an estuary with frequent extreme fluctuations in salinity and dissolved nutrients. Flows reported at S79 are generally indicative of the regional climate conditions, but also reflect water management actions based on evolving protocols for budgeting water to control Lake Okeechobee levels, regulating estuarine salinities, etc. Target minimum flows to the Caloosahatchee Estuary are 300 cubic feet per second (CFS) (8.5 $m^3 s^{-1}$) and operational flexibility has allowed 450 CFS. The optimal minimal flows supportive of valued ecosystem components is 300-800 CFS (SFWMD 2000). Maximum flows that exceed 2,800 CFS (79.3 $m^3 s^{-1}$) can affect oyster spat settlement (*Crassostrea virginica*) and other euhaline species such as *Thalassia testudinum* (CERP RECOVER 2007, Milbrandt et al. in press). The proportion of flow from the watershed versus Lake Okeechobee changes from year-to-year depending on several factors including; rainfall, water storage, the level of Lake Okechobee, climatological predictions, and water use in the Caloosahatchee basin.

Extensive mangrove-lined shorelines in Pine Island Sound and San Carlos Bay are closely associated with oyster reefs and large areas of mixed seagrass species. The biodiversity in seagrass areas is high (Heck 1979, Poulakis et al. 2003) and supportive of a variety of recreational fishery activities with significant economic value (Beever and Walker 2013). The extensive seagrass areas are threatened by shifts in water quantity and quality from the

Caloosahatchee Estuary, and variation in salinity has been identified as a major stressor (Orlando and Douglass 2014). However, exactly how the frequency and duration of lower salinity periods and subsequent reductions in water clarity affect surrounding seagrass areas is poorly understood. The Technical Advisory Committee (TAC) of the Charlotte Harbor National Estuary Program (Janicki et al. 2010, Dixon and Wessel 2014) has established optical targets for protection of existing seagrass, estimated from the data of a large, multi-agency water quality monitoring network (http://www.chnep.wateratlas.usf.edu). The optical targets are based on light attenuation coefficients (Kirk 1983), both measured and derived from optical properties (fluorescent dissolved organic matter, chlorophyll *a*, turbidity). Fluorescent dissolved organic matter (FDOM) is strongly negatively correlated with salinity and positively correlated with total nitrogen (TN) in the Caloosahatchee Estuary (Bailey et al. 2009). This suggests that exceeding the upper threshold of 2,800 CFS (79.3 $m^3$ $s^{-1}$) not only lowers salinities below thresholds of harm for certain seagrass species (Irlandi 2006), but also introduces higher nutrient loading and increased phytoplankton productivity, or macroalgal overgrowth (Milbrandt unpublished data, Douglass 2013). While light attenuation observations and targets have provided some initial guidance about water and resource management in this system, an evaluation was needed of the frequency and duration of low salinity events in Pine Island Sound and San Carlos Bay, and their relationship to optical water quality and seagrass health.

## Materials and Methods

**Monitoring region and sites.** Barrier islands, including southern Pine Island, Sanibel, and Captiva Islands, form the border of a large, protected system of bays and sounds in lower Charlotte Harbor. Pine Island Sound and San Carlos Bay are shallow, mangrove-lined estuaries that are affected by freshwater from the Caloosahatchee River (including the tidal watershed, the watershed of the Caloosahatchee upstream of S79, and Lake Okeechobee).

The River, Estuary, and Coastal Observing Network (RECON) is composed of real-time sensor platforms at fixed sites in the Caloosahatchee Estuary, San Carlos Bay, Pine Island Sound, and the Gulf of Mexico (Figure 1). The Shell Point RECON fixed site, established in November 2007, is at the confluence of the Caloosahatchee Estuary and San Carlos Bay and experiences strong tidally-driven mixing with a large range of salinities with each tidal cycle. Shell Point is near several oyster reef and seagrass indicator sites targeted for northern Everglades restoration monitoring (Douglass 2013, Volety and Haynes 2013). McIntyre Creek and Tarpon Bay are fixed RECON locations at the confluence of lower Pine Island Sound and J.N. “Ding” Darling National Wildlife Refuge. McIntyre Creek was established by the U.S. Geological Survey (USGS) in January 2008 and was added to RECON in 2013. Tarpon Bay was established in 2007 but data were not collected from 2008-2010. These sites are moderately influenced by freshwater inputs from the Caloosahatchee River but also have local sources of freshwater from Sanibel Island. They are typically euryhaline (30-35) during dry periods (November-June), and occasionally experience lower salinities during tropical storms or high rainfall and freshwater flow periods (July-October). McIntyre Creek and Tarpon Bay are in well-protected areas close to shore. Blind Pass was a fixed site from 2007-2012 located on the Intracoastal Waterway (ICW) in open water in the middle of the south end of Pine Island Sound. The site was discontinued in 2012 due to a boat strike and the instrumentation was moved to McIntyre Creek. Redfish Pass and Gulf of Mexico are

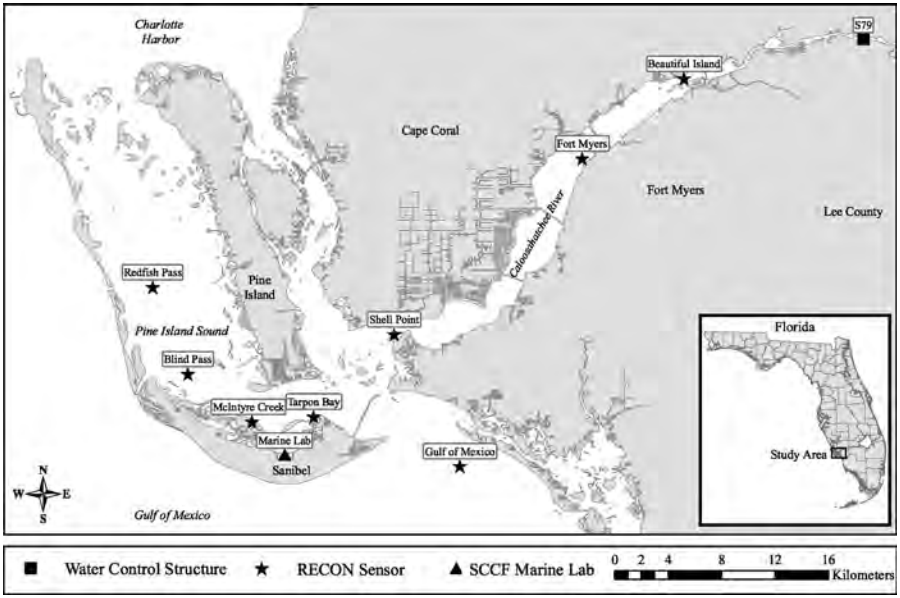


Figure 1. Map of the study area. The map shows the locations of the River, Estuary and Coastal Observing Network (RECON) and the location of the Sanibel-Captiva Conservation Foundation (SCCF) in Southwest Florida.

geographically distant from the Caloosahatchee Estuary and are euryhaline. Redfish Pass and the Gulf of Mexico RECON sites are in open water secured to U.S. Coast Guard channel markers.

**RECON monitoring methods.** Several biological, chemical, and physical parameters are measured hourly at each RECON station and all data are autonomously sent back to shore and made web-accessible in near real time with Seabird Coastal LOBO instruments (Table 1) Instruments are deployed and maintained from small boats with a maximum service interval of 1-2 months. The Seabird Coastal LOBO instrument packages are each attached to pilings at depths of at least 1.5 m below MLLW (Mean Lower Low Water). Configurations, tolerances, sensitivity, and variability specifications of the sensors are available from the sensor manufacturers (Seabird Coastal) and on the RECON website (http://recon.sccf.org). Each instrument package was scheduled to record and transmit data to the shore at 1-hour intervals. Data were downloaded into AQUARIUS, a software program for water data management, for analyses.

**Kd PAR.** During monthly service and maintenance visits to RECON sites, a data logger and sensor (Biospherical) were deployed to simultaneously measure surface and subsurface flux of photosynthetically active radiation (PAR, μmol $m^{-2}$ $s^{-1}$). The data were collected for 60 s at 1 hz intervals at 0.70 m offset depths. The Biospherical BICs are multi-channel radiometers (PAR, 400-700 nm) and are deployed from the boat at a fixed width (0.6 m) using a steel lowering frame. For each sampling event at each station, a light attenuation coefficient (Kd) was calculated as:

$$Kd = \ln(I_1/I_2)/(Z_2\text{-}Z_1), \quad (1)$$

where Kd is light attenuation coefficient, $I_1$ and $I_2$ are downwelling irradiances at depths at $Z_1$ and $Z_2$ (Kirk 1983). The data were entered into a Microsoft Access database along with other RECON field Quality Assurance/Quality Control discrete data. Light sensors were calibrated by Biospherical in 2007, 2008, 2009, and 2012.

Table 1. Parameters measured at RECON locations.

| Parameter | Instrument | Manufacturer |
|---|---|---|
| Temperature | WQM | Seabird Coastal |
| Salinity | WQM | Seabird Coastal |
| FDOM | ECO FLS | Seabird Coastal |
| Chlorophyll *a* | WQM | Seabird Coastal |
| Turbidity | WQM | Seabird Coastal |
| Dissolved Oxygen | WQM | Seabird Coastal |
| Nitrate | ISUS V3 | Seabird Coastal |

**S79 flow.** The proportion of flow from the watershed versus Lake Okeechobee varies from year-to-year depending on a large number of factors, including climatic factors and lake management. For the purposes of this article, the water conditions in the lower estuary are influenced directly by S79 flows. The flow from S79 is the cumulative flow from Lake Okeechobee and the Caloosahatchee watershed (Figure 2). While inexact, the proportion of flow from Lake Okeechobee can be derived by subtracting flows coming out of the Lake (S77) and dividing by the total flow from S79. Daily flows from S77 and S79 were downloaded from DBHydro and were provided by the South Florida Water Management District (SFWMD).

**Seagrass transects.** Ten transects are sampled at the end of the dry period (May) as part of an annual evaluation of seagrass within and near the Ding Darling National Wildlife Refuge. For each transect, a quadrat is placed at the shallow edge, a quadrat is placed at the deep edge, and quadrats are taken at two randomly selected positions along the transect. Shoot density, percent cover, and canopy height are determined for each seagrass species present and for macroalgae (Thompson et al. 2013).

**Statistical analysis.** The frequency and duration of low salinity events, defined by salinities lower than 25, was analyzed using AQUARIUS data parsing tools. A monthly summary by site of the number of days, average salinity, and sample size was calculated. A monthly summary of flow from S77 and S79 was also tabulated along with the proportion of flow from S77 as a percentage. Kendall seasonal trend analysis (WQStatPlus, Sanitas Technologies) was performed on light attenuation, salinity and optical properties for all RECON sites (Hirsch et al. 1982). A General Linear Model (GLM) analysis tested whether chlorophyll *a* (chl *a*), turbidity and FDOM were significantly different between salinities above or below 25. Seasons were defined as wet from June 15 to October 15 and dry from Oct 16 to June 14 based on rainfall data from 2004-2014. Pearson correlation analysis (MINITAB) was used to relate salinity with optical properties. Descriptive statistics (MINITAB) were used on seagrass shoot density data.

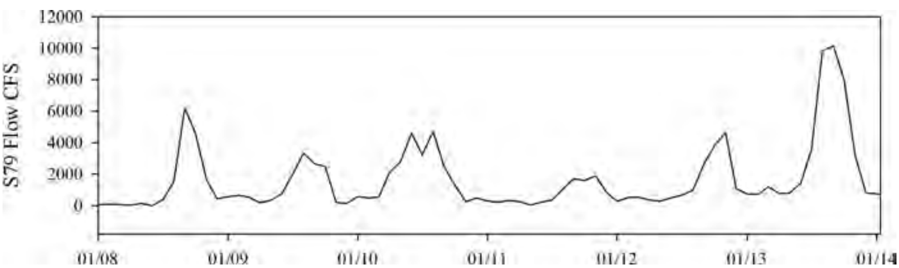


Figure 2. Monthly mean flow from S79 during the study period. The landfall of TS Faye in 2008 and the high flows in 2013 resulted in low salinity periods in the lower estuary, as measured by RECON. Daily flow data were provided by DBHYDRO and the SFWMD.

## Results

**Shell Point.** Salinities were routinely below 25 at Shell Point during the study period (Figure 3), especially during the summer wet period each year. There were 4 years where average monthly salinities during the dry period did not go below 25 (2008, 2009, 2011, 2012). In the remaining 3 years (2010, 2013, 2014), monthly salinities were below 25 during the dry period for an average of 10.77 days. Wet period monthly salinities were lowest in 2013 (10 days) and highest in 2012 (27 days), when comparing wet periods in all years. The duration of low salinity periods during the wet period was between 7-9 days per month for most years. The exception was in 2013 where salinities were below 25 for an average of 20 days per month and ranged from 9 days in June 2013 to 30 days in September 2013. Light attenuation (Kd PAR) ranged from 0.48-2.57 $m^{-1}$ with a mean of 1.13 ($n = 41$) during the study period (Figure 4). There were no significant differences in the optical properties between salinities $< 25$ and $> 25$. Photosynthetically active light attenuation (Kd PAR) did exhibit a positive trend (Seasonal Kendall, WQ Stats) from 2007 to 2012, but was not significant.

**Tarpon Bay and McIntyre Creek.** These two sites were grouped together because of their similarity in salinity responses to S79 flows. Salinities at these two sites did not go below 25 from January through June from the start of monitoring until 2014 (Figure 3). However, Tarpon Bay salinities were not collected from 2008-2010. During 2008-2009, salinities at McIntyre Creek (USGS 2014) were above 25 during both the wet and dry seasons. Salinities were below 25 for moderate periods (10-20 days) at the end of the wet period and beginning of the dry period (August-December) at McIntyre Creek and Tarpon Bay. In 2010, salinities at McIntyre Creek were below 25 in both August and October (13 days) with average monthly salinities from 27-31. In 2011, salinities at McIntyre Creek were below 25 for 10 days. In 2012, mean monthly salinities at McIntyre Creek were 24.5-31.3 for 2 months (October-November) and salinities were below 25 for 16 and 18 days, respectively. In 2012 in Tarpon Bay, salinities were below 25 for the same months and for slightly longer duration. Salinities at both McIntyre Creek and Tarpon Bay sites were lower and lasted for longer duration in 2013 because of high flows from S79. At McIntyre Creek, salinities were below 20 for all of September and October 2013 and most of August and November. Similarly, salinities at Tarpon Bay were below 20 for extended periods in September (29 days) and October (24 days) during the high flows from S79 in 2013. There was not sufficient light attenuation data at either site to conduct the seasonal trend analysis.

**Blind Pass.** In 2008, salinities at Blind Pass (Figure 3) were below 25 in August 2008 for only 3 days (mean salinity 21.24) and in September for 5 days (23.59). From August 27 to September 3, 123 of 193 observations (64%) were below 25. Significantly higher FDOM ($F = 20.42$, $p < 0.01$) and chl *a* ($F = 9.10$, $p < 0.01$) were found in the water where salinity was below 2. Salinities at Blind Pass were below 25 for brief periods from May-June 2008. Relatively short periods (3-7 days) of salinities less than 25 occurred again in July-September 2010. Periods of salinity below 25 were very short, lasting less than 1 day in 2011. The periods below 25 in 2012 occurred from August through October and were short (September, 6 days) duration events. In all observations where salinity was below 25, FDOM and chl *a* were significantly higher in low salinity water than FDOM and chl *a* in water where salinity was > 25 during the same month. Light attenuation at Blind Pass (Figure 4) from 2007-2012 ranged from 0.28-2.00 $m^{-1}$ with a mean of 0.96 $m^{-1}$ (MINITAB, n = 47). There was a significant increasing trend (95% confidence) in light attenuation during the study period (Kendall, WQStat Plus). Trend analysis indicated a 0.01 $m^{-1}$ increase annually in Kd PAR. There is no data from Blind Pass after 2012 because the piling it was attached to was struck by a large vessel and the instrument was moved to McIntyre Creek.

**Redfish Pass and Gulf of Mexico.** In 2008, salinity at Redfish Pass was below 25 a day in September after Tropical Storm Fay (Figure 3). Salinities at the Gulf of Mexico were also below 25 for 1 day (1 d) in August 2008 related to the same high flow event. During periods of salinity below 25 at the Gulf of Mexico site (n = 13) compared to periods above 25 (n = 191), there were significantly higher FDOM (MINITAB, GLM, $F = 180$, $p < 0.01$), and significantly higher chl *a* ($F = 41.37$, $p < 0.01$). However, turbidity was not significantly different. At Redfish Pass, optical properties were similarly compared during this event. There were significantly higher FDOM (GLM, Minitab, $F = 59.68$, $p < 0.01$), significantly higher chl *a* ($F = 28.36$, $p < 0.01$) and significantly higher turbidity ($F = 4.27$, $p < 0.05$) in the lower salinity periods. Another short period below 25 occurred at Redfish Pass in October 2008.

The other notable period where salinities were below 25 at the most marine sites (Redfish and Gulf) occurred from August-October, 2013. Salinities at the Gulf of Mexico in 2013 were frequently below 25 for several consecutive months (August-December, 2013), but only for short periods, usually at low tide for 2-5 hours. During those periods, significant differences in optical properties were found when salinities below 25 were observed. From July 21-27, 2013 there were 6 occurrences of salinity below 25 at the Gulf of Mexico site. Significantly higher FDOM (GLM, Minitab, $F = 73.24$, $p < 0.01$) and chl *a* ($F = 47.57$, $p < 0.01$) were associated with salinities less than 25, while differences in turbidity were not significant. At Redfish Pass, 150 of 2183 observations (7%) were below 25 from July 7, 2013 to October 5, 2013. Most of the below 25 observations occurred for short periods at low tide. During these observations where salinity was below 25, FDOM was significantly higher (GLM, Minitab, $F = 236.42$, $p < 0.01$), along with chl *a* ($F = 267.49$, $p < 0.01$) and turbidity ($F = 4.26$, $p < 0.05$). Light attenuation (Kd PAR) at the Gulf

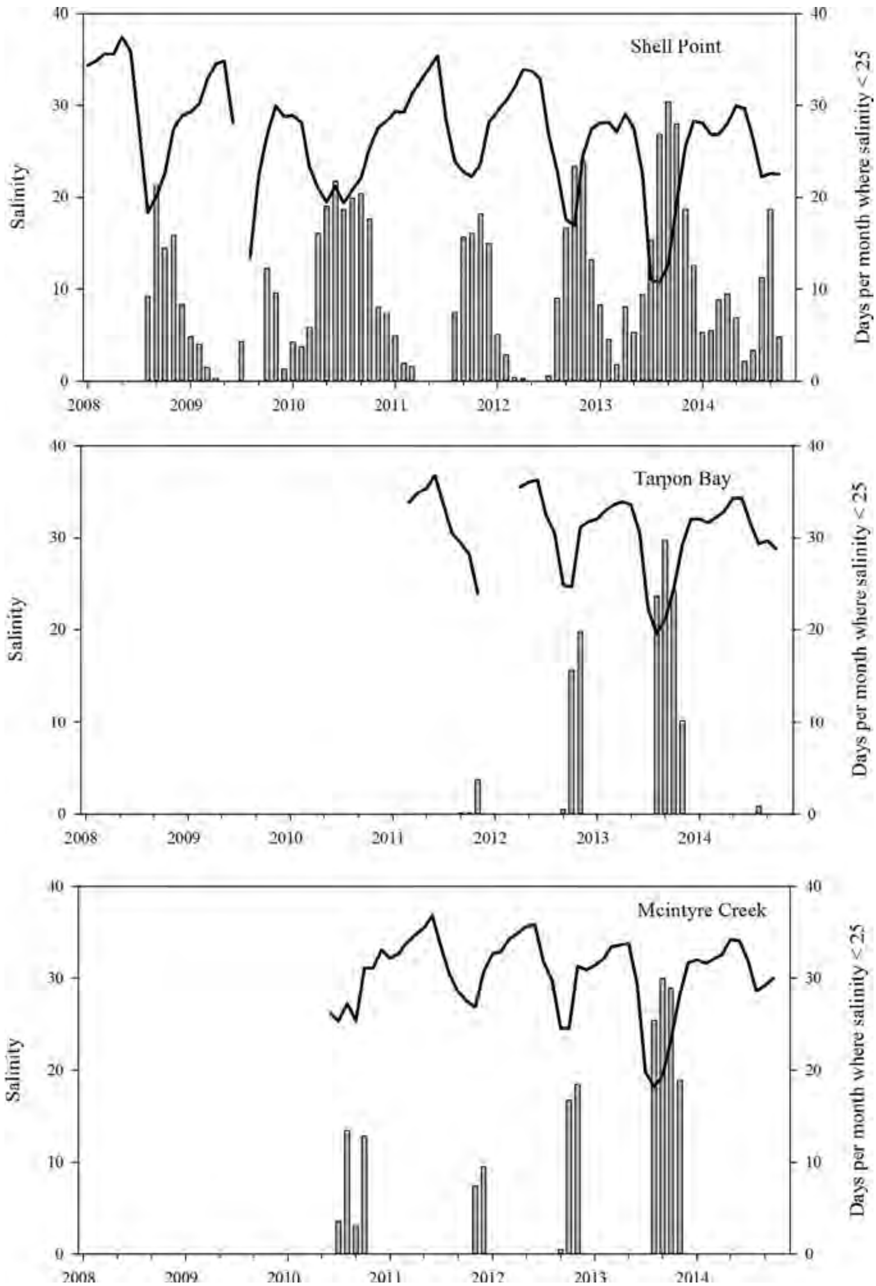


Figure 3. Frequency and duration of low salinity events recorded at RECON locations. Mean monthly salinity (line graph) and the cumulative number of days per month (bars) when salinity below 25 was measured. Sites are listed, generally, from upstream to downstream.

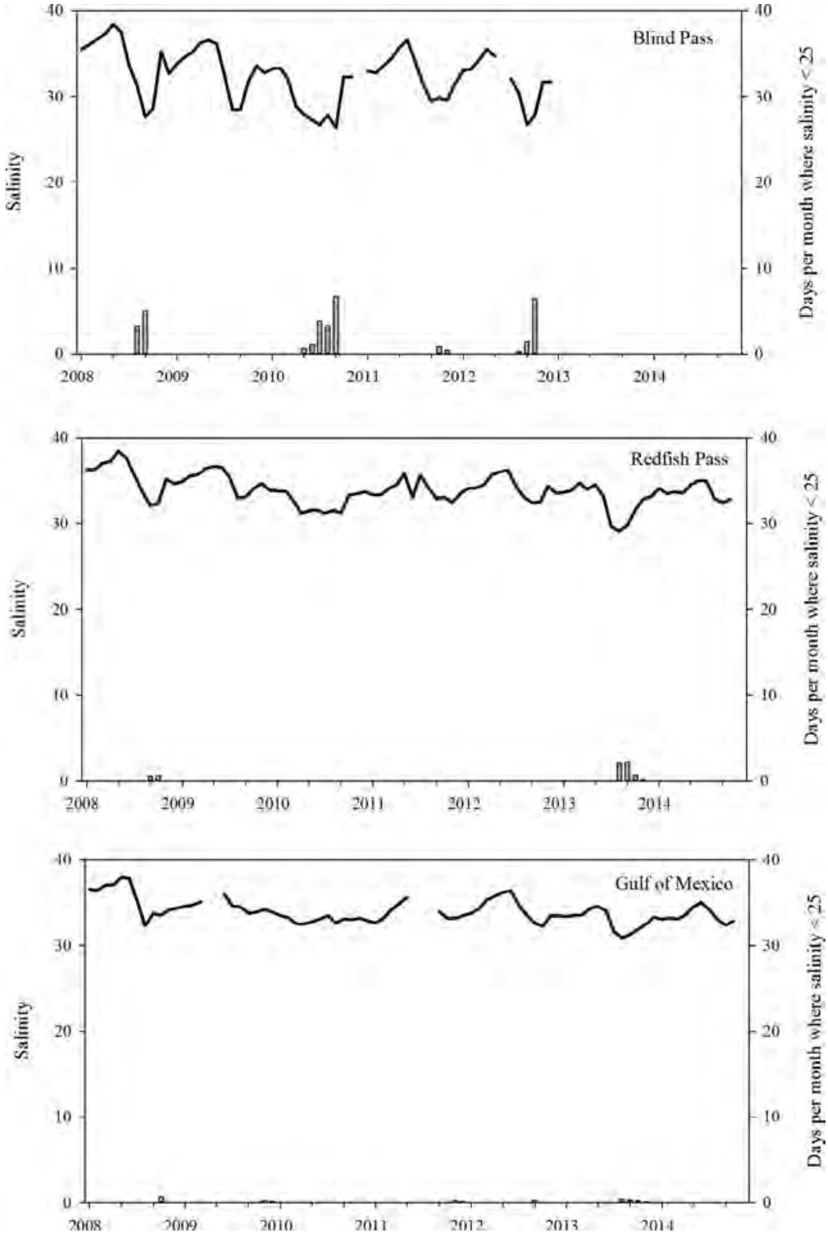


Figure 3. Continued.

of Mexico site ranged from 0.35-1.83 $m^{-1}$ with a mean of 0.82 $m^{-1}$ (Minitab, n = 30). No trends in light attenuation were found when similar seasons were grouped and analyzed at the Gulf of Mexico location (Seasonal Kendall, WQ Plus).

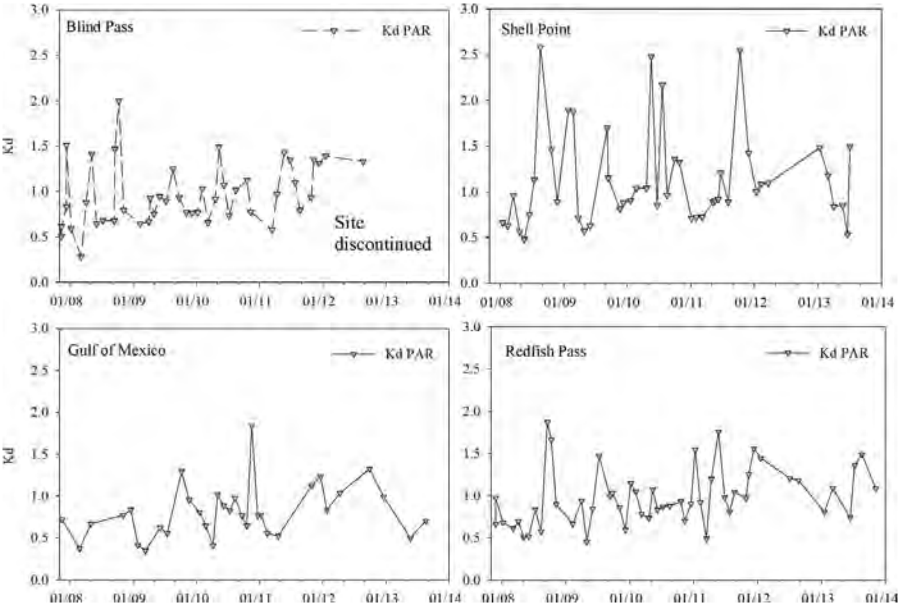


Figure 4. Light attenuation coefficients (KdPAR) recorded during monthly RECON maintenance and calibration service trips. Light attenuation is calculated from 2 underwater sensors on a lowering frame. KdPAR is the photosynthetically active radiation (400 nm-700 nm). Significant increasing trends in Kd were found at 95% confidence at Redfish Pass and Blind Pass but not at the Gulf of Mexico or Shell Point.

Light attenuation at Redfish Pass (Figure 4) ranged from 0.45-1.86 $m^{-1}$ with a mean of 0.97 $m^{-1}$ (Minitab, n = 46). There was a significant increasing trend (Kendall, WQStat Plus) in Kd PAR during the study period (2007-2012, 95% confidence).

