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Modeling the impacts of pulsed riverine inflows on hydrodynamics and water quality in the Barataria Bay estuary

Anindita Das

Louisiana State University and Agricultural and Mechanical College

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**MODELING THE IMPACTS OF PULSED RIVERINE INFLOWS ON
HYDRODYNAMICS AND WATER QUALITY IN THE BARATARIA BAY
ESTUARY**

A Dissertation

Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
in partial fulfillment of the
requirements for the degree of
Doctor of Philosophy

in

The Department of Oceanography and Coastal Sciences

by

Anindita Das

B.S., University of Calcutta, 1991

M.S., University of Calcutta, 1993

PGDEM, Indian School of Social Welfare and Business Management, Kolkata, 1999

M.S., Louisiana State University, 2003

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DEDICATION

To my parents, Anjali and Dr. Brig. Shyama Charan Dutta, for always going that extra mile, and encouraging me to do the same.

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ABSTRACT

Eutrophication and coastal wetland loss are the major environmental problems affecting estuaries around the world. In Louisiana, controlled diversions of the Mississippi River water back into coastal wetlands are thought to be an important engineering solution that could reverse coastal land loss. There are concerns, however, that freshwater diversions may increase nutrient inputs and create severe eutrophication problems in estuaries and wetlands adjacent to the diversion sites.

My dissertation research concerns modeling the effects of the observed and hypothetical freshwater diversion discharges on the hydrodynamics, salinity and water quality in the Barataria estuary, a deltaic estuary in south Louisiana. This estuary receives freshwater and nutrient discharges from the Davis Pond diversion, the world's largest freshwater diversion project. I have implemented two Barataria Bay simulation models of differing complexity, a simple 6-box mass-balance model and a high resolution two-dimensional (2-D) coupled hydrology-hydrodynamic- water quality model. Model results have shown that the Barataria estuary imports nitrogen and exports carbon to the coastal ocean. Compared to the lower Mississippi River, the Barataria estuary appears to be a very small source of total organic carbon for the northern Gulf of Mexico and is unlikely to have a significant influence on the development of the Gulf's hypoxia. Model simulations pointed out that the effects of different diversion discharges on salinity are most apparent in the middle and lower sections of the Barataria estuary. Further, tracer simulation experiments have shown that residence times differ markedly at different locations within the same water body due to differences in small scale hydrodynamics. Model simulations clearly demonstrated the importance of residence times for the overall functioning of the estuary. Model simulations also pointed out the differences in spatial

patterns in phytoplankton response to distributed freshwater and nutrient inflows, reflecting the near-field control of nutrients and far-field control of residence times on phytoplankton standing stock. The models reiterate the fact that there are significant tradeoffs in using freshwater diversions in coastal restoration efforts, namely tradeoffs between hydrologic restoration and water quality effects.

CHAPTER 1

BARATARIA ESTUARY MODELING – BACKGROUND, RATIONALE AND OBJECTIVES

BACKGROUND AND RATIONALE

An estuary is a semi-enclosed body of water connected with the open sea, within which sea water is measurably diluted with fresh water derived from land drainage (Pritchard, 1967). Estuaries occupy 17% of the total land surface (USGCRP, 2000) and are among the most fertile and productive ecosystems on earth. This fertility makes estuaries both economically and ecologically important.

The economic importance of estuaries is evident from the fact that 22 of the 32 world's largest cities are located on estuaries (Ross, 1995), probably because estuaries afford safe harbors for water borne industries. Human population at or near estuaries is estimated to increase by 13% from 1999 to 2010 (Bricker et al., 1999), and these ecosystems come under increasing pressure from human occupation and development. Important natural resources are supported by estuaries. Estuarine-dependent species comprised about 46% by weight and 68% by value of all U.S. commercial fisheries landings from 2000 to 2004. This translates to over \$2 billion based on the value of the commercial fisheries catch in 2006 (\$4 billion; Lellis-Dibble et al., 2008). Additionally, recreational fishing generates billions of dollars through small businesses such as boat charters, tackle shops, restaurants and hotels (Van Voorhees and Pritchard, 2007). The economic value of estuaries is further enhanced by the fact that estuarine dependent species comprise about 80% of the recreational fisheries landings.

Estuaries are ecologically important because they provide habitat to a large number of animal and plant species. Many species (for example crabs, fish and shrimp) use estuaries as nursery areas during a part of their life cycle (USEPA, 1993; McLusky and Elliott, 2004). They

also serve as migratory pathways for both birds and fishes. Estuarine wetlands help to filter out pollutants (White et al., 2006) and other materials like nutrients from the water (Soetaert et al., 2006) before it flows into the sea (USEPA, 1993). Wetlands act as buffers against storm surges, thus protecting coastal shorelines and neighboring inland areas from the devastating effects of hurricanes (NOAA, 2005; Lopez, 2009).

Estuarine ecosystems are fragile and vulnerable to disturbances in part because of their transitional nature between the land and the sea ecosystems. For example, small disturbances in their salt, temperature, nutrient balance and sediment delivery can have a profound impact on the physiography, hydrology and nature of estuarine ecosystems. These disturbances can be caused by natural factors (e.g., winds, tides, subsidence) and anthropogenic factors (e.g., coastal development, pollution, construction of canals, dams, and levees, introduction of non-native species) (Turner, 1997), or interacting natural and human caused factors.

