PROJECT DESCRIPTION

INTRODUCTION

Estuaries and coastal ecosystems experience a range of stresses, both natural and anthropogenic. Human activities have greatly altered the timing, magnitude, and properties of inputs to coastal systems. Eutrophication affects most coastal ecosystems in the United States, and as a result most coastal ecological research has been conducted in systems that are [to some degree] eutrophic--including both LTER and LMER sites (e.g. Chesapeake Bay, Waquoit Bay, Plum Island, Virginia Coast Reserve, Columbia River, Tomales Bay, Georgia Rivers). *We propose to add a coastal Everglades site to the LTER network for several key reasons.* First, this system is oligotrophic, making it extremely sensitive to and responsive to anthropogenic stresses. As a result, the Everglades is an excellent venue to study ecological structure and function in coastal ecosystems because, in low-nutrient ecosystems, both tend to respond rapidly and often quite dramatically. Second, the Everglades is the focus of the largest watershed restoration effort ever implemented. The goal of this $8 billion, 50 year project (known as the Everglades Restudy) is to restore the quantity, quality, timing, and distribution of water flow to the Everglades. Third, the tropical climate of south Florida is particularly conducive to quantifying how large variability in precipitation and freshwater flow controls ecological interactions at the land margin interface. And fourth, a coastal Everglades LTER will take advantage of a number of ongoing projects and databases by synthesizing and enhancing existing research. For most, both project descriptions and data are accessible on the Internet (and access to a web browser may facilitate review of this proposal). As a result, with a coastal Everglades LTER we will be able to quantify ecological responses to interannual variability and disturbances which affect other coastal LTER and LMER sites while also studying the effects of a major perturbation (Everglades restoration). Our long term research program will focus on the following central idea and hypotheses:

Regional processes mediated by water flow control population and ecosystem level dynamics at any location within the coastal Everglades landscape. This phenomenon is best exemplified in the dynamics of an estuarine oligohaline zone where fresh water draining phosphorus-limited Everglades marshes mixes with water from the more nitrogen-limited coastal ocean.

Hypothesis 1: In nutrient-poor coastal systems, long-term changes in the quantity or quality of organic matter inputs will exert strong and direct controls on estuarine productivity, because inorganic nutrients are at such low levels.

Hypothesis 2: Interannual and long-term changes in freshwater flow control the magnitude of nutrient and organic matter inputs to the estuarine zone, while ecological processes in the freshwater marsh and coastal ocean control the quality and characteristics of those inputs.

Hypothesis 3: Long-term changes in freshwater flow (manifest through management and restoration in the coastal Everglades) will interact with long-term changes in the climatic and disturbance regimes to modify ecological pattern and process across coastal landscapes.

We propose to test these hypotheses along freshwater to marine gradients represented by landscape transects in two Everglades drainage basins located in Everglades National Park (ENP; Fig.1). The Shark River Slough transect (SRS) will be anchored at canal inflow points along the Tamiami Trail and will extend through the mangrove estuary to Florida's southwest coast. Historically, most of the water draining the "River of Grass" flowed through this system. The Taylor Slough/ENP Panhandle transect (TS/Ph) will be anchored at two main canal inflow points, and will extend through the Florida Bay estuary to the same coastal ocean endpoint. This is a smaller, more localized drainage basin. We have observed a clear productivity peak in the oligohaline zone of the SRS transect, but not along the TS/Ph transect. This dynamic is observable in soils and water nutrient concentrations, in biomass and productivity data from marsh plants, mangroves, and phytoplankton, and in secondary consumer biomass data (Fig.2). We propose to address the 3 primary hypotheses above by quantifying: 1) primary productivity; 2) consumer dynamics; 3) organic matter accretion and turnover in soils and sediments, and; 4) concentrations and turnover dynamics of inorganic nutrients and organic matter (particularly dissolved organic matter; DOM) along the SRS and TS/Ph transects. The central hypotheses (above) may also be represented as long-term scenarios (detailed below) with differential increases in freshwater flow, DOM, and nutrient inputs to these two coastal drainage basins. We will focus our specific research questions on these long-term scenarios.

CONCEPTUAL APPROACH AND JUSTIFICATION

It is a well established paradigm that freshwater inputs are an important exogenous control on ecological structure and function in estuaries (Day et al., 1989) and even oceans (Weaver et al., 1991). Variability in those inputs, and in the controls on those inputs, also plays a strong role in estuarine dynamics (i.e. the Pulsing Paradigm; Odum et al., 1995). In most coastal systems, man's influence greatly affects variability in freshwater inputs--most often by altering land use or watershed hydrologic patterns (Hopkinson and Vallino, 1995). Elucidating how exogenous forces [and the variation in those forces] control estuarine and coastal dynamics requires long data records (Hudson et al., 1989; Smith et al., 1989; Houde and Rutherford, 1993; Neinhuis, 1993; Stanley et al., 1993).
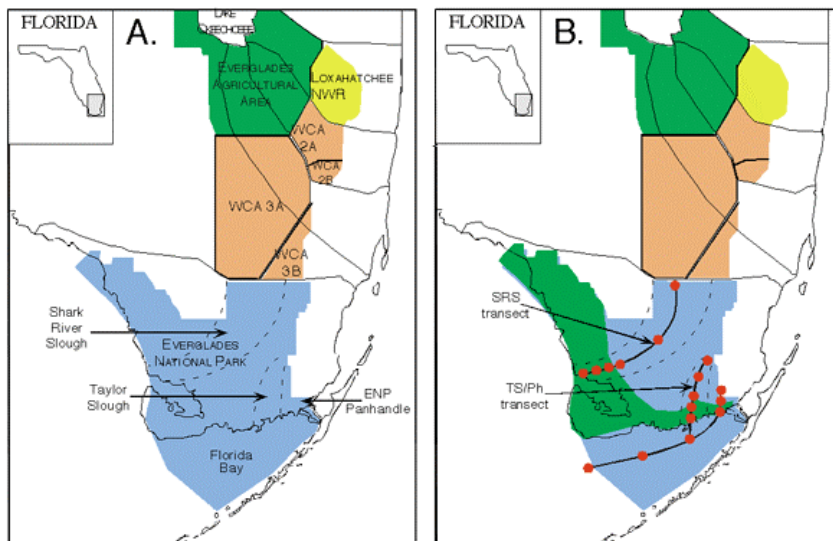

Figure 1: Map of south Florida showing (a) major components of the current and historical Everglades; note locations of Shark River Slough, Taylor Slough, the ENP Panhandle, and Florida Bay, all in ENP, and (b) showing locations of our coastal Everglades transects with study sites shown as red dots (6 along the SRS transect and 11 along the TS/PH transect, including 3 in Florida Bay); blue represents freshwater marsh and Florida Bay, green represents mangrove forest.

The coastal Everglades landscape is a particularly appropriate site for long-term research investigating how freshwater flow controls estuarine dynamics because: 1) the entire system is oligotrophic (Amador and Jones, 1993; Fourqurean et al., 1992, 1993); 2) the range in interannual variability in precipitation nearly equals mean annual rainfall (Duever et al., 1994; Rudnick et al., 1999), and; 3) freshwater inputs are being modified by Everglades restoration efforts (see http://www.restudy.org). Ecological structure and function of oligotrophic systems are extremely sensitive to exogenous inputs of nutrients, and responses to these inputs are both dramatic and easily quantified (Caraco et al., 1992; Carpenter et al., 1995; others). This is an excellent setting to study the importance of DOM to coastal ecosystems (Hypothesis 1) because DOM is the dominant form of organic matter and, potentially, is a large nutrient source in oligotrophic Everglades waters (Fourqurean et al., 1993; Boyer and Jones, 1999; Rudnick et al., 1999). The hydrologic perturbation provided by Everglades restoration will facilitate long-term study of how management interacts with both regional climatic variability and the regional disturbance regime (Hypothesis 3). Notably, all coastal LTER and LMER sites are affected by a disturbance regime of sea level rise and changing storm frequency/intensity, and by interannual variability in climatic controls.