**Seagrass shoot density.** A complete analysis of all nine transects is available in Thompson et al. (2013). The results from one transect in southern Pine Island are shown (Figure 5) and is located near the SCCF Marine Lab and the Tarpon Bay RECON site. Shoot density was variable by year. Shoot densities were highest for *Thalassia testudinum* in 2013 and increased from 2010-2013. There was a clear decrease in *Thalassia* shoot densities 2014. The shoot densities for *Halodule wrightii*, in contrast, were highest in 2010 and decreased to the lowest in 2012.

## Discussion

The high resolution salinity and optical properties (chl *a*, FDOM, turbidity) for the Caloosahatchee Estuary and Pine Island Sound provided by RECON were used to determine the frequency, duration, and effects of low salinity periods in the lower estuary. High volume flows from S79 as a result of tropical storms and/or water management in south Florida resulted in large-scale periodic decreases in salinity, increases in FDOM, and increases in chlorophyll *a*. Given

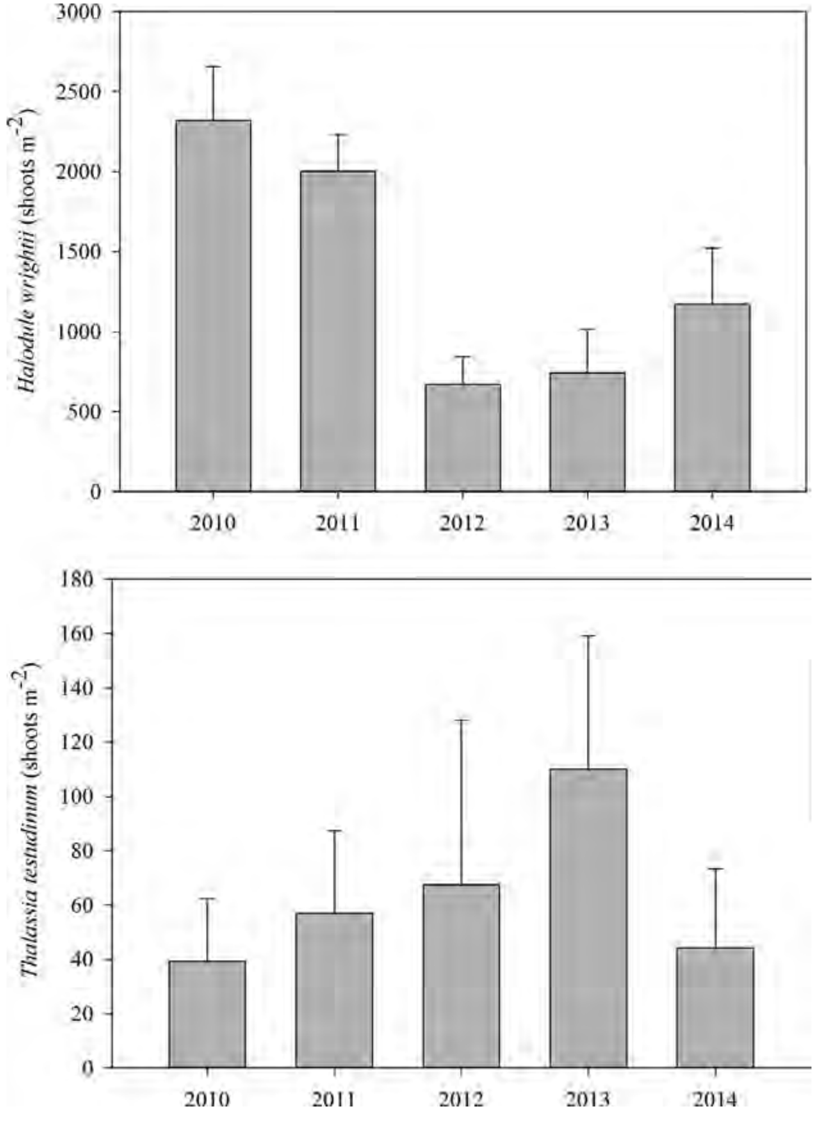


Figure 5. Seagrass shoot density for *Halodule wrightii* and *Thalassia testudinum* at transect NWR2 in southern Pine Island Sound. Annual seagrass monitoring within J.N. "Ding" Darling National Wildlife Refuge occurs at 10 sites (Thompson et al. 2013). These are the results from 1 site near the Tarpon Bay RECON.

that there are likely to be delays in the diversion of water to the Everglades and the creation of water storage projects, an analysis of how often and how long these events occur was needed along with a synthesis of the extent that these events affect water quality indicators (such as chlorophyll *a* and light attenuation) and biological indicators (such as SAV and oysters).

Salinities at Shell Point are often below 25, especially during the wet season because it is near the mouth of the Caloosahatchee which is relatively narrow and conveys water from S79 and the tidal watershed. Shell Point is also surrounded by oyster reefs that require periods of lower salinity (Volety et al. 2008). Typical wet season periods where salinities were below 25 were between 15-20 days per month. An exceptionally wet year with high flows occurred in 2013 where salinities were below 25 for an average of 20-30 days during wet season months. Large releases from S79 during 2013 combined with tidal basin flows likely resulted in slower oyster growth, poor spat production and excessive valve closure as salinities were below 14 (Volety et al. 2003). Monitoring sites near Shell Point (Volety and Haynes 2013) during this period indicated 100 percent mortality of spat.

While low salinity periods are common at Shell Point, it remained unclear whether and how high discharges from S79 and high tidal basin flows affect locations in Pine Island Sound or the Gulf of Mexico. High frequency data collected by RECON allowed for the determination of the frequency and duration of low salinity events. Two events were evident in this analysis; the landfall of Tropical Storm Faye (August 2008) and the wet season of 2013. The S79 average flow during August 2008 was 6,202 CFS with a maximum occurring on August 20, 2008 of 18,139 CFS. By looking at RECON sites and data, the event propagated throughout the lower estuary over the course of August and September, depending on the distance from the Caloosahatchee, with greater distances seeing low salinity events at a later time. At the Gulf of Mexico site, the Tropical Storm Faye event caused salinities to fall below 25 for a total of less than 1 day during September 2008 (Figure 3). The low salinity event was recorded at other RECON locations in 2008. Salinities at Blind Pass were below 25 in August 2008 for 3 days and in September for nearly 5 days. During the last week of August and first week of September of 2008, 64 percent of the salinity observations at Blind Pass were below 25. This large pulse of water from Tropical Strom Faye was widely distributed but fairly short lived. The water associated with the low salinity pulse had significantly higher FDOM and chlorophyll *a* which would affect the light field for seagrass but the short duration likely had little to no effect.

The wet season of 2013 brought a long duration event where low salinities occurred in consecutive months (July-November). The flows recorded at S79 were well-above recommended maximum flows (2,800 CFS) the maximum monthly average occurring in September (10,160 CFS) with 56 percent coming from S77 (Lake Okeechobee) and 44 percent from the Caloosahatchee watershed upstream of S79. This event was caused by the lack of water storage in South Florida combined with a record-setting rainfall throughout south Florida. The Caloosahatchee and St. Lucie Rivers are 'relief valves' for excess water and high lake stage levels in Lake Okeechobee. This event was most evident in lower Pine Island Sound RECON sites such as McIntyre Creek and Tarpon Bay. At both of these sites, salinities were around 20 for several months. The extended low salinity period is thought to cause seagrass leaf losses in July and August and resulted in unusually large wrack piles on nearby shorelines (Milbrandt, personal observation). The scale of the 2013 wet season low salinity event also was evident at Redfish Pass and the Gulf of Mexico RECON sites, despite the long distances from the mouth of the Caloosahatchee. At Redfish Pass, low salinity periods were up to 2 days, while at the Gulf of Mexico site, low salinity events lasted 2-5 hours. The longer durations likely reflect greater mixing as the saltwater and freshwater mix in route to Redfish Pass. However, there is limited data available on stratification throughout the Charlotte Harbor and lower Caloosahatchee Estuary to completely understand the duration and frequency of low salinity periods from RECON's fixed depth platforms.

All of the low salinity events had significantly higher FDOM and higher chlorophyll *a* than higher salinity events that occurred during the same time. These properties are potential threats to *Thalassia* and *Syringodium* (Irlandi 2006), which comprise a large proportion of the mapped seagrass in the region (Brown and Stafford 2011). Salinities below 25 are known to have harmful effects on seagrasses and they face additional sublethal effects of lower light availability (Dennison et al. 1993).

While the RECON sensors provide high frequency data about salinity and optical properties, the maintenance and calibration activities that occur monthly have generated a monthly dataset that includes the empirical measurement of the light field in the upper 1 m of the water column. Light attenuation showed significant increasing trends at several RECON locations, including Shell Point, Blind Pass, and Redfish Pass. The salinities at Blind Pass and Redfish Pass also showed significant decreasing trends when the data are considered by season (wet, dry). Correlations confirmed that lower salinities were also associated with higher FDOM and chlorophyll *a* and offer a cause for the increasing trends in light attenuation.

An increasing trend in light attenuation should be evaluated carefully by researchers and resource managers. Seagrass aerial extent as determined by photointerpretation has limitations (Robbins 1997) and may not be sensitive enough to detect incremental declines (or increases) in seagrass habitat. The lack of sufficient ground-truthing of the interpreted aerial images for a large area like Pine Island Sound means that any losses of habitat and degradation of habitat function may not be detected. Results from one transect in Tarpon Bay suggest that shoot densities are not following a linear response to either salinity or the light field. Instead, there appears to be a biological interactive effect between species where after periods of low salinity the community shifts from a *Thalassia*-dominated to a *Halodule*-dominated or macroalgal-dominated community. This pattern was also found by Orlando and Douglass (2014). The role of macroalgae as a competitor for space and light and as a false positive in aerial interpretation of SAV cover should be addressed to better interpret seagrass transect data collected by the SCCF, SFWMD, SWFWMD, FDEP and others. While the presence of macroalgae in seagrass ecosystems is not a rare occurrence (Brown and Stafford 2011), others have shown macroalgae to be an indicator of nutrient loading in Buzzards Bay, MA

(Valiela et al. 1997). A manipulative, hypothesis-based approach to understanding these interactions in lower Pine Island Sound is needed. Additional ground-truthing of aerial imagery through hydroacoustic, deep-edge mapping or by drop camera mapping is recommended.

Literature suggests that sustained periods of low salinity and low light threaten seagrasses (Irlandi 2006) but it is unclear how high flow events are affecting seagrasses. It is possible that the decreased shoot densities of *Thalassia* in 2014 represents an eventual response to extended low salinity periods in 2013. The nearest RECON site is Tarpon Bay which had 4 consecutive months in 2013 of 20 or more days of salinities $< 25$. It is also possible that seagrasses retreated to shallower depths after 2013 but the data shown here do not consider the deep edge. The field experiment was not designed to examine changes in depth distributions during low salinity events. Shoot densities vary due to biological and physical factors and this study demonstrates that further investigation of deep edges and seagrass areas around the RECON sites is needed.

The establishment of optical targets (Corbett and Hale 2006, Janicki et al. 2010) represents a supplemental method to monitoring using aerial images to meet the goal of long term sustainability and maintenance of the seagrass resources. The optical targets are more sensitive to incremental changes in water quality and optical properties. Significant increasing trends in FDOM at Redfish Pass should serve as an indication that water clarity is declining (Dixon and Wessel 2014) and that nitrogen loads are increasing, given the positive linear correlation between FDOM and total nitrogen (Baldwin et al. 2009). Increased storage of storm water and the diversion of water south to Florida Bay through the Everglades as envisioned by the Comprehensive Everglades Restoration Plan (CERP) can stabilize light attenuation trends and potentially result in restoration of SAV habitats and habitat function.

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# Analysis of nutrients and chlorophyll relative to the 2008 fertilizer ordinance in Lee County, Florida

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**Abstract** Research and qualitative observations suggest that a major contributor to algae growth in stormwater ponds is the nitrogen (N) and phosphorus (P) contained in fertilizers that enter the ponds via runoff from lawns and impervious surfaces. In May 2008, the Lee County Board of Commissioners enacted a fertilizer ordinance with the objective of lessening discharge of nutrients to stormwater ponds. It prohibits applying fertilizers containing N and P during the four wet summer months (June through September). The Pond Watch Program is a citizen volunteer monitoring program that helps understand and manage community ponds. This study examines Pond Watch data to compare N, P and chlorophyll *a* levels in similar urban stormwater ponds during the wet months of 2004 through 2008 (prior to the fertilizer ordinance enforcement) compared to 2009 through 2013 (after enactment). The results showed a statistically significant difference in the reduction of levels between pre- and post-ordinance in total phosphorus and chlorophyll *a.* This was not the case for total nitrogen. The study suggests that the fertilizer ordinance may have had a positive effect on the reduction of nutrient concentrations in some stormwater ponds, which may contribute reducing the abundance of planktonic algae.

**Keywords** Chlorophyll, fertilizer ordinance, nutrients, Pond Watch, volunteer monitoring

## Introduction

During the summer months in southwest Florida, stormwater ponds exhibit from time to time frequent algae blooms (Greening et al. 2014). Crane and Xian (2006) reported that a major contributor to these algae blooms was the increased amount of nitrogen (N) and phosphorus (P) by anthropogenic activities associated with urban sprawl. N and P contained in fertilizer may enter stormwater ponds from lawns and landscapes, runoff from impervious surfaces and water that percolates through predominantly sandy soils. Ultimately, these waters containing excess nutrients may enter natural water bodies and coastal waters (FDEP 2009).

In 2008, Lee County Board of Commissioners enacted a fertilizer ordinance (Lee County Ordinance 08-08) (Lee County Florida 2008) after technical recommendations from the South West Florida Regional Planning Council and several non-governmental organizations (SWFRPC 2007). These laws became mandatory during the wet months of 2009. Among many of the

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Table 1. Nine ponds selected from the Pond Watch Program to evaluate nutrient contents.

| No. | Pond Watch ID | Subdivision | Area (ha) | Longitude | Latitude |
|---|---|---|---|---|---|
| 1 | 1 | Stone Bridge | 0.4 | −81.896515 | 26.506573 |
| 2 | 4 | Peppertree Pointe | 2.1 | −81.949689 | 26.519921 |
| 3 | 14 | South Pointe South | 2.0 | −81.909797 | 26.547372 |
| 4 | 35 | Corkscrew Woodland | 7.5 | −81.777317 | 26.424099 |
| 5 | 37 | Wellington | 2.5 | −81.900010 | 26.502472 |
| 6 | 42 | Wyldewood Lakes | 0.2 | −81.888100 | 26.562310 |
| 7 | 47 | South Wind | 3.1 | −81.897330 | 26.485810 |
| 8 | 54 | Candlewood Lake | 6.3 | −81.966610 | 26.509190 |
| 9 | 57 | Caloosa Creek | 4.0 | −81.969470 | 26.513330 |

best management practices, the ordinance prohibits the application of nitrogen (N) and phosphorus (P) (from fertilizers) during the four wet summer months (June 1 through September 30) with the stated objective of lessening nutrients in stormwater ponds and other waters that run into major bodies of water.

The Lee County Hyacinth Control District created the Pond Watch Program in the early 1990s to educate citizens in the management of ponds and aquatic weeds. The program is a citizen volunteer monitoring initiative that involves sampling and analyzing numerous stormwater ponds for water quality chemistry on a monthly basis. This paper examines Pond Watch data to compare the amount of nitrogen, phosphorus and chlorophyll *a* in relation to the implementation of the fertilizer ordinance.

This paper investigates whether or not there is a difference in the concentration of nutrients present in selected stormwater ponds in the summer months of 2009 through 2013 compared to the years before the Lee County fertilizer ordinance was adopted (2004 through 2008).

## Methods

**Sampling location.** The volunteer monitoring program Pond Watch receives water samples for analysis every month from community ponds in Lee County, Florida. In the interest of having a representative sample of ponds for comparison, we selected ponds with the following basic characteristics: 1) pond surface area between 0.2 to 7 hectares; 2) pond depth not greater than 3.6 meters; 3) pond surrounded by single family housing; 4) lawns maintained by the homeowner and/or private landscaping contractors; and 5) water quality data available from 2004 to 2013. Nine ponds met these criteria (locations provided in Table 1).

**Sample collection.** Samples were collected monthly by volunteers following the sampling protocol of the Pond Watch Program. Chemical analysis of the pond water was conducted at the Water Quality Laboratory of the Lee County Hyacinth Control 3District (DOH Certification # E25945, Florida USEPA ID. FL01214). Total phosphorus (TP) was determined using the ascorbic acid method (Standard Methods SM 4500PE) (APHA 1998). Total Kjeldahl Nitrogen (TKN) was determined using the block digestion procedure (SM 4500ND) followed by the phenolic method of ammonia determination (SM 4500NH3F). Nitrite and nitrate (NOx) were determined using the cadmium reduction method (SM 4500NO3E). The relative total nitrogen (RTN) reported was calculated by adding the TKN and the NOx concentrations. Chlorophyll *a* (Chl *a*) analysis was determined by acetone extraction and with fluorometric analysis (EPA Method 445.0-1 and 446.0) (Arar 1997, Arar and Collins 1997). The data for the pre-ordinance period was from 2004 through

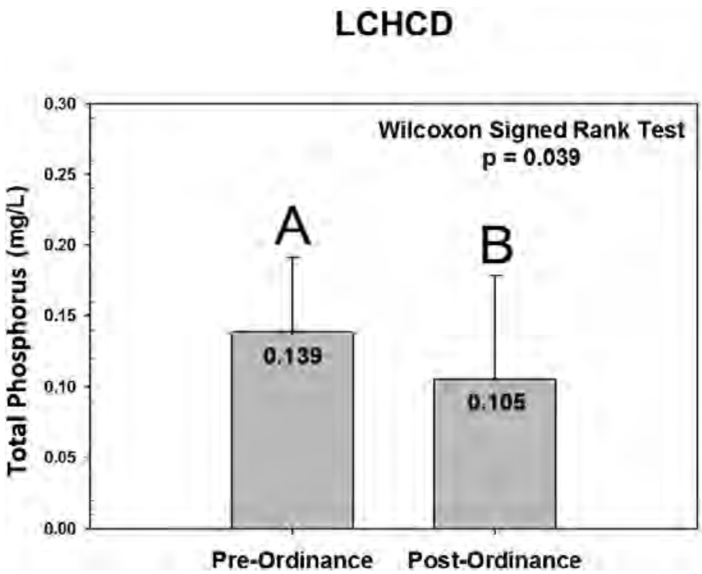


Figure 1. Graphic comparison of Total Phosphorus values pre- and post-fertilizer ordinance.

2008 because TKN analysis was introduced in 2004. The post-ordinance period was from 2009 through 2013.

**Statistical analysis.** A paired t-test and a Wilcoxon signed rank test were conducted using SigmaPlot 10.0 and SigmaStat 3.5 software (Systat Software 2014) to determine if there was a significant difference between the wet season months (June 1 through September 30) of pre-ordinance and post-ordinance average values for total phosphorus, relative total nitrogen, and chlorophyll *a*. The analyses conducted on the observations combined the data of all nine ponds. However, we also performed statistical analyses for each individual pond since volunteers wanted to answer questions regarding their behavioral change.

## Results

There was a statistically significant difference ($P < 0.05$) between the pre- and post-ordinance average for total phosphorus (Figure 1) and chlorophyll *a* (Figure 2) in the water of all nine ponds evaluated. The reduction in the overall combined concentration of total phosphorus (from 0.139 mg/L to 0.105 mg/L) and chlorophyll *a* (from 27.02 μg/L to 17.83 μg/L) was statistically significant. Relative total nitrogen did not present a statistical significant difference ($P > 0.05$) between the pre- and post-ordinance levels (Figure 3).

For total phosphorus, eight out of nine (8/9) stormwater ponds demonstrated a decrease in the concentration of phosphorus when comparing the wet months of pre- and post-ordinance periods (Figure 4). Although not statistically significant, relative total nitrogen in six out of the nine (6/9) ponds showed a decrease in the concentration of nitrogen (Figure 5). All nine (9/9) ponds demonstrated a decrease in the concentration of chlorophyll *a*, which is an indicator of the relative abundance of suspended planktonic algae in the water column (Figure 6).

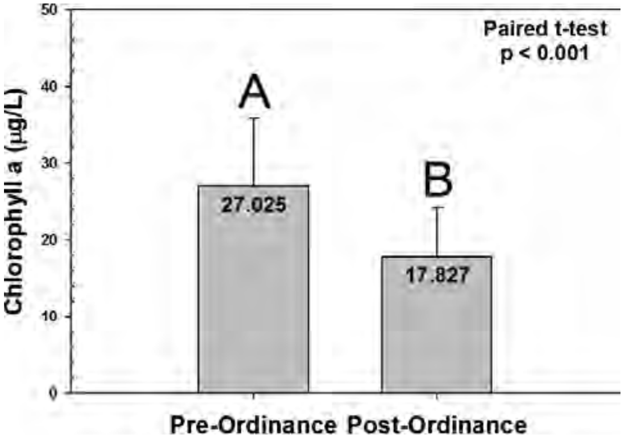


Figure 2. Graphic comparison of chlorophyll *a* values pre- and post-fertilizer ordinance.

Table 2 presents a summary of the nine stormwater ponds evaluated indicating the change (decrease, increase or no change) by pond for each parameter analyzed between the pre- and post-ordinance experimental groups.

We must clarify that the values from Pond 14 may have added a bias error. This pond has been treated between 2009 and 2013 with a special dye (AQUASHADE$^{TM}$) to minimize light penetration to control underwater submerged plants. This dye consists of small particles that increase the amount of phosphorus present in the water column; therefore, the values of TP increase in this pond.

## Discussion

The Lee County ordinance prohibiting the application of N and P in fertilizers during the rainy season from June through September went into effect in May of 2008. The Pond Watch Program of the Lee County Hyacinth Control

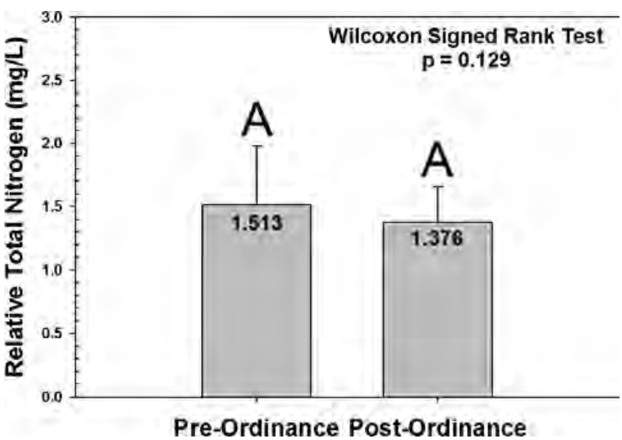


Figure 3. Graphic comparison of relative Total Nitrogen values pre- and post-fertilizer ordinance.

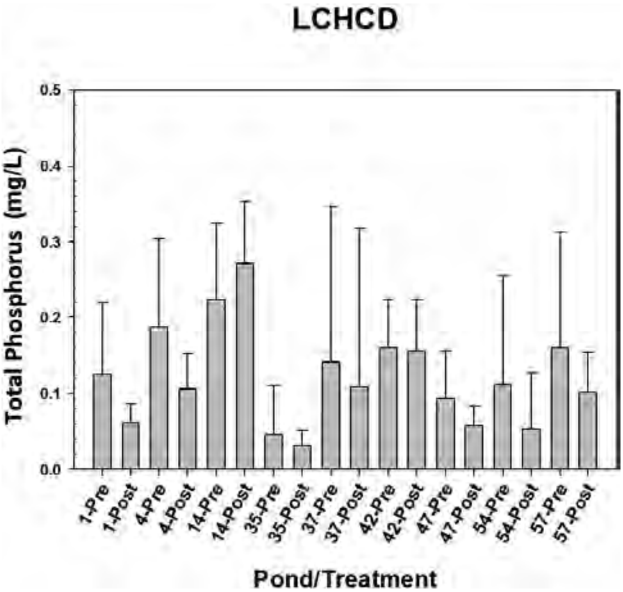


Figure 4. Graphic comparison of Total Phosphorus values pre- and post-ordinance per pond studied.

District has been collecting data in stormwater ponds throughout Lee County since 1992; however, it was only in 2004 when the laboratory added TKN analyses to determine total nitrogen (TN) with complement measurements of other species of nitrogen, such as proteins, ammonia, nitrites and nitrates. Rainfall varies in Florida between the wet months of June through September

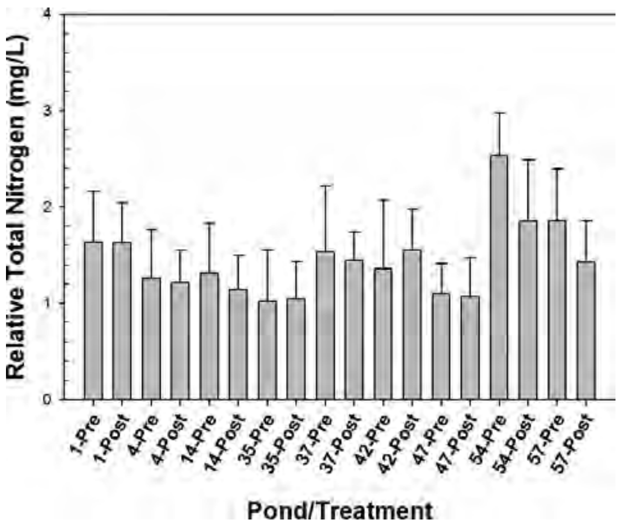


Figure 5. Graphic comparison of relative Total Nitrogen values pre- and post-ordinance per pond studied.

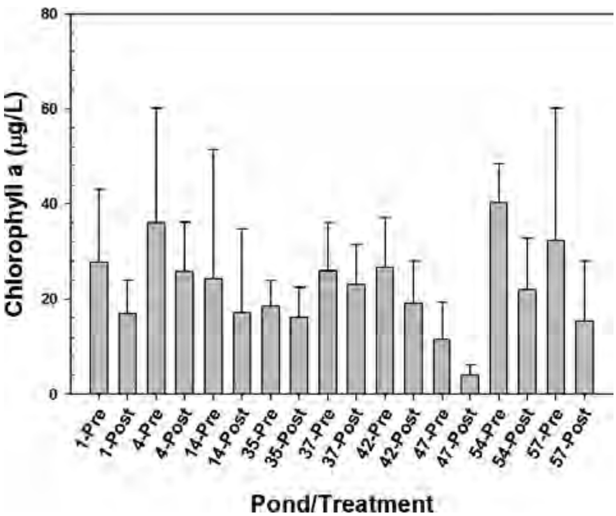


Figure 6. Graphic comparison of chlorophyll *a* values pre- and post-ordinance per pond studied.

as compared to the dry months of October through May (Sculley 1986). Thus, the comparison of pre-ordinance and post-ordinance reflects only the wet months. It has been suggested that the pre-ordinance period was not representative of a normal rain pattern for South West Florida since it registered one of the wettest year (2005) and the driest year (2007) in the study period (2004-2013) (data from the Lee County Rainfall Gauge Data website). However, a statistical analysis between the average rainfall between the wet seasons from 2004-2008 vs. 2009-2013 periods presented no significant difference ($P > 0.05$).