Eutrophication and wetland loss are among the many problems being faced by estuaries. Eutrophication is manifested in a number of harmful and unwanted effects that typically include reduced water clarity associated with elevated chlorophyll *a* levels (Boynton et al., 1982; Nixon and Pilson, 1983), noxious and toxic algal blooms (ORCA, 1992; Rabalais et al., 1996), reduced dissolved oxygen concentrations (Whitledge, 1985; Gerlach, 1990), loss of submerged aquatic vegetation (McGlathery, 2001; Twilley et al., 1985; Burkholder et al., 1992), change in the benthic fauna (Dauer et al., 1992; Rabalais et al., 2001), loss of benthic secondary production (Diaz and Rosenberg, 2008), and modification of benthic predator-prey interactions (Breitberg et al., 1997; Sagasti et al., 2001).

Excessive inputs of nitrogen and phosphorus from anthropogenic sources have been recognized as the main factor affecting estuarine eutrophication (Rabalais et al., 2002; Bricker et al., 2008). Elevated nitrogen and phosphorus concentrations and changes in their relative

proportions can alter trophic interactions and trigger algal blooms that can lead to hypoxic and/or anoxic conditions that negatively affect aquatic food webs. Noxious or toxic blooms, such as the *Pfiesteria* outbreak in the Neuse and Pamlico rivers in 1992 and in the Chesapeake Bay in 1997, can harm commercial and recreational fisheries (Carpenter, 1998; Howarth et al., 2000) and cause human health problems (Howarth et al., 2000). Other human activities can also affect estuarine eutrophication. Dredging and construction of channels and artificial levees, for example, alter estuarine turbidity, hydrology, flushing rates, and the processing of nutrients in estuaries. Eutrophic conditions have been observed with increasing frequency in estuaries all over the world. In the U.S., at least 65% of all estuaries can be classified as eutrophic, with highly eutrophic conditions prevailing in estuaries along the Gulf of Mexico and the middle Atlantic U.S. coasts (Bricker et al., 2008).

Estuarine and coastal wetlands are disappearing at an alarming rate. In the 19th and early part of the 20th century, extensive areas of wetlands (~38%; Good et al., 1998) were drained and filled mainly for agricultural purposes. In the mid-1970s the U.S. had an estimated $4.3 \times 10^{11} \text{ m}^2$ of total wetlands. By the mid-1980s, the wetland area had decreased by $1.1 \times 10^{10} \text{ m}^2$, to $4.2 \times 10^{11} \text{ m}^2$. Among the remaining wetlands, $3.9 \times 10^{11} \text{ m}^2$ were fresh water wetlands and $2.2 \times 10^{10} \text{ m}^2$ were estuarine wetlands.

Various wetland protection measures since the mid-1980s have helped to slow down the wetland loss rate in the country (Dahl and Johnson, 1991). The rate of wetland loss from the mid 1950s to the mid 1970s was about $1.9 \times 10^9 \text{ m}^2$ per year. The rate of loss started decreasing from the mid 1970s to the mid 1980s: the rate of loss was about $1.2 \times 10^9 \text{ m}^2$ per year and from 1986 to 1997, the rate of loss of wetlands was $2.4 \times 10^8 \text{ m}^2$ per year (net estuarine wetland loss was about $4 \times 10^6 \text{ m}^2$ per year).

Historically, the Louisiana coastal zone has been the site of the massive wetland loss amounting to about a quarter of the nearly $1.9 \times 10^{10} \text{ m}^2$ of wetlands existing at the beginning of the 20th century (Gagliano et al., 1981), and a loss of $3.5 \times 10^9 \text{ m}^2$ from 1956 to 2006 (Barras, 2008). The majority of wetland loss occurred due to the vegetated wetlands giving way to open water bodies (Dahl and Johnson, 1991). Wetland loss in Louisiana is a function of both natural and man-made factors. Natural land loss is attributed to subsidence and net erosion in abandoned river deltas, loss of barrier islands and widening of inlets (Craig et al., 1979) and sea level rise (Boesch et al., 1994; Day et al., 2000). The man-made factors include levee construction, channelization and spoil bank construction, land reclamation, impoundments for agricultural development, and urban development (Craig et al., 1979; Boesch et al., 1994; Dahl, 2006). Leveeing of the Mississippi River stopped the inflow of sediments from the river that naturally compensated for natural subsidence occurring in the wetlands, and the current accretion due to organic matter from marsh plants is not enough to counter the subsidence rates (Craig et al., 1979; Boesch et al., 1994). Loss of barrier islands and channelization has caused loss in habitat and land loss, due to altered hydrology, saltwater intrusion and marsh deterioration (Craig et al., 1979; Boesch et al., 1994; Reddy and DeLaune, 2008).

Methods used to restore wetlands include hydrologic restoration, planting appropriate vegetation, marsh and shoreline creation, sediment trapping, stabilization of barrier islands, and river diversions (Milano, 1999; CWPRA 2006). Restoring tidal inundation is important in salt marsh restoration. For example, tidal exchange was restored to impounded saltwater marshes in Florida's Indian River Lagoon, resulting in a recovery of native plant, invertebrate and fish communities (Brockmeyer et al., 1997). Restoration of tidal flow in New England salt marshes led to increased water levels, increased salinities and increased use by estuarine fish (Burdick et al., 1997).