Low salinity regions of many estuaries are sites of primary productivity maxima (Malone et al., 1980; Sharp et al., 1984; Boyer et al., 1993). This is most often caused by the confluence of freshwater phosphorus (P) inputs by the watershed and nitrogen (N) inputs from the estuary itself. The magnitude of freshwater flow thus influences the location of the productivity peak by controlling the position of this mixing zone. In the oligotrophic coastal Everglades, the estuarine headwaters are P depauperate but N rich. Therefore, estuaries of the coastal Everglades provide a unique model for inverse nutrient mixing, as P is supplied by coastal waters while N is supplied by freshwater flow (Hypothesis 2). The productivity gradients along the SRS and TS/Ph transects (Fig.2a) are well founded in a range of datasets (Fig.2b-f). Both transects are anchored upstream at canal inputs, where water inputs are usually characterized by total P (TP) concentrations slightly higher than ambient levels and by high total N (TN) concentrations (Boyer and Jones, 1999; Rudnick et al., 1999). Both transects meet at a marine endmember, where marine water inputs contain relatively more TP and less TN (for data see: http://www.fiu.edu/~serp/jrpp/wqmn/datamaps/datamaps.html). The productivity peak in the oligohaline region of the SRS basin is not found in the TS/Ph basin. This occurs because the Florida Bay estuary scavenges all available P from coastal waters before it reaches this oligohaline zone (Fourqurean et al., 1993; Boyer et al., 1998). In freshwater regions of SRS, soil P is greatest at the downstream site nearest the oligohaline region while in the TS/Ph basin the lowest soil P is at the downstream freshwater site (J.Trexler, unpubl.data). In SRS, soil P continues to increase seaward through the mangrove zone (Fig.2b; Chen and Twilley, 1999b), tracking the marine P source. In TS/Ph, low soil P values continue through the mangrove zone (Fig.2b; Oehm et al., in prep.) but increase seaward through Florida Bay (Fourqurean et al., 1993), also tracking the Gulf P source. Our water quality data from SRS estuarine and marine zones indicate 1) a non-conservative decline in DOM along the salinity gradient towards offshore; 2) an increase in inorganic N and P concentrations in the oligohaline region relative to both freshwater and marine water inputs; and 3) consistently high standing stocks of phytoplankton (measured as chlorophyll *a*) in this region (Fig.2c,d; http://www.fiu.edu/~serp/jrpp/wqmn/datamaps/datamaps.html).
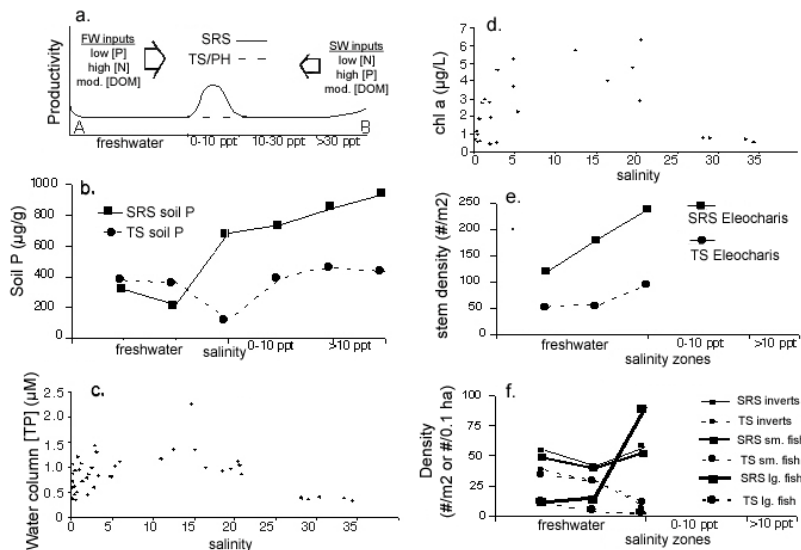
Figure 2: (a) conceptual simplification of oligohaline productivity peak in the SRS basin but not observed in the TS/Ph basin--as supported by existing data (b-f). Salinity axes for existing data show either actual salinities (estuarine data) or salinity zones (combination data) **(b)** wetland soil P concentrations from SRS transect, squares (freshwater data=J.Trexler, unpubl. data; SRS mangrove data=Chen and Twilley, in press); TS mangrove data =Oehm et al., in prep. **(c&d)** water [TP] and [chl a] from SRS mangrove zone (Boyer and Jones, unpubl. data) **(e)** spikerush stem density from SRS and TS/Ph freshwater marsh transects (J.Trexler, unpubl. data) **(f)** secondary producer densities from SRS and TS/Ph freshwater marsh transects, in # indiv./m^2 for aquatic invertebrates and small fish and # indiv./0.1 ha for large fish (J.Trexler, unpubl. data)

In the freshwater marsh, standing stocks of algae, some vascular plants, invertebrates, and fishes reflect spatial patterns in nutrients that differ between the SRS and TS/Ph basins. The floating periphyton mat is most dense at the downstream site in TS/Ph and least dense at the downstream site of SRS, consistent with experimental studies of the response of periphyton mats to nutrient variation (McCormick et al., 1996). Spikerush (*Eleocharis* sp.) stem density is highest in SRS and increases towards the estuaries in both basins (Fig.2e). Densities of aquatic invertebrates, small fish (<8 cm), and large fish (>8 cm) decrease downstream in TS/Ph; in SRS, invertebrates and small fish are equally abundant at all sites while large fish are substantially more abundant at the downstream site (Fig.2f). This pattern is consistent with a trophic cascade where increased productivity at the downstream site leads to increased biomass and turnover of the top trophic levels. We present a qualitative summary of how some ecosystem attributes compare and contrast along the SRS and TS/Ph transects in Table 1.

Our long-term research will address four probable scenarios that will modify the low salinity productivity maximum (Fig.2a). Notably, the ordinate axis is salinity and not geographical location because the low salinity zone changes position between wet and dry season. For an example of how high wet season discharge affects the estuaries of both basins as

Table 1: General description of various ecosystem attributes comparing and contrasting the 2 basin transects in qualitative and relative terms. TS/Ph=Taylor Slough/ENP Panhandle transect; SRS=Shark River Slough transect. Mod.=moderate; Min.=minimal.

Ecosystem Attribute	TS/Ph freshwater	TS/Ph mangrove	TS/Ph Subtidal est.	SRS freshwater	SRS mangrove
[P] in water	low	low	Low→mod.	low	Low→mod.
[N] in water	high	high	High→mod.	high	High→mod.
[DOM] in water	high	high	moderate	high	moderate
[P] in soils	low	low	Low→mod.	low	Moderate
Soil type	1° marl	peat	carbonate	peat	Silty peat
1° producer biomass	moderate	low	moderate	Moderate	high
Water residence time	short	long	long	short	short
Degree of tidal influence	none	minimal	Min.→mod.	none	Moderate
Magnitude of FW flow	moderate	moderate	low	high	moderate

well as the southwest Florida coastal zone, see http://www.fiu.edu/~serp/jrpp/wqmn/datamaps/sal-13.gif. Also, several long-term factors may act to change the location of the oligohaline zone, including increased freshwater flow (seaward displacement) and sea level rise (landward displacement).

1. Long-term scenario 1 - Increased freshwater flow (the optimal Everglades restoration scenario; Fig.3a): Freshwater inputs at the upstream ends of both the SRS and TS/Ph transects are not accompanied by related increases in nutrient or organic matter concentrations. We anticipate a small decrease in marine nutrient inflows, particularly in Florida Bay, due to simple conservation of mass. Increased freshwater will force the oligohaline productivity peak in SRS seaward during the wet season, but make little or no change in its dry season position. The result will be a broadening of this productivity peak in SRS, but no real change in the TS/Ph gradient.

2. Long term-scenario 2 - Scenario 1 plus increased marine nutrient inputs (the coastal eutrophication scenario; Fig.3b): The most dramatic effect of this scenario will be an increase in the magnitude of the SRS productivity peak and a productivity increase at the marine endmembers. If Florida Bay is unable to scavenge all of this additional marine P, the TS/Ph gradient will show a small oligohaline productivity peak as well, following the SRS model.

3. Long term-scenario 3 - Scenario 1 plus increased freshwater nutrient inputs (the direct terrestrial eutrophication scenario; Fig.3c): In this undesirable scenario, Everglades restoration results in nutrient inputs disproportionate to water flow inputs. The freshwater marshes will be most immediately affected by this, but as this new P progresses downstream, the result will be a higher productivity peak in the SRS gradient and initiation of a peak in the TS/Ph gradient, but no marine endmember increase along either transect.

4. *Long term-scenario 4 - Scenario 1 plus increased DOM inputs (the direct terrestrial eutrophication scenario; Fig.3d): The effect of this scenario will be quite different from Scenario 3, because nutrient remineralization from DOM(perhaps via a microbial loop) is less efficient. Thus, much of this new DOM will flow through the freshwater marshes relatively unchanged. However, much of this DOM may become bioavailable in the oligohaline zone by the combination of physical processes, such as flocculation, and biotic community changes. Thus, this scenario represents an increased input of terrestrial DOM that will stimulate estuarine productivity in the oligohaline zone of both the SRS and TS/Ph basins.*
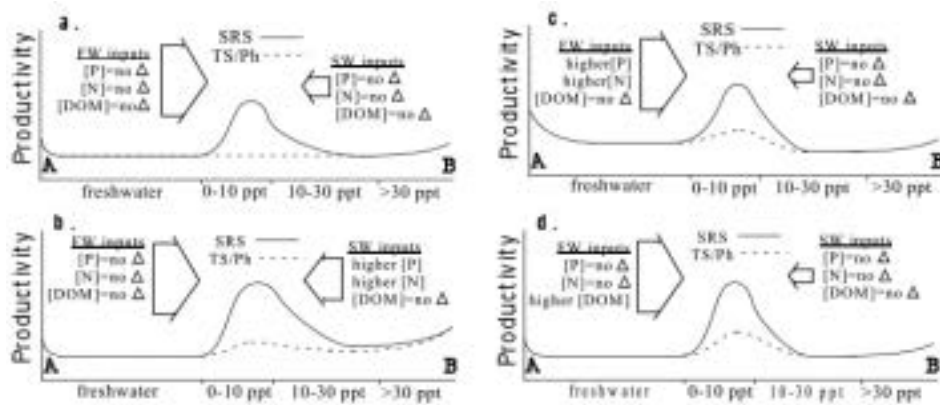

Figure 3: Hypothesized long-term scenarios. Arrows represent the hydrologic control; water quality conditions for each are shown relative to conditions in Fig.2a. A=canal input transect anchors; B=offshore marine transect anchors. (a) Scenario 1=increased freshwater flow; (b) Scenario 2=#1+coastal eutrophication; (c)#1+increased freshwater nutrient loading; (d)#1+increased freshwater DOM loading.

Ongoing Freshwater Research: Our group at FIU is currently involved with several large research efforts in the freshwater marshes of both the SRS and TS/Ph basins. The largest project is investigating how pristine Everglades marshes are changed by increased P concentrations. In this study, a multidisciplinary group of researchers at FIU is using three flumes in central SRS, each with three 100 m long P addition channels and 1 control channel, to quantify the effects of P additions on all major ecosystem components of freshwater Everglades marshes (R.Jones, D.Childers, and J.Trexler, Managing PIs). Phosphorus is added continuously to the 3 experimental channels at each flume to increase ambient concentrations by 0.17, 0.5, and 1.0 µM (Childers et al., 1999a; http://www.fiu.edu/~ecosyst/dosing.html). Two other studies are investigating how increased water inputs from canals affect nutrient dynamics and ecosystem structure in downstream marshes of both Taylor Slough and the ENP Panhandle. These studies are quantifying water quality, macrophyte productivity, and soil and periphyton dynamics along canal-estuary transects throughout the TS/Ph basin (http://www.fiu.edu/~ecosyst/c111.html; D.Childers, PI).