According to the results, total phosphorus (TP) was significantly reduced by 24.5% and chlorophyll *a* was significantly reduced by 34.0%, whereas the reduction for relative total nitrogen (RTN) of 9.2 % was not significant. These statistically significant values demonstrate that the reduction was considerable for the overall values of P and chlorophyll *a*. Furthermore, it was of greater value for the individual communities to see whether their landscape practices had an effect on the condition of the stormwater pond (see Table 2). Home

Table 2. Summary of changes per community.

| Pond | Location | TP | RTN | Chl *a* |
|---|---|---|---|---|
| 1 | Stone Bridge | DECREASE | No Change | DECREASE |
| 4 | Peppertree Point | DECREASE | decrease | decrease |
| 14 | South Point S. | increase | decrease | decrease |
| 35 | Corkscrew W. | decrease | increase | decrease |
| 37 | Wellington | DECREASE | decrease | decrease |
| 42 | Wyldewood L. | decrease | increase | decrease |
| 47 | South Wind | decrease | decrease | DECREASE |
| 54 | Candlewood L. | DECREASE | DECREASE | DECREASE |
| 57 | Caloosa Creek | decrease | decrease | decrease |

Note: Significant reduction indicated by capital letters.

owner associations, community development districts and neighborhood associations play an important role enforcing and educating their communities (Hartman et al. 2008). According to Greening et al. (2014), one factor addressing the impact in watershed-based nutrient management in Tampa Bay, among many others, was the citizen involvement reducing the residential fertilizer use and collaborative actions with local regulatory programs.

The results of this study suggest that the fertilizer ordinance may have had a positive effect on the reduction of nutrient concentrations in some stormwater ponds, which may have contributed to the reduction of the relative abundance of planktonic algae. We hope that the fertilizer ordinance contributes to the improvement of water quality conditions in runoff water reaching larger water bodies.

**Acknowledgements** The authors would like to thank all the volunteers that collected and continue to collect samples for their communities as a volunteer effort to improve the condition of their ponds and the waters that affect them. A big thank you to Dr. Dave Karlen for his assistance in the statistical analysis of the data.

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# Groundwater seepage nutrient loading in a recently dug wet detention stormwater pond

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**Abstract** Most of the 10,000 urban wet stormwater ponds found in southwest Florida were constructed post 1980 to protect State waters. Ideally, the impounded polluted water runoff is detained, phyto-remediated and released at the right time at mandated acceptable pollutant levels to the above and underground hydrosystems. However, most studies seem to show the poor performance for these ponds to treat surface runoffs while their groundwater connection is very largely overlooked. This one-year study examined, at high spatiotemporal resolution, the groundwater seepage nutrient loading of a 0.47 ha karstic wet detention pond which is nutrient rich despite a small mostly pervious underused watershed. This 7,000 $m^3$ water body behaved as a seepage pond (i.e. the pond did not overall recharge the aquifer) with groundwater fluxes typical of other lakes in Florida. Fluxes were positively correlated with rainfall during the dry season but not during the rainy season. Higher fluxes in the northeastern portion than in the southwestern portion were in agreement with the subsurface flow pattern in the region. Groundwater nutrient concentrations were high for total phosphorus and typical for total nitrogen. Groundwater nutrient loading could explain the resulting eutrophic conditions of the pond.

**Keywords** Florida, groundwater seepage, nutrient loading, stormwater pond

## Introduction

Since the early 1980s, in the face of past significant environmental degradation (e.g. Porter and Porter 2002), the State of Florida mandated that actions be taken to reestablish natural hydropatterns and prevent pollutants (especially nutrients) from reaching natural hydrosystems. In particular, Chapter 62-40 of the Florida Administrative Code was enacted so that stormwater runoff was slowed down in order to i) prevent erosion, ii) allow siltation/sedimentation prior to reaching natural hydrosystems, iii) promote soil filtration for pollutant removal, and iv) promote aquifer recharge. Through Chapter 62-40, stormwater pollutants were to be reduced by 80% with respect to the State Water Quality Standards. This figure was changed to a 95% reduction when such stormwater emptied into an Outstanding Florida Waterway (OFW). Several different types of stormwater management systems exist and range from swales to dry and wet detention and retention ponds placed judiciously to intercept stormwater and provide flood protection as well as fill for construction. In Lee

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County, Florida, these systems are widely used since this heavily developed region borders the coasts and large (e.g. Caloosahatchee River) or small coastal rivers (Imperial River) emptying into OFWs (e.g. Estero Bay). Although the total number of these systems is unknown, a census of all the wet urban and agricultural wet ponds accounted for 7,632 water bodies in 2012 (e.g. Thomas 2014) for a total surface area of 57.2 $km^2$ or 1.8% of the County's footprint. It is unknown whether these ponds function adequately as studies are rare (e.g. Harper and Baker 2007) or not readily available (e.g. non-disclosed studies or unpublicized ones). Harper and Baker (2007) report that wet detention ponds are the least efficient and often do not meet the 80% pollutant reduction while wet retention ponds overall meet that requirement but that dry retention ponds are better. A more recent study on wet ponds in the City of Naples, located in the adjacent southern Collier County (AMEC 2012), showed that most ponds were well below 80% pollutant reduction for Total Phosphorus (TP) and Total Nitrogen (TN) with often negative retention rates, especially for copper. This suggests that stormwater ponds could export more pollutants than they receive as they age.

Residential ponds (e.g. ponds in London, UK) (Birch and McCaskie 1999) were often built so as to increase the shoreline available for adjacent houses leading to excessive nutrient loading from fertilized lawns and impervious surfaces. These nutrients led to the development of macrophytes and microphytes, which were then suppressed with herbicides and algaecides, especially copper based algaecides. Deposition of dead macrophytes and microphytes and shoreline erosion due to a lack of rooted macrophytes in the littoral zone led to decreased water storage and overall pond life (e.g. Thomas 2014). It is noteworthy that pond pollutant and hydraulic retention are based on surface water runoff only (e.g. AMEC 2012) but the underground components are often not taken into consideration. This might explain why dry and wet retention ponds appear to be better at sequestering pollutants (Harper and Baker 2007).

Groundwater has a significant, but unseen influence on the ecology of natural lakes and coastal environments since it can be an important source of water and nutrients to these systems (Kang et al. 2005). This is especially true south of Lake Okeechobee where soils are porous (e.g. karstic) and groundwater tables shallow (e.g. Schiffer 1998, Lee et al. 2014). Most groundwater seepage studies involving direct seepage measurement are rare and rely on hydrologic data (e.g. water level), precise topographic maps and modeling (e.g. Grubbs 1995) but uncertainties remain (e.g. Lee and Swancar 1997). Direct measurements in Florida include natural lakes located north of Lake Okeechobee since very few natural lakes exist south of it. Examples include Lake Tohopekaliga (Belanger and Mikutel 1985, Belanger et al. 1985), Lake Conway and Apopka (Fellows and Brezonik 1980) and more recently Lake Jessup (Harper 2011). Information from indirect groundwater seepage measurements made on detention ponds are only available for three detention ponds in West Central Florida (Pinellas County, Fernandez

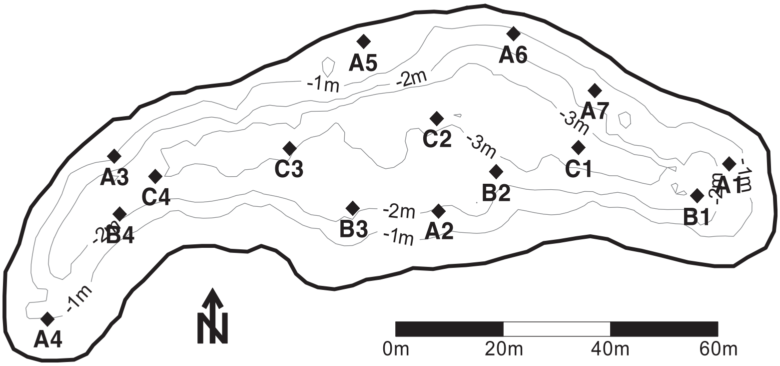


Figure 1. Bathymetry of the pond studied with the 0 m isobaths delineating the pond when full (3/3/2013). Closed diamonds represent the positions of the seepage meters. Isobaths units are in meters. Note that meters A5 through A7 included were out of the water during the dry season.

and Hutchinson 1993) and one pond in Central Florida (Orange County, McCann and Olson 1994).

Directly measuring groundwater fluxes can be difficult, but groundwater seepage meters are valuable using "less inventive water budgets" (Deevey 1988) even though spatial variation can be high within each system (e.g. Brock et al. 1982, Shaw and Prepas 1990).

This study aims to examine with groundwater seepage meters (Lee 1977) groundwater fluxes in a 3 year old eutrophic wet detention pond and to document whether the pond acts as a source of water and nutrients to the surficial aquifer or as a sink for water and nutrients. Furthermore, groundwater seepage is generally heterogeneous spatially and temporally. This study thus also purposes to examine spatiotemporal variability using numerous seepage meters per acre of pond and various temporal scales ranging from monthly to seasonal.

## Materials and Methods

**Study site.** The wet detention pond surveyed from 03/2013 to 02/2014 is an oblong, East-West oriented, 0.47 ha, 3 years old eutrophic detention pond located within Florida Gulf Coast University campus (17N 422079mE 2927505mN, Figure 1 and Figure 2). The pond is bordered to its north and west by a 16-acre flat lot paved with pervious crushed limestone that makes the foundations of the solar panels field. It is bordered to its south by preservation lands high enough for scattered slash pines to be established and low enough for the establishment of a short hydroperiod wetland (Figure 2). To the east, the pond is immediately bordered by an elevated dirt road. The pond has a shoreline development index of 1.49, a volume of 8,400 $m^3$, a mean depth of 1.8 m and a maximum depth of 3.8 m (Figure 1). Prior to being dug, the excavation site roughly laid within the center of an 84 ha zone delineated by roads which includes disturbed pinelands scattered with short-hydroperiod wetlands (Figure 2). At a larger scale, the main campus, with 13 manmade ponds, is bordered to its north by a retired borrow mine pit of about 210 ha and 5.6 m in mean depth (Lakes Miromar and Como). Its shores are heavily urbanized by single family homes (Figure 2). The rest of the constructed portion of

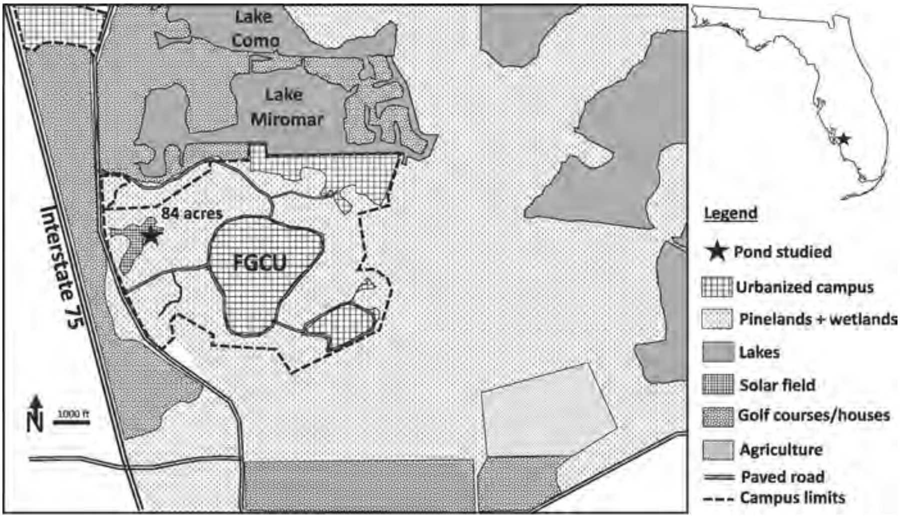


Figure 2. Location of the studied pond and description of the land uses surrounding it. Redrawn to scale from Google Earth Pro satellite imaging (3/31/2014).

the campus is built within disturbed pinelands and short-hydroperiod wetlands. To the east, these lands are bordered by borrow pit mine lakes and to the south and the west by golf course communities (Figure 2).

**Seepage meters location.** Water fluxes whether positive (i.e. groundwater influx or seepage) or negative (i.e. groundwater recharge) were assessed from March 2013 to February 2014 using 12 to 15 seepage meters. Because it was expected that most groundwater influxes would occur within the littoral zone (e.g. Fellows and Brezonik 1980, Brock et al. 1982), seven meters (A1-A7) were placed nearly equidistant from each other within the 0-1 m depth zone (Figure 1). This shallow zone utilized all seven meters when water levels were high from August to November 2014, six meters (all but A6) in December 2014 when water levels began to drop and four meters (A1 through A4) during the rest of the study. These four meters were always used as they were submerged during the entire year and positioned in the southern ¾ portion of the pond (meters A1-A4, Figure 1). The littoral zone of the northern quarter of the pond was much steeper and rockier and provided an inadequate seal of the meters with the pond bed in the median portion of this shallow 0-1 m depth zone. Locations further up the bank, however, provided an adequate seal for the meters during the rainy season when the pond water level was higher. The depth zones between 1-2 m and >2 m each received four meters placed at regular distance intervals (meters B1-B4 and C1-C4 respectively, Figure 1).

**Meter construction and positioning.** Groundwater seepage meters have been used to understand the interconnection between groundwater and surface water in various water bodies including lakes (e.g. Boyle 1994), wetlands (e.g. Harvey at al. 2000), rivers (e.g. Libelo and Macintyre 1994), and coastal marine systems (e.g. Lewis 1987). A variety of designs have been used, dating back to the 1940s (e.g. Israelson and Reeve 1944) but it was not until Lee's 1977 study that an easy, affordable, and accurate way to consistently measure groundwater seepage was developed. Each meter consisted of a 55 gallon steel drum half encapsulating 0.26 $m^2$ of pond bed (Lee 1977). Each meter was cut unevenly at the base so that when the base was driven down into the pond's bed to create a good seal, the top of the meter was slightly tilted. This allowed gas bubbles to vent out through a 1.27 cm diameter outlet connected to a clear vinyl tube which was positioned at the highest point of the meter with respect to the pond's bed. Water fluxes were assessed by rubber strapping a clear 15 cm×30 cm plastic bag (1.5 mm thick) to the end of the vinyl tube. Collection bags were thin walled to reduce resistance to flow (Shaw and Prepas 1989, Asbury 1990, Murdoch and Kelly 2003). Harvey et al. (2000) found that larger diameter connection materials reduced resistance of flow allowing for more accurate measurements. Bags were placed via SCUBA or snorkel to prevent disturbance of surrounding sediment. Each bag was prefilled with 500 ml of deionized water and care was taken so that no bubbles were trapped in the bag when it was connected to the vinyl tube. Bag deployments were set to 24 h for the monthly seepage meter assessment. Bags were retrieved by SCUBA and chilled in an ice chest. The water volume in each bag was then measured to the nearest ml and, in the event of a net water gain from May 2013 to February 2014, water was transferred into a 100 ml PTFE bottle which was filled to the top and subsequently frozen at $-18°C$.

**Nutrients analysis.** Nutrient analyses were performed for all bags which received groundwater with the exception of March and April 2013. Once thawed, the water from the collection bag was analyzed within 6 months with a Cary-100 spectrophotometer for TP (APHA 2012) and TN (Bachmann and Canfield 1996). A long holding time of the samples in the freezer does not affect the analyses of TP (Lambert et al. 1992) and TN (Bachmann and Canfield 1996). The water concentration of the groundwater influx was then corrected from the dilution occurring in the prefilled bag with deionized water. The groundwater nutrient influxes (mass loading) were calculated by multiplying the water flux by the nutrient concentration ($mg/m^2/d$).

**Weather.** A Davis Vantage Pro2 weather station located at about 400 m to the east of the pond logged rainfall every 10 min.

**Computations and statistics.** Water flux (m/day or $1000\times l/m^2/d$) was calculated as $\Delta V/(\Delta t\times A)$, where $\Delta V$ ($m^3$) and $\Delta t$ (days) were the changes in net volume in the bag and net time, respectively, while A was the surface area encapsulated by the seepage meter ($m^2$). A correction coefficient of 1.25 was applied to all flux velocities to account for the resistance to water flow the meter's components generate (e.g. Asbury 1990, Harvey et al. 2000, Murdoch and Kelly 2003). The spatial variability of the water flux was assessed through mapping with Surfer 12 (www.goldensoftware.com) using the Kriging method and the adequate variogram model to spatially interpolate the data. Surfer was also used to calculate the net daily water gain or loss (l/d) for the pond within the polygon of interpolation. The daily net gain or loss of water and nutrient influx for the whole pond was calculated through extrapolation outside the polygon of interpolation using the correction factor "whole pond/polygon of interpolation" surface areas. The average water flux was determined by dividing the daily water gain or loss by the surface area of pond bed that day. Each daily flux was then expressed on a monthly basis by multiplying the daily flux by the number of days of the month. All resulting twelve monthly fluxes were then summed to estimate the yearly net groundwater exchange. This net exchange was then further divided by the yearly average volume of the pond to determine the yearly percent groundwater contribution. Daily nutrient influxes for the meters were computed by multiplying the daily water flux by the concentration of groundwater nutrients (i.e. nutrient mass loading in mg TP or $TN/m^2/day$). The mass of nutrients entering the pond was then computed daily by multiplying the average nutrient concentration by the daily groundwater flux to the pond (mg/d). This daily groundwater flux was finally divided by the pond volume to get the additional equivalent nutrient concentration (mg/l) brought to the pond. The cumulative monthly direct rainfall collected on the planar surface of the pond was calculated by multiplying the planar surface area of the pond for a given month and the total rainfall amount for the same month. All cumulative monthly rainfall volumes were then combined to estimate the yearly total volume of rainfall. Histograms, scatterplots and regressions were conducted in Microsoft Excel 2010 and averages are presented along with their respective standard deviation.

## Results

**Water exchanges.** Water fluxes ranged from $-2.0$ l/m$^2$/d (10/2013, meter A4) to 7.04 l/m$^2$/d (01/2014, meter A2) when all data are pooled together over space and time. With one exception, all maps of water fluxes exhibited higher values in the north to northeastern portion of the pond (Figure 3) than its south to southwestern portion. The one exception was in 01/2014 when higher water fluxes were observed at meters A4 and A2. This seepage event was marked with significant rainfall prior to and during the water seepage collection period (50.8 mm accounting for 78% of the precipitation in January). When water fluxes were averaged over the pond surface area, water fluxes ranged from $-0.28$ l/m$^2$/d (06/2013) to 1.47 l/m$^2$/d (01/2014, Figure 4). These figures equate to $-0.18$‰ and 0.89‰ water exchanges per day with the overall pond water volume. Water fluxes were positive 9 out of 12 months. They were positive during the dry season when there was low precipitation and negative or minimal during the rainy season (Figure 5). The resulting yearly groundwater recharge was 531 m$^3$ (i.e. pond water losses) and groundwater influx (i.e. pond water gains or seepage) was 984 m$^3$ or 13.6% of the average pond volume for the year. The net water exchange between the pond and the groundwater was 453 m$^3$ or 6.3% of the average pond volume. The yearly volume of rainfall over the pond surface was 5,280 m$^3$ or 73% of the average pond volume. There was a positive linear correlation between the amount of rainfall during the dry season and the average water flux for the whole pond ($P=0.02$, Figure 5). In contrast, no correlation was found during the rainy season (Figure 5).

**Groundwater nutrient loading.** Groundwater nutrient concentrations did not exhibit a clear seasonal pattern (Figure 6). It was 2.83+2.96 mg/L and 0.27+0.23 mg/L for TN and TP, respectively. Nitrogen groundwater seepage loading increased from 05/2013 with 2.01+1.23 mg/m$^2$/d to 4.93+5.60 mg/m$^2$/d in 11/2013 (Figure 7). TP groundwater seepage loading similarly increased from 0.21+0.09 mg/m$^2$/d in 05/2013 to 0.92+0.49 mg/m$^2$/d in 02/2014 (Figure 7). Thus the additional nutrient loading on a liter of pond water basis for the period 05/2013-02/2014 was 0.019+0.021 mg/l/d and 0.0014+0.0018 mg/l/d for TN and TP, respectively (Table 1).

## Discussion

This study is unique since groundwater seepage was measured at high spatiotemporal resolution with 32 seepage meters per hectare and in a 0.47 ha wet manmade detention pond instead of a lake. As a comparison, Harper (2011) used 0.006 meters per hectare (40 in Lake Jesup with 6,475 ha) while this figure was 0.003 meters per hectare (25 in Lake Tohopekaliga with

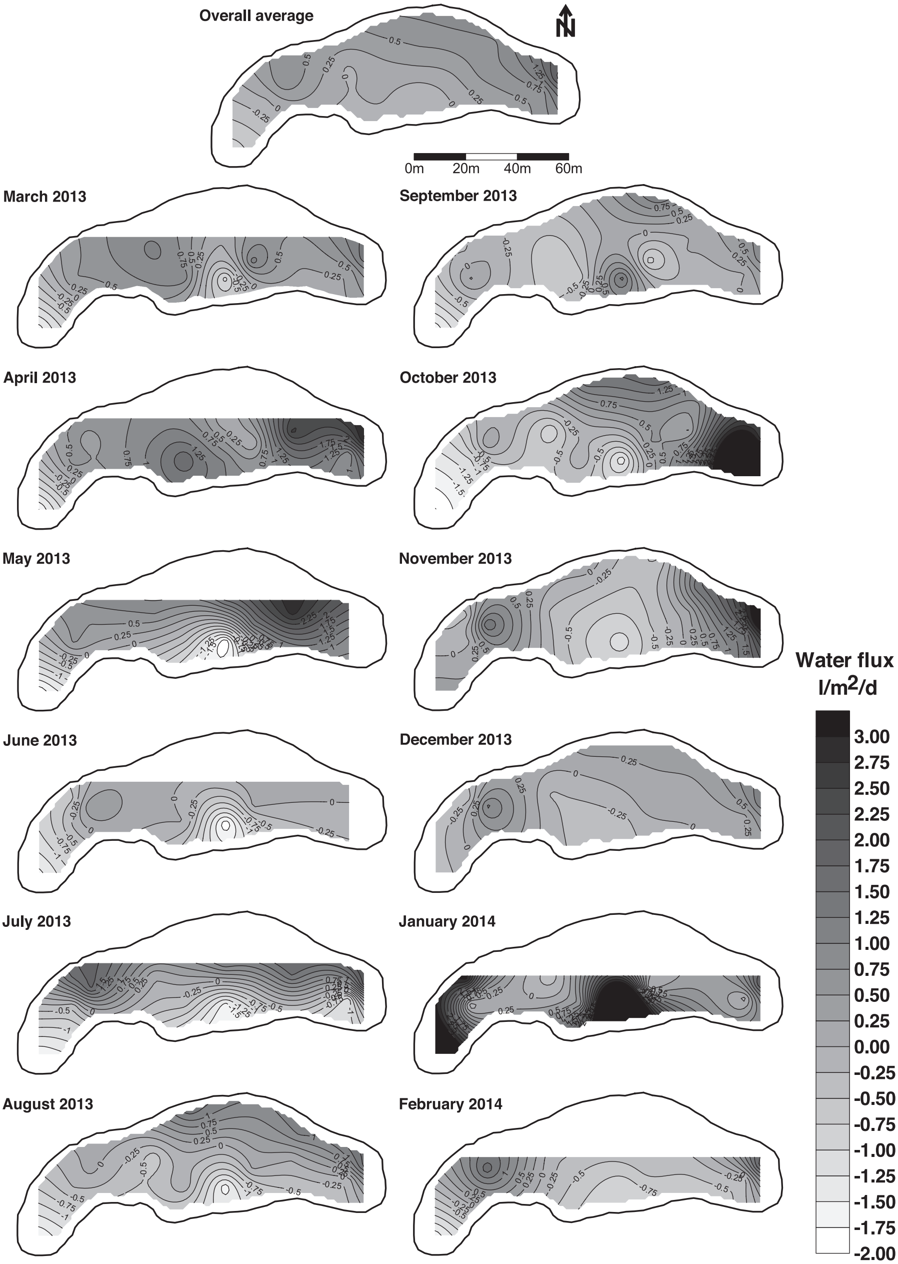


Figure 3. Spatial variations of water fluxes from March 2013 through February 2014. The overall average fluxes are also depicted in the top map.

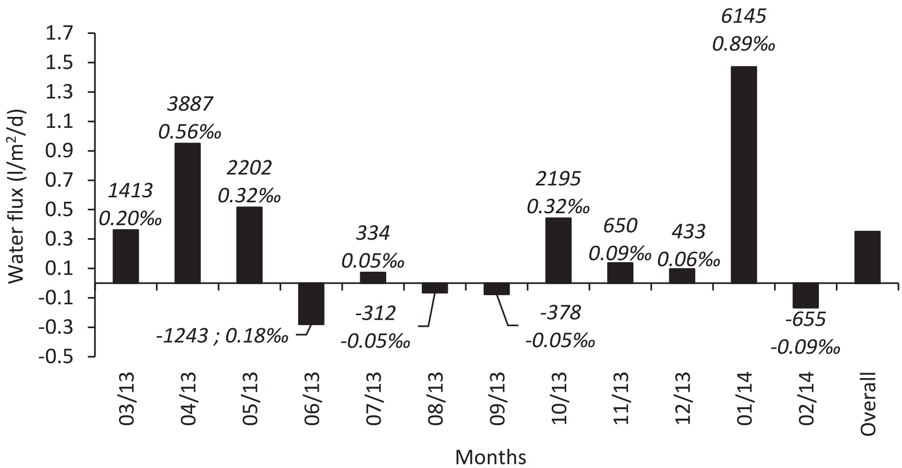


Figure 4. Average water fluxes (column graph) as measured over a 24 h period and each month from March 2013 to February 2014. The overall average water fluxes for all the months combined are also shown. Data values in italics above each column represent the overall water gained or lost in l/d and the ‰ gain or loss of the overall pond water volume per day.

9,186 ha) in Belanger et al. (1985) and even less in Fellows and Brezonik (1980) investigations (Lake Apopka with 12,464 ha and Lake Conway with 711 ha). The high density of seepage meters allowed precise mapping of groundwater movement in a northeastern to southwestern pattern. This pattern is in agreement with the general groundwater flow direction in the region (e.g. Bennett 1992). Overall, the pond was a seepage pond (as defined in Lee et al. 2014) because it had a positive net groundwater inflow. However, this groundwater inflow was about 12 times less than direct rainfall onto the planar surface of the pond. According to Lee et al. (2014), most lakes of the peninsular

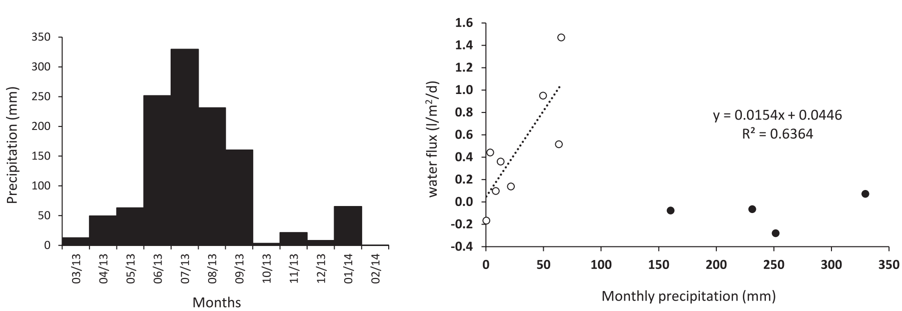


Figure 5. Monthly precipitation at FGCU campus from 03/13 to 02/2014 (left) and regression between the water flux and the monthly precipitation for the dry months (open circles, P=0.018) and wet months (closed circles, P>0.05).