In Louisiana, wetland restoration efforts include river diversions, marsh restoration using sediment slurry, reopening distributaries, creation of marsh terraces and shoreline protection (Turner and Streever, 2002; Day et al., 2007). River diversions are thought to be effective in reducing high salinities caused by salt water intrusions and reduced tidal inundation (Boesch et al., 1994). They are also thought to stimulate marsh growth by providing sediments and nutrients, and supply iron that can precipitate toxic sulfides (DeLaune et al., 2003). Studies on the effect of pulsed fresh water discharges from the Caernarvon Diversion on the Breton Sound Estuary have shown that diversion decreases estuarine water residence times, tempers the impact of drought on the marshes by reducing saltwater intrusion, increases marsh water interactions by increasing overland flow, and increases sediment delivery, especially during the rising river stage (Day et al., 2009). Changes in nutrient concentration and nutrient stoichiometric ratios during pulsed events have also been observed (Lane et al., 2004) and phytoplankton tend to be flushed out during high discharge events due to short residence times (Day et al., 2009). Nevertheless, there are a number of potential problems associated with fresh water diversions, including increased eutrophication and increased potential for development of harmful algal blooms (Turner et al., 2004; Ren et al., 2009), development of hypoxia in receiving water bodies, bioaccumulation of heavy metals, and possible weakening of marsh substrate and destruction of belowground biomass (Darby and Turner, 2008a, b; Swarzenski et al., 2008).

My dissertation research concerns the effects of freshwater diversions on the hydrodynamics, salinity and water quality of the Barataria estuary, a deltaic estuary located in the Mississippi River delta. Most of the data used in my dissertation were collected as a part of the NOAA-funded project “Multistress: Coastal Stressors in the Northern Gulf of Mexico” (2003-2008; principal investigator, R. E. Turner). This project examined the effects of eutrophication and associated secondary impacts (e.g. salinity changes, altered productivity and

species composition of phytoplankton) associated with diversions of Mississippi River water through the Davis Pond diversion. The objectives of the Multistress project were to quantify the effects of eutrophication in concert with other anthropogenic and natural stressors, identify indicators of cumulative stress at individual, population, and ecosystem levels, evaluate the effectiveness of potential mitigation strategies, and extend the approaches, results, models and techniques developed in this project to other coastal ecosystems.

Within the overall Multistress project, my specific objectives were:

1. to develop and calibrate a 6-box mass-balance model (BOX) of the Barataria estuary,
2. to develop and calibrate a fully coupled hydrodynamic-biological two dimensional (2-D) water quality model,
3. compare and crosscheck the outputs of BOX and 2-D models against measured values and other published models,
4. use BOX and 2-D models to describe estuarine-shelf exchanges and the coupling between fresh water and nutrient inflows, residence times, and phytoplankton dynamics in the estuary, and,
5. predict probable changes in the hydrodynamics and water quality of the Barataria estuary for a number of historical and hypothetical diversion scenarios.

My study seeks to answer the following questions:

1. How are fresh water diversions from the Mississippi River affecting the salinity and the residence times of the various waterbodies in the Barataria estuary?
2. How are the diversions affecting estuarine-shelf exchanges of water, nutrients and carbon?
3. How are the diversions affecting phytoplankton productivity and standing stock?

4. Will the estuary become increasingly eutrophic if the Davis Pond diversion discharge is increased or additional diversions are implemented?

STUDY SITE

The Barataria estuary (Figure 1.1) is located in the north-central Gulf of Mexico, just to the west of the Mississippi River Delta. The estuary is about 120 km long and angles northwest to southeast. The average depth is about 2 m. The Barataria estuary was formed about 2000 years ago due to the process of delta switching (Rejmanek et al., 1987). All its lakes and wetlands are built on a number of overlapping delta lobes of the Mississippi River deltaic plain (Madden et al., 1988). Alluvial floodplains in the north transition to coastal marshes in the south, and the landscape is fragmented by many bayous and their natural levees. The average annual rainfall is about 150 cm with a yearly evaporation of about 75 cm (Rabalais et al., 1995). The estuary is surrounded by agricultural and urban areas and so receives a fair amount of nutrients from anthropogenic sources (Rabalais et al., 1995). The lower end of the basin is influenced by the Gulf of Mexico and the Mississippi River plume. Nutrient concentrations, phytoplankton production, chlorophyll *a* concentrations and turbidity show a gradient along the axis of the Barataria estuary (Rabalais et al., 1995). Water bodies in the upper and middle basin are the most eutrophic (Witzig and Day, 1983; Ren et al., 2009).

Artificial flood control levees have been constructed along the Mississippi River during the last 100 years, thereby obstructing freshwater flow, sediments and nutrients from the Mississippi river into the adjacent estuaries (Boesch, 1996; Reed and Wilson, 2004). Channelization of the basin has changed the hydrography of the basin and has reduced its ability to assimilate high nutrient loadings (Gael and Hopkinson, 1979; Reed and Wilson, 2004; Boesch, 2006). Fresh water enters the Barataria estuary mainly from four sources: rainfall, stream runoff, man-made diversions, and from the Gulf Intracoastal Waterway. Only a small amount of

riverine input is introduced into the basin's wetlands through the Naomi and West Pointe a la Hache siphons. Freshwater is also being introduced through the Davis Pond freshwater diversion (up to $300 \text{ m}^3 \text{ s}^{-1}$) that started operating in 2002.

The Davis Pond freshwater diversion structure (Figure 1.2) is located in St. Charles Parish, Louisiana, on the west bank of the Mississippi river. It is the largest freshwater diversion in the world (LaCoast, 2002). The primary purpose of Davis Pond was salinity control in the Barataria estuary. It is expected that introducing freshwater from the Mississippi River into the Barataria estuary through the Davis Pond structure would help control salinity and increase the production of seafood (oysters, shrimps, crabs, etc) as well as provide habitat for other animals and birds (USACE, 2004; Lopez, 2009).