Several studies of consumers are ongoing, including an extensive effort to describe the dynamics of spatial distribution and abundance of fishes and macroinvertebrates in Everglades National Park and Water Conservation Areas 3A and 3B. This study is linked to research on fish dispersal and population genetic structure. Experimental studies of predator-prey dynamics and community regulation using enclosure and exclusion cages are also underway in conjunction with that sampling study (J.Trexler, PI). We also have extensive data from fish gut content analyses, collected as part of a mercury bioaccumulation and trophic dynamics study.

Ongoing Estuarine and Marine Research: Our group at FIU is currently involved with research and monitoring efforts in the estuarine zones of both the SRS and TS/Ph basins. One process-level project in the mangrove zone of the TS/Ph basin is quantifying the exchange of water and nutrients between these mangrove wetlands and Florida Bay. This project applies an hierarchical process approach to understand the ecological dynamics that influence this exchange, and is generating data on water quality, wetland fluxes, soil dynamics, and mangrove productivity (http://www.fiu.edu/~ecosyst/mangrove.html; D.Childers, PI).

The Southeast Environmental Research Center (SERC) at FIU operates a network of >330 water quality monitoring stations in coastal South Florida that includes 28 stations in Florida Bay, 22 stations in Whitewater Bay, 25 stations in the mangrove estuaries of Ten Thousand Islands, 28 stations from Marco Island to Pine Island Sound, 49 offshore stations, 25 stations in Biscayne Bay, and 154 stations in the Florida Keys National Marine Sanctuary (FKNMS). The purpose of this network is to assess and interpret status and trends in surface water quality of the coastal South Florida ecosystem (for a project overview and data to date, see: http://www.fiu.edu/~serp/jrpp/wqmn/datamaps/datamaps.html; R. Jones and J.Boyer, co-PIs).

We are also conducting ecosystem and community level research in the seagrass-dominated ecosystems of Florida Bay (J. Fourqurean, PI; http://www.fiu.edu/~seagrass/). Current projects include: 1) monitoring of seagrass community composition, biomass and productivity; 2) statistical modelling of the habitat requirements of seagrass-associated plants and animals; 3) the role of herbivores in seagrass ecosystems; 4) food web structure; 5) mass-balance modelling of water and solutes in Florida Bay; 6) the role of disturbance in determining the outcome of competitive interaction in seagrass meadows, and; 7) sediment-water column biogeochemical coupling.

PROJECT SYNTHESIS

Information synthesis is a key objective of our coastal Everglades LTER program. This is particularly important because this work will be interacting closely with a number of ongoing research and monitoring efforts. In fact, we envision this LTER as the hub that brings together many spokes of a large research wheel while also adding critical spokes of its own. The hub will be an hierarchical modelling approach that integrates data and spatial information from our empirical research. The first tier will involve development of process-based simulation models in real time with ongoing research. In this way, we will synthesize data as they are collected and use our models as adaptive management tools for our research program. Fig.4 conceptualizes the framework for this first tier of synthesis (the ecosystem components shown as boxes in Fig.4 correspond to the columns in Table 1). It includes all biotic parameters and exogenous forcings important to our proposed research, and identifies which components will be affected by our disturbance/perturbation regime. We have already developed a freshwater marsh periphyton model (Buzzelli et al., in review) and a marsh macrophyte-soil model is currently under development. We will work closely with R.Twilley to parameterize his individual-based mangrove models to our SRS and TS/Ph basin transects (http://www.ucs.usl.edu/~rrt4630/mangrove-restudy.htm; Twilley and Chen, 1998; Chen and Twilley, 1999a). These mechanistic models will be particularly critical to the integration of our process-based research. For example, we will use C and P models to link our DOM-microbial loop research with our secondary consumer dynamics, food web, and grazing studies.

In the second tier of our modelling effort, we will develop water budget models to estimate water residence times for the freshwater and estuarine zones of both study basins (we represent this tier of modelling as the water flow arrows in Fig.4). As in most coastal systems, water residence time couples internal ecological processes to variability in exogenous forcings. In addition to the hydrologic data we will collect, we have access to a number of relevant databases, including water level data collected throughout Everglades National Park by ENP hydrologists, water flow data from a number of key locations throughout both SRS and TS/Ph basins by USGS hydrologists (see http://www.sflorida.er.usgs.gov/realtime.html;), and canal and water control structure information from the South Florida Water Management District (SFWMD, see http://www.sfwmd.gov/org/omd/rt.html). We will take advantage of the existing mass balance hydrologic model for Florida Bay (FATHOM; J.Fourqurean, PI; http://www.fiu. edu/~seagrass/ fathom/), but we do not intend to extensively model estuarine hydrodynamics (as per Kjerfve et al., 1991; Kjerfve et al., 1992, for example). Rather, we will use these hydrologic datasets to parameterize box models of the components of our coastal Everglades system in order to approximate water residence times for the areas in which our sampling will be concentrated. From these residence times, we will calculate constituent fluxes, turnover rates for various components, nutrient spiralling parameters (Newbold et al., 1981, 1983; Elwood et al., 1983; Stream Solute Workshop, 1990), and constituent budgets dynamic in both space and time (Childers et al., 1993a).

The upper tier of our modelling and synthesis work will integrate both levels described above into a spatial modelling platform. Burke and Lauenroth (1993) evaluated the extent to which the Central Plains Experimental Range LTER actually represented shortgrass steppe ecosystems, and concluded that such site→region extrapolations are best accomplished by applying simulation models to regional spatial databases. Spatially explicit models are the most efficient means of accomplishing this scaling up from the site level (Sklar and Costanza, 1991; Sklar et al., 1994). Thus, spatial models will facilitate the integration of our GIS database with

our data synthesis efforts. Thus, There are two major efforts currently underway to develop spatial models of the entire Everglades landscape. The first is the Everglades Landscape Model (ELM; http://kabir.umd. edu/ Glades/ELM.html), being developed by scientists at the SFWMD (F.Sklar, SFWMD, pers.comm.). The second is the ATLSS (Across Trophic Level Simulation System), which is being developed by a group based at the USGS BRD (D.DeAngelis, PI; http://atlss.org/). We will work with both groups on our spatial modelling efforts for the SRS and TS/Ph basins.
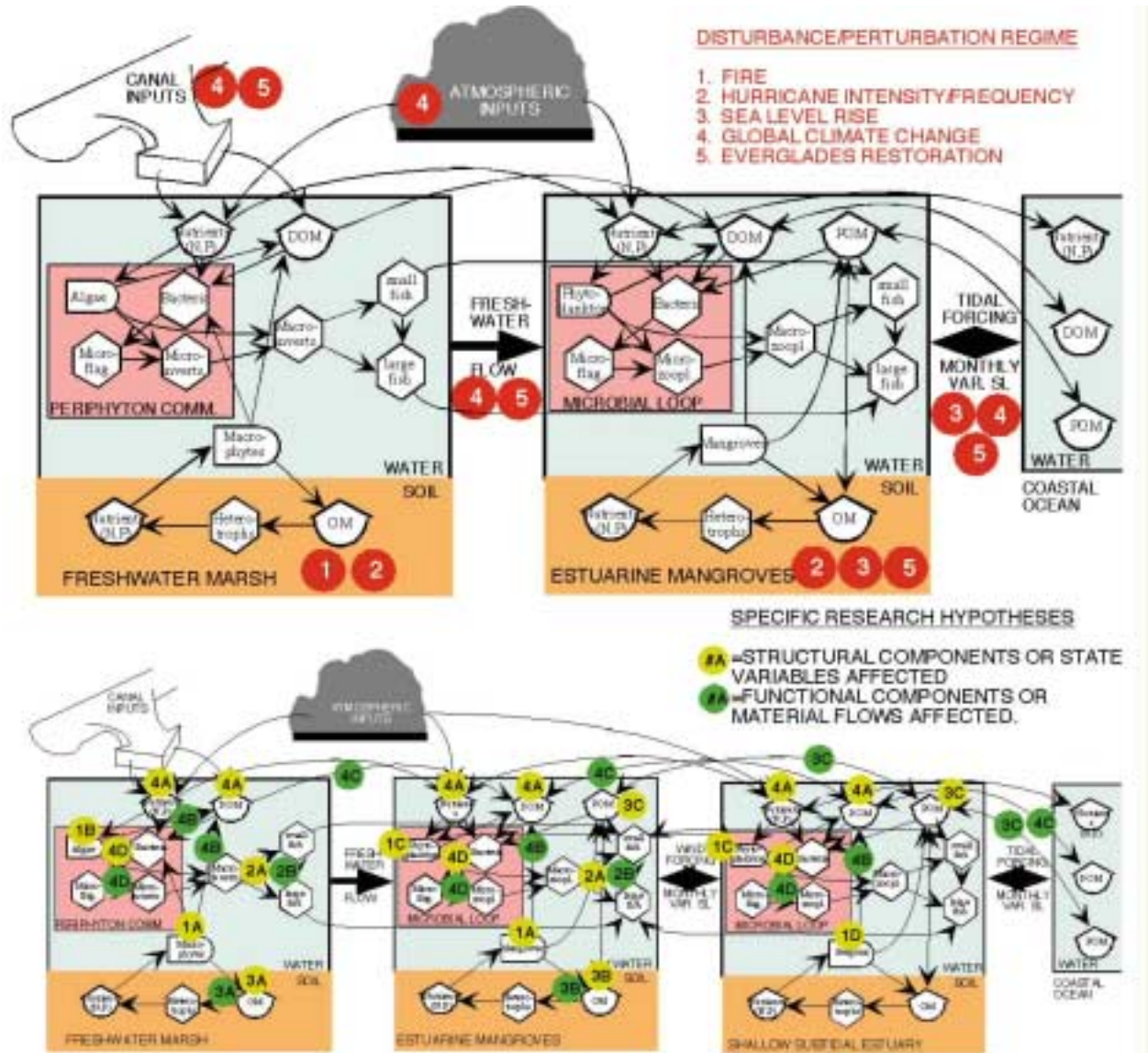

Figure 4: Conceptual diagram of the ecosystem components and exogenous forcings on which our coastal Everglades LTER research will focus. Arrows generally represent material flows. Note the conceptual similarities across ecosystem components (e.g. between periphyton in the freshwater marsh and the microbial loop in the estuarine compartments) Top panel=the SRS transect; Bottom panel=the TS/Ph transect. We show where the 5 primary factors of our disturbance/perturbance regime will have direct effects with the red circles (top panel) and we represent the focus of each Specific Research Hypothesis with yellow circles, for structural components, or green circles, for functional components (bottom panel).