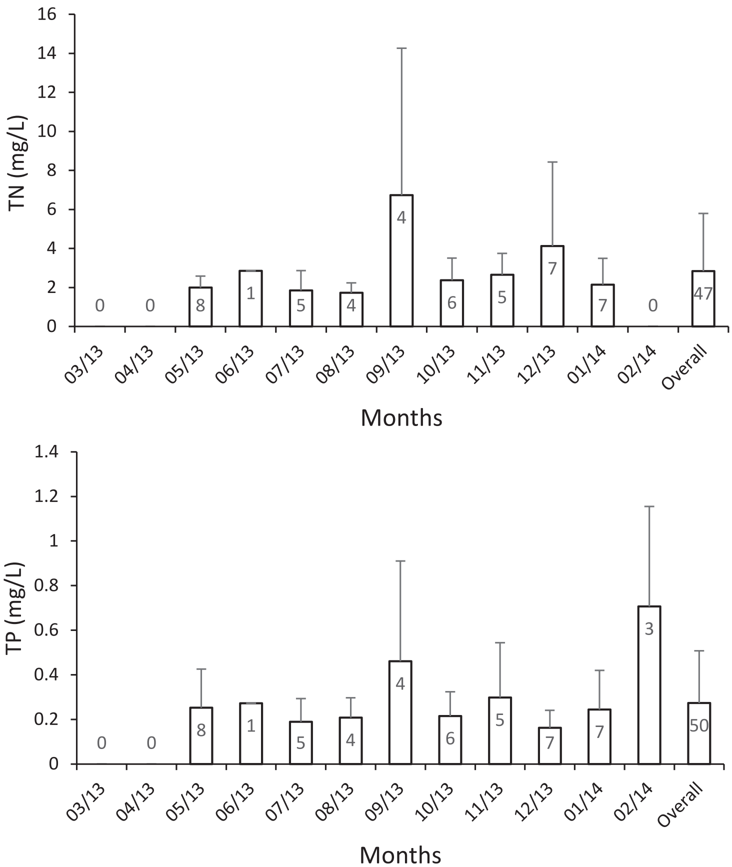


Figure 6. Average TN (top) and TP (bottom) water concentrations coming from the seepage meters and calculated from one 24 h assessment each month from May 2013 to February 2014 (the number of replicates is encapsulated in the column, zero means that nutrients were not considered early in the project).

Florida have more evaporation than precipitation, i.e. a negative net precipitation which is compensated by a positive net groundwater inflow. A water budget for this pond would have been useful to concretely verify these assertions, but evaporation pans (or a net radiometer), as well as a precise water level recorder, were not available. We attempted to conduct statistics to compare water fluxes of the shallow versus deeper portion of the ponds with the expectation that shallow meters would have higher values than that of the deeper meters (e.g. Belanger et al. 1985). This proved to be challenging to do because of the aforementioned spatial northeastern to southwestern groundwater flow pattern. Also, the absence of meter data during the dry season (meters were out of the water) made it difficult to run a repeated measure ANOVA using time as the repeating factor and the various meters per depth

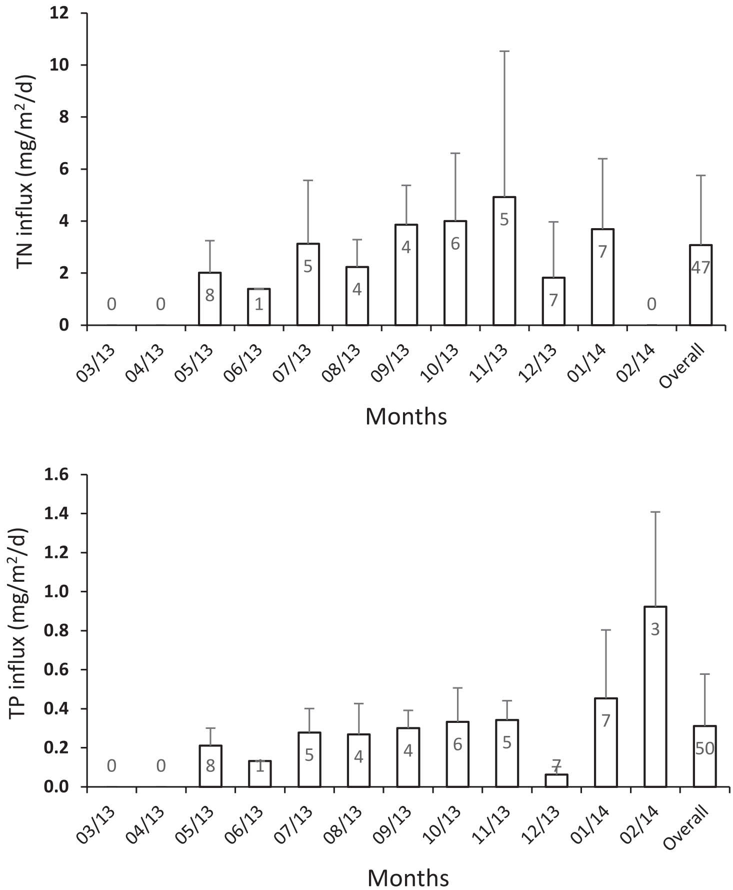


Figure 7. Average TN (top) and TP (bottom) influxes calculated from one 24 h assessment each month from May 2013 to February 2014 (the number of replicates is encapsulated in the column, zero means that nutrients were not considered early in the project).

as the group subjects. Nevertheless, there was enough data collected to assert that this pond is a seepage pond and, as such, it does not recharge the aquifer as mandated.

Most studies focusing on groundwater seepage establish a hydrological budget and in particular assess the percentage of groundwater contribution to the water sources (and water sink) of the water body considered. In this study, because no hydrological budget was made, the amount of water exchanged compared to the volume of the pond was used to give a percentage of water exchange per day over time. Our percentages are thus difficult to compare with those in the literature. To put the study pond into perspective, a high-end estimate of percentage groundwater contribution could be calculated by not including rainfall runoff to the pond. The pond is mostly surrounded by

Table 1. Contribution of groundwater TN and TP on a per liter basis to the pond water per month and overall for the pond water capacity that month.

| | Average Additional [TN] | | Average Additional [TP] | |
|---|---|---|---|---|
| Month | mg/l/month | g/month | mg/l/month | g/month |
| 03/13 | N/A | N/A | N/A | N/A |
| 04/13 | N/A | N/A | N/A | N/A |
| 05/13 | 0.019 | 131.4 | 0.0017 | 11.5 |
| 06/13 | no influx | no influx | no influx | no influx |
| 07/13 | 0.002 | 18.5 | 0.0001 | 1.0 |
| 08/13 | no influx | no influx | no influx | no influx |
| 09/13 | no influx | no influx | no influx | no influx |
| 10/13 | 0.019 | 155.8 | 0.0009 | 7.1 |
| 11/13 | 0.007 | 51.6 | 0.0006 | 4.8 |
| 12/13 | 0.007 | 53.5 | 0.0001 | 1.0 |
| 01/14 | 0.059 | 394.8 | 0.0049 | 32.4 |
| 02/14 | no influx | no influx | no influx | no influx |

pervious flat surfaces and an elevated dirt road to its east which blocks runoff. Only its south portion should thus receive runoff, but it is believed that this would be limited to the rainy season when water levels are high enough (which is an event we did not visually notice). The groundwater inflow contribution could then be calculated as the ratio between the groundwater seepage divided by the sum of the volumes of direct rainfall precipitation falling over the planar surface area of the pond and the groundwater inflow. This calculation yields an estimate of 15.7% groundwater contribution to the pond. Despite the value representing a maximum estimate of groundwater contribution (because runoffs were not included), it is within the order of magnitude found in Florida lakes (e.g. Belanger et al. 1985).

The correlation of rainfall with net groundwater inflow during the dry season, and the lack of such a relationship during the rainy season, was particularly conspicuous. Seepage lakes in Florida increase in water level nearly immediately after precipitation over their watershed (e.g. Lee et al. 2014). The lack of such a relationship in the study pond during the rainy season may be due in part to water table elevation differences between the two seasons. During the dry season, rapid evaporation may have drawn down the pond level below the level of the surrounding water table, creating a positive hydraulic gradient, drawing water into the pond through the littoral areas (Darcy's Law). Sporadic rain events on the surrounding landscape could have increased, via rain infiltration, the water table level even more, creating a larger hydraulic gradient and more groundwater inflow. Conversely, during the wet season, surface runoff and direct precipitation occurred often and this potentially raised the level of the pond high enough to reduce the head difference and even temporarily increase pond levels above the surrounding water table, allowing slight seepage. The implementation of piezometers around the pond along transects across elevation gradients could determine the head differences between the pond water level and surrounding water table, which would test this hypothesis.

The contribution of nutrient loading via groundwater seepage compared to the pond's overall nutrient load is difficult to estimate because of the lack of a hydrological budget and the analysis of water column nutrients during the study. As such, a nutrient budget was not made. Based on unpublished investigations by the authors involving measurements of Secchi disk depths and water total chlorophyll concentrations, the pond surveyed is high mesotrophic to eutrophic on the Carlson trophic status index scale (1977) but its immediate watershed is fertilizer free and very likely reduced in size. Thus, it is hypothesized that groundwater inflow should be responsible for its relatively high nutrient status especially for a young wet detention pond. The nutrient concentration in the groundwater inflow recorded is high for TP when compared to other wet ponds of ages ranging from 1 to 30 years and surrounded by residential and commercial lands (Fernandez and Hutchinson 1993). However, note that groundwater samples were taken from wells (their study) instead of seepage meters bags (this investigation). Groundwater inflow collected from groundwater seepage meters often overestimate nutrients inputs (e.g. Belanger et al. 1985) since meters create a sediment confinement potentially leading to sediment anoxia, which promotes the dissociation of orthophosphates from the reduced iron. As such, adjacent groundwater wells should have been used to draw groundwater samples. This additional phosphorus input was found to be twice higher in the Belanger et al. study (1985), but even if this correction were to be used, our TP in groundwater would still remain high and could explain the pond's nutrient status. This source of phosphorus in groundwater remains unknown since the campus lands (including lawns) northeast of the pond are not fertilized with a few exceptions such as the soccer and football fields which are 1,000-2,000 m away. It is doubtful that the heavily vegetated lands and wetlands found northeast of the pond would release nutrients to the groundwater as the labile pool of nutrients should be sequestered by the plants and algae thus creating a refractory nutrient pool. Further, the land immediately east-northeast of the pond was scraped down to the limestone to create a short hydroperiod oligotrophic reclaimed wetland. Finally, because of the direction of the groundwater flow found in this study, the nearby western and southern golf courses cannot be considered as groundwater nutrient sources.

In conclusion, this study provides a detailed insight into connections between a wet detention pond and its groundwater. Overall, the water body studied was a seepage pond and this assertion should be verified for other ponds in the region. The large number of meters and sampling effort documented the high spatial heterogeneity of the groundwater flux (fluxes were higher in the northeastern portion of the pond than its southwestern portion which agreed with the general subsurface flow pattern in the area). This study would have greatly benefited from a hydrological budget which would have included all the water sources and sinks to the pond as well as the use of piezometers judiciously placed to validate the general groundwater flow pattern. Shoreline meters should also have ideally been relocated as the shoreline was receding during the dry season. The establishment of a nutrients budget including nutrients loading via runoff, rainfall and pond water nutrients as well as, ideally, the groundwater nutrients taken from a groundwater well adjacent to each seepage meter would have best determined why this pond is nutrient rich despite the efforts undertaken at Florida Gulf Coast University (FGCU) to prevent its eutrophication. These types of studies can be helpful in understanding why some ponds remain nutrient rich despite management efforts to reduce surface nutrient loading via runoff and internal loading via sediment dredging. Studies of groundwater nutrient loading in urban stormwater ponds are also important because the numbers of these ponds are steadily increasing over time.

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Biological Sciences

# The ability of barley straw, cypress leaves and L-lysine to inhibit cyanobacteria in Lake Hancock, a hypereutrophic lake in Florida

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**Abstract** Lake Hancock is a hypereutrophic lake in Central Florida dominated by nitrogen-fixing cyanobacteria. In this study, we conducted a replicated aquarium study to determine if additions of barley straw, cypress leaves or the amino acid L-lysine could reduce levels of cyanobacteria in Lake Hancock, as measured by changes in chlorophyll *a*. Additions of L-lysine brought about the quickest reduction in chlorophyll *a* compared to controls. However, the effect of L-lysine levelled off after the first week, while aquaria treated with barley straw and cypress leaves showed increased benefits over time. At the end of the three week experiment, the addition of cypress leaves resulted in an 87 percent reduction in chlorophyll *a* concentrations compared to controls. Prior assessments of water quality have found that lakes that have retained hydrologic connections to their fringing wetlands have higher levels of colored dissolved organic matter (CDOM) than lakes that have been lowered below the elevations of their historical contiguous wetlands. Furthermore, lakes with higher CDOM levels are less susceptible to the influence of nutrient supply. This preliminary study suggests that CDOM may provide lakes with compounds that may moderate the growth of cyanobacteria in lakes with healthy intact wetland fringes.

**Keywords** Chlorophyll *a*, cyanobacteria, hydrology, Lake Hancock, nutrients

## Introduction

Water quality of the various lakes within the Peace River watershed is important not only to those lakes themselves, but also to the downstream waters of Charlotte Harbor. The combined watersheds of the Winter Haven Chain of Lakes (WHCOL), Banana Lake and Lake Hancock are just over 1,000 square kilometers in size, comprising approximately 17 percent of the Peace River watershed. Water quality within Lake Hancock is so poor, due to elevated levels of nutrients and cyanobacteria, that the Peace River Manasota Regional Water Supply Authority is able to detect impacts from lake discharges into the Peace River at its intake structure more than 130 km downstream. The poor water quality within Lake Hancock is such that it is unlikely that it can be restored (Tomasko et al. 2009), which has led to the development of the Lake Hancock Outfall Treatment Marsh. The influence of the polluted waters of Lake Hancock is so extensive that the Lake Hancock Outfall Treatment Marsh alone is expected to meet the Pollutant Load Reduction Goal for the downstream waters of Charlotte Harbor by treating

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discharges out of the lake prior to the water entering the upper Peace River (SWFWMD 2000).

In addition to the influence of Lake Hancock, the 24 inter-connected lakes of the WHCOL system are connected to the Peace River through the Peace Creek and Wahneta Farms drainage canals. The watershed of the WHCOL is approximate 83 square kilometers in size (FDEP 2007) and many of the lakes within the WHCOL system have been negatively impacted through historical point and non-point sources of pollution and have been listed as impaired by FDEP.

Recent studies conducted for the City of Winter Haven and Polk County have shown that water quality in local lakes is often more strongly correlated with lake levels than nutrient concentrations (Atkins 2008, Tomasko et al. 2009, Atkins 2010), which is consistent with a state-wide assessment conducted by Terrell et al. (2000). In particular, lakes that have been lowered below their historical condition via regional drainage networks were shown to be more likely to be impaired for water quality than lakes without such impacts, regardless of the degree of urbanization of their watersheds (Atkins 2010). Levels of colored dissolved organic matter (CDOM) in lakes, which correlate with the amount of forested wetlands along the lake boundary (Atkins 2008), appear to moderate the transformation of nutrients into phytoplankton (Atkins 2008, Atkins 2010), a finding consistent with the results obtained by Terrell et al. (2000).

At present, there is no clear understanding of the precise basis for the ability of wetland-associated CDOM to moderate algal populations in Florida lakes. However, research related to the link between decomposition of barley straw and nuisance algal blooms (mostly cyanobacteria) may shed some light on the processes involved. As early as the 1970s, decomposing barley straw had been used to control the growth of cyanobacteria in freshwater lakes in the UK and elsewhere (Newman and Barrett 1993). Researchers in Scotland (Barrett et al. 1996), Minnesota, USA (McComas and Stuckert 2009) and Iran (Rajabi et al. 2010) have shown that decomposing barley straw is capable of reducing the abundance of nuisance algae, particularly cyanobacteria, in lakes in different climates. However, Lembi (2002) and others have pointed out that water quality in lakes does not always respond in a favorable manner to treatment with barley straw.

In addition, researchers in Japan (Hehmann et al. 2002) and Mississippi, USA (Zimba et al, 2001) have found that the amino acid L-lysine was capable of reducing the abundance of nuisance cyanobacteria. Additions of L-lysine appeared to have no impact on the populations of more benign species of chlorophytes, leading Zimba et al. (2001) to postulate that L-lysine might have practical applications for hypereutrophic lakes dominated by nitrogen-fixing cyanobacteria.

In this study, a replicated aquarium experiment was conducted to determine if additions of barley straw and L-lysine might be able to reduce the abundance of nuisance cyanobacteria in Lake Hancock, which has been classified as the most polluted large lake in Florida (FDEP 2005). In addition to these two treatments, an experimental treatment with leaves from cypress trees (*Taxodium distichum*) was included to determine if cypress leaves might have a similar ability to reduce cyanobacteria populations as had been previously documented for barley straw.

### Materials and Methods

The experimental design was set up with three replicate 39-L aquaria for each of four treatments: 1) addition of barley straw, 2) addition of cypress leaves, 3) addition of L-lysine, and 4) controls.

Recommended application rates for barley straw include areal application rates (e.g. Lembi 2002, McComas and Stuckert 2009) as well as volume-based application rates (e.g. Barrett et al. 1996). Based on previously published reports (Tomasko et al. 2009 and references within), areal and volume-based application rates were normalized for Lake Hancock with a chosen dosage rate of 40 grams of dry barley straw per cubic meter of water. Scaling dosage rates to a 39-L aquarium, 1.51 grams of dry barley straw were added in a mesh bag to each of three replicate aquaria. The barley straw was purchased from a commercial pond and landscape store. Cypress (*Taxodium distichum*) leaves were collected from a central Florida wetland. Only leaves that had fallen to the ground were used, with the leaves dried to room temperature until they reached constant weight. Cypress leaves were added at the same dosage rate as barley straw, 1.51 grams dry weight per each 39-L aquarium, and they also were placed in a mesh bag at the aquarium surface.

Hehmann et al. (2002) assessed water quality responses to L-lysine at dosage rates of 0.6 to 5 mg/L, while Zimba et al. (2001) used a dosage rate of 1 mg L-lysine/L. For this study, the highest range value was used (5 mg L-lysine/L) as it appeared that cyanobacteria levels in Lake Hancock were higher than those in the experiments carried out in prior studies. In addition to the three treatments described above, three aquaria were designated as controls, with no additions of barley straw, cypress leaves or L-lysine.

The aquarium study was conducted over a three week period from May to June of 2013. Aquaria were located outdoors on property owned by the City of Winter Haven. Aquaria were located under an awning that protected them from rainfall and also were covered with clear plastic to reduce water loss through evaporation and to keep wildlife from disturbing the aquaria. The aquaria were placed on a south-facing side of the covered site and received abundant sunlight throughout the day. Each aquarium included an aeration stone fed by an air pump to ensure circulation of the contents of the aquarium for the entire three-week duration of the experiment. Water for the experiment was collected from the surface waters of Lake Hancock. Water was transported to the aquaria for use the same day as the study was initiated.

Water chemistry was measured for all aquaria at the start of the experiment and then each week for the next three weeks. Water clarity was not quantified at the start of the experiment, but it was measured for the remaining weeks through the use of a miniature Secchi disk, which was moved horizontally away from an observer at the front of the aquarium until it was no longer visible. The disk was then moved farther away still, and then brought back toward the observer, with the distances where the disk disappeared and then reappeared recorded for each aquarium. Water quality data were collected for further analysis by Polk County's water quality laboratory. The parameters, methods and practical quantification limits are listed in Table 1.

## Results

All results are shown first as raw data, then for each of the three treatments for each date and then as compared to the control aquaria. Displayed raw data are the means of the three replicate aquaria for each date. When results from treatment criteria are displayed as a percentage of the mean values of controls, the data displayed are the means of the treatment aquaria for each date compared to the mean of the control aquaria for that same sampling event.

Table 1. Water quality parameters, analytical methods used, and practical quantification limits. Method standard operating procedure refers to specific edition of EPA or Standard Method used.

| Parameter | Method | Units | Practical Quantification Limit |
|---|---|---|---|
| Chlorophyll *a* (corrected) | 10200 H | μg/L | 3 |
| Color | 2120 B | Pt-Co | 5 |
| Total Kjeldahl nitrogen | 351.2 | mg/L | 0.40 |
| Nitrate plus nitrite | 4500NO3-I (21ed) | mg/L | 0.04 |
| Total nitrogen | 351.2&4500NO-3-I | mg/L | 0.44 |
| Ortho phosphorus | 4500 P-G (21-ed) | mg/L | 0.04 |
| Total phosphorus | 365.4 | mg/L | 0.04 |

Water clarity in the control aquaria varied from 5.2 cm after one week to 16.6 cm after three weeks (Figure 1). At the end of the experiment, water clarity was greatest in the aquaria with the cypress leaves, while the lowest water clarity was in the control aquaria. At the end of the experiment, aquaria with the cypress leaves treatment had the greatest improvement in water clarity, with a mean value 67 percent higher than control aquaria (Figure 2).

Concentrations of chlorophyll *a* in the control aquaria varied from 677 μg/L at the start of the experiment to 160 μg/L after three weeks (Figure 3). Concentrations of chlorophyll *a* in the control aquaria remained essentially the same for the first week of the experiment, but declined by 76 percent over the next two weeks. At the end of the experiment, chlorophyll *a* concentrations were lowest in the aquaria with the cypress leaves, while the highest chlorophyll *a* values were in the control aquaria. At the end of the experiment, aquaria with

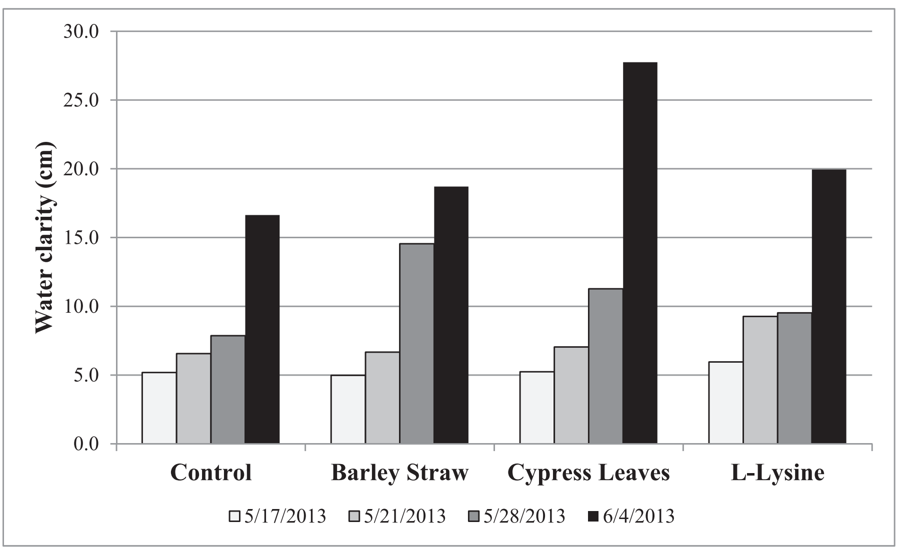


Figure 1. Water clarity (cm) in controls and aquaria with Barley straw, aquaria with Cypress leaves, and aquaria with L-lysine treatments. Values are means of n = 3.

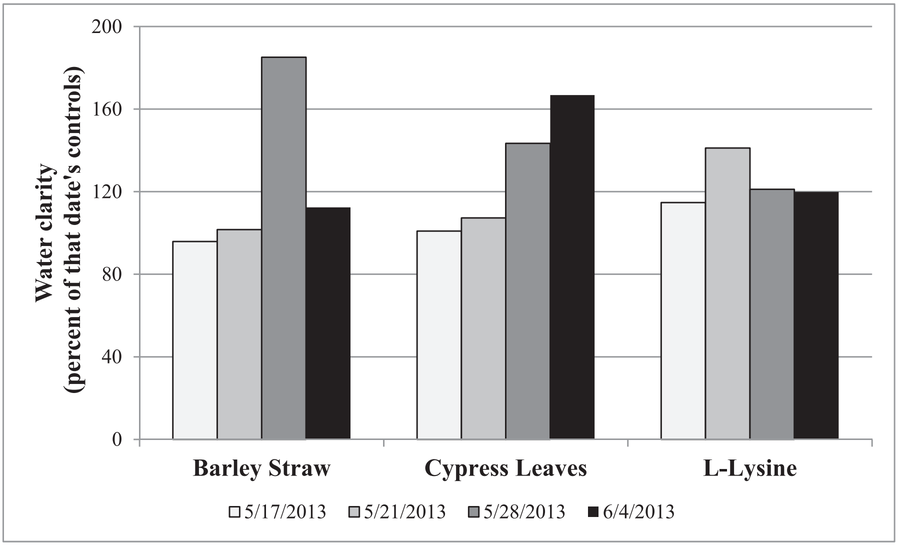


Figure 2. Water clarity (cm) in aquaria with Barley straw, aquaria with Cypress leaves, and aquaria with L-lysine treatments as a percentage of values from controls on the same date. Values are based on comparison of means of n = 3.

the cypress leaves had the greatest reduction in concentrations in chlorophyll *a*, with a mean value more than 87 percent lower than control aquaria (Figure 4).

Concentrations of total nitrogen in all aquaria exceeded 12 mg/L at the start of the study, likely reflecting the high rates of nitrogen fixation previously

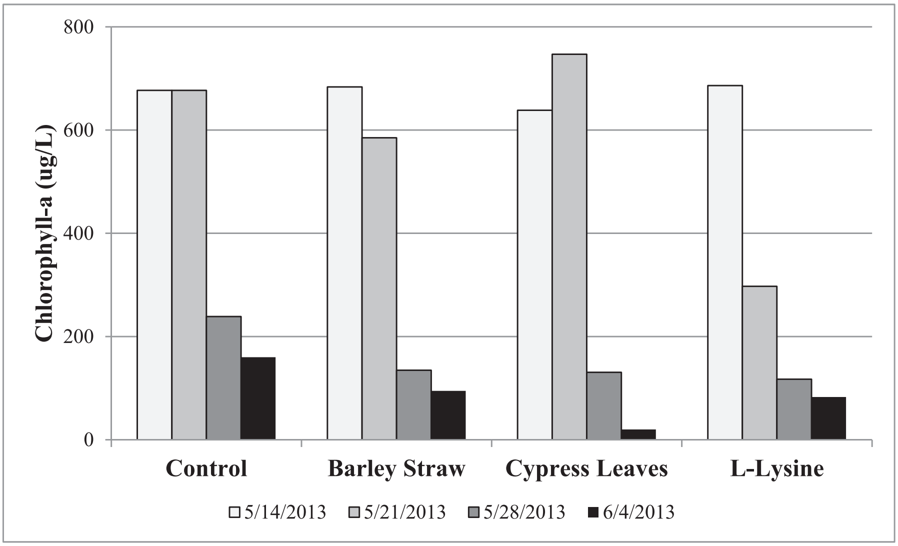


Figure 3. Chlorophyll *a* concentrations (µg Chlorophyll *a*/L) in controls and aquaria with Barley straw, aquaria with Cypress leaves, and aquaria with L-lysine treatments. Values are means of n = 3.