There are differing opinions on whether or not river diversions are effective means of wetland restoration (e.g., Day et al., 2009; Turner, 2009). The arguments supporting the use of fresh water diversions in coastal restoration may be summarized as follows: (1) water residence times are reduced by increased flushing, thereby decreasing the potential for algal blooms (2) impact of drought years is diminished, and (3) over-marsh flow of the diverted water increases nitrogen removal via denitrification (Day et al., 2009). The arguments against diversions are: (1) there are few data showing that decreasing salinity will help reduce wetland loss, (2) there are no scientific estimates of land gain or loss from diversion inflows, (3) the excess nutrients coming in from the river decrease belowground biomass, which is considered to be important in sustaining marshes (Turner, 2009), and (4) increased nutrients from the diversions could increase the occurrence of algal blooms (Ren et al., 2009).

THE ROLE OF MODELING

Simulation modeling is a powerful tool when it comes to helping develop strategies for



Figure 1.1. Satellite image of the Barataria estuary showing the major water bodies. Arrows denote locations of the fresh water diversion Davis Pond, and fresh water siphons Naomi and Pointe a la Hache.

best management practices for restoration of estuaries, which is especially true in cases where measurements are expensive and impractical, or when the time scales are too long, or where the temporal domain extends into the future beyond when the experimental data can be collected (Soetaert and Herman, 2009). Numerical models can also be used to test the validity of the existing hypotheses based on historical data, as well as to guide future monitoring efforts. Proper model calibration is crucial, and model results should be expected to be reasonable and useful for management decisions only to the extent that actual future external forcing functions match those in the forecast scenarios.

Simple models (i.e., box models) and complex models (i.e., 2-D and 3-D models) each have their own unique advantages and drawbacks. An advantage of simple models is that their data requirements for inputs and calibration/validation are much less extensive than for complex models. One consequence of this advantage is that simple models (e.g., Justić et al., 2002; Scavia et al., 2003) can often be applied/tested using data from much longer periods of record than complex models (e.g., Bierman et al., 1994). The ability to test simple models for long periods of record confers them with a degree of robustness that strengthens their ability to forecast future conditions, subject to the above caveats. The offset for this advantage is that simple models provide information on only a limited number of parameters, e.g., average oxygen concentrations at a single station (Justić et al., 2002), or summer average hypoxic area (Scavia et al., 2003; Turner et al., 2005; Turner et al., 2006; Turner et al., 2008). Consequently, simple models can indeed be valuable forecasting tools, but they do so at the expense of providing understanding of the multiple cause-effect mechanisms governing a particular system.

An advantage of complex models is that they can generally provide a better understanding of the progression of events and their causal mechanisms that are impossible to derive solely from observational data (e.g., mass balance components for carbon, oxygen and nutrients; relative importance of light, temperature and nutrients in limiting primary production; transport and fate of organic carbon). A disadvantage of complex models is the extensive data requirements for inputs, calibration and validation. Consequently, it is much more difficult to apply complex models for long periods of record and confirm their robustness over the full dynamic ranges of their external forcing functions (e.g., Bierman et al., 1994 model calibration periods - three summer snapshots, 1985, 1988 and 1990). Complex models are very effective as forecasting tools, but they are expensive to develop, calibrate and validate because of the

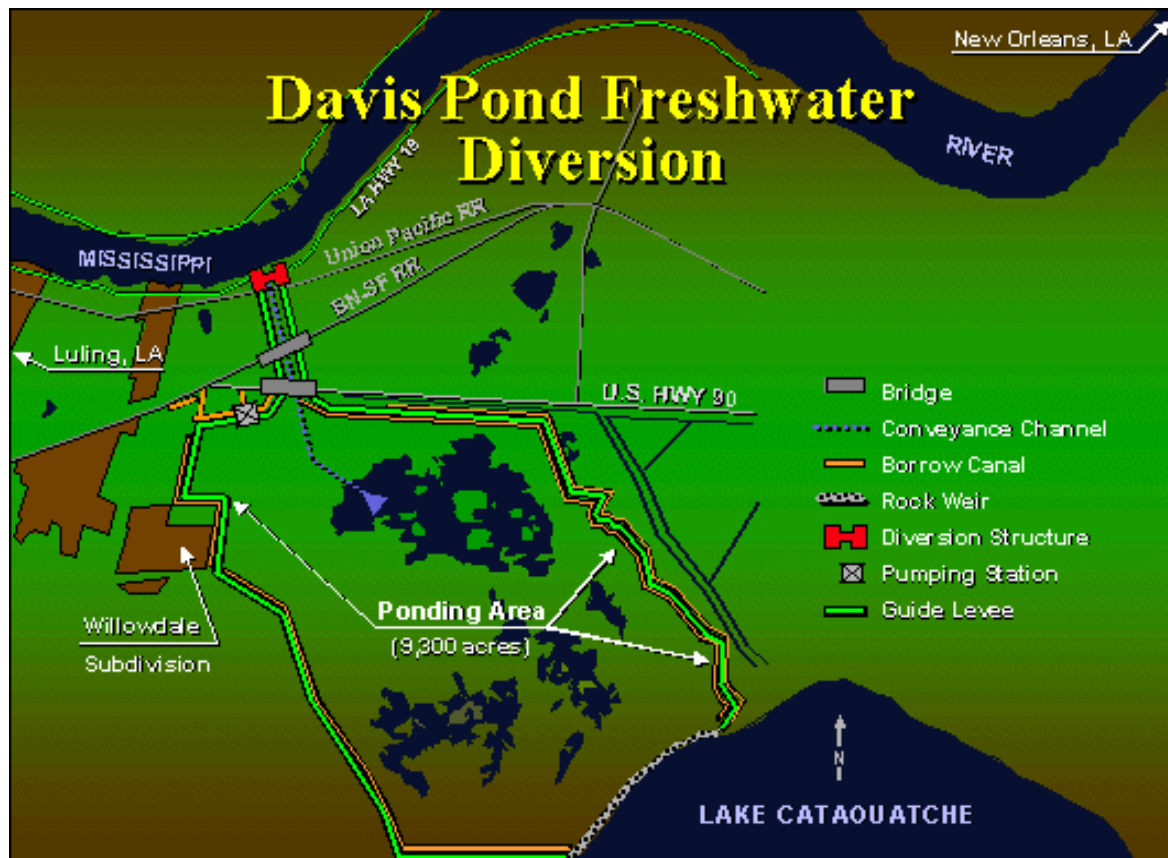


Figure 1.2. Construction features in the Davis Pond Diversion area (Source: USACE).

difficulty in demonstrating their robustness over the full range of conditions for which they were designed.