BACKGROUND AND SITE DESCRIPTION

The Everglades, an International Biosphere Reserve, is one of the largest freshwater wetland landscapes in North America. The Everglades landscape is highly oligotrophic, with P limiting ecosystem productivity and biomass accumulation (Amador and Jones, 1993). Everglades waters typically contain undetectable amounts of dissolved P (Walker, 1991; others). Because P follows a sedimentary cycle in ecological systems, hydrologic inputs presumably dictate its supply and transport in Everglades wetlands. Furthermore, ecosystems have no biotically-mediated mechanism for removing P, in the way that denitrification removes N from wetlands and estuaries. This lack of internal control of P cycling suggests that the oligotrophic wetlands of the Everglades are particularly susceptible to changes in P loading (Caraco, 1993). *Because of these characteristics, the coastal Everglades is a unique venue for investigating how nutrient availability and cycling interacts with the hydrologic regime to potentially control ecosystem structure and function.*

The Everglades is part of a greater watershed that includes over 28,000 km^2 of central and southern Florida and extends from near Orlando to Florida Bay. The expansive freshwater marshes, known popularly as the "River of Grass", dominate the portion of this landscape south of Lake Okeechobee (Fig.1a). Before the turn of the century, water moved through the Everglades wetland landscape as a slow but pervasive surface flow from Lake Okeechobee through a mosaic of sawgrass marshes, wet sloughs, and forested islands. The topographic gradient from the lake to Florida Bay was about 1 m per 56 km (Light and Dineen, 1994). Water flow was seasonal, driven by wet season precipitation (June through November) and overflow from the lake. In a typical dry season, many of the marshes along the slightly higher fringes were not inundated.

In addition to the hydrologic manipulations of the last 100 years, the south Florida climate provides a uniquely variable template of climatic forcing. In an average year, about 130 cm of rain falls on our study area. Interannual variability in annual rainfall, however, shows a range of over 100 cm between wet years and dry years. There is a decadal periodicity to this wet-dry year cycle, and a regular pattern in which decades with extreme interannual variability alternate with far less variable decades (Duever et al., 1994). Thus, the climatic regime of the coastal Everglades is at least as variable from year to year as it is within a given year--verifying the value of long term datasets with less temporally intensive research.

Humans have drastically altered the hydrologic regime of the Everglades. Over 2500 km of drainage canals and levees and hundreds of water control structures now dissect the region into numerous sub-basins. The 3059 km^2 Everglades Agricultural Area (EAA) has been completely drained. To the south, Everglades National Park (ENP) includes over 4300 km^2 of the watershed; surface water flow in the park is regulated primarily by four water control structures along its northern border. Between the EAA and ENP are five large Water Conservation Areas (WCA) that impound over 3500 km^2 of Everglades marsh (Light and Dineen, 1994; Fig.1a). The primary objective of water management in these areas--to minimize flood risk during the wet season and maximize dry season water storage--is at odds with natural hydrologic patterns and has increased the frequency and intensity of disturbance events, such as floods/droughts and fire (DeAngelis and White, 1994; Gunderson and Snyder, 1994). Additionally, canals, levees, and unnaturally deep water have greatly reduced the wetland connectedness in the WCAs. Everglades National Park is the only remaining sector of the original Everglades where the hydrologic regime is somewhat similar to pre-management conditions. For this reason, *we propose to conduct our research in Everglades National Park.*

Patterns of flow, inundation, and P delivery in Everglades marshes have all been altered. The EAA is a major source of P to the oligotrophic marshes, often by point-source canal inputs (Coale et al., 1994a,b; Davis, 1994; Doren et al., 1997). Hydroperiod gradients have been greatly reduced. Because the WCAs are hydrologically isolated from each other, only vestiges of the original sheetflow remain. Where the hydrologic regime remains somewhat natural (i.e. ENP, Loxahatchee National Wildlife Refuge), the source of the water is not upstream wetlands but canals, and rainfall and canal inputs are often surprisingly uncoupled. The balance between exogenous P inputs via canal inflows versus atmospheric sources (which track the climatic variability) is critical to nutrient cycling and ecosystem dynamics. We combined estimated areas of the SRS and TS/Ph basins with annual rainfall and P loading from canals to demonstrate the relative importance of these two water sources (Table 2). Notably, data on P loading via canal structures exist (see Rudnick et al., 1999), while data on atmospheric P inputs (via rainfall and dry deposition) are often suspect. In dry years (e.g. 1990), roughly two-thirds of the exogenous P inputs are via rainfall, while in wet years (e.g. 1995) canals supply about half of all P inputs. Table 2 suggests that P inputs from canals are considerably more variable than rainfall inputs, in spite of the large interannual variability. Boyer and Jones (1999) found a similar pattern in their review of freshwater and P inputs to the Florida Bay estuary. P concentrations in water reaching Florida Bay are so low that the ratio of ambient total P concentrations to P inputs is nearly 1:1-- far lower than in other estuaries (Boyer and Jones, 1999).

One major difference between our 2 study transects is that TS/Ph includes Florida Bay while there is no shallow estuarine basin along the SRS transect (Table 1, Fig.4). Florida Bay is separated from the Atlantic Ocean and the Straits of Florida by the Florida Keys, which form a nearly continuous barrier. Carbonate mudbanks compartmentalize Florida Bay into a network of

Table 2: Estimated phosphorus load budgets for rainfall and point-source canal inputs. Rainfall and canal input data are from Rudnick et al. (1999). Point source P loads from canals were distributed over the entire area of both systems for the sake of a comparative budget only; in fact, the system does not behave this way at all (i.e. see Doren et al., 1997).   Mean rainfall P concentration of 0.3 µM (10 ppb) was asumed (R.Jones, pers. comm.).

Basin	estimated area	Rainfall 1990	Rainfall 1995	Rain P input 1990	Rain P input-1995	canal P input-1990	canal P input-1995
SRS	840 km^2	105 cm	180 cm	8.8 m.tons	15 m.tons	4 m.tons	17 m.tons
TS/Ph	200 km^2	100 cm	175 cm	2 m.tons	3.5 m.tons	1 m.ton	3 m.tons

shallow basins, and circulation between them is restricted. Mixed semidiurnal tides along the western margin generate a mean tidal range of 61 cm, but tidal energy is quickly attenuated by the mud banks and there is essentially no lunar tide over most of central and northeastern Florida Bay (Turney and Perkins 1972; Holmquist et al. 1989b). Intra-annual variability in mean monthly water levels follow a typical coastal cycle (Kjerfve et al., 1978), with highest water levels from August to November, lowest water levels from February to May, and a typical range of 30 cm (Holmquist et al. 1989b). Freshwater inputs to Florida Bay are dominated by rainfall and flow through the TS/Ph basin. Groundwater inputs are generally not important (Corbett et al. 1999).

The coupling of primary productivity and nutrient availability is strong in the oligotrophic Florida Bay, where phytoplankton production is strongly P-limited (Fourqurean et al., 1993; Phlips and Badylak, 1996; Lavrentyev et al., 1998), however other resources (e.g. light, N, Si) may also partially control productivity in some areas of the bay (Lavrentyev et al. 1998). Dissolved inorganic P concentrations are very low (20 nM), while DIN (dominated by ammonium) concentrations are often relatively high (Fourqurean et al., 1993;

Boyer et al., 1997; Boyer et al., 1999). Seagrass communities dominate the bottom of Florida Bay. Of the 2000 km^2 of Florida Bay, 95% of the bottom is covered by seagrass. Seagrass standing crop increases from 30 g d.w. m^{-2} in northeastern Florida Bay to 125 g d.w. m^{-2} along the western margin (Zieman et al. 1989). This east-west gradient is driven by gradients in sediment accumulation (Zieman et al. 1989) and P availability, which limits seagrass growth in Florida Bay (Fourqurean et al., 1992). Generally, animal density and diversity also decreases from west to east (Holmquist et al., 1989; Sogard et al., 1989; Thayer and Chester, 1989), although Florida Bay supports a diverse avifauna and a number of threatened or endangered vertebrates.

PROPOSED RESEARCH

We will focus our research around five research questions that parallel the LTER core areas. We propose to address these questions from freshwater and estuarine/coastal perspectives. Conceptually, this is analogous to a Lagrangian approach in which a parcel of water is tracked as it flows through freshwater Everglades marshes, the mangrove estuaries, and Florida Bay to offshore. Along the way, we will quantify patterns and processes in both the water itself, and in the ecosystem through which it is flowing, using long-term sampling and short-term process-based studies at both population and ecosystem levels. We will develop hydrologic models to simulate water residence times along both landscape transects, and we will synthesize data from this LTER research and from other related projects with an hierarchical simulation modelling effort. A primary product of this synthesis effort will be an integrated landscape model of both the SRS and TS/Ph drainage basins.