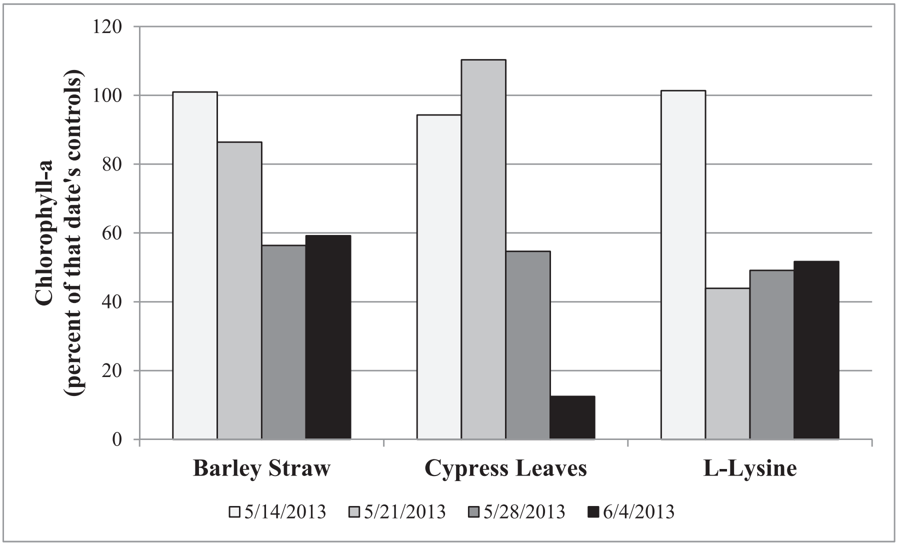


Figure 4. Chlorophyll *a* concentrations (µg Chlorophyll *a*/L) in aquaria with Barley straw, aquaria with Cypress leaves, and aquaria with L-lysine treatments as a percentage of values from controls on the same date. Values are based on comparison of means of n = 3.

documented for Lake Hancock cyanobacteria by Tomasko et al. (2009). Concentrations of total nitrogen increased over the three weeks of the experiment, suggesting that nitrogen fixation was ongoing throughout the experiment in all four treatments (Figure 5). At the end of the experiment,

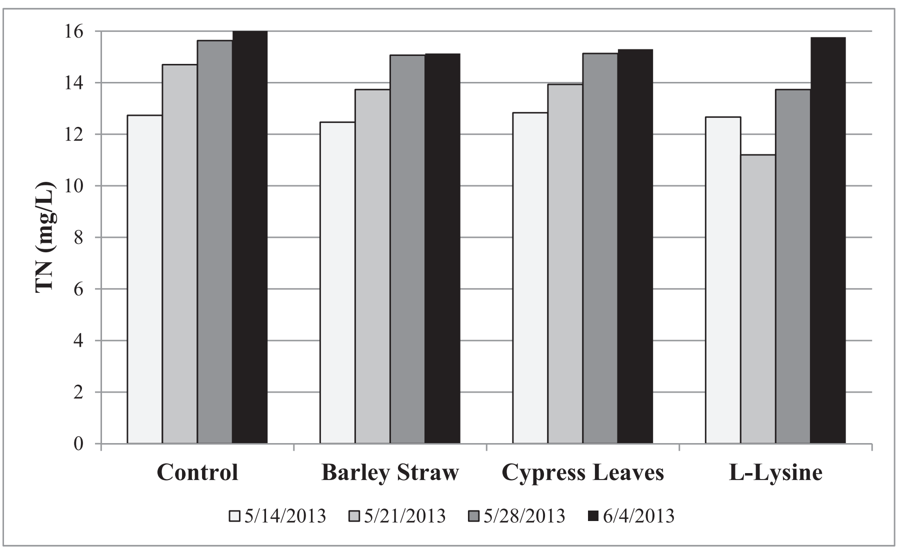


Figure 5. Total nitrogen concentrations (mg TN/L) in controls and aquaria with Barley straw, aquaria with Cypress leaves, and aquaria with L-lysine treatments. Values are means of n = 3.

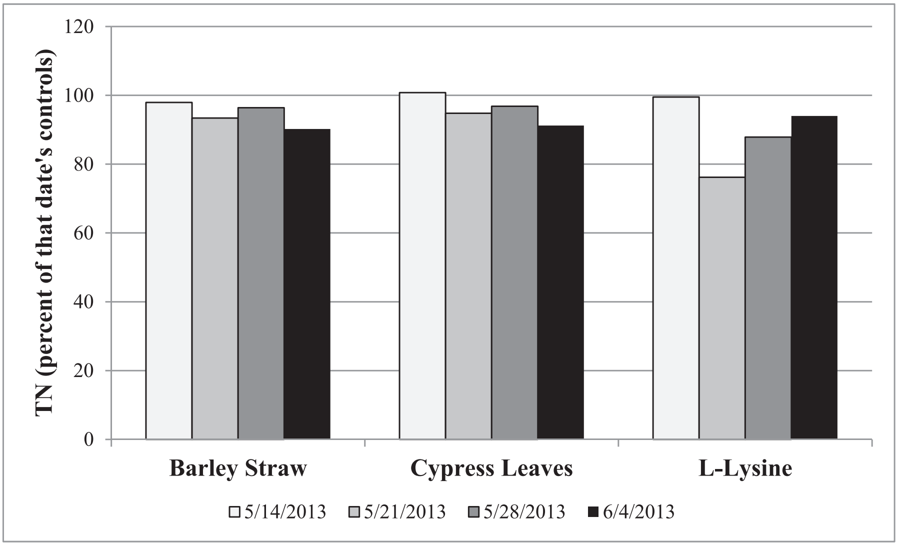


Figure 6. Total nitrogen concentrations (mg TN/L) in aquaria with Barley straw, aquaria with Cypress leaves, and aquaria with L-lysine treatments as a percentage of values from controls on the same date. Values are based on comparison of means of n = 3.

total nitrogen concentrations were similar in all aquaria, with mean values in the three treatments within 10 percent of values from the control aquaria (Figure 6).

At the beginning of the experiment, concentrations of total phosphorus in all aquaria exceeded 0.40 mg TP/L, reflecting the very high concentrations of phosphorus that characterize Lake Hancock (FDEP 2005 and references within). In contrast to total nitrogen, concentrations of total phosphorus declined in all treatments over the three weeks of the experiment (Figure 7), suggesting that phosphorus was settling out of the water column, perhaps as dead and dying phytoplankton (as suggested by the concurrent trend of decreasing concentrations of chlorophyll *a* in the water column). At the end of the experiment, total phosphorus concentrations ranged from 15 percent lower than controls for the aquaria with cypress leaves to values 28 percent higher than controls for aquaria with L-lysine treatments (Figure 8).

Values of true color, a surrogate for CDOM, ranged between 90 and 100 platinum-cobalt units at the start of the experiment. After one week, there appeared to be an increase in color values in all aquaria. However, laboratory results became unreliable after the first week of the experiment, perhaps due to interactions between dissolved substances associated with the very high levels of phytoplankton (as indicated by chlorophyll *a*) and the techniques for quantifying levels of CDOM. Results are not shown, as the laboratory manager did not feel the technique employed was useful in such hypereutrophic conditions.

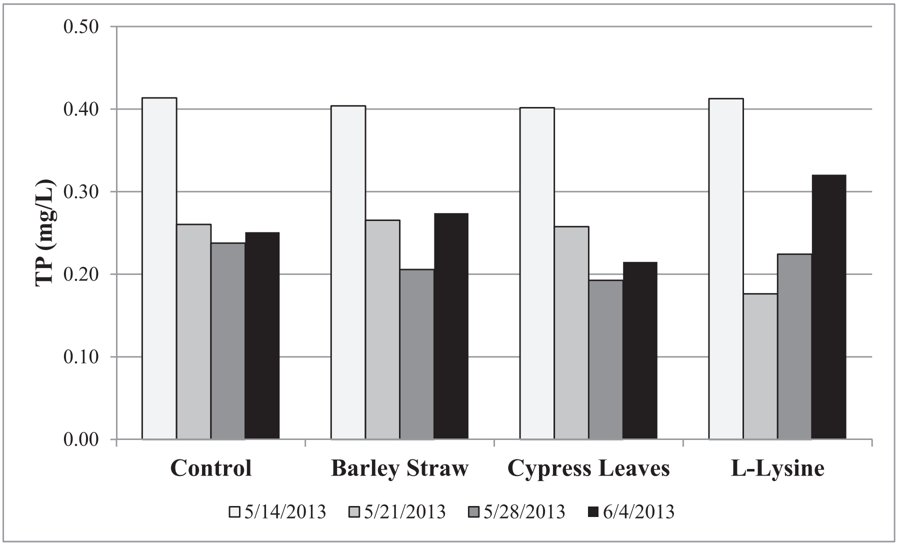


Figure 7. Total phosphorus concentrations (mg TP/L) in controls and aquaria with Barley straw, aquaria with Cypress leaves, and aquaria with L-lysine treatments. Values are means of n = 3.

## Discussion

As outlined by both FDEP and the US Environmental Protection Agency, the explicit approach to restoring water quality in Lake Hancock, the WHCOL system and elsewhere is that local, regional, state and federal resources should

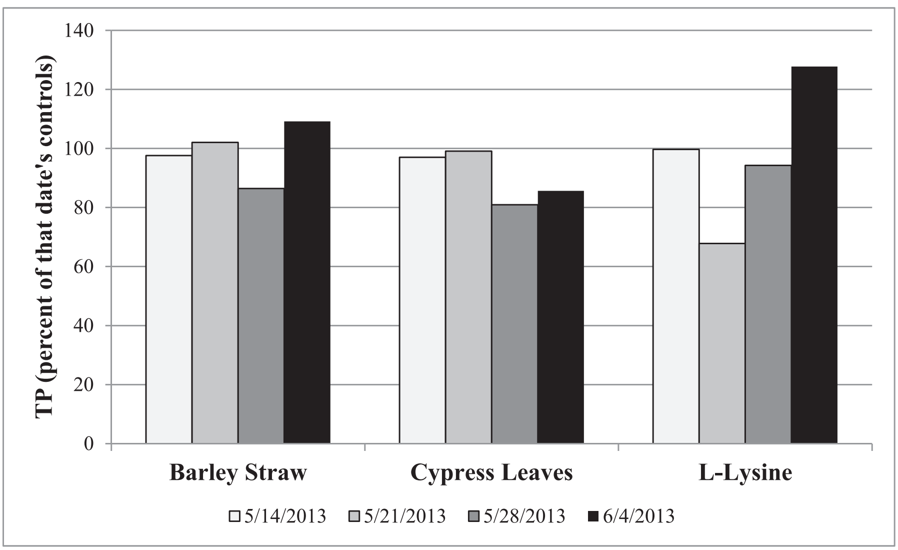


Figure 8. Total phosphorus concentrations (mg TP/L) in aquaria with Barley straw, aquaria with Cypress leaves, and aquaria with L-lysine treatments as a percentage of values from controls on the same date. Values are based on comparison of means of n = 3.

be directed at reducing the external loads of nutrients to impaired waterbodies through various regulatory and non-regulatory programs. While this approach is broadly consistent with prior successful water resource management efforts, external nutrient loads are sometimes not the primary stressor to water quality in Florida lakes (e.g. Terrell et al. 2000). Thus, acting on external nutrient loads alone has not always brought about improvements in water quality in Florida lakes (e.g. Atkins 2008, Atkins 2010).

Within the WHCOL system, Lakes Lulu, May and Shipp have all fully met their regulation-required reductions in stormwater loads of total phosphorus (TP), yet none of these three lakes showed signs of improving water clarity or decreased concentrations of chlorophyll *a* (Atkins 2008). As disappointing as the results are for Lakes Lulu, May and Shipp, they are consistent with the conclusions reached by Terrell et al. (2000) who examined water quality data from 127 lakes located throughout Florida over the period of 1967 to 1997. Over the time period examined, the overall trend in Florida lakes was that of decreasing concentrations of TP, the nutrient most commonly linked with eutrophication in freshwater systems (FDEP 2005 and 2007), and no trend in Total Nitrogen (TN). Despite the downward trend in TP and the lack of a trend in TN, there was a clear trend of increasing levels of chlorophyll *a* in those same lakes. The authors therefore concluded that altered hydrology and impacts from management efforts to control *Hydrilla verticillata* through herbicide applications were important influences on water quality.

Within the City of Winter Haven, lakes that have been lowered below their historical elevations (most often through the construction of canals and regional drainage systems) are more than four times as likely to have impaired water quality than lakes that are at their approximate historical elevations, regardless of the degree of urbanization of their watersheds (Atkins 2011).

In the Chain of Lakes Water Quality Management Plan (Atkins 2010) developed for the WHCOL system, it was determined that the lowering of lakes below their historical elevations appeared to be responsible for “disconnecting” a number of lakes from their historical wetland fringes (Atkins 2008, Atkins 2010). For example, Lake Lulu appears to be approximately 1.2 meters lower now than in its historical condition and the lake’s former swamp shoreline now lies perched above the open waters of the lake. As a result, Lake Lulu has much lower levels of CDOM than Lake Henry, where its open waters and swamp shoreline are still hydrologically connected (Atkins 2008). Lake Henry has CDOM levels more than 10 times as high as Lake Lulu and has lower concentrations of chlorophyll *a*, despite having significantly higher concentrations of both TN and TP (Atkins 2008, Atkins 2010). CDOM is mostly comprised of a mixture of tannin, lignin and humic substances (McDonald et al. 2004). Tannin is a complex molecule found in soils and decomposing vegetation, while lignin is mostly associated with woody parts of trees. Like tannin, humic substances are mostly related to the decomposition of organic matter. All three materials, tannin, lignin and humic substances, can be present through entirely natural processes (McDonald et al. 2004).

Within Florida, there appears to be a disconnect between recent lake management research and approaches to lake management encompassed within current regulatory programs. While there are locations where increased availability of nutrients is the primary stressor to a lake's water quality, the importance of stressors such as hydrologic alterations and the potentially detrimental impacts of eradication efforts focused on *Hydrilla verticillata* are usually not included in the State of Florida's regulatory approach to lake management. As the implementation of regional stormwater treatment projects for Lakes Lulu, May and Shipp did not bring about the intended benefits to water quality (Atkins 2008, Atkins 2010), a more holistic approach to water quality management appears warranted for many lakes. The reestablishment of historical hydrologic connections between lakes and their adjacent wetland fringes should be given at least as much attention as is currently focused on external nutrient loads.

While not specifically related to the topic of wetland influences, the use of barley straw for algal control may be related to wetland influences and the role of CDOM in moderating the transformation of nutrients into phytoplankton biomass. The barley straw treatment promoted by some researchers could be related to algistatic compounds released during aerobic decomposition of barley, which has an elevated concentration of lignin (Newman and Barrett 1993). While the exact mechanisms are unknown, the decomposition of lignin-rich compounds under oxygenated conditions has been postulated as bringing about the release of secondary metabolites that may interfere with the growth of cyanobacteria (Barrett et al. 1996).

Based on the results of this aquarium study, it appears that cypress leaves are capable of producing the same or better benefits as have been noted with the use of barley straw. After three weeks, water clarity in aquaria with cypress leaf additions was 67 percent greater than in controls. The increase in water clarity is likely due to the 87 percent reduction in concentrations of chlorophyll *a* compared to controls. In aquaria with barley straw and cypress leaves, the reduction in chlorophyll *a* concentrations was much greater than the decline in phosphorus concentrations, suggesting that algal growth was being reduced through mechanisms other than nutrient availability alone. Also, as the aquaria were continuously circulated and located in an outdoor setting, it is unlikely that phytoplankton reductions were due to reduced water clarity in aquaria with the added barley straw or cypress leaves.

The finding that cypress leaves have a similar, and actually greater, inhibitory effect on cyanobacteria than was found with barley straw is encouraging and consistent with studies that have concluded that disconnecting lakes from their historical wetland fringes might have contributed to increased phytoplankton populations due to the loss of this moderating influence (e.g. Terrell et al. 2000, Atkins 2008, Atkins 2010). For those lakes with lowered lake levels that could potentially be elevated again to their historical elevations (e.g. Cypress Lake in Osceola County), hydrologic restoration projects might be able to also bring about restoration of water quality through increased wetland influences on the water column. For those lakes that are lower than they were historically, but which are now surrounded by development, deployment of floating bales of barley straw or cypress leaves at appropriate application rates and with suitable techniques could be a way of bringing the lost influence of previously connected wetland fringes back to lakes that have been altered in such a manner.

The addition of L-lysine does not appear to be a good candidate in terms of long-term management techniques for dealing with hypereutrophic lakes. Those aquaria with the L-lysine treatment had the quickest response in terms of water clarity and concentrations of chlorophyll *a*, but after three weeks those aquaria lagged behind the benefits seen in the aquaria with either barley straw or cypress leaves. Also, while the barley straw and cypress leaves techniques are consistent with a larger narrative of lost wetland influences, there does not yet appear to be a nature-based paradigm that would warrant the use of widespread application of amino acids to hypereutrophic Florida Lakes.

Finally, these results should be viewed as preliminary in nature and in need of further assessments to determine their applicability outside of an aquarium setting. In particular, it would be useful to follow up this study with an outdoor and in-lake mesocosm study similar to the techniques used to study the benefits to Lake Hancock of sediment removal (e.g. Tomasko et al. 2009). If a larger-scale and in-lake mesocosm study results in similar findings, an experimental approach on a whole-lake level would be a logical next step, with system responses monitored through the use of a Before and After, Control and Impact experimental design.

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Biological Sciences

# Fishes of the Lemon Bay estuary and a comparison of fish community structure to nearby estuaries along Florida's Gulf coast

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**Abstract** Lemon Bay is a narrow, shallow estuary in southwest Florida. Although its fish fauna has been studied intermittently since the 1880s, no detailed inventory has been available. We sampled fish and selected macroinvertebrates in the bay and lower portions of its tributaries from June 2009 through April 2010 using seines and trawls. One hundred three fish and six invertebrate taxa were collected. Pinfish *Lagodon rhomboides*, spot *Leiostomus xanthurus*, bay anchovy *Anchoa mitchilli*, mojarras *Eucinostomus* spp., silver perch *Bairdiella chrysoura*, and scaled sardine *Harengula jaguana* were among the most abundant species. To place our information into a broader ecological context, we compared the Lemon Bay fish assemblages with those of nearby estuaries. Multivariate analyses revealed that fish assemblages of Lemon and Sarasota bays differed from those of lower Charlotte Harbor and lower Tampa Bay at similarities of 68–75%, depending on collection gear. These differences were attributed to greater abundances of small-bodied fishes in Lemon and Sarasota bays than in the other much larger estuaries. Factors such as water circulation patterns, length of shoreline relative to area of open water, and proximity of Gulf passes to juvenile habitat may differ sufficiently between the small and large estuaries to affect fish assemblages.

**Keywords** Estuarine fishes, Lemon Bay, southwest Florida

## Introduction

Estuaries in southwest Florida support productive biological systems with diverse and valuable fish populations (Comp and Seaman 1985). If these systems and their fish communities are to be protected, it is important to have accurate and detailed information on the fish fauna of the estuaries (Able and Fahay 1998, Franco et al. 2008). The ecological integrity of many estuaries worldwide is being increasingly taxed by impacts of human population growth such as habitat destruction, altered freshwater inflow, excess nutrient input, and introduction of nonnative organisms (Kennish 2002). Estuarine fish populations can also be negatively affected by overfishing and by natural events such as, in southwest Florida, hurricanes, cold kills, and toxic algal (red tide) blooms (Hammett 1990, Flaherty and Landsberg 2011). As a starting point for tracking changes in a region's ichthyofauna, a basic species inventory is needed.

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A southwest Florida estuary for which the fish fauna has not been thoroughly described is Lemon Bay. Henshall (1891) first investigated the fish fauna of southwest Florida, including Lemon Bay, in 1889. Subsequently, marine research has been conducted at facilities adjacent to Lemon Bay (Bass Biological Laboratory 1931–1944, Cape Haze Marine Laboratory 1955–1960). Wang and Raney (1971) sampled fish by trawl at four sites in Lemon Bay during April 1968. A comprehensive species inventory of fish in this estuary, however, has been unavailable. We sampled fish in Lemon Bay from June 2009 through April 2010 to develop such an inventory and to describe the fish fauna.

To place our information into a broader ecological context, we also compared species compositions and relative abundances of fish assemblages in Lemon Bay with those of several similar nearby polyhaline estuarine areas (lower Charlotte Harbor, Sarasota Bay, and lower Tampa Bay). Fish assemblages in all these estuaries might be expected to share similarities due to the influence of shared regional factors, including climatic variations and variations in recruitment of transient species from common offshore spawning sites. Such similarities have been demonstrated among proximate European estuaries, where fish assemblages were shown to be functionally similar to one another and to exhibit a shared seasonal pattern in diversity (Franco et al. 2008, Selleslagh et al. 2009, Cardoso et al. 2011). Differences between neighboring estuaries in characteristics such as watershed land use (e.g. agricultural vs. urban), connectivity to open ocean waters, freshwater input, and composition and quality of aquatic habitat, can nevertheless result in estuary-specific differences in the fish assemblages (Paperno et al. 2001, Franco et al. 2008). Identifying interbay differences in fish assemblages would help resource managers in evaluating the relative value of an area as fish habitat.

## Materials and Methods

**Study site.** Lemon Bay is a long (21 km), narrow (<2 km), shallow estuary on the coast of southwest Florida (FDNR 1992) (Figures 1 and 2). It is located behind a series of barrier islands where the town of Englewood Beach and several smaller coastal communities are situated and is connected to the Gulf of Mexico by Stump and Gasparilla passes. Fresh water enters the estuary from several small tributaries on the mainland that pass through the urban areas of Englewood and Venice. Historically, the Lemon Bay estuarine system consisted of a series of shallow linear lagoons. In the early 1960s these embayments were connected and joined with Charlotte Harbor to the south and Sarasota Bay to the north as part of the Intracoastal Waterway (Antonini et al. 1998, FDEP 2005). The study area included the estuarine waters of Lemon Bay from the Boca Grande Causeway north to Alligator Creek, and the lower (1 km) portions of the six major tributaries (Alligator, Forked, Gottfried, Ainger, Oyster, and Buck creeks).

The climate in the Lemon Bay area is subtropical, with rainy summers and a fall to spring dry season (Hammett 1990, FDNR 1992). Occasional events such as freezes, tropical cyclones, and blooms of toxic phytoplankton (red tides) affect the ecosystem (Hammett 1990, Flaherty and Landsberg 2011). Mangroves dominate the shorelines of the estuary, although small stretches have been seawalled where the Intracoastal Waterway passes close to shore. Seagrasses, principally shoal grass *Halodule wrightii*, turtle grass *Thalassia testudinum*, and manatee grass *Syringodium filiforme*, are common throughout the shallow waters of Lemon Bay and are present to some degree in the relatively shallow (<2 m) tributaries (FDNR 1992).

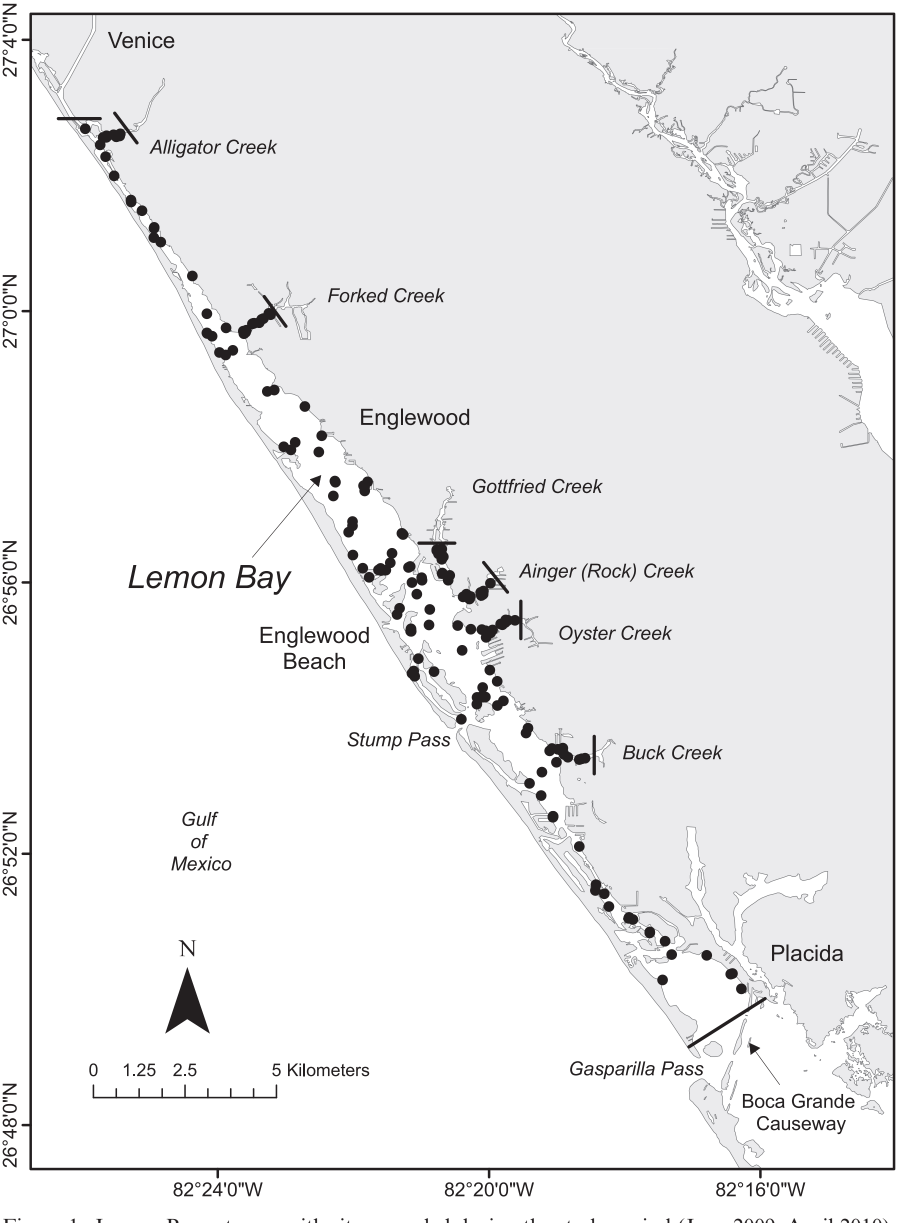


Figure 1. Lemon Bay estuary, with sites sampled during the study period (June 2009–April 2010). Solid vectors denote boundaries of study area.

The Lemon Bay estuary has undergone considerable urbanization since the 1950s, and "significant artificial alterations have been made to the natural shorelines and drainage patterns of the Lemon Bay area through dredging and filling" (FDNR 1992). It has been estimated that about 70% of the historic wetlands and 55% of the mangroves in the watershed remain (Conservancy of Southwest Florida 2011). Overall water quality in this system (e.g. chlorophyll *a*, fecal coliform,

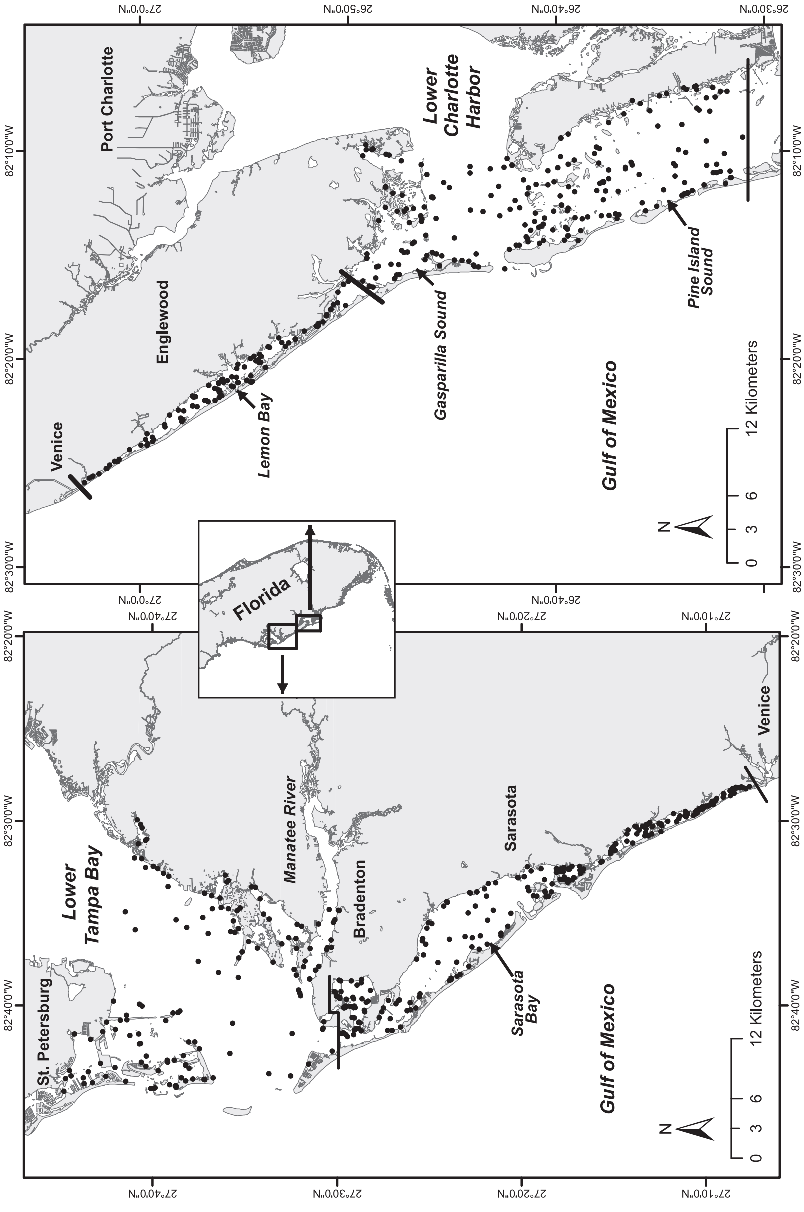


Figure 2. Lemon Bay and nearby estuaries on the coasts of central and southwest Florida (Lower Tampa Bay, Sarasota Bay, and Lower Charlotte Harbor), with sites sampled during the study period (June 2009–April 2010). Solid vectors denote boundaries of study areas.

dissolved oxygen) has been categorized as fair to below average, with the poorest quality in the upper bay and at the creek mouths (FDEP 2005, FDEP 2007).