A number of physical and simulation models were developed for the Barataria estuary and the surrounding Louisiana estuaries. One of the earliest physical modeling experiments was performed by von Arx (1957) to study the effects of local winds, tides, river discharge, stratification, and bottom and internal frictions on the circulation patterns in shallow water bays and estuaries using the Barataria estuary as an example. Hacker (1973) developed a two-dimensional (2-D), vertically averaged transport model of the Barataria estuary that described water level variations, velocity profiles, temperature and salinity distributions. Hopkinson and Day (1977) developed a biological model of the Barataria estuary that simulated annual changes

in nitrogen and carbon concentrations. The model was run for 10 years, and the results suggested that the model reached a steady-state after the fourth year. Hart (1978) used a numerical model of the Chandeluer-Breton Sound Estuary to simulate tide- and wind-induced circulation over several tidal cycles. Wiseman and Swenson (1989) developed a tidal prism model of the Barataria estuary and Terrebonne-Timbalier Basin to estimate the flushing characteristics of these estuaries and the degree to which produced waters contribute to increasing the salinity levels in the estuaries. They used tidal ranges to calculate tidal excursion of a particle and used that to determine segment boundaries in the model. They also used this model to determine the fate of a contaminant introduced into a specific segment. Park (1998, 2002) developed a two-dimensional, depth-integrated hydrodynamic model to simulate estuarine processes in the Barataria estuary and examined the impact of freshwater dispersion by controlling the freshwater sources. The time period for the model simulations was three weeks. Moffat and Nichols (2005) modeled the Barataria estuary using a 2-D finite element TABS-2 model, a numerical modeling system developed by the U.S. Army Corps of Engineers. They modeled salinity, water levels, current velocities and fresh water discharges from Davis Pond. The simulations were carried out over a period of 10 days. Three dimensional modelling of the Barataria estuary has also been attempted. One such example is by McCorquodale and Georgiou (2006) who used the Finite Volume Coastal Ocean Model (FVCOM) to model the hydrodynamics and transport in the Barataria estuary.

All the hydrodynamic models described above were run only over short time intervals, ranging from a few days to a few weeks. It is very difficult to acquire accurate numerical results from such short runs, because model spin-up times (i.e., response to model initial conditions) are often considerably longer. Much more accurate numerical results can be obtained if a model is run for an entire year. This not only alleviates problems with model spin-up time but also takes

into account both daily and seasonal variability in the model forcing functions. For that reason, I have developed and implemented two models of differing complexities that simulate conditions in the Barataria estuary over an annual cycle. This approach, in most cases, was computationally very intensive and required the use of supercomputers.

In Chapter 2, I describe a novel tidal prism 6-element model that was developed to examine estuarine-shelf exchanges in the Barataria estuary. The model was used to calculate the fluxes of water, nitrogen and carbon through the Barataria estuary passes and to estimate the importance of estuarine derived nitrogen and carbon for the overall carbon budget and development of hypoxia in the northern Gulf of Mexico.

In Chapter 3, I compare the skills of two different coupled hydrodynamic-water quality models that were developed for the Barataria estuary, a simple 6-box (6 element) model and a complex high resolution 2-D model (~1.3 million elements), and discuss the advantages and shortcomings of each of the models.

In Chapter 4, I look at how salinities will vary across the Barataria estuary (near field versus far field) in response to different fresh water diversion scenarios using the high resolution 2-D model.

In Chapter 5, I describe a simple nutrient-phytoplankton-zooplankton (NPZ) model that was used in conjunction with the 2-D hydrology-hydrodynamic model. I use this fully coupled biological-physical model to simulate the complex patterns of phytoplankton biomass dynamics (represented by the concentration of chlorophyll *a*) in the Barataria estuary. In this chapter, I also carry out a number of simulated tracer experiments to determine the residence times of various water bodies in the Barataria estuary under different diversion scenarios. Tracer experiments are also used to estimate the travel time for a phytoplankton bloom originating in the upper estuary to reach the oyster beds in the middle and lower estuary.

In Chapter 6 (Summary), I present an overview of the results and conclusions from the entire study.

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CHAPTER 2

MODELING ESTUARINE-SHELF EXCHANGES IN A DELTAIC ESTUARY: IMPLICATIONS FOR COASTAL CARBON BUDGETS AND HYPOXIA

INTRODUCTION

Eutrophication has been a growing problem in many estuarine and coastal ecosystems around the world (Rosenberg, 1985; Diaz and Rosenberg, 1995; Nixon, 1995; Rabalais et al., 2007). Eutrophication is often characterized as an increase in the supply of organic matter (Nixon, 1995), whose manifestations include enhanced primary productivity, noxious phytoplankton blooms and bottom water hypoxia (Officer et al., 1984; Smayda, 1990; Rabalais et al., 2007). The extent and severity of these phenomena in the coastal waters worldwide have increased during the late 20th century (Justić et al., 1987; Andersson and Rydberg, 1988; Cooper and Brush, 1991; Hickel et al., 1993; Turner and Rabalais, 1994), coincidentally with increased use of fertilizer in the watersheds and higher riverine concentrations of nitrogen and phosphorus (Justić et al., 1995; Howarth et al., 1996; Turner et al., 2007).