Our experimental design will entail permanent sites located along the SRS and TS/Ph transects (Fig.1b). Most of these sites overlap with ongoing research efforts, but some will be new. There will be 3 freshwater marsh and 3 mangrove wetland sites along the SRS transect. In the TS/Ph basin, our freshwater sampling will follow 2 arms--one through Taylor Slough and one through the ENP Panhandle region. We will sample 3 freshwater marsh sites and 2 mangrove sites in Taylor Slough, and 2 marsh sites + 1 mangrove site in the much shorter ENP Panhandle transect. They will join in Florida Bay, where we will sample 3 sites. Although we are proposing only minimal process-based research in the coastal ocean, both transects conceptually meet at the same water quality monitoring station on the southwest Florida shelf.

Each of the following research questions has specific research hypotheses directed at both freshwater and estuarine locations of the coastal Everglades landscape. In most cases, we will address each hypothesis at each site (see Table 3 for summary). We note how each hypothesis relates to our long-term scenarios for the coastal Everglades (Fig.3) and our project conceptualization (Fig.4).

QUESTION 1: HOW ARE PATTERNS AND MAGNITUDES OF PRIMARY PRODUCTION CONTROLLED BY FRESHWATER FLOW AND THE CONCENTRATIONS AND CHARACTERISTICS OF NUTRIENTS AND ORGANIC MATTER IN THE SOURCE WATER?

Specific Research Hypothesis 1A: Primary productivity by freshwater marsh macrophytes will increase with increasing nutrient inputs, but not with increased freshwater flow or DOM inputs, while mangrove productivity will be sensitive to all three (Long-term Scenarios 1-4, Fig.3).

Specific Research Hypothesis 1B: Periphyton community composition will be altered and productivity will decline where any process increases nutrient concentrations (Long-term Scenario 3, Fig.3c).

Specific Research Hypothesis 1C: The spatial domain of phytoplankton primary production will be altered with increasing freshwater flow (Long-term Scenarios 1-4, Fig.3).

Specific Research Hypothesis 1D: Increased fresh water flow will result in a decrease in estuarine benthic primary productivity (Long-term Scenarios 1-4, Fig.3).

QUESTION 2: HOW DOES FRESHWATER FLOW OR THE CONTENT OF SOURCE WATER CONTROL CONSUMER AND TROPHIC DYNAMICS?

Specific Research Hypothesis 2A: Increasing freshwater flow will change the spatial pattern of standing stocks, and in particular the location of peak biomass along the estuarine zone. This change will be more apparent in the SRS basin than the TS/Ph basin, as per Long-term Scenarios 1-4 (Fig.3).

Specific Research Hypothesis 2B: Increasing freshwater flow will change the spatial pattern of consumer dynamics and food web structure This change will be more apparent in the SRS basin than the TS/Ph basin, as per Long-term Scenarios 1-4 (Fig.3).

QUESTION 3: HOW DO CHANGES IN FRESHWATER FLOW OR THE CONTENT OF SOURCE WATER CONTROL ORGANIC MATTER ACCUMULATION IN FRESHWATER AND MANGROVE WETLAND SOILS?

Specific Research Hypothesis 3A: Any process that introduces nutrients to freshwater marshes will have a negative effect on soil elevation change, as the nutrient stimulation of soil respiration exceeds organic matter inputs via enhanced macrophyte (peat marshes) or periphyton (marl marshes) production (Long-term Scenario 3; Fig.3c).

Specific Research Hypothesis 3B: Any process that introduces nutrients or enhances tidal exchange in the mangrove zone will have a positive effect on soil elevation change, by stimulation of mangrove productivity and enhanced inputs of allochthonously derived particulate matter (Long-term Scenario 2; Fig.3b).

Specific Research Hypothesis 3C: The sources, transport processes and environmental fate of estuarine SOM will reflect the balance between transport processes (freshwater flow vs. tidal energy), mixing phenomena (storms), and local processes (sources of primary production, soil diagenetic pathways; Long-term Scenarios 1-4; Fig.3).

QUESTION 4: HOW IS THE QUALITY AND/OR QUANTITY OF DOM OR THE QUANTITY OF INORGANIC NUTRIENTS IN SOURCE WATER ALTERED BY CHANGING FRESHWATER FLOW VERSUS INTERNAL PROCESSES OCCURRING AT A GIVEN LOCATION IN THE LANDSCAPE? HOW ARE LOCAL ECOSYSTEM PROCESSES CONTROLLED BY CHANGES IN SOURCE WATER DOM OR INORGANIC NUTRIENTS?

Specific Research Hypothesis 4A: Nutrient and DOM concentrations will be low throughout the coastal Everglades landscape except near freshwater inflows and in the oligohaline zone of the SRS system. All 4 Long-term Scenarios (Fig.3) reflect how these concentration patterns will respond to changes in regional controls (freshwater flow, nutrient and DOM inputs).

Specific Research Hypothesis 4B: Periphyton and sawgrass are the primary sources of DOM in freshwater Everglades marshes while mangroves and seagrass are the primary sources of DOM in the estuaries.

Specific Research Hypothesis 4C: The source of DOM is an important controller of the fate of DOM in both the freshwater and the estuarine systems. DOM from more oligotrophic areas will be less bioavailable than from more productive areas. DOM from sawgrass and mangroves will be less bioavailable than DOM from periphyton and seagrass due to increased lignin content.

Specific Research Hypothesis 4D: Microbial loop dynamics in periphyton mats are functionally different from those in the estuaries because of the spatial compression of the mat community. Secondly, increased community heterotrophy will result in increased microbial loop activity at the expense of traditional grazing pathway

---

> **QUESTION 5: HOW DO LONG-TERM CHANGES IN FRESHWATER FLOW (PRIMARILY MANIFEST THROUGH EVERGLADES RESTORATION) INTERACT WITH LONG-TERM CHANGES IN THE CLIMATIC AND DISTURBANCE REGIMES TO MODIFY ECOLOGICAL PATTERN AND PROCESS IN COASTAL LANDSCAPES?**

Specific Research Hypothesis 5A: The long-term effects of sea level rise and Everglades restoration will be secondary to any long-term change in the climatic regime, and its characteristic cycles of variation (Long-term Scenarios 1-4; Fig.3).

Specific Research Hypothesis 5B: Everglades restoration (=increased freshwater flow) will expand the oligohaline productivity zone while sea level rise will tend to push the entire estuarine zone landward. This landward migration will be driven by storm surges, and thus will be a punctuated rather than continuous process (Long-term Scenarios 1-4; Fig.3).

Specific Research Hypothesis 5C: Fires and hurricanes will have local effects at scales that depend upon disturbance size, but will have little impact on long-term changes affected by sea level rise, climate change, and Everglades restoration (Long-term Scenarios 1-4; Fig.3).

LITERATURE CITED

Amador, J.A., and R.D. Jones. 1993. Nutrient limitations on microbial respiration in peat soils with different total phosphorus content. Soil Biol. Biochem. 25:793-801.

Amon R.M.W., and Benner R., 1996, Bacterial utilization of different size classes of dissolved organic matter, Limnol. Oceanogr. 41, 41-51.

Azam, F., T. Fenchel, J.G. Field, A. Meyer-reil l, and F. Thingstad f., 1983. The ecological role of water-column microbes in the sea. Mar. Ecol. Prog. Ser. 10:257-263.

Bachoon, D. and R.D. Jones, 1992. Potential rates of methanogenesis in sawgrass marshes with peat and marl soils in the Everglades. Soil Biol. Biochemistry 24(1): 21-27.

Battin T.J., 1998, Dissolved organic matter and its optical properties in a blackwater tributary of the upper Orinoco river, Venezuela, Org. Geochem., 28, 561-569.

Belanger, T.V., D.J. Scheidt, and J.R. Platko, 1989. Effects of nutrient enrichment on the Florida Everglades. Jour. Lake Reservoir Man. 5(1):101-112.

Benner R., Peele E.R. and Hodson R.E., 1986, Microbial utilization of dissolved organic matter from leaves of the Red Mangrove, Rhizophora mangle, in the Fresh Creek Estuary, Bahamas, Est. Coast. Shelf Sci., 23, 607-619.

Bianchi T.S., Argyrou M. and Chippett H.F., 1999, Contribution of vascular- plant carbon to surface sediments across the coastal margin of Cyprus (eastern Mediterranenan), Org. Geochem., 30, 287-298.

Biber M.V., Gülaçar F.O. and Buffle J., 1996, Seasonal variations in principal groups of organic matter in a eutrophic lake using pyrolysis/GC/MS, Environ. Sci. Technol., 30, 3501-3507.

Boumans, R. M. J. and J.W. Day, Jr., 1993. High precision measurements of sediment elevation in shallow coastal areas using a sedimentation-erosion table. Estuaries 16(2): 375-380.

Boyer J.N., Christian R.R., and Stanley D.W., 1993, Patterns of phytoplankton primary productivity in the Neuse river estuary, North Carolina, Mar. Ecol. Prog. Ser. 97, 287-297.

Boyer J.N., and Groffman P.M., 1996, Bioavailability of water extractable organic carbon fractions in forest and agricultural soil profiles. Soil Biol. Biochem. 28, 783-790.

Boyer, J.N., J.W. Fourqurean, and R.D. Jones. 1997. Spatial characterization of water quality in Florida Bay and Whitewater Bay by multivariate analyses: zones of similar influence. Estuaries 20: 743-758.

Boyer, J.N. and R.D. Jones, 1999. Effects of freshwater inputs and loading of P and N on the water quality of eastern Florida Bay. In: [K.R. Redd, G.A. O'Connor, and C.L.Schelske, eds] Phosphorus Biogeochemistry in Subtropical Ecosystems. Lewis Publ., pp. 545-561.

Boyer, J.N., J.W. Fourqurean, and R.D. Jones. 1999. Seasonal and long-term trends in water quality of Florida Bay (1989-1997). Estuaries 22 (in press).

Boynton W.R., Kemp W.M., and Keefe C.W, 1982, A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. In: Kennedy, Ed. Estuarine Comparisons. Academic Press, pp. 69-90.