**Sample collection.** We sampled Lemon Bay and its major tributaries from June 2009 through April 2010 as part of the Florida Fish and Wildlife Conservation Commission's Fisheries-Independent Monitoring (FIM) program, which uses standardized procedures and protocols to collect data on fish and selected macroinvertebrates in estuarine systems throughout Florida. A stratified-random sampling design and use of multiple types of sampling gear enabled collection of fish and macroinvertebrates from a variety of habitats and in a range of sizes and life history stages. The resulting data can be used to examine patterns of spatial and temporal abundance.

Three types of sampling gear were used: 1) a 21.3-m, 3.2-mm stretched-mesh, center-bag seine was used to collect small-bodied fish (typically $<$80 mm standard length) in shallow ($<$1.8 m) areas; 2) a 6.1-m otter trawl with a 3.2-mm mesh cod-end liner was used to collect small-bodied fish in deeper (1.8–7.6 m) areas; and 3) a 183-m haul seine with 38-mm stretched mesh was used to collect large-bodied fish (typically $>$100 mm standard length). Seine samples in the Lemon Bay tributaries (river seines) were collected in areas to a depth of 1.8 m by deploying the 21.3-m seine from the stern of a boat in a shallow arc along the shore and hauling it directly to shore, encompassing an area of ~68 $m^2$. Samples in Lemon Bay proper (bay seines) were collected by deploying the 21.3-m seine in areas to a depth of 1.5 m along shorelines and on offshore flats, sampling an area of 140 $m^2$. The 6.1-m trawl was used in areas 1.8 to 7.6 m deep and towed for 5 min at ~0.6 m/s, giving a typical tow distance of 180 m and an approximate area sampled of 720 $m^2$. The 183-m haul seine was deployed by boat in a standardized rectangular shape in areas to a depth of 2.5 m and hauled to shore, enclosing an area of 4,120 $m^2$ (see Paperno et al. 2001 for a more detailed description of sampling techniques). During the study period (June 2009–April 2010), 180 samples were collected (Figure 1). Thirty samples (twelve 21.3-m river seines, ten 21.3-m bay seines, four 6.1-m bay trawls, and four 183-m haul seines) were collected during each bimonthly sampling event.

Concurrently with each net deployment, physicochemical parameters including temperature (°C), salinity (psu), and dissolved oxygen (mg/l) were profiled with a water quality datasonde (measurements taken at 0.2 m from the surface, every meter thereafter, and at 0.2 m from the bottom). A variety of qualitative habitat assessments were also made, such as characteristics of the shoreline (e.g. vegetation type and distance of inundation), substrate (e.g. sediment type, quantity and type of submerged aquatic vegetation), and bycatch (total volume and composition). All sampling was conducted during the day.

Fish and selected macroinvertebrates were identified to the lowest practical taxonomic level (nomenclature follows Page et al. 2013 and WoRMS 2012). Selected macroinvertebrates (i.e. those of commercial or recreational importance, such as pink shrimp *Farfantepenaeus duorarum*, spiny lobster *Panulirus argus*, portunus crabs *Portunus* spp., blue crab *Callinectes sapidus*, and stone crab *Menippe* spp., are included in the terms "fish" and "fish assemblage" throughout this paper. When large numbers ($>$1,000) of fish of a single taxon were captured, the total number was estimated by fractional expansion of subsampled portions of the total catch split with a modified Motoda box splitter (Winner and McMichael 1997).

Due to hybridization of local species (Dahlberg 1970, Duggins et al. 1986) menhaden *Brevoortia* spp. and silversides *Menidia* spp. were treated as single taxonomic groups. Due to the extreme difficulty of distinguishing between species at small size, mojarras of the genus *Eucinostomus* ($<$40 mm SL), gobies of the genus *Gobiosoma* ($<$20 mm SL), and small *Strongylura marina* and *S. timucu* ($<$100 mm SL) were also treated as single taxonomic groups.

**Data analysis.** We compared Lemon Bay fish assemblages with those of nearby estuarine systems using univariate (ANOVA) and nonparametric multivariate analyses with PRIMER v6 software (PRIMER-E Ltd., UK) (Plymouth Routines in Multivariate Ecological Research, Clarke and Warwick 2001). Data used in the analysis included those from Lemon Bay, lower Charlotte Harbor, Sarasota Bay, and lower Tampa Bay. Together these water bodies represent the entire chain of polyhaline bays behind the barrier island complex from Charlotte Harbor north through Tampa Bay (Figure 2). Comparisons used FIM program collections during the same time period and months, and included the same gear types and techniques, as those collected in Lemon Bay. Data collected with each gear type were treated separately; data from Lemon Bay tributaries were not included in the interbay analysis. Abundance was standardized to number/sample for seine samples. Trawl sample abundance was standardized to number/100 $m^2$ to account for varying tow distances.

Total fish densities were compared between estuaries using ANOVA. Total fish densities were ln-transformed to improve normality and better satisfy the assumptions of the statistical test. Comparisons were deemed significant at $p < 0.05$. Significant differences were analyzed further using Tukey's HSD test.

To compare the fish assemblage structure of Lemon Bay with those of nearby estuarine systems, a one-way analysis of similarity (ANOSIM, Clarke and Warwick 2001) was performed for each type of gear, with estuarine system as the factor of interest. Before ANOSIM was performed, Bray-Curtis similarity matrices were calculated for data averaged by estuary and sampling event (month-year) to include an appropriate level of variability in the statistical test. Nonmetric multidimensional scaling (MDS, Clarke and Warwick 2001) and hierarchical agglomerative cluster analysis (CLUSTER, Clarke and Warwick 2001) were used to graphically depict relative differences in fish assemblages among the estuarine systems. Before performing MDS and CLUSTER, Bray-Curtis similarity matrices were calculated for data averaged by estuarine system in an effort to allow for better visual interpretation of the factor of interest. Similarity percentage analysis (SIMPER, Clarke and Warwick 2001) was used to identify species representative of dissimilarities between groups determined from MDS and CLUSTER. Species that contributed >2% to the total average dissimilarity between groups were considered distinguishing. All abundances were square-root transformed prior to analysis to reduce the influence of highly abundant taxa.

## Results

Climatic conditions during the latter part of our study period were atypical. A severe freeze occurred in January 2010, and minimum air temperatures remained below freezing for two weeks; fish kills of predominately subtropical species resulted (Adams et al. 2012). Precipitation did not differ appreciably from the typical yearly pattern.

A total of 96,647 fishes (103 taxa) and selected invertebrates (6 taxa) were collected in 180 Lemon Bay samples (Table 1). Pinfish *Lagodon rhomboides*, bay anchovy *Anchoa mitchilli*, spot *Leiostomus xanthurus*, and mojarras *Eucinostomus* spp. comprised a large portion (81.4%) of the catch made in shallow areas of Lemon Bay proper with the 21.3-m seine (bay seines). Spot and pinfish were the dominant species in deeper habitats sampled with 6.1-m trawls (composing 72.8% of the catch). Pinfish, silver perch *Bairdiella chrysoura*, and scaled sardine *Harengula jaguana* accounted for 79.4% of the catch made with the 183-m haul seine. Spot and bay anchovy were the most numerous species in the tributaries, composing 55.9% of the tributary catch (river seines). No introduced fish species were represented in our samples. Detailed collection data, including seasonality, size, and density information, are available in Stevens et al. (2010a).

Physicochemical conditions were similar among the bay areas we examined during each sampling event (Figure 3). Mean water temperatures (based on water column–averaged values at each site) ranged from 16° to 31°C during the study period. Mean salinities were relatively high and stable, ranging from 28 to 38 psu. Lower Tampa Bay had slightly lower mean salinities during most sampling events, reflecting the influence of the Manatee River, a large tributary

Table 1. Fish and selected macroinvertebrate taxa collected during Lemon Bay bimonthly sampling, June 2009–April 2010. Values refer to number of individuals collected. Effort, or the total number of hauls, is labeled *E*. Taxa are arranged phylogenetically.

| Species | Common Name | River Seine E=72 | Bay Seine E=60 | Bay Trawl E=24 | Bay Haul Seine E=24 | Totals E=180 |
|---|---|---|---|---|---|---|
| Penaeidae | | | | | | |
| *Farfantepenaeus duorarum* | pink shrimp | 108 | 731 | 47 | . | 886 |
| Panuliridae | | | | | | |
| *Panulirus argus* | spiny lobster | . | . | 1 | . | 1 |
| Portunidae | | | | | | |
| *Portunus spp.* | portunus crabs | . | 1 | 47 | 2 | 50 |
| *Callinectes ornatus* | shelligs | . | 1 | 7 | 2 | 10 |
| *Callinectes sapidus* | blue crab | . | 7 | 170 | 33 | 210 |
| Xanthidae | | | | | | |
| *Menippe spp.* | stone crabs | 1 | . | 101 | . | 102 |
| Ginglymostomidae | | | | | | |
| *Ginglymostoma cirratum* | nurse shark | . | . | . | 1 | 1 |
| Dasyatidae | | | | | | |
| *Dasyatis americana* | southern stingray | . | . | 2 | 1 | 3 |
| *Dasyatis sabina* | Atlantic stingray | . | . | 2 | 2 | 4 |
| Elopidae | | | | | | |
| *Elops* spp. | ladyfish | . | . | 1 | 94 | 95 |
| Ophichthidae | | | | | | |
| *Myrophis punctatus* | speckled worm eel | . | 1 | . | . | 1 |
| Engraulidae | | | | | | |
| *Anchoa hepsetus* | striped anchovy | 27 | 5 | 23 | . | 55 |
| *Anchoa mitchilli* | bay anchovy | 7,740 | 7,382 | 1 | . | 15,123 |
| Clupeidae | | | | | | |
| *Brevoortia* spp. | menhadens | 2 | . | . | . | 2 |
| *Harengula jaguana* | scaled sardine | 128 | 719 | 2 | 957 | 1,806 |
| *Opisthonema oglinum* | Atlantic thread herring | . | 42 | . | 3 | 45 |
| *Sardinella aurita* | Spanish sardine | . | 93 | . | . | 93 |
| Ariidae | | | | | | |
| *Ariopsis felis* | hardhead catfish | . | 1 | 49 | 103 | 153 |
| *Bagre marinus* | gafftopsail catfish | . | . | 1 | . | 1 |
| Synodontidae | | | | | | |
| *Synodus foetens* | inshore lizardfish | 18 | 51 | 14 | 25 | 108 |
| Phycidae | | | | | | |
| *Urophycis floridana* | southern hake | . | 11 | 5 | | 16 |
| Batrachoididae | | | | | | |
| *Opsanus beta* | Gulf toadfish | 4 | 1 | 5 | 9 | 19 |
| Ogcocephalidae | | | | | | |
| *Ogcocephalus cubifrons* | polka-dot batfish | . | . | 13 | . | 13 |
| Mugilidae | | | | | | |
| *Mugil cephalus* | striped mullet | 1,734 | 1,153 | . | 44 | 2,931 |
| *Mugil curema* | white mullet | 1 | . | . | 19 | 20 |
| *Mugil gyrans* | whirligig mullet | 23 | 740 | . | 31 | 794 |
| Atherinopsidae | | | | | | |
| *Membras martinica* | rough silverside | . | 1 | . | . | 1 |
| *Menidia spp.* | silversides | 964 | 804 | . | . | 1,768 |
| Belonidae | | | | | | |
| *Strongylura marina* | Atlantic needlefish | . | . | . | 5 | 5 |

Table 1. Continued.

| Species | Common Name | River Seine E=72 | Bay Seine E=60 | Bay Trawl E=24 | Bay Haul Seine E=24 | Totals E=180 |
|---|---|---|---|---|---|---|
| *Strongylura notata* | redfin needlefish | 118 | 21 | . | 52 | 191 |
| *Strongylura spp.* | needlefishes | 2 | 2 | . | . | 4 |
| *Strongylura timucu* | timucu | 1 | 1 | . | . | 2 |
| Hemiramphidae | | | | | | |
| *Hyporhamphus meeki* | false silverstripe halfbeak | . | 5 | . | 3 | 8 |
| *Hyporhamphus unifasciatus* | Altantic silverstripe halfbeak | . | 1 | . | . | 1 |
| Fundulidae | | | | | | |
| *Fundulus similis* | longnose killifish | . | 9 | . | . | 9 |
| *Lucania parva* | rainwater killifish | 1,195 | 3,183 | . | . | 4,378 |
| Poeciliidae | | | | | | |
| *Gambusia holbrooki* | eastern mosquitofish | 1 | . | . | . | 1 |
| *Poecilia latipinna* | sailfin molly | . | 3 | . | . | 3 |
| Cyprinodontidae | | | | | | |
| *Floridichthys carpio* | goldspotted killifish | 30 | 38 | . | . | 68 |
| Syngnathidae | | | | | | |
| *Hippocampus erectus* | lined seahorse | . | . | 2 | . | 2 |
| *Hippocampus zosterae* | dwarf seahorse | 3 | 14 | . | . | 17 |
| *Syngnathus floridae* | dusky pipefish | . | 5 | . | . | 5 |
| *Syngnathus louisianae* | chain pipefish | . | 6 | 10 | . | 16 |
| *Syngnathus scovelli* | Gulf pipefish | 6 | 115 | 12 | . | 133 |
| Scorpaenidae | | | | | | |
| *Scorpaena brasiliensis* | barbfish | . | 1 | 4 | . | 5 |
| Triglidae | | | | | | |
| *Prionotus scitulus* | leopard searobin | 2 | . | 27 | 1 | 30 |
| *Prionotus tribulus* | bighead searobin | . | 1 | 17 | . | 18 |
| Centropomidae | | | | | | |
| *Centropomus undecimalis* | common snook | 17 | 3 | . | 115 | 135 |
| Serranidae | | | | | | |
| *Centropristis striata* | black sea bass | . | 1 | . | . | 1 |
| *Diplectrum formosum* | sand perch | . | . | 4 | . | 4 |
| *Mycteroperca microlepis* | gag | . | 2 | . | 4 | 6 |
| *Serraniculus pumilio* | pygmy sea bass | . | . | 2 | . | 2 |
| *Serranus subligarius* | belted sandfish | . | . | 11 | . | 11 |
| Carangidae | | | | | | |
| *Caranx hippos* | crevalle jack | 3 | 3 | . | 3 | 9 |
| *Oligoplites saurus* | leatherjack | 5 | 8 | . | 3 | 16 |
| *Trachinotus carolinus* | Florida pompano | . | . | . | 2 | 2 |
| *Trachinotus falcatus* | permit | . | . | . | 93 | 93 |
| Lutjanidae | | | | | | |
| *Lutjanus griseus* | gray snapper | 17 | 62 | 1 | 53 | 133 |
| *Lutjanus synagris* | lane snapper | 4 | 18 | 11 | 20 | 53 |
| Gerreidae | | | | | | |
| *Diapterus auratus* | Irish pompano | 1 | . | . | 4 | 5 |
| *Eucinostomus gula* | silver jenny | 549 | 772 | 222 | 152 | 1,695 |
| *Eucinostomus harengulus* | tidewater mojarra | 408 | 83 | 5 | 51 | 547 |
| *Eucinostomus spp.* | mojarras | 4,348 | 5,906 | 188 | . | 10,442 |
| *Eugerres plumieri* | striped mojarra | 5 | . | . | 9 | 14 |

Table 1. Continued.

| Species | Common Name | River Seine E=72 | Bay Seine E=60 | Bay Trawl E=24 | Bay Haul Seine E=24 | Totals E=180 |
|---|---|---|---|---|---|---|
| Haemulidae | | | | | | |
| *Haemulon aurolineatum* | tomtate | . | 1 | . | . | 1 |
| *Haemulon plumierii* | white grunt | . | 6 | . | 5 | 11 |
| *Orthopristis chrysoptera* | pigfish | . | 118 | 95 | 199 | 412 |
| Sparidae | | | | | | |
| *Archosargus probatocephalus* | sheepshead | 9 | 60 | 11 | 122 | 202 |
| *Diplodus holbrookii* | spottail pinfish | . | 12 | . | 8 | 20 |
| *Lagodon rhomboides* | pinfish | 2,482 | 25,394 | 1,489 | 3,081 | 32,446 |
| Sciaenidae | | | | | | |
| *Bairdiella chrysoura* | silver perch | 1 | 623 | 14 | 1,400 | 2,038 |
| *Cynoscion nebulosus* | spotted seatrout | . | 55 | 2 | 16 | 73 |
| *Leiostomus xanthurus* | spot | 8,769 | 6,911 | 2,083 | 55 | 17,818 |
| *Menticirrhus americanus* | southern kingfish | . | . | 3 | . | 3 |
| *Pogonias cromis* | black drum | . | . | . | 1 | 1 |
| *Sciaenops ocellatus* | red drum | 10 | 5 | . | 2 | 17 |
| Scaridae | | | | | | |
| *Nicholsina usta* | emerald parrotfish | . | 33 | 4 | 7 | 44 |
| Blenniidae | | | | | | |
| *Chasmodes saburrae* | Florida blenny | 1 | 12 | 2 | 1 | 16 |
| *Hypleurochilus caudovittatus* | zebratail blenny | . | . | 10 | . | 10 |
| *Hypsoblennius hentz* | feather blenny | . | . | 1 | . | 1 |
| Gobiidae | | | | | | |
| *Bathygobius soporator* | frillfin goby | . | . | 3 | . | 3 |
| *Gobiosoma bosc* | naked goby | 2 | . | . | . | 2 |
| *Gobiosoma longipala* | twoscale goby | . | . | 2 | . | 2 |
| *Gobiosoma robustum* | code goby | 9 | 217 | 4 | . | 230 |
| *Gobiosoma spp.* | gobies | 19 | 82 | 10 | . | 111 |
| *Microgobius gulosus* | clown goby | 129 | 365 | 3 | . | 497 |
| Ephippidae | | | | | | |
| *Chaetodipterus faber* | Atlantic spadefish | . | . | 1 | 8 | 9 |
| Sphyraenidae | | | | | | |
| *Sphyraena barracuda* | great barracuda | 9 | 1 | . | 10 | 20 |
| *Sphyraena borealis* | northern sennet | . | 5 | . | . | 5 |
| Scombridae | | | | | | |
| *Scomberomorus maculatus* | Spanish mackerel | . | . | . | 1 | 1 |
| Paralichthyidae | | | | | | |
| *Ancylopsetta quadrocellata* | ocellated flounder | . | 1 | 3 | 1 | 5 |
| *Citharichthys macrops* | spotted whiff | . | . | 10 | . | 10 |
| *Etropus crossotus* | fringed flounder | . | . | 18 | . | 18 |
| *Paralichthys albigutta* | Gulf flounder | . | 6 | 30 | 17 | 53 |
| Achiridae | | | | | | |
| *Achirus lineatus* | lined sole | 4 | 5 | 6 | . | 15 |
| *Trinectes maculatus* | hogchoker | . | . | 1 | . | 1 |
| Cynoglossidae | | | | | | |
| *Symphurus plagiusa* | blackcheek tonguefish | . | 1 | 7 | 2 | 10 |
| Monocanthidae | | | | | | |
| *Aluterus schoepfii* | orange filefish | 1 | . | . | . | 1 |
| *Monacanthus ciliatus* | fringed filefish | . | . | 4 | . | 4 |
| *Stephanolepis hispidus* | planehead filefish | 1 | 41 | 15 | 8 | 65 |

Table 1. Continued.

| Species | Common Name | River Seine E=72 | Bay Seine E=60 | Bay Trawl E=24 | Bay Haul Seine E=24 | Totals E=180 |
|---|---|---|---|---|---|---|
| Ostraciidae | | | | | | |
| *Acanthostracion quadricornis* | scrawled cowfish | . | 3 | 8 | . | 11 |
| Tetraodontidae | | | | | | |
| *Sphoeroides nephelus* | southern puffer | . | 15 | 7 | 2 | 24 |
| Diodontidae | | | | | | |
| *Chilomycterus schoepfii* | striped burrfish | . | 2 | 54 | 6 | 62 |
| Totals | | 28,911 | 55,981 | 4,907 | 6,848 | 96,647 |

that discharges into lower Tampa Bay. Mean dissolved oxygen remained above 5 mg/l, and each bay followed the same trend, with the lowest values in summer and the highest values in winter.

Some patterns were evident in overall mean fish abundances among the bays (Figure 4). The abundances of small-bodied fish sampled with 21.3-m bay seines differed significantly among the bays (ANOVA, $P < 0.001$). Lemon and Sarasota bays had greater abundances than lower Tampa Bay and lower Charlotte Harbor (Tukey's pairwise comparisons, $P < 0.01$). Although the abundance pattern for fish collected with the 6.1-m trawl was similar to that collected with the 21.3-m seine (i.e. greater abundances in Lemon and Sarasota Bays), the differences between estuaries were not significant (ANOVA, $P > 0.05$). There were no obvious trends or significant differences (ANOVA, $P > 0.05$) among the bays in abundances of fish collected with 183-m seines.

Differences in fish assemblages were also apparent among the bay areas, with Lemon and Sarasota bays distinguished from Charlotte Harbor and Tampa Bay. ANOSIM comparisons among the estuaries did not indicate significant assemblage differences ($P > 0.05$). The MDS plots, however, show that the Lemon and Sarasota Bay fish assemblages separated from the Charlotte Harbor and Tampa Bay fish assemblages collected with each gear type at Bray-Curtis similarity percentages ranging from 68 to 75 (Figure 5). Moreover, mean abundances of most of the taxa that distinguished these two groups, for each gear type, were greater in Lemon and Sarasota bays than in Charlotte Harbor and Tampa Bay (SIMPER, Figure 6). All of the 12 distinguishing species collected with 21.3-m bay seines were more abundant in Lemon and Sarasota bays. The bay anchovy, for example, was more than four times more abundant in those two estuaries. Seven of the top 12 distinguishing taxa collected with trawls were more abundant in Lemon and Sarasota bays (i.e. pinfish, silver jenny *Eucinostomus gula*, stone crabs, mojarras, spot, blue crab, and pink shrimp). The majority of the distinguishing species collected with 183-m seines also had greater abundances in Sarasota and Lemon bays (i.e., silver jenny, silver perch, pigfish *Orthopristis chrysoptera,* sheepshead *Archosargus probatocephalus*, common snook *Centropomus undecimalis*, hardhead catfish *Ariopsis felis*, gray snapper *Lutjanus griseus*, and ladyfishes *Elops* spp.).

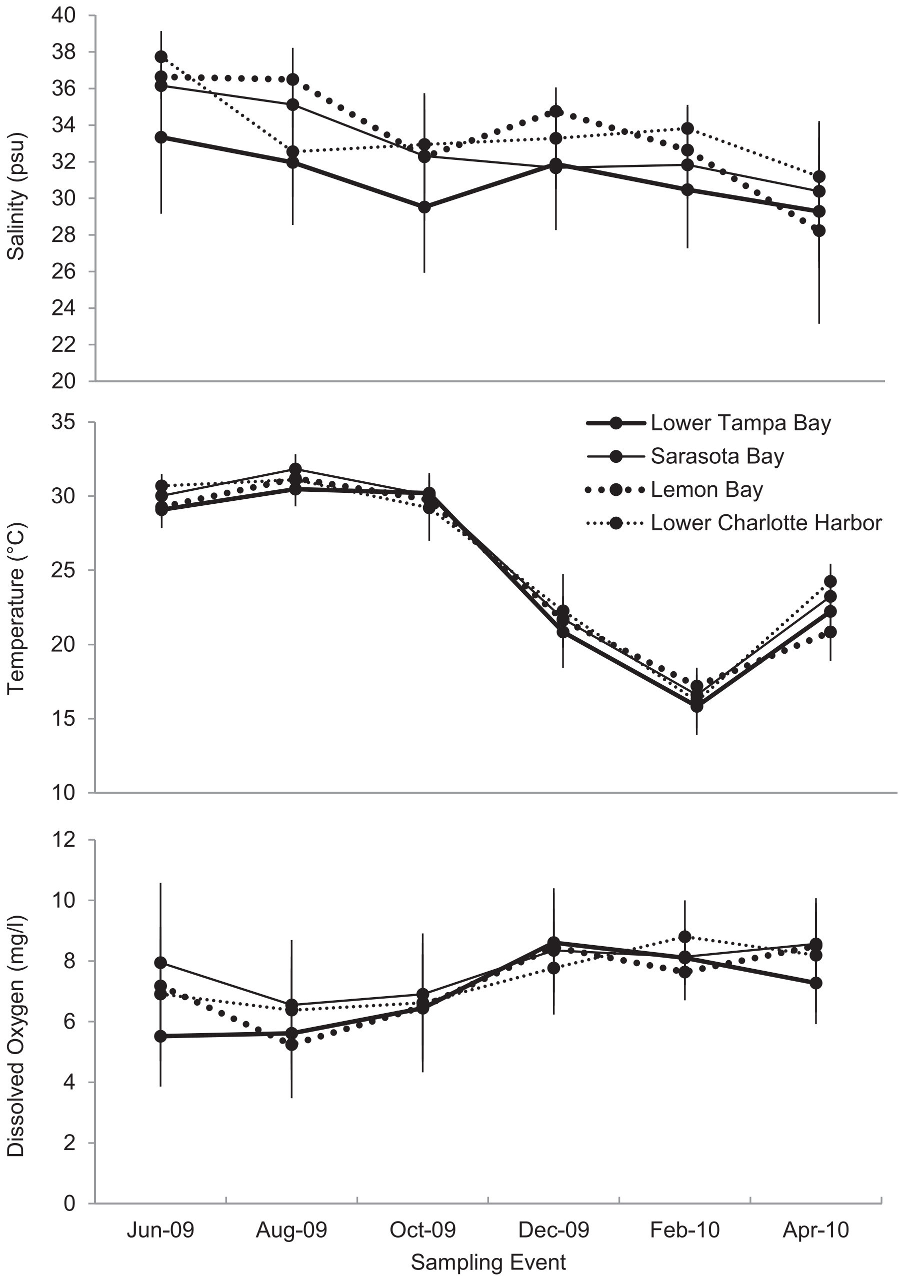


Figure 3. Mean (± standard deviation) water temperature, salinity, and dissolved oxygen taken during the study period (June 2009–April 2010) in Lemon Bay and nearby estuaries on the coasts of central and southwest Florida. Values calculated from water column averages collected during each sample.