In the northern Gulf of Mexico, widespread hypoxia has been documented for over 20 years, with present areal extent of up to 22,000 km² (Figure 2.1). Hypoxia typically occurs from March through October in waters below the pycnocline, and extends between 5 and 60 m depth offshore (Rabalais et al., 2007). Model hindcasts suggest that large hypoxic regions were not likely to have been present prior to the mid-1970s and that the size of those regions grew steadily until the mid 1980s (Scavia et al., 2003; Turner et al., 2006, 2007). Hindcasts of oxygen levels below the pycnocline (Justić et al., 2002) suggest that summertime oxygen minima in the central section of the Gulf's hypoxic zone between 1955 and 1969 were fairly constant, always >2 mg l⁻¹ and most often > 4 mg l⁻¹. The oxygen concentrations decreased during the 1970s, and have

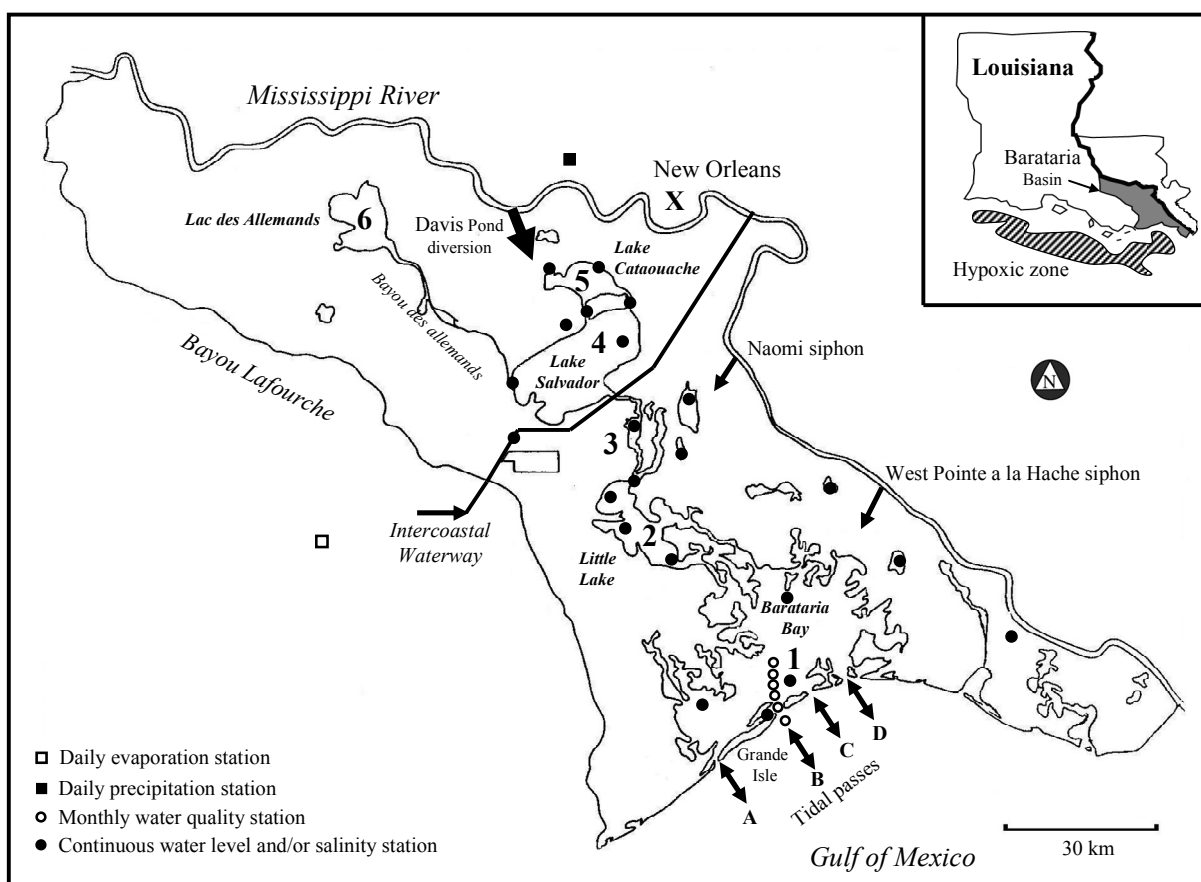


Figure 2.1. Map of the Barataria estuary showing major water bodies, sampling stations, and the locations of the Mississippi River freshwater diversions (block arrows). The tidal passes are indicated by the bi-directional block arrows (A = Barataria Pass, B = Caminada Pass, C = Pass Abel, and D = Quatre Bayou Pass). The water bodies corresponding to the model boxes are indicated by the numbers 1-6. The insert map shows the extent of the Gulf's hypoxic zone during 1993.

remained consistently lower than 2 mg l^{-1} in most years since. Model results are consistent with the limited historical oxygen concentration data collected between 1970 and 1985, before the shelfwide surveys began (Turner and Allen, 1982; Rabalais et al., 1999, 2002). These model results are additionally supported by retrospective analyses of sedimentary records, including organic carbon accumulation rates (Eadie et al., 1994), biogenic silica content (Turner and Rabalais, 1994), and stratigraphic records of benthic foraminifera (Sen Gupta et al., 1996; Platon and Sen Gupta, 2001; Platon et al., 2005).