Browder, J.A., P.J. Gleason, and D.R. Swift. 1994. Periphyton in the Everglades: Spatial variation, environmental correlates, and ecological implications. Pages 379-418 in S.M. Davis and J.C. Ogden, editors. Everglades: The Ecosystem and its Restoration. St. Lucie Press, Delray Beach, Florida.

Bruchet A., Rousseau C. and Mallevialle J., 1990, Pyrolysis-GC/MS for investigating high-molecular-weight THM precursors and other refractory organics, J.Am. Water Works Assoc., Sept., 66-74.

Burke, I. C. , Lauenroth, W.K., 1993. What do LTER results mean - extrapolating from site to region and decade to century. Ecological Modelling 67(1): 19-35.

Buzzelli, C.P., D.L. Childers, Q. Dong, and R.D. Jones, in review. Simulation of periphyton dynamics and P related water quality in Everglades National Park. Ecol. Mod.

Cabana, G., and J. B. Rasmussen. 1994. Modelling food chain structure and contaminant bioaccumulation using stable isotopes. Nature 372:255-257.

Cahoon, D. R. and R.E.Turner, 1989. Accretion and canal impacts in a rapidly subsiding wetland. II. Feldspar marker horizon technique. Estuaries 12(4): 260-268.

Canuel E.A., Freeman K.H. and Wakeham S.G., 1997, Isotopic composition of lipid biomarker compounds in estuarine plants and surface sediments, Limnol. Oceanogr., 42, 1570-1583.

Caraco, N. F. , J.J. Cole, and G.E. Likens, 1992. New and recycled primary production in an oligotrophic lake: Insights for summer phosphorus dynamics. Limnology and Oceanography 37(3): 590-602.

Caraco, N. F., 1993. Disturbance of the Phosphorus Cycle - A Case of Indirect Effects of Human Activity. Trends in Ecology & Evolution 8(2): 51-54.

Caron, D.A., 1994. Inorganic nutrients, bacteria, and the microbial loop. Microb. Ecol. 28:295-298.

Carpenter, S. R. , S.W. Chisholm, C.J. Krebs, D.W. Schindler, and R.F. Wright, 1995. Ecosystem experiments. Science 269: 324-327.

Chanton, J.P., G.J.Whiting, J.D.Happell, and G.Gerard, 1993. Contrasting rates and diurnal patterns of methane emission from emergent aquatic macrophytes. Aquatic Bot. 46:111-128.

Chen, R. and R.R. Twilley, 1999a. A simulation model of organic matter and nutrient accumulation in mangrove wetland soils. Biogeochem. 44:93-119.

Chen, R. and R.R. Twilley, 1999b. Patterns of mangrove forest structure and soil nutrient dynamics along the Shark River estuary, Florida. Estuaries (in press).

Chick, J. H. and C.C. McIvor. 1994. Patterns in the abundance and composition of fishes among different macrophytes: viewing a littoral zone as a landscape. Canadian Journal Fisheries Aquatic Sciences 51:2873-2882.

Chick, J. H., S. Coyne, and J. C. Trexler. 1999. Effectiveness of airboat electrofishing for sampling fishes in shallow vegetated habitats. North American Journal of Fisheries Management, in press

Childers, D.L., H.N.McKellar, Jr., R. Dame, F.Sklar, and E.Blood, 1993a. A dynamic nutrient budget of subsystem interactions in a salt marsh estuary. Est. Coast. Shelf Sci. 36:105-131.

Childers, D.L., F.H.Sklar, B.Drake, and T.Jordan, 1993b. Seasonal measurements of sediment elevation in three Mid-Atlantic estuaries. Jour. Coastal Res. 9(4):986-1003.

Childers, D.L., R.D. Jones, J. Trexler, C. Buzzelli, S. Dailey, A.L. Edwards, E. Gaiser, K. Jayachandaran, A. Kenne, D. Lee, J. Meeder, M.Nair, J. Pechman, A. Renshaw, J. Richards, M. Rugge, L. Scinto, P. Sterling, and W. Van Gelder, 1999a. Quantifying the effects of low-level phosphorus enrichment on unimpacted Everglades wetlands with *in situ* flumes and phosphorus dosing. In: [K.Porter and J.Porter, eds.], The Everglades Hydroscape, St. Lucie Press. In press.

Childers, D.L., S.Davis, V.Rivera-Monroy, and R.R.Twilley, 1999b. Wetland-water column interactions and the biogeochemistry of estuary-watershed coupling around the Gulf of Mexico. In [T.S. Bianchi and R.Twilley, eds.]: Biogeochemistry of Gulf of Mexico Estuaries. John Wiley & Sons, New York, NY. pp. 211-235.

Childers, D.L., J.W. Day, Jr., and H.N.McKellar, Jr, 1999c. Twenty more years of marsh and estuarine flux studies: Revisiting Nixon (1980). In: [M.P. Weinstein and D.Q. Kreeger, eds.] Concepts and Controversies in Tidal Marsh Ecology, in press.

Coale, F. J. , F.T. Inuzo, and A.B. Bottcher, 1994a. Phosphorus in drainage water from sugarcane in the Everglades Agricultural Area as affected by drainage rate. J. Environ. Qual. 23(1): 121-126.

Coale, F. J. , F.T. Inuzo, and A.B. Bottcher, 1994b. Sugarcane production impact on N and P in drainage water from an Everglades histosol. J. Environ. Qual. 23(1): 116-120.

Corbett, D.R., J. Chanton, W. Burnett, K. Dillon, C. Rutkowski, and J.W. Fourqurean. 1999. Patterns of groundwater discharge into Florida Bay. Limnology and Oceanography 44**:** 1045-1055.

Craft, C.B. and C.J.Richardson, 1993. Peat accretion and P accumulation along a eutrophication gradient in the northern Everglades. Biogeochem. 22:133-156.

Cranwell P.A., 1982, Lipids of aquatic sediments and sedimenting particulates, Prog. Lipid Res., 21, 271-308.

Daoust, R. and D.L.Childers, 1998. Quantifying aboveground biomass and estimating productivity in nine Everglades wetland macrophytes using a non-destructive allometric approach. Aquatic Botany. 62:115-133.

Daoust, R. and D.L.Childers, 1999. Controls on emergent macrophyte composition, abundance, and productivity in freshwater Everglades wetland communities. Wetlands 19:262-275.

Daoust, R. and D.L. Childers, in review. The ecosystem-level effects of low-level phosphorus enrichment on freshwater wetlands of the Florida Everglades, USA. Oecologia.

Davis, S.E. III, D.L. Childers, J.W. Day, Jr., D.T. Rudnick, and F.H. Sklar, in prep. Factors affecting the concentration and flux of C, N, an dP in a non-tidal dwarf mangrove forest. Est. Coast. Shelf Sci..

Davis, S.M., 1991. Growth, decomposition, and nutrient retention of *Cladium jamaicense* and *Typha domingensis* in the Florida Everglades. Aquat. Bot. 40:203-224.

Davis, S.M., 1994. P inputs and vegetation sensitivity in the Everglades. In [S.M. Davis and J.C.Ogden, eds]: Everglades: The ecosystem and its restoration. St. Lucie Press, FL, p. 357-378.

Day, J. W. , Jr., C. Coronado-Molina, F.R. Vera-Herrera, R. Twilley, V.H. Rivera-Monroy, H. Alvarez-Guillen, R. Day, and W. Conner, 1996. A 7 year record of above-ground net primary production in a southeastern Mexican mangrove forest. Aquatic Botany 55: 39-60.

Day, J. W. , C.A.S. Hall, W.M.Kemp, and A.Yanez-Arancibia, 1989. Estuarine Ecology John Wiley & Sons.

DeAngelis, D.L. and P.S.White, 1994. Ecosystems as products of spatially and temporally varying driving forces, ecological processes, and landscapes: A theoretical perspective. In [S.M. Davis and J.C.Ogden, eds]: Everglades: The ecosystem and its restoration. St. Lucie Press, FL, p. 9-28.

De Leeuw J.W., Rijpstra I.C. and Nienhuis P.H., 1995, Free and bound fatty acids and hydroxy fatty acids in the living and decomposing eelgrass Zoztera marina L., Org. Geochem., 23, 721-728.

Doren, R.F., T.V. Armentano, L.D. Whiteaker, and R.D. Jones, 1997. Marsh vegetation patterns and soil phosphorus gradients in the Everglades ecosystem. Aquatic Botany 56:145-163.

Duarte C.M., Gasol J.M., and Vaque D., 1997. Role of experimental approaches in marine microbial ecology. Aquat. Microb. Ecol. 13, 101-111.

Duever, M.J., J.F. Meeder, L.C. Meeder, and J.M. McCollom, 1994. The climate of south Florida and its role in shaping the Everglades ecosystem. In [S.M. Davis and J.C.Ogden, eds]: Everglades: The ecosystem and its restoration. St. Lucie Press, FL, p. 225-248.

Elwood, J.W., J.D.Newbold, R.V.O'Neill, and W.Van Winkle, 1983. Resource spiralling: An operational paradigm for analyzing lotic ecosystems. In: [T.D.Fontaine III and S.M.Bartell, eds.] Dynamics of Lotic Ecosystems, Ann Arbor Science, Ann Arbor, MI, pp. 3-27.

Fourqurean, J.W., J.C. Zieman, and G.V.N. Powell. 1992a. Phosphorus limitation of primary production in Florida Bay: Evidence from the C:N:P ratios of the dominant seagrass *Thalassia testudinum.* Limnology and Oceanography 37**:** 162-171.

Fourqurean, J.W., J.C. Zieman, and G.V.N. Powell. 1992b. Relationships between porewater nutrients and seagrasses in a subtropical carbonate environment. Marine Biology 114**:** 57-65.