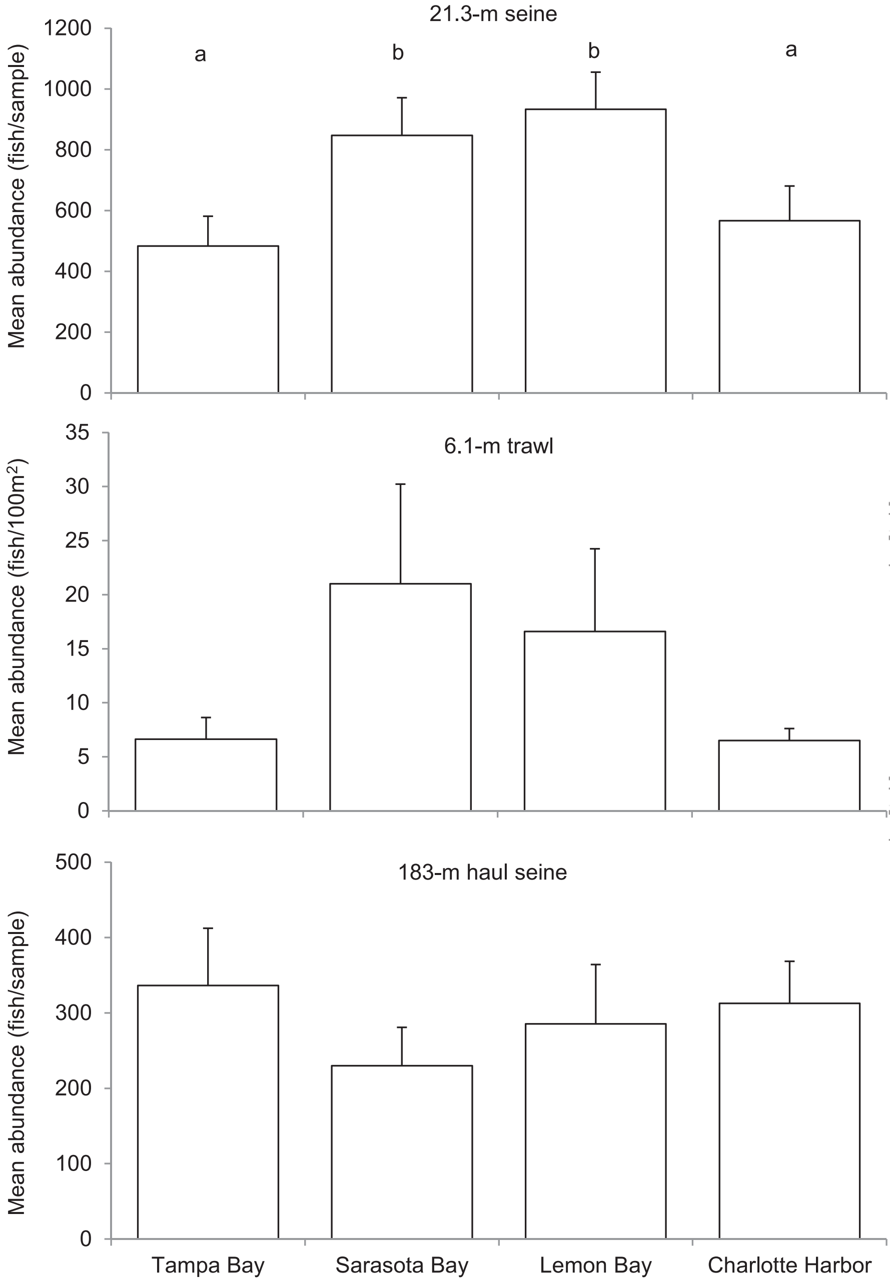


Figure 4. Mean abundance (± standard error) of fish collected in Lemon Bay and nearby estuaries on the coasts of central and southwest Florida. Letters above bars in 21.3-m seine plot represent groups identified as significantly different (ANOVA, $P < 0.001$, Tukey post hoc test, b > a); significant differences ($P < 0.05$) were not found among fish abundances based on 6.1-m trawls or 183-m seine collections.

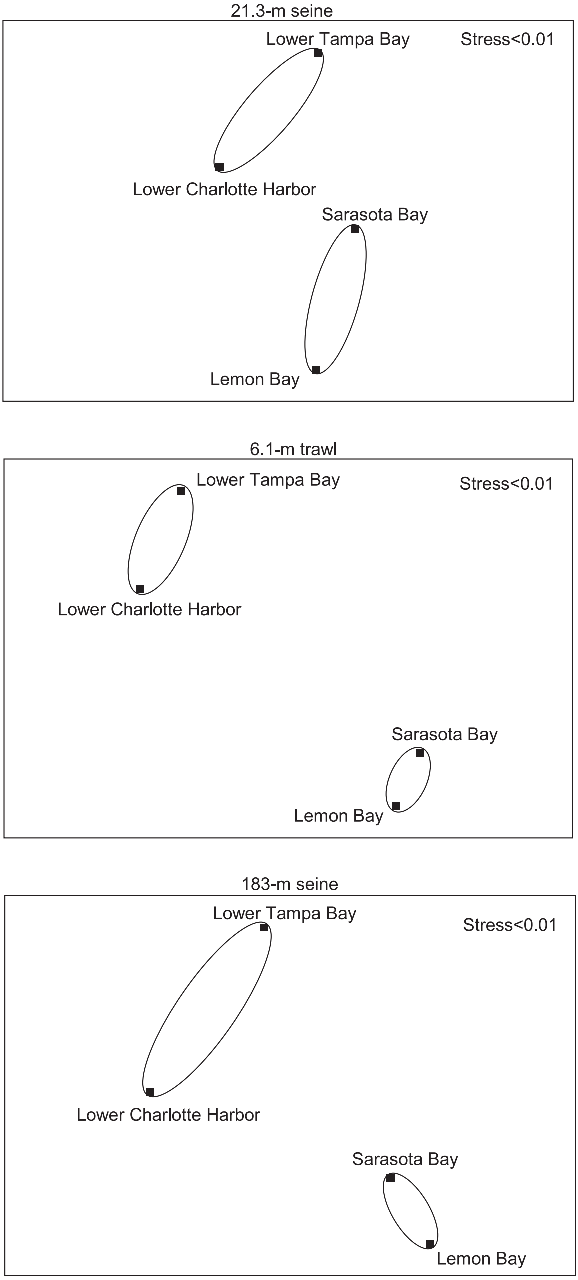


Figure 5. Two-dimensional nonmetric scaling ordination (MDS) of fish assemblages collected in Lemon Bay and nearby estuaries on the coasts of central and southwest Florida during bimonthly sampling (June 2009–April 2010). Ellipses denote groups identified using Bray-Curtis similarity percentages of 68-75 from hierarchical agglomerative cluster analysis.

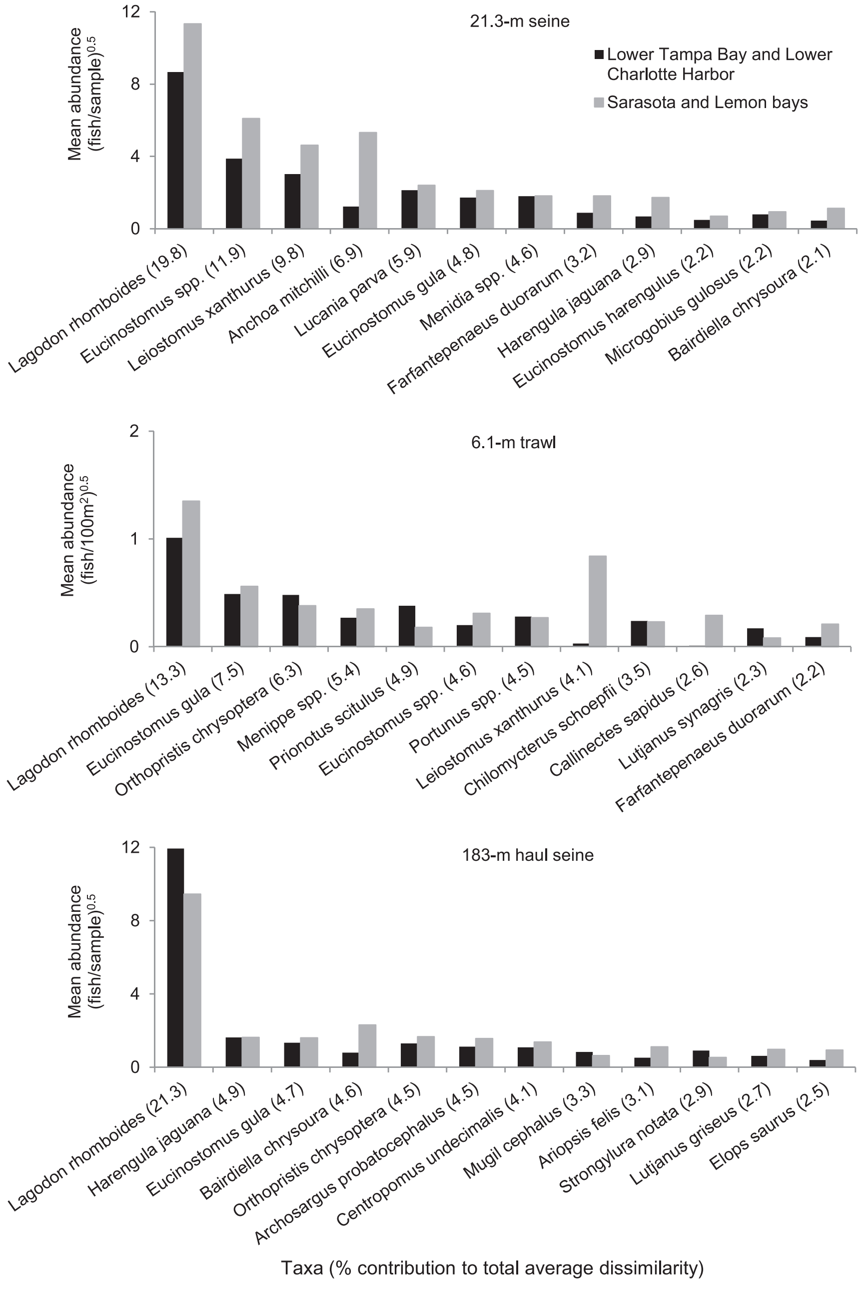


Figure 6. Similarity percentage analysis (SIMPER) showing fishes that distinguished bay groups depicted in MDS ordination. Abundance is the square root of average abundance (fish/sample or fish/100 m$^2$) as output from SIMPER. Total average dissimilarities between groups ranged from 68.5 to 79.1%. The contribution of each species to the total average dissimilarity is shown in parentheses.

## Discussion

The list of taxa collected in Lemon Bay was consistent with a comprehensive list of fishes reported for the adjacent Charlotte Harbor estuarine system (Poulakis et al. 2004) and with results of studies examining habitat use of fish in Charlotte Harbor (Poulakis et al. 2003, Idelberger and Greenwood 2005, Greenwood et al. 2006, Greenwood et al. 2007, Stevens et al. 2010b). It is noteworthy that nonnative species were not collected in Lemon Bay during the study period. High rainfall that occurs during summer would likely have increased the probability of collecting nonnative freshwater and euryhaline fishes such as cichlids, which tend to move downstream with high freshwater flows (Adams and Wolfe 2007). These species, however, may have been present farther upstream in the tributaries. Similar studies in larger bay systems to the north (Paperno et al. 2008, Greenwood 2012) and south (Stevens et al. 2008a, Stevens et al. 2008b, Champeau et al. 2009, Stevens et al. 2010c, Idelberger et al. 2011) have reported established populations of nonnative fishes.

The tributaries of Lemon Bay were found to be a juvenile habitat for many fish species. Juvenile spot, bay anchovy, striped mullet *Mugil cephalus*, red drum *Sciaenops ocellatus*, and common snook were at least twice as abundant in the tributaries than in Lemon Bay proper (fish density estimates are included in Stevens et al. 2010a). These fishes spawn either in the ocean or in the estuary downstream of their juvenile habitats, and early life history stages (larvae to juveniles) must swim or be transported upstream. Shallow-water refuge, high levels of productivity, and the salinity gradient in the tributaries likely provide physicochemical conditions that are favorable for juveniles of these species. Since these early life stages often use selective tidal stream transport and olfactory navigation to travel to nursery grounds (e.g. Schultz et al. 2003), maintenance and timing of freshwater inflows within the range expected under natural conditions are important for ensuring optimal access to these habitats.

The fish assemblages of Lemon and Sarasota bays appeared to differ from those of lower Tampa Bay and lower Charlotte Harbor. In particular, there were significant differences among the estuaries in total fish abundances of the small-bodied fishes collected with 21.3-m bay seines (ANOVA, Tukey's HSD test). Although ANOSIM did not detect differences in fish assemblage structure among the estuaries (perhaps due to relatively low sample size), the additional multivariate analyses (MDS, CLUSTER followed by SIMPER) seemed to corroborate the findings of the univariate tests. Fish assemblages collected with the 21.3-m bay seine differed and all the distinguishing species were more abundant in Lemon and Sarasota bays than in lower Charlotte Harbor and lower Tampa Bay. Although total fish abundances did not differ significantly for collections made with the other types of sampling gear, the multivariate analyses indicated a similar trend: the distinguishing species, as determined by SIMPER, were generally more abundant in Lemon and Sarasota bays than in lower Charlotte Harbor and lower Tampa Bay.

The differences in fish abundance and community structure among the bay systems we examined may be related to differences in their overall size. The same types of sampling gear and techniques were used in all systems, and physical and biological factors (e.g. water conditions, sediment types, submerged vegetation) appeared to be similar. The most striking difference is the geomorphologies of the bays; Lemon and Sarasota bays are considerably smaller and narrower than Charlotte Harbor and Tampa Bay. Factors such as current and water circulation patterns, water depth, the length of shoreline relative to area of open water, and proximity of Gulf passes to juvenile habitat may differ sufficiently between the small and large bays to affect fish recruitment patterns. For example, the differences in fish assemblages we found between the small and large estuaries were most pronounced for the small fishes collected with 21.3-m seines, with larger numbers of juveniles of many Gulf-spawning species (e.g. pinfish and spot) found in the smaller bays. If larval supplies entering Gulf passes are roughly similar, then the densities of juveniles occupying relatively small bays could be higher than those of larger bays where settlement may occur across a much broader area. Also, narrow bay systems present more shoreline, with its associated shallow seagrass flats, in proximity to the Gulf passes. Shallow shorelines and seagrass beds have been shown to be important habitat for juveniles of many fish species (Bell and Pollard 1989, Arrivillaga and Baltz 1999, Travers and Potter 2002, Acosta et al. 2007). Proximity of suitable habitat to Gulf passes has been found to be an important factor in determining settlement densities of ocean-spawning species (Hannan and Williams 1998, Brown et al. 2005).

While proximity to inlet habitat may have had a positive effect on recruitment or survival of offshore-spawning fishes, conditions in the smaller bays also seemed to favor many small resident estuarine species, including bay anchovy and rainwater killifish *Lucania parva*. These results illustrate that the factors accounting for the greater abundances of fishes in the smaller bay systems are more numerous and complex than inlet dynamics and bay geomorphology alone. It is possible that Lemon and Sarasota bays have greater fish production than that of lower Charlotte Harbor and Tampa Bay (i.e. the resident estuarine species have greater reproductive output, growth, and survivorship). Increases in fish production can be caused by increases in nutrient loads into a receiving body (Deegan and Peterson 1992), with smaller water bodies possibly more sensitive to such changes. The increased production can be viewed as a benefit initially, so long as the receiving body does not reach a tipping point. A tipping point can occur when nutrient loads reach a threshold where conditions can change suddenly and become difficult to reverse (Caddy 2000, Hagy et al. 2004). These sudden changes often take the form of excessive algal blooms with cascading effects (e.g. seagrass losses from shading by algae, fish kills resulting from low dissolved oxygen). Comparative studies across proximate estuaries that include estuarine fauna, as was conducted here, may be of interest to resource managers modeling the effects of nutrient loading into estuaries.

Our year-long survey indicated that the Lemon Bay estuary supports a fish fauna that is similar to those of other bay systems in southwest Florida. More extensive sampling would likely collect additional species that are less common or more difficult to collect. Overall fish abundances were greater in Lemon Bay and nearby Sarasota Bay than in the larger bay systems. The factors accounting for these differences are uncertain, but may be related to general differences in the overall size of these bays.

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# Total ecosystem services values (TEV) in southwest Florida: The ECOSERVE method

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**Abstract** Ecosystem services are the multitude of resources and processes that are supplied by natural ecosystems. This includes a wide range of natural processes that help sustain and fulfill human life, such as: purifying air and water, detoxifying and decomposing wastes in water, pollinating natural vegetation, cycling and moving nutrients, protecting coastal shores from erosion by waves, moderating weather extremes and their impacts, and providing aesthetic beauty and intellectual stimulation that lift the human spirit. A total ecosystem services quantification using the ECOSERVE mapping methodology was developed by the Southwest Florida Regional Planning Council in partnership with the Charlotte Harbor National Estuary Program (CHNEP) and the Sanibel-Captiva Conservation Foundation in the project "Estimating and Forecasting Ecosystem Services within Pine Island Sound, Sanibel Island, Captiva Island, North Captiva Island, Cayo Costa Island, Useppa Island, Other Islands of the Sound, and the Nearshore Gulf of Mexico" (Beever and Walker 2013), linking the derived ecosystem function measurements with a geo-spatially positioned ecosystem services database, derived from wetland functional analyses that are utilized to forecast, evaluate and report trends in wetlands ecosystem services.

**Keywords** Ecosystem services, GIS, Pine Island Sound, Sanibel, decision-making, sea level rise

## Introduction and Background

The natural world, its biodiversity, and its constituent ecosystems are critically important to human well-being and economic prosperity, but are consistently undervalued in conventional economic analyses and decision-making. (CAVSARTE 2004, Samuelson 1983) Ecosystems and the services they deliver underpin our very existence. Humans depend on these ecosystem services to produce food, regulate water supplies and climate, and breakdown waste products. Humans also value ecosystem services in less obvious ways: contact with nature gives pleasure, provides recreation and is known to have positive impacts on long-term health and happiness (UK National Ecosystem Assessment 2011). Human societies get many benefits from the natural environment. In Southwest Florida, we are well aware of how important eco-tourism, sport and commercial fishing, and natural products such as locally produced fruits, vegetables and honey are to our regional economy. The natural environment also provides, for free, services that we would otherwise have to pay for, in both capital outlay, and operation and maintenance costs.

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The United Nations 2004 Millennium Ecosystem Assessment grouped ecosystem services into four broad categories: Provisioning, such as the production of food and water; Regulating, such as the control of climate and disease; Supporting (Habitat), such as nutrient cycles and crop pollination; and Cultural (Socio-economic), such as spiritual and recreational benefits.

Ecosystem services values can be useful in justifying grant funding and in leveraging restoration dollars. These values can also used by decision-makers when establishing and maintaining conservation lands, siting utilities, making development decisions, putting numbers to the impacts associated with decisions, and adding data when critical trade-offs are discussed. Discussions on retaining local government land acquisition programs in Lee County and Collier County are examples of the use of ecosystem services values in decision-making.

Recognition of how ecosystems provide complex services to mankind are documented in Western culture to at least Plato (c. 400 BC) (Marsh 1965). The term 'environmental services' was introduced in a 1970 report of the Study of Critical Environmental Problems (SCEP 1970), which listed services including insect pollination, fisheries, climate regulation and flood control. In following years, variations of the term were used, but eventually 'ecosystem services' became the standard in scientific literature (Ehrlich and Ehrlich 1981). Modern expansions of the ecosystem services concept include socio-economic and conservation objectives (de Groot et al. 2012).

There has been some resistance, particularly from life-science academics and environmentalists, to establishing monetary values for ecosystem services because it is difficult to capture the total value and there is always the potential to risk under-valuing the services. However, assigning value to ecosystem services is a necessary and important tool to demonstrate the economic values being lost to society. Ecosystem evaluation is a field that requires great amounts of innovation; developing communication tools that can relate tangible value to ecosystem services will be meaningful in protecting healthy watersheds. The important message is that conservation provides myriad economic and social benefits at the local level. Protecting these systems will provide society with greater economic security, healthy, bountiful fisheries, a higher quality of life and clean drinking water (Dlugolecki 2012).

**Materials and Methods**

The project locations assessed by the ECOSERVE method, so far include the greater Pine Island system, the Lee County Conservation 2020 lands, and the Conservation Collier lands.

Pine Island Sound Project - Pine Island Sound is located in Lee County, Florida, lying between Pine Island and the barrier islands of Sanibel Island, Captiva Island, North Captiva Island and Cayo Costa, which separate the Sound from the Gulf of Mexico (Figure 1). The Sound connects to Gasparilla Sound and Charlotte Harbor to the north, and to San Carlos Bay and the Caloosahatchee River to the south. The Sound is conterminous with the Pine Island Sound Aquatic Preserve, which was established in 1970 and consists of 220 $km^2$ (54,000 acres) of submerged land. Important habitats in the Sound include mangrove forests, seagrass beds, salt marshes, oyster reefs and tidal flats. Pine Island Sound has the most extensive sea grass beds in the greater Charlotte Harbor complex. Large areas of oyster reef-hard bottom communities are comparatively rare in the

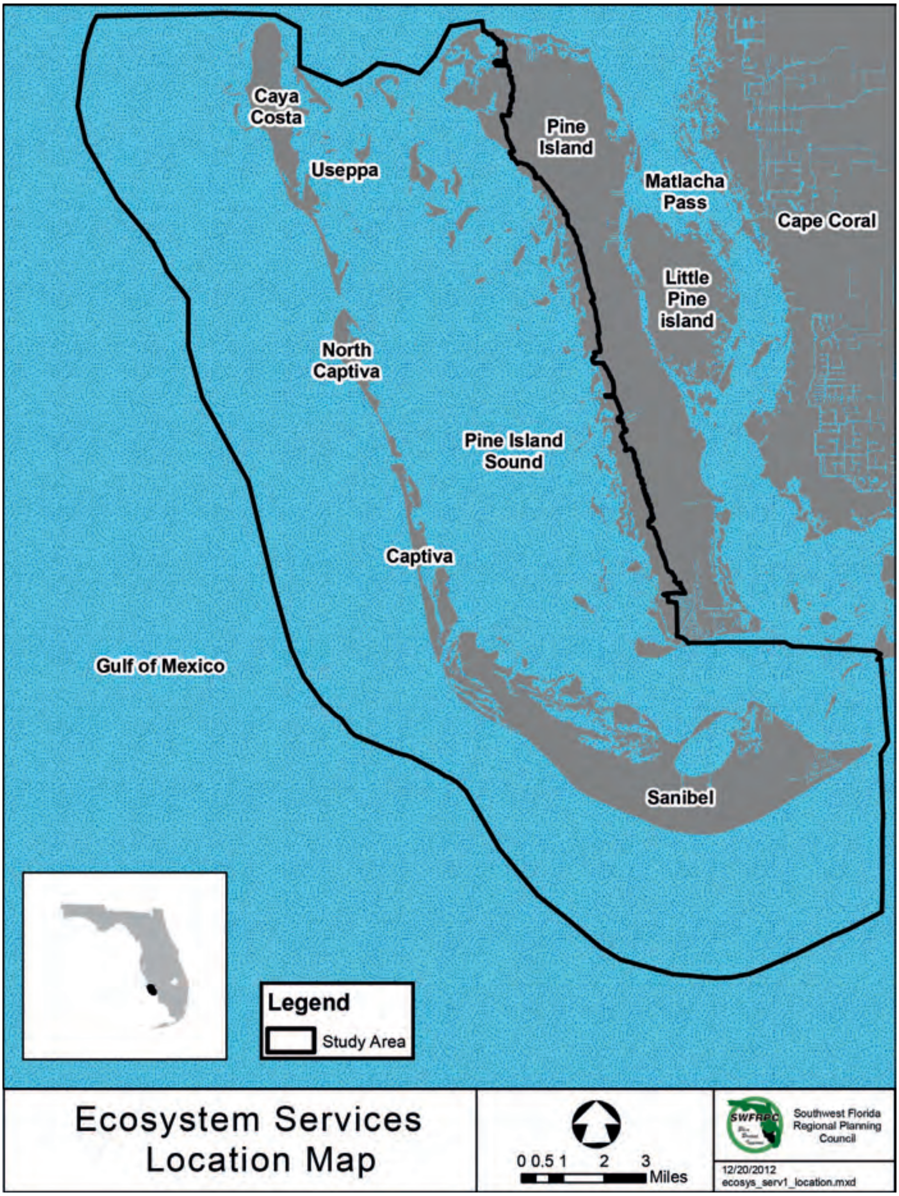


Figure 1. Pine Island Sound Project Study Area.

estuary, and are often found associated with the shoreline, yet at a distance from it. The total estimated mangrove acreage for the Pine Island Sound/Matlacha sub basin is 7,732 hectares (19,107 acres). The mangroves in this region are extensive and fringe all of the protected shorelines of the barrier islands. The mosaics of mangroves in the southern portion of this region, on the northern coast of Sanibel Island, are particularly noted for the living resources they support, such as large populations of endangered roseate spoonbills. The establishment of ecosystem services values for this ecologically rich area was among the first valuations of this type in this region and served as a model for later evaluations of conservation lands in Lee and Collier Counties.

For Pine Island Sound we identified all the existing habitat types found in the study area through GIS analysis of existing aerial imagery. The most recent available GIS layers were utilized included the NOAA bathymetry (CHNEP 2011), the CHNEP Benthic Habitat Map (Photo Science 2007), the SFWMD seagrass mapping (FWC 2011), the SFWMD land use map (2008), and the 2011 salt marsh by type map created by the SWFRPC in the salt marsh study (Beever et al. 2012).

Functional assessment methods utilized in Beever et al. (2013) including UMAM, HGM, and WRAP, were utilized, linking the derived ecosystem function measurements with geo-spatially positioned ecosystem services information. The combined land and bottom cover map was constructed in the following order beginning with the NOAA bathymetry layer, the CHNEP Benthic Habitat Map, the SFWMD seagrass map, the SFWMD land-use map and finally the salt marsh by type map. Later layers in the sequence took priority over earlier layers. For small edges where no land use was indicated, usually at the meeting of the benthic layer and land cover, the blank area was assigned the value of the nearest adjacent benthic or bathymetric value.

The range and quantity of ecosystem services provided by existing habitats was estimated, including the marine, estuarine and freshwater wetlands, and associated native uplands of the islands were estimated. Dollar values for ecosystem services were obtained either directly or through calculation from the following: Allsopp et al. (2008), Beever III and Cairns (2002), Beever III (2011), Beever III et al. (2012), Bolund and Hunhammar (1999), Casey and Kroeger (2008), Committee on Assessing and Valuing the Services of Aquatic and Related Terrestrial Ecosystems (CAVSARTE) (2004), Conservation International (2008), Costanza et al. (1997), Costanza (2008), Costanza et al. (2008), Dale and Polasky (2007), Dlugolecki (2012), Engeman et al. (2008), Goulder and Kennedy (2007), Goulder and Kennedy (2011), Hazen and Sawyer (1998), Henderson and O'Neil (2003), Isaacs et al. (2009), Krieger (2001), Kroeger and Casey (2007), Kroeger et al. (2008), Lee County Visitor and Convention Bureau (2002), Losey and Vaughan (2006), Lugo and Brinson (1979), McKee (2011), McLeod and Salm (2006), Metzger et al. (2006), Morales (1980), Paling et al. (2009), Pidwirny (2006), Vo et al. (2012), Sathirathai (2003), South Florida Water Management District (2007), Spaninks and van Beukering (1997), American Sportfishing Association (2006), USFWS (2007), UK National Ecosystem Assessment (2011), Weisskoff (2012), and Wells et al. (2006).