The strong temporal association between the magnitude of the Mississippi River nutrient fluxes and areal extent of hypoxia suggests that riverborne nutrients play a dominant role in the development of hypoxia in the northern Gulf of Mexico (Rabalais et al., 2007; Turner et al., 2008). High riverine nutrient inputs lead to high surface primary productivity, which is also manifested in a high carbon flux to the sediments. Recently, several researchers have pointed to deteriorating coastal wetlands as another potential source of carbon for the Gulf's hypoxia region (Dagg et al., 2007). The Louisiana coastal zone inshore of the hypoxia region (Figure 2.1) is the site of massive wetland loss amounting to about a quarter of the nearly 2 million ha of wetlands existing at the beginning of the 20th century (Gagliano et al., 1981). The coastal wetland loss rate in Louisiana was about 77 km² yr⁻¹ from 1978 to 2000 (Barras et al., 2003). This loss is attributed to a complex interaction of factors, including altered wetland hydrology, channelization, sea-level rise, and elimination of riverine sediment input to coastal wetlands due to flood control levees on the Mississippi River (e.g. Day et al., 1997; Turner, 1997). While deteriorating wetlands have a potential to release large amounts of nutrients and carbon into the surrounding bays and estuaries, the export of these materials to the coastal Gulf of Mexico has not been quantified.

In this paper, we examine estuarine-shelf exchanges in the Barataria estuary, a deltaic estuary located in the north-central Gulf of Mexico (Figure 2.1). This estuary had the highest historical land loss rates in coastal Louisiana, averaging nearly 25 km² yr⁻¹ from 1978 to 2000 (Barras et al., 2003). Our objectives are twofold: (1) to calculate the fluxes of water, nitrogen and carbon through the Barataria passes, and, 2) to estimate the importance of estuarine derived nitrogen and carbon for the overall carbon budget and development of hypoxia in the northern Gulf of Mexico.

STUDY SITE

The Barataria estuary (Figure 2.1) is located in the north-central Gulf of Mexico, just to the west of the Mississippi River Delta. The estuary is about 120 km long and angles southeast towards the Gulf of Mexico. The average depth is about 2 m. The estuarine basin is bounded on the east by the levee of the Mississippi River, on the west by a former channel of the Mississippi River, Bayou Lafourche, and on the south by the Gulf of Mexico. A chain of barrier islands separates the estuary from the Gulf of Mexico. The northern half of the basin contains several large lakes. The southern half of the basin contains tidally influenced marshes interconnected by ponds, lakes, and channels that finally empty into a large bay system behind the barrier islands.

Artificial flood control levees have been constructed along the Mississippi River during the last 100 years thereby obstructing freshwater flow into the estuary. Freshwater enters the Barataria estuary mainly from four sources: rainfall, stream runoff, man-made diversions, and from the Gulf Intracoastal Waterway. Only a small amount of riverine input is introduced into the basin's wetlands through the Naomi and West Pointe a la Hache siphons. Freshwater is also being introduced through the Davis Pond freshwater diversion which started operating in July, 2002. Although the structure has a design capacity of up to $300 \text{ m}^3 \text{ s}^{-1}$, it was only operated 29% of the year during 2002 with a mean flow of $15 \text{ m}^3 \text{ s}^{-1}$ and a maximum flow of $64 \text{ m}^3 \text{ s}^{-1}$.

The estuary is connected to the Gulf of Mexico through four tidal passes (Barataria, Caminada, Abel and Quatre Bayou). The tropic diurnal tide range is approximately 0.35 m at the coastal endpoint, but decreases by an order of magnitude as the tide progresses up the estuary. This gradient is attributed to the energy loss as the tide moves through the highly frictional deltaic landscape (Snedden et al., 2007). Salinities range from near zero in the upper reaches of the estuary to about 25 in the southernmost section of the estuary. The coastal waters adjacent to the Barataria estuary are strongly influenced by the Mississippi River. In terms of the size of the

watershed, freshwater discharge, and sediment discharge, the Mississippi River ranks the third, sixth, and seventh in the world, respectively (Milliman and Meade, 1983). The 1817-2002 average discharge rate for the lower Mississippi River was around $16,000 \text{ m}^3 \text{ s}^{-1}$ (Turner et al., 2007). The Mississippi River delta has prograded to the shelf break and much of the water discharges into deep waters (Wiseman et al., 1999). As a result, the buoyant freshwater plume lifts off the bottom and expands rapidly as soon as it leaves the river mouth (Wiseman and Garvine, 1995). The Mississippi River plume initially flows in a clockwise direction until encountering the Louisiana coast where it mostly becomes a part of the westward flowing Louisiana Coastal Current.

MODEL FORMULATION

The model domain was divided into six boxes that correspond to major water bodies in the Barataria estuary: Barataria Bay (Box 1), Little Lake (Box 2), Bayou Perot-Rigolettes (Box 3), Lake Salvador (Box 4), Lake Cataouatche (Box 5) and Lac des Allemands (Box 6) (Figure 2.2). Surface water areas and wetland areas, volumes and tidal prisms of individual model boxes are given in Table 2.1. We developed a variation of a tidal prism model that calculates volumes and water level variations in response to hydrodynamic and hydrologic forcings.