Fourqurean, J.W., R.D. Jones, and J.C. Zieman. 1993. Processes influencing water column nutrient characteristics and phosphorus limitation of phytoplankton biomass in Florida Bay, FL, USA: inferences from spatial distributions. Estuarine, Coastal and Shelf Science 36**:** 295-314.

Fry B., Hopkinson C.S Jr., Norrman B. and Zweifel U.L., 1996, Long-term decomposition of DOC from experimental diatom blooms, Limnol. Oceanogr., 41, 1344-1347.

Fuhrman, J.A. and F. Azam, 1980. Bacterioplankton secondary production estimates for coastal waters of British Columbia, Antarctica, and California. Appl. Environ. Microbiol. 39:1085-1095.

Gaffney J.S., Marley N.A. and Clark S.B., 1996, Humic and fulvic acids and organic colloidal materials in the environment, in: Humic and Fulvic Acids: Isolation, Structure and Environmental Role, ACS Symposiun Series 651, Gaffney et al. Eds., American Chemical Society, pp. 2-16.

Gjessing E.T., Alberts J.J., Bruchet A., Egeberg P.K., Lydersen E., McGowon L.B., Mobed J.J., Münster U., Pempkowiak J., Perdue M., Ratnawerra H., Rybacki D., Takacas M. and Abbt-Braun G., 1998, Multi-method characterization of natural organic matter isolated from water: Characterization of reverse osmosis-isolates from water of two semi-identical dystrophic lakesbasins in Norway, Wat. Res., 32, 3108-3124.

Goldsborough, L.G., and G.G.C. Robinson. 1996. Pattern in wetlands. Pages 78-117 in R.J. Stevenson, M.L. Bothwell, R.L. Lowe, editors. Algal Ecology, Freshwater Benthic Ecosystems. Academic Press, New York.

Gunderson, L.H. and J.R.Snyder, 1994. Fire patterns in the southern Everglades. In [S.M. Davis and J.C.Ogden, eds]: Everglades: The ecosystem and its restoration. St. Lucie Press, FL, p. 291-305.

Hedges J.I. and Ertel J.R., 1982, Characterization of lignin by gas capillary chromatography of cupric oxide oxidation products, Anal. Chem., 54, 174-178.

Holmquist, J.G., G.V.N. Powell, and S.M. Sogard. 1989a. Decapod and stomatopod communities of seagrass-covered mud banks in Florida Bay: inter- and intra-bank heterogeneity with special reference to isolated subenvironments. Bulletin of Marine Science 44**:** 251-262.

Holmquist, J.G., G.V.N. Powell, and S.M. Sogard. 1989b. Sediment, water level and water temperature characteristics of Florida Bay's grass-covered mud banks. Bulletin of Marine Science 44**:** 348-364.

Hopkinson, C.S. and J.J. Vallino, 1995. The relationships among man's activities in watersheds and estuaries: A model of runoff effects on patterns of estuarine community metabolism. Estuaries 18(4):598-621.

Houde, E. D. and E.S.Rutherford, 1993. Recent trends in estuarine fisheries: Predictions of fish production and yield. Estuaries 16(2): 161-177.

Hubert, W. A. 1983. Passive capture techniques, pp. 95-122. In: L. A. Nielsen, D. L. Johnson, and S. S. Lumpton. Fisheries Techniques. American Fisheries Society, Bethsda, MD.

Hudson, J. H. , G.V.N. Powell, M.B. Robblee, and T.J. Smith, 1989. A 107-year-old coral from Florida Bay: Barometer of natural and man-induced catastrophes? Bull. Mar. Sci. 44(1): 283-291.

Jaffé R., Chen L., Huang C.W., Hajje N., Furton K.G. and Sicre M.A, 1995, An investigation on hydrocarbon speciation in ancient sediments by stepwise supercritical carbon dioxide extraction at various temperatures:

Preliminary results, *Organic Geochemistry: Developments & Applications in Energy, Climate, Environment and Human History,* 1995, pp. 796.

Jaffé R., Diaz D., Hajje N., Chen L., Eckardt C. and Furton K.G., 1997, Hydrocarbon speciation in ancient sediments studied by stepwise supercritical carbon dioxide extraction,*Org. Geochem.,* **26**, 59-65.

Jordan, C. F., S. Coyne, and J. C. Trexler. 1997. Sampling fishes in heavily vegetated habitats: the effects of habitat structure on sampling characteristics of the 1-m^2 throw trap. Transactions of the American Fisheries Society 126:1012-1020

Kjerfve, B. L. B.Miranda, and E.Wolanski, 1991. Modelling water circulation in an estuary and intertidal salt marsh system. Neth. Jour. Sea Res. 28(3): 141-147.

Kjerfve, B. H. E. Seim, A.F. Blumberg, and L.D. Wright, 1992. Modelling of the residual circulation in Broken Bay and the Lower Hawkesbury River, NSW. Australian Journal of Marine and Freshwater Research 43: 1339-1357.

Koch, M.S. and K.R.Reddy, 1992. Distribution of soil and plant nutrients along a trophic gradient in the Florida Everglades. Soil Sci. Soc. Amer. J. 56:1492-1499.

Koch, M. S., 1997. Rhizophora mangle L. seedling development into the sapling stage across resource and stress gradients in subtropical Florida. Biotropica 29(4): 427-439.

Kolbowski J. and Schreiber U., 1995, In: Mathis, Ed. Photosynthesis: from light to biosphere, Kluwer Academic, pp. 825-828.

Kushlan, J.A. 1981. Sampling characteristics of enclosure fish traps. Transactions of the American Fisheries Society 110:557-562.

Landry M.R., Kirshtein J., and Constantinou J., 1995. A refined dilution technique for measuring the community grazing impact of microzooplankton, with experimental tests in the central equatorial Pacific. Mar. Ecol. Prog. Ser. 120, 53-63.

Lavrentyev, P.J., H.A. Bootsma, T.H. Johengen, J.F. Cavaletto, and W.S. Gardner. 1998. Microbial plankton response to resource limitation: insights from the community structure and seston stoichiometry in Florida Bay, USA. Marine Ecology Progress Series 165**:** 45-57.

Light, S.S. and J.W. Dineen, 1994. Water control in the Everglades: A historical perspective. In [S.M. Davis and J.C.Ogden, eds]: Everglades: The ecosystem and its restoration. St. Lucie Press, FL.

Loftus, W.F., and A.M. Eklund. 1994. in S.M. Davis and J.C. Ogden, editors. Everglades: The Ecosystem and its Restoration. St. Lucie Press, Delray Beach, Florida.

Loftus, W.F., and J.A. Kushlan. 1987. Freshwater fishes of southern Florida. Bulletin of the Florida State Museum, Biological Sciences 31:147-344.

Lorenz, J.J., G.V.N. Powell, C.C. McIvor, and P.C. Frederick. 1997. A drop net and removable walkway for sampling fishes over wetland surfaces. Wetlands 17:346-359.

Malone T.C., Neale P.J., and Boardman D., 1980, Influences of estuarine circulation on the distribution and biomass of phytoplankton size fractions, In: Kennedy, Ed. Estuarine Perspectives, Academic Press, pp. 249-262.

Maul, G.A. and D.M. Martin, 1993. Sea leve rise at Key West, FL, 1846-1992: America's longest instrument record? Geophys. Res. Letters 20(18):1955-1958.

McCormick, P.V., and M.B. O'Dell. 1996. Quantifying periphyton responses to phosphorus in the Florida Everglades: A synoptic experimental approach. Journal of the North American Benthological Society 15:450-468.

McCormick, P.V., P.S. Rawlik, K. Lurding, E.P. Smith, and F.H. Sklar, 1996. Periphyton-water quality relationships along a nutrient gradient in the northern Florida Everglades. J. N. Am. Benthol. Soc. 15(4):433-449.

McKnight D.M., Andrews E.D., Spaulding S.A. and Aiken G.R., 1994, Aquatic fulvic acids in algal-rich antarctic ponds, Limnol. Oceanogr., 39, 1972-1979.

Meyers P. and Ishiwatari R., 1993, Lacustrine organic geochemistry – and overview of organic matter sources and diagenesis in lake sediments, Org. Geochem., 20, 867-900.

Newbold, J.D., J.W.Elwood, R.V.O'Neill, and W.Van Winkle, 1981. Measuring nutrient spiralling in streams. Can.J.Fish.Aquat.Sci. 38:860-863.

Newbold, J.D., J.W.Elwood, R.V.O'Neill, and A.L.Sheldon, 1983. Phosphorus dynamics in a woodland stream ecosystem: A study of nutrient spiralling. Ecology 64(5):1249-1265.

Newman, S. J. B. Grace, and J. W. Koebel, 1996. Effects of Nutrients and Hydroperiod on Typha, Cladium, and Eleocharis: Implications for Everglades Restoration. Ecological Applications 6(3): 774-783.

Nienhuis, P. H., 1993. Nutrient cycling and foodwebs in Dutch estuaries. Hydrobiologia 265(1-3): 15-44.

Norman B., Zweifel U.L., Hopkinson C.S. Jr. and Fry B., 1995, Production and utilization of dissolved organic carbon during an experimental diatom bloom, Limnol. Ocean., 40, 898-907.

Odum, W.E., E.P. Odum, and H.T. Odum, 1995. Nature's pulsing paradigm. Estuaries 18(4):547-555.

Oehm, N., D.L. Childers, and R.D. Jones, in prep. Soil carbon remineralization in the oligotrophic mangrove soils of the southeast Everglades, USA. Mangroves & Salt Marshes.

Parrish C.C., 1988, Dissolved and particulate marine lipid classes: A review, Marine Chem., 23, 17-40.

Phlips, E.J., and S. Badylak. 1996. Spatial variability in phytoplankton standing crop and composition in a shallow inner-shelf lagoon, Florida Bay, Florida. Bulletin of Marine Science 58: 203-216.