For developed land use types (Florida Land Use Cover Classification System (FLUCCS) codes 100, 200, and 800), the Total Ecosystem Services Value (TEV) calculation involved the estimation of the amount of non-impervious surface on the specific land use type and the vegetation type on that lands use. This information was obtained from Thompson et al. (2011), the Sanibel Plan (City of Sanibel 2012), the U.S. Census Bureau (2010) and the Sanibel-Captiva Conservation Foundation.

We produced a current map of ecological services value topographies (ECOSERVE) using combined GIS map and the total estimated ecosystem services value for each habitat type. This provides a visual representation of the geographic distribution of the TEV within the study area.

We then calculated the TEV for the total acreage of each habitat type within the study area. Each dollar value for ecosystem service provided by a particular habitat was specified for its year of estimation. The dollar value of the ecosystem service estimate was then normalized using the inflation rate from the consumer price index (Bureau of Labor Statistics 2012) to a 2012 dollar value using the appropriate inflation multiplier. The resulting ecosystem service value per acre was then multiplied by the number of acres of that habitat type to obtain the total ecosystem services value for that habitat type in the study area. All of the habitat values were then summed to obtain a total ecosystem services value for the entire Pine Island Sound study area (Beever and Walker 2013).

An ecosystem services topography (ECOSERVE) geographic information system (GIS) layer was generated from the TEV value per acre mapped within each habitat. This geographic representation of the TEV for the study area provides a visual representation of where the highest value habitats are and how different changes on the landscape can change and transform the value and nature of the ecosystem services provided by the estuary and barrier islands (Figure 2).

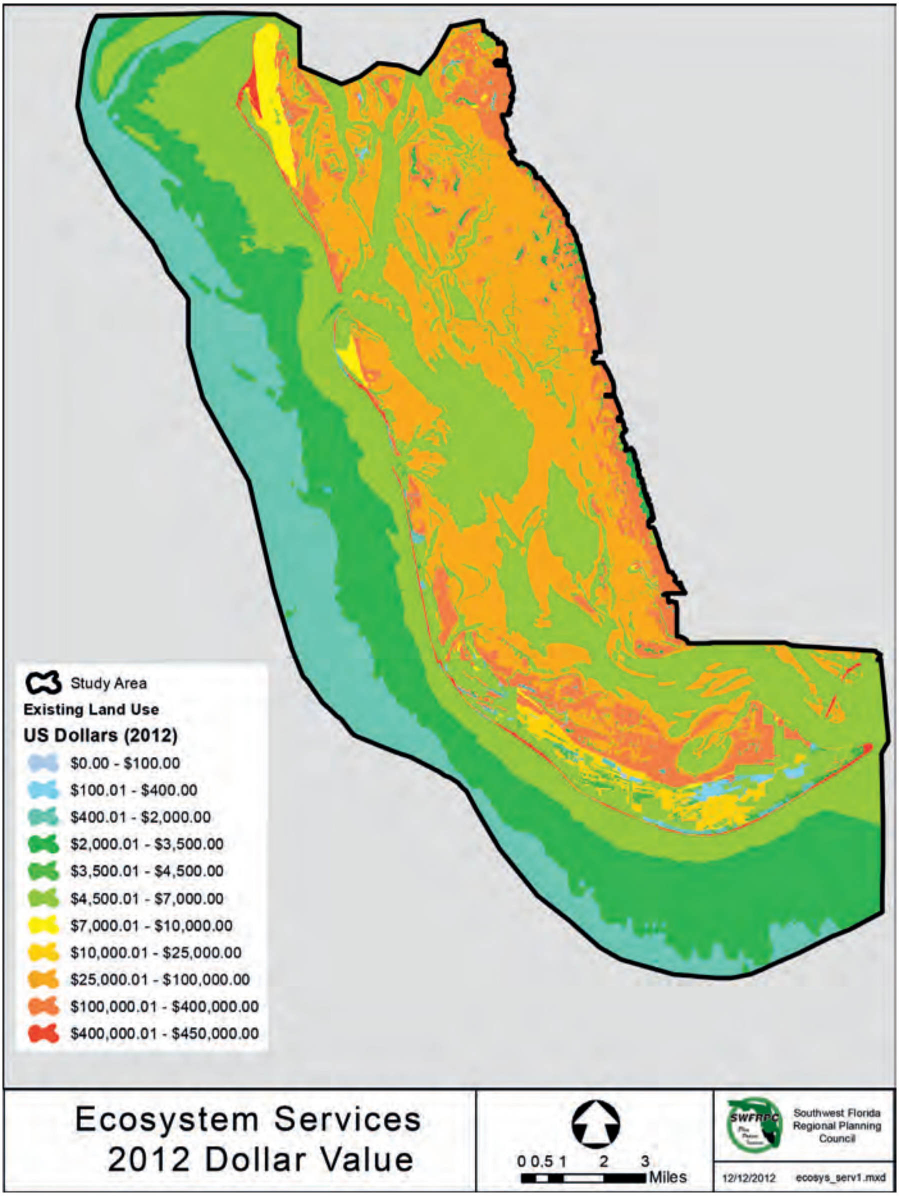


Figure 2. Ecosystem Services 2012 Dollar Values in the Year 2012 (Baseline Condition) for the Study Area.

The ECOSERVE map can be combined with other geographic information system (GIS) layers for functional analyses by service type, by geographic boundary and in combination. This process is a tool that can generate projections of ecosystem services that may result from land use changes, anticipated climate changes, natural and man-made disasters, the implementation of alternative wetland protection and land conservation programs or the landscape scenario which

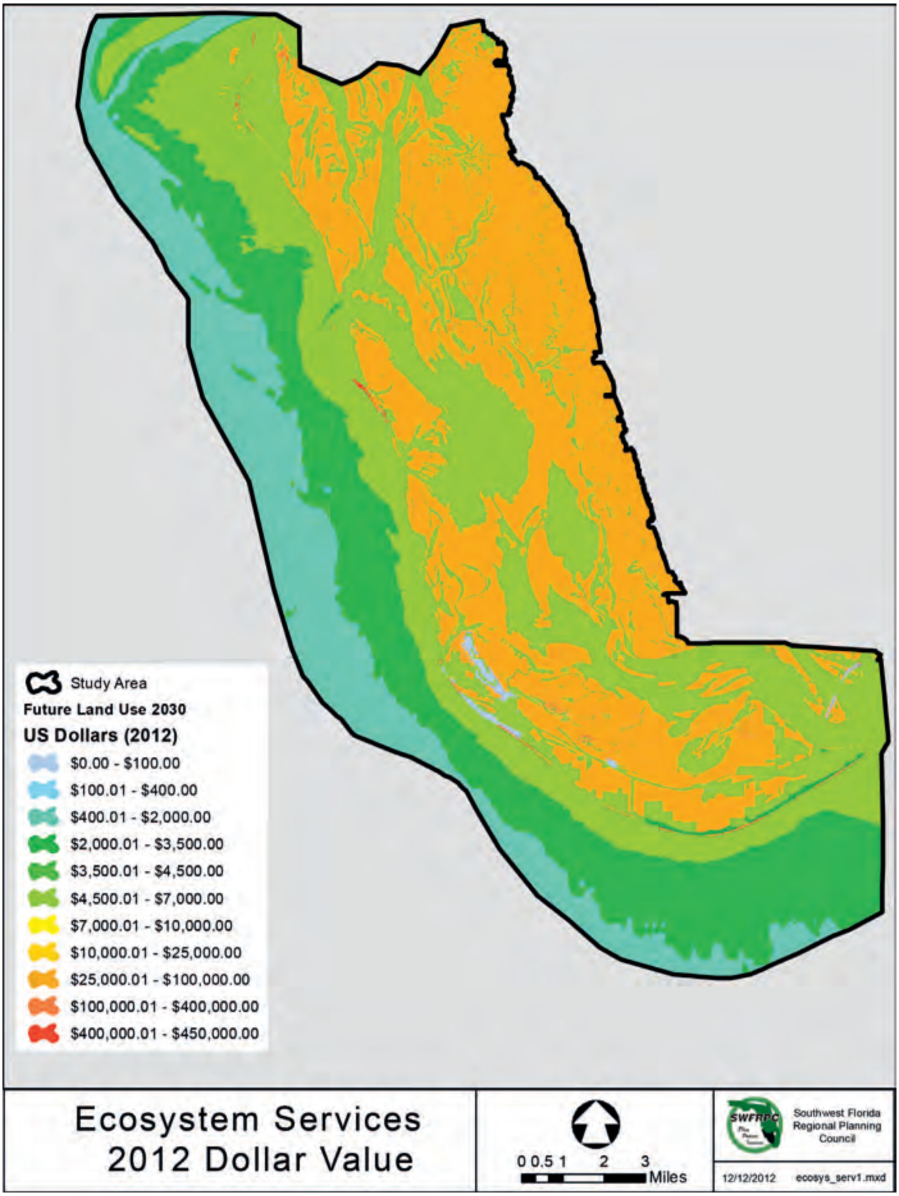


Figure 3. Ecosystem Services 2012 Dollar Values Projected for the Year 2030 (Future Land Use Map Condition) for the Study Area.

would reflect the eventualities resulting from making no changes to current land use, management or regulatory policy.

We generated two alternate future ECOSERVE topographies related to the anticipated land use changes that come with the future land use projection for the year 2030 (Figure 3) and for a one-foot sea level rise in the study area (Figure 4).

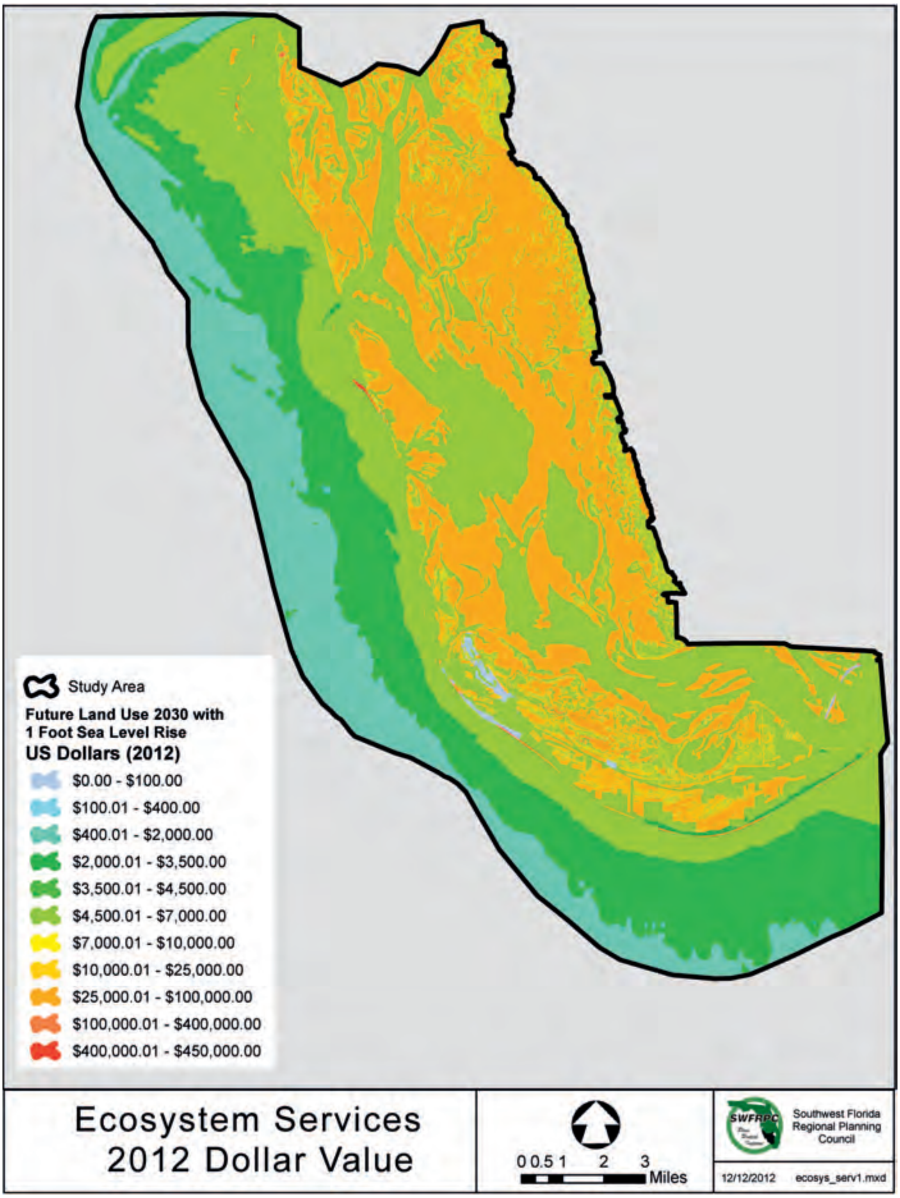


Figure 4. Integrated Ecosystem Services 2012 Dollar Values Projected for the Year 2100 (80% IPCC probability) with 1 foot of Sea Level Rise for the Study Area.

**Lee County Conservation 2020 study.** All the 76 existing habitat types found on Conservation 2020 lands were identified by Lee County staff. The most recent available tabulation was utilized. The total area of Conservation 2020 lands is 10,088 hectares (24,928 acres). The largest habitat type is Mesic Pine Flatwoods which constitutes 19.3% of all Conservation 2020 lands. Mesic Pine Flatwoods, Wet Flatwoods, Disturbed Mesic Pine Flatwoods, Mangrove Swamp, Disturbed Wet

Pine Flatwoods, and Strand Swamp make up 51.5 % of all the Conservation 2020 lands. For the Conservation 2020 lands project the range and quantity of ecosystem services provided by existing habitats was estimated utilizing the methods developed by Beever and Walker (2013) described above. For habitats not included in the Pine Island Sound study (Beever and Walker 2013), dollar values for ecosystem services were obtained either directly or through calculation from the same sources (Beever 2013a). When a habitat was indicated as disturbed a 50% valuation of the full TEV for that habitat type was utilized based on consultation with Lee County staff concerning the extent of disturbance. To estimate ecosystem service values, we developed a table analogous to the Pine Island Sound study, including 2012 dollar normalizations (Beever 2013a).

**Conservation Collier.** As of February 2014, Conservation Collier lands made up approximately 0.02% of Collier County's land, with 19 properties totaling 4,054.7 acres. All the 57 existing habitat types found on Conservation Collier lands were identified by Collier County staff. The most recent available tabulation was utilized. The total area of Conservation Collier lands is 1,640.9 hectares (4,054.7 acres). The largest habitat type is Improved Pasture which constitutes 17.5% of all Conservation Collier lands. Pine Flatwoods are the most common type of native habitat constituting 8.3% of Conservation Collier Lands. Disturbed depression marsh is the most common freshwater wetland habitat (7.8%) and mangroves are the most common saltwater wetland habitat (7.6%). Improved Pasture, Pine Flatwoods, disturbed Depression Marsh, Mangrove Swamp, Upland Mixed Forest, Mixed Wetland Hardwoods, Wetland Scrub, and Cypress Swamp make up 64.4% of all the Conservation Collier lands.

All the 57 existing habitat types found on Conservation Collier lands were identified by Collier County staff. The most recent available tabulation was utilized. The total area of Conservation Collier lands is 1,641 hectares (4,055 acres). The largest habitat type is Improved Pasture which constitutes 17.5% of all Conservation Collier lands. Pine Flatwoods are the most common type of native habitat constituting 8.3% of Conservation Collier Lands. Disturbed depression marsh is the most common freshwater wetland habitat (7.8%) and mangroves are the most common saltwater wetland habitat (7.6%). Improved Pasture, Pine Flatwoods, disturbed Depression Marsh, Mangrove Swamp, Upland Mixed Forest, Mixed Wetland Hardwoods, Wetland Scrub, and Cypress Swamp make up 64.4% of all the Conservation Collier lands. The methods of the Pine Island Study and Conservation 2020 Study were applied here, including the production of a final summary table (Beever 2013b).

## Results

These studies identify the range and quantity of ecosystem services provided by marine, estuarine and freshwater wetlands and native upland habitat and to determine how the functional types of wetlands and native uplands, their distribution and position in the landscape, and their ecological condition affects ecosystem services within the Pine Island Sound, and on Sanibel Island, Captiva Island, North Captiva Island, Cayo Costa Island, Useppa Island and Islands of the Sound; the Lee County Conservation 2020 lands, and the Collier Conservation lands.

Based on current calculations of Total Ecosystem Services Value (TEV) for the Pine Island Sound Study Area the 2012 TEV is $7,033,362,634.63. (Figure 2) (Beever and Walker 2013). It is notable that the majority (98.6%) of the TEV is found in the top seven habitats including mangrove swamp (38.3%), continuous seagrass beds (36.5%) estuarine embayments (10.7%), swimming beaches (5.4%), the nearshore Gulf of Mexico (3.7%), discontinuous sea grass beds (2.3%), and unvegetated shallow subtidal bottoms (1.8%). These seven habitats make up 83.9% of the physical area of the study area.

Projecting to the build-out scenarios envisioned on the Future Land Use (FLU) Map for the study area which projects to a future at 2030 and beyond it is possible to see using the ECOSERVE what the future anticipated ecosystem services value would be. The future land use map is not as detailed in specific development and conservation land covers and uses simplified land use covers. Subsequently some cover types are subsumed into large categories such as Coastal Rural, Conservation Lands Upland, Conservation Lands Wetland, Outer Island, Outlying Suburban, Public Facilitates, Rural, Suburban, Urban Community, and Wetlands. For these larger land use categories mean TEV per acre were derived from the specific land uses included in that category (Beever and Walker 2013). The resulting FLU 2030 map indicated loss of native upland and wetland habitat, some conversions of existing developed land uses to more intense developed land uses, and the elimination of most exotic plant communities with their development into human land uses (Figure 4). In the land use changes associated with the 2030 build out the following land use categories are no longer present: mobile home parks, dry prairie, pine flatwoods, Brazilian pepper, upland melaleuca, saltwater ponds, shrub black mangrove, freshwater marsh, algal marsh, and saltern. The resulting landscape has a reduced TEV Future Land Use Projection (2030) (TEV) for the Pine Island Sound area of \$5,146,537,673.59 measured in 2012 dollars. (Beever and Walker 2013). This constitutes a 26.83% loss of 2012 TEV. If a projected level of inflation between 2012 and 2030 is applied then the dollar value of the TEV would increase. This study does not have a projection for what that inflation rate might be since in the prior 18 year period inflation rates have ranged from 0.03 to 4.3 per year with an average of $2.47 \pm 1.02$ (U.S. Bureau of Labor Statistics 2012). If one assumes the same rates of inflation (which the authors do not consider likely) then inflation would make up the loss of TEV for a total of \$7,434,688,323.27 in 2030 dollars. Of course those 2030 dollars would be worth \$0.69 in 2012 currency.

Projecting to a future with the build-out scenarios envisioned on the Future 2030 Land Use Map and a one foot sea level rise in the study area it is possible to project the future anticipated ecosystem services values of the resulting landscape. The point at which one-foot of additional sea level will occur in the project study area depends on several variables that influence the local relative sea level rise, including; global sea level rise from thermal expansion, global sea level rise from non-replaced land ice melt; local sediment deposition; local accretion from wetland plant activity; local accretion from storm effects; local erosion from storm effects and long term erosive forces; human mediated sediment loss including shoreline hardening, disruption of coastal dynamics, reduction of alluvial deposition by dams and water control structures: plate tectonic lift, recession and tilt; and the geomorphic migration of barrier islands.

The current measured sea level rise rate for Lee County is approximately 9 inches in 100 years. Assuming this rate continued without acceleration then a one-foot sea level rise above 2012 levels would be attained in the year 2162.

The resulting one-foot sea level rise map (1FSLR) map indicates significant loss of native upland and wetland habitat, some conversions of existing developed land uses to open water, and the elimination of most exotic plant communities (Beever and Walker 2013). The resulting landscape has a Future Land Use Projection of Total Ecosystem Services Value (TEV) for the Pine Island Sound study area with the Future Land Use Projection (2030) and a one foot of sea level rise of $4,184,956,813.96.

The effect of sea level rise varies with the habitat type. It is important to remember that while a habitat may change from a current above water land cover to an open water submerged condition that the new open water habitat has an ecosystem services values that must be accounted for. If only the loss of above water habits to open water is accounted for than the TEV loss in the study area for a 1 foot sea level rise is $4,019,726,568.16. However, the gain of open water generates $165,230,245.80 of TEV with 1 foot sea level rise. Therefore the net loss of TEV from sea level rise in the study area for 1 foot of sea level rise separate from the 2030 land use changes is $1,126,811,105.43. This is a 16.0% loss of 2012 TEV from the sea level rise alone. Combining the sea level rise of 1 foot with the future land use changes results in a $3,013,636,066.47 loss of TEV. This constitutes a 42.9% loss of 2012 TEV.

The establishment of ecosystem services values for the ecologically rich Conservation 2020 lands was the second valuation using ECOSERVE in Lee County. These ecosystem services values can be used by decision-makers when establishing and maintaining conservation lands, siting utilities, or making development decisions, putting numbers to the impacts associated with those decisions, and adding data when critical trade-offs are being discussed. These values will also be useful in justifying other grant funding and in leveraging future restoration dollars. The output of this project is an assessment of the total ecosystem services provided by all habitat types on the Conservation 2020 lands in Lee County, Florida. This assessment is available to the local governments and the public to assist in planning for use in developing conservation plans. This work was intended to identify the range and quantity of ecosystem services provided by all the land covers types on Conservation 2020 lands including marine, estuarine and freshwater wetlands and native upland habitat, and disturbed habitats.

Based on current calculations the 2012 TEV of the Conservation 2020 lands is $628,865,027.93 (Beever 2013a). It is notable that the majority (92.5 %) of the TEV is found in the top nine habitats including mangrove swamp (63.3%), mesic flatwoods (15.1%) wet flatwoods (3.7%), mesic flatwoods - disturbed (2.6%), mangrove swamp - disturbed (2.2%), strand swamp (2.0%), scrubby flatwoods (1.47%), depression marsh (1.1%), and wet flatwoods - disturbed (1.8%). These nine habitats make up 54.6% of the physical area of the Conservation 2020 lands.

Estimating ecosystem services values for the diverse Conservation Collier lands was the third valuation, following Beever and Walker (2013) and Beever (2013a) using ECOSERVE in southwest Florida. The output of this project is an assessment of the total ecosystem services provided by all habitat types on the Conservation Collier properties, including a detailed accounting of how mangrove habitat value was derived. This assessment will be made available to the local government and the public to assist in planning for use in developing the Collier County conservation plans. This work is intended to identify the range and quantity of ecosystem services provided by all the land covers types on Conservation Collier lands including marine, estuarine and freshwater wetlands and native upland habitat, and disturbed habitats.

The calculations the 2013 TEV of the Conservation Collier property is \$144,988,312.22 per year (Beever 2013b Table 1). Note that this only includes the Collier County owned lands in the Conservation Collier program, a small subset of the total of all the conservation lands in Collier County. The majority (90.6 %) of the TEV for Conservation Collier property is found in the top twelve habitats including mangrove swamp (58.1%), pine flatwoods (7.3%) upland mixed forest (5.6%), mixed wetland hardwoods (3.1%), cypress (2.9%), wetland scrub (2.6%), depression marsh (2.1%), wetland forest mixed (2.0%), upland mixed forest disturbed (1.6%), depression marsh disturbed (1.6%), bottomland forest (1.4%), cypress- cabbage palm-pine (1.2%) and saltwater marsh (1.1%). These twelve habitats make up 64.6% of the physical area of the Conservation Collier property.

## Discussion and Conclusions

The output of this project is an assessment of the total ecosystem services provided by all habitat types in the Pine Island Sound, Sanibel Island, and Captiva Island; Lee County Conservation 2020 lands; and the Conservation Collier lands. These assessments have been made available to local governments, state agencies, federal entities and the public for use in developing wetlands and conservation planning, restoration and enhancement plans.

In addition, ecosystem services topography (ECOSERVE) layers were generated that can be combined with other ecosystem services layers for functional analyses by geographic boundary (watershed, municipality, county, etc.). Projections of alternate futures of ecosystem services resulting from land use changes and anticipated climate changes were completed.

The ECOSERVE method can be utilized to forecast and back cast alternate future and past landscapes. With more time and funding we could look at increased sea level rise extents, the benefits and costs of different land acquisitions, the consequences in terms of ecosystem services of various changes in wetland and upland extents resulting from restoration or development plans, the consequences of natural and man-made disasters, the implementation of alternative wetland protection and land conservation programs, as well as the potential impacts of making no changes to current land use, management, or regulatory policy. Utilization of the ECOSERVE layers will allow permit reviewers to evaluate the impact, for example, of alternative project site designs including reduction of areas of wetland impacts.

Subsequently the tables associated with the ECOSERVE mapping can include: quantification of observed habitat condition information linked to ecosystem services and their contributions to human well-being; quantification of the pollution prevention or mitigation services (e.g. chemical pollutant removal, sediment removal) provided by ecosystems with a comparison to the cost of providing them through built infrastructure; quantification by habitat of the amount of food or fiber produced per unit area in well-protected areas versus that of poorly protected or unprotected areas; the economic value of recreational opportunities provided by a specific habitat provision or by protection of fishable/swimmable water; construction costs avoided by the presence of habitats that slow and absorb floodwaters (flow mitigation or flood control).

Given more time and resources the maps and tables of this project could be improved by a detailed mapping of mangrove forest type to better estimate the ecosystem services provided by each type and better represent the relative functions of each forest type in location and landscape. As indicated by an internal separate analysis of salt marsh combined average versus salt marsh by detailed type estimates significant TEV differences can be obtained. We would expect the difference for a detailed mangrove forest type could be even more pronounced. Another refinement would be to apply a sea grass extent light extinction model to predict future sea grass extent losses as estuarine waters deepen. In this analysis the level of sea level rise (one foot) would not cause major sea grass bed losses as new shallow water is generated. With higher level of sea level rise the deeper edge of sea grass beds would move landward as light attenuation losses occurred in the deeper waters.

Another potential future application could be the ecosystem services values of the conservation easements (Beever and Walker 2015). This could demonstrate the ecosystem services provided by easements in contrast to public acquisitions that may have additional values such as of public access recreation.

More alternate futures could be evaluated with additional climate change perturbations, alternate land use plans, and regulatory environments. The differential benthic habitats in the Gulf of Mexico could be further refined and mapped with methods utilized in identifying the source locations of benthic drift algae.

This development of the ArcGIS-friendly ECOSERVE protocol for statistical and geographical analysis and interpretation can be used with the types of information generated by surveys of ecological condition indicators to quantify ecosystem services. ECOSERVE can be used to quantify the relative importance of perturbation stressors (e.g. land clearing, hydrologic alteration, development, climate change) that impact habitats and the ecosystem services they provide, ECOSERVE is a GIS tool that can be used to develop regionally relevant ecosystem services measurement and assessment programs and that can be used to assist in implementing efficient and effective decision-making by local and regional regulatory, mitigation, enforcement programs. The ECOSERVE method protocol is applicable elsewhere southwest Florida in the southeastern United States, and around the Gulf of Mexico, provided that the ecosystem services values are recalibrated to the specific conditions of the subject watershed.

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