Prahl F.G., Ertel J.R., Goni M.A., Sparrow M.A. and Eversmeyer B., 1994, Terrestrial organic matter contributions to sediments on the Washington margin, Geochim. Cosmochim. Acta., 58, 3035-3048.

Prartono T. and Wolff G.A., 1998, Organic geochemistry of lacustrine sediments: A record of the changing trophic status of Rostherne Mere, UK, Org. Geochem., 28, 729-747.

Rader, R.B., and C.J. Richardson. 1992. The effects of nutrient enrichment on algae and macroinvertebrates in the Everglades: a review. Wetlands 12:121-135.

Reddy, K.R., W.F.DeBusk, Y.Wang, R.DeLaune, and M.Koch, 1991. Physico-chemical properties of soils in Water Conservation Area 2 of the Everglades. Final report to the SFWMD, West Palm Beach, FL. 118. pp.

Reed, D. J., 1989. Patterns of sediment deposition in subsiding coastal saltmarshes, Terrebonne Bay, LA: The role of winter storms. Estuaries 12(4): 222-227.

Rivera-Monroy, V. H. , J.W. Day, R.R. Twilley, F Vera-Herrera, and C. Coronado-Molina, 1995. Flux of nitrogen and sediment in a fringe mangrove forest in Terminos Lagoon, Mexico. Est. Coast. Shelf Sci. 40: 139-160.

Ronnback, P., M. Troell, N. Kautsky, and J.H. Primavera. 1999. Distributional patterns of shrimps and fish among *Avicennia* and *Rhizophora* microhabitats in the Pagbilao mangroves, Philippines. Estuarine, Coastal and Shelf Science 48: 223-234.

del Rio J.C. and Hatcher P.G., 1996, Structural characterization of humic substances using thermochemolysis with tetramethylammonium hydroxide, in: Humic and Fulvic Acids: Isolation, Structure and Environmental Role, ACS Symposiun Series 651, Gaffney et al. Eds., American Chemical Society, pp. 78-95.

Rozas, L.P. 1992. Bottomless lift net for quantitatively sampling nekton on intertidal marshes. Marine Ecology-Progress Series 89:287-292.

Rudnick, D.T., Z. Chen, D.L. Childers, J.N. Boyer, and T.D.I. Fontaine. 1999. Phosphorus and nitrogen inputs to Florida Bay: the importance of the Everglades watershed. Estuaries 22 (in press).

Saiz-Jimenez C and De Leeuw J.W., 1986, Lignin pyrolysis products: Their structures and their significance as biomarkers, Org. Geochem., 10, 869-876.

Scheidt, D.J., M.D.Flora, and D.R.Walker, 1989. Water quality management for Everglades National Park. Wetlands: Concerns and Successes, AWRA 132:377-390.

Servais, P., A. Anzil, and C. Ventresque, 1989. Simple method for determination of biodegradable dissolved organic carbon in water. Appl. Environ. Microbiol. 55:2732-2734.

Sharp J.H., Pennock J.R., Church T.M., Tramontano J.M., and Cifuentes L.A., 1984, The estuarine interaction of nutrients, organics, and metals: A case study in the Delaware Estuary, In: The Estuary as a Filter. In: Kennedy, Ed. Academic press, pp. 241-258.

Sklar, F. H. and R.Costanza, 1991. The development of dynamic spatial models for landscape ecology: A review and prognosis. In [M.G.Turner and R.H.Gardner, eds] Quantitative Methods in Landscape Ecology Ecol. Series 82, p.239-288

Sklar, F. H. , K.K. Gopu, T. Maxwell, and R. Costanza, 1994. Spatially explicit and implicit dynamic simulations of wetland processes. Global Wetlands: Old World and New : 537-554.

Smith, T. J. , J.H. Hudson, M.B. Robblee, G.V.N. Powell, and P.J. Isdale, 1989. Freshwater flow from the Everglades to Florida Bay: A historical reconstruction based on fluorescent banding in the coral Solenastrea bournoni. Bull. Mar. Sci. 44(1): 274-282.

Smith, III T. J., K.G. Boto, S.D. Frusher, and R.L. Giddins, 1991. Keystone species and mangrove forest dynamics: the influence of burrowing by crabs on soil nutrient status and forest productivity. Estuarine, Coastal, and Shelf Science 33: 419-432.

Snedaker, S.E., 1989. Overview of ecology of mangroves and information needs for Florida Bay. Bull. Mar. Sci. 44(1):341-347.

Sogard, S.M., G.V.N. Powell, and J.G. Holmquist. 1989. Spatial distribution and trends in abundance of fishes residing in seagrass meadows on Florida Bay mudbanks. Bulletin of Marine Science 44: 179-199.

South Florida Water Management District, 1992. Draft Surface Water Improvement and Management Plan for the Everglades, Supporting Information Doc., SFWMD, West Palm Beach, FL. 472 pp.

Stanley, D. W., 1993. Long-term trends in Pamlico River estuary nutrients, chlorophyll, dissolved oxygen, and watershed nutrient production. Water Resources Research 29(8): 2651-2662.

Stream Solute Workshop, 1990. Concepts and methods for assessing solute dynamics in stream ecosystems. J.N.Am.Benthol. Soc. 9(2):95-119.

Tezuka, Y., 1990. Bacterial regeneration of ammonium and phosphate as affected by the carbon:nitrogen:phosphorus ratio of organic substrates. Microb. Ecol. 19:227-238.

Thayer, G.W., and A.J. Chester. 1989. Distribution and abundance of fishes among basin and channel habitats in Florida Bay. Bulletin of Marine Science 44: 200-219.

Tranvik L.J. 1997, Rapid fluorometric assay of bacterial density in lake water and seawater Limnol. Oceanogr. 42, 1629-1634.

Trexler, J. C., W. F. Loftus, C. F. Jordan, J. Chick, K. L. Kandl, and O. L. Bass. Ecological scale and its implications for freshwater fishes in the Florida Everglades. In J. W. Porter and K. G. Porter (eds.) Linkages between Ecosystems in the South Florida Hydroscape. CRC. In review.

Turner, A., and J. C. Trexler. 1997. Sampling invertebrates from the Florida Everglades: a comparison of alternative methods. Journal of the North American Benthological Society 16:694-709

Turner, A. M., J. C. Trexler, F. Jordan, S. J. Slack, P. Geddes, and W. Loftus. 1999. Targeting ecosystem features for conservation: Standing crops in the Florida Everglades. Conservation Biology, in press

Turney, W.J., and B.F. Perkins. 1972. Molluscan distribution in Florida Bay. Sedimenta III. RSMAS, University of Miami. 37 pp. .

Twilley, R. R., 1985. The exchange of organic carbon in basin mangrove forests in a SW Florida estuary. Est. Coast. Shelf Sci.` 20: 543.

Twilley, R. R. , A.E. Lugo, and C. Patterson-Zucca, 1986. Litter production and turnover in basin mangrove forests in southwest Florida. Ecology 67(3): 670-683.

Twilley, R. R. , R.H. Chen, and T. Hargis, 1992. Carbon sinks in mangroves and their implications to carbon budget of tropical coastal ecosystems. Water, Air, and Soil Pollution 64: 265-288.

Twilley, R.R. and R. Chen, 1998. A water budget and hydrology model of a basin mangrove forest in Rookery Bay, FL. Mar. Freshwater Res. 49:309-23.

Vallino J.J., Hopkinson C.S. Jr. and Hobbie J.E., 1996, Modeling bacterial utilization of dissolved organic matter: Optimization replaces Monod growth kinetics, Limnol. Ocean., 41, 1591-1609.

Vance, D.J., M.D.E. Haywood, D.S. Heales, R.A. Kenyon, N.R. Loneragan, and R.C. Pendrey. 1996. How far do prawns and fish move into mangroves? Distribution of juvenile banana prawns *Penaeus merguiensis* and fish in a tropical mangrove forest in northern Australia. Marine Ecology - Progress Series 131:115-124.

Volkman J.K., Barrett S.M. and Blackburn S.I., 1999, Eustigmatophyte microalgae are potential sources of C_{29} sterols, C_{22}-C_{28} n-alcohols and C_{28}-C_{32} n-alkyl diols in freshwater environments, Org. Geochem., 30, 307-318.

Vymazal, J., C. B. Craft & C. J. Richardson. 1994. Periphyton response to nitrogen and phosphorus additions in the Florida Everglades. Algol. Stud. 73: 75-97.

Walker, W.W., 1991. Water quality trends at inflows to Everglades National Park. Water Res. Bull. 27(1):59-72.

Weaver, A. J. , E.S. Sarachik, and J. Morotze, 1991. Freshwater flux forcing of decadal and interdecadal oceanic variability. Nature 353: 836-838.

Wetzel R.G., Hatcher P.G. and Bianchi T.S., 1995, Natural photolysis by ultraviolet irradiance of recalcitrant dissolved organic matter to simple substrates for rapid bacterial metabolism, Limnol. Ocean., 40, 1369-1380.

Wu, Y. F. H. Sklar, and K. Rutchey, 1997. Analysis and simulations of fragmentation patterns in the Everglades. Ecological Applications 7(1): 268-276.

Zieman, J.C. 1974. Methods for the study of the growth and production of turtle grass, {UThalassia} {Utestudinum} K"nig. Aquaculture 4: 139-143.

Zieman, J.C., J.W. Fourqurean, and T.A. Frankovich. 1999. Seagrass dieoff in Florida Bay (USA): long-term trends in abundance and growth of turtle grass, *Thalassia testudinum*. Estuaries 22 (in press).

Zieman, J.C., J.W. Fourqurean, and R.L. Iverson. 1989. Distribution, abundance and productivity of seagrasses and macroalgae in Florida Bay. Bulletin of Marine Science 44: 292-311.

Zumstein J. and Buffle J., 1989, Circulation of pedogenic and aquagenic organic matter in an eutrophic lake, Wat. Res., 23, 229-239.