The mass balance equations for volumes in boxes 1-6 were:

$$\partial V_i / \partial t = F_i + P_i + R_i - E_i + Q_i$$

where V_i is the segment volume (m^3), F_i is the influx (or outflux) of water due to sea level variations in the Gulf of Mexico ($\text{m}^3 \text{ hr}^{-1}$), P_i is direct precipitation over the box area ($\text{m}^3 \text{ hr}^{-1}$), R_i is runoff from the adjacent wetland areas ($\text{m}^3 \text{ hr}^{-1}$), E_i is evaporation ($\text{m}^3 \text{ hr}^{-1}$) and Q_i is runoff from the Mississippi River diversions ($\text{m}^3 \text{ hr}^{-1}$) (Box 5 only). The F_i was evaluated as a product

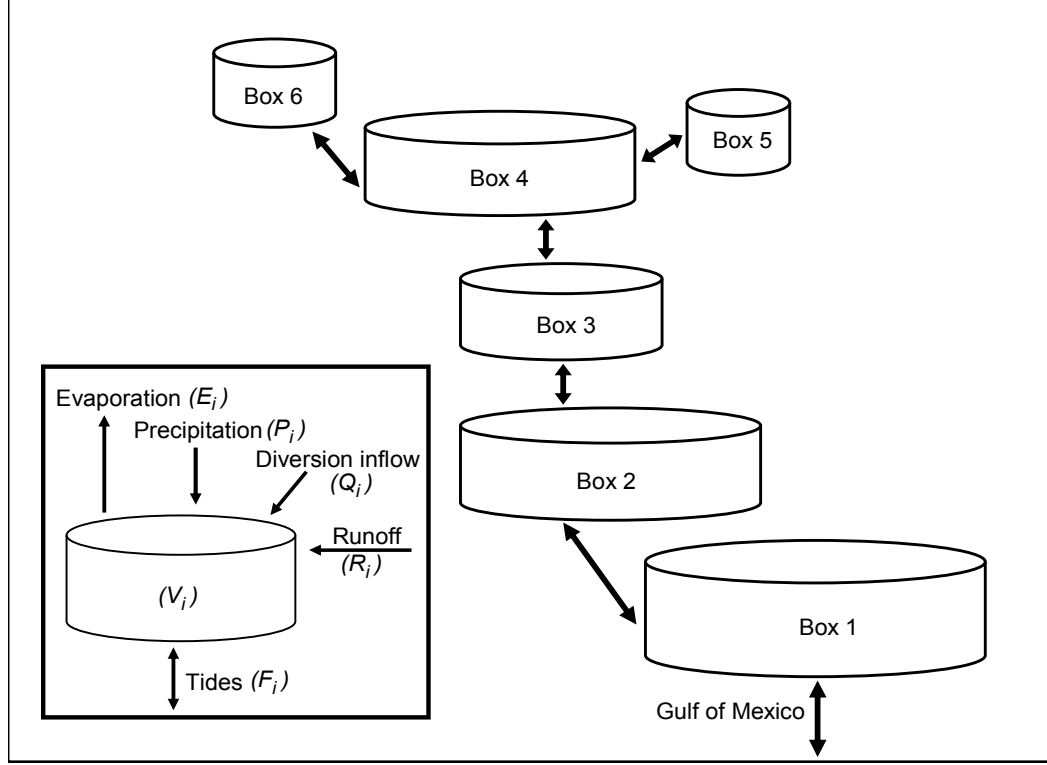


Figure 2.2. Conceptual model showing connections between model boxes and representative forcing functions.

of the rate of sea level change ($\partial L / \partial t$), box water area (S_i) and tidal attenuation coefficient (Ψ_i):

$$F_i = (\partial L / \partial t) S_i \Psi_i.$$

The tidal attenuation coefficient (Table 2.1) is expressed as the ratio of the mean tidal amplitude in a given box and mean tidal amplitude at the offshore endpoint station at Grand Isle (0.35 m) and it represents the fraction of total tidal prism that will be contained within a specific model box during the rising tide. This formulation proved inadequate in describing low frequency and high amplitude water level variations due to frontal passages and tropical disturbances. These non-tidal variations in water level propagate through the estuary with smaller attenuation compared to tides, so a different formulation for F_i was used to describe those events:

$$F_i = (\partial L / \partial t) \zeta_i S_i \Psi_i \quad \text{if } L > 0.35.$$

In the above equation ζ_i is a scaling constant that was estimated by calibration. The freshwater runoff (R_i) was calculated as

$$R_i = P_i - ET_i$$

where P_i ($\text{m}^3 \text{ hr}^{-1}$) and ET_i ($\text{m}^3 \text{ hr}^{-1}$) are direct precipitation and evapotranspiration, respectively, over the wetland area associated with box i (Table 2.1). The model equations were solved using the Runge-Kutta integration method of the fourth order, and an integration step of 0.1 hour.

Table 2.1. Characteristics of individual model boxes.

Box No.	Water Body	Water area (m^2)	Wetland area (m^2)	Total Volume (m^3)	Tidal amplitude (m)	Tidal attenuation coefficient	Tidal prism (m^3)
1	Barataria Bay	8.5×10^8	3.0×10^8	1.7×10^9	0.30	0.86	2.5×10^8
2	Little Lake	4.2×10^8	3.5×10^8	8.4×10^8	0.12	0.34	5.1×10^7
3	Perot-Rigolettes	2.1×10^8	3.3×10^8	4.2×10^8	0.08	0.23	1.7×10^7
4	Lake Salvador	2.6×10^8	4.9×10^8	5.2×10^8	0.02	0.06	5.1×10^6
5	Lake Cataouatche	6.3×10^7	2.6×10^8	1.3×10^8	0.03	0.09	1.9×10^6
6	Lac des Allemands	1.0×10^8	1.0×10^9	2.0×10^8	0.01	0.03	1.0×10^6

DATA

The input data set included hourly data on precipitation, evaporation, evapotranspiration, sea level variations at the coastal station Grand Isle, and Davis Pond discharge (Figure 2.3).

Hourly water level data were obtained from recording gages (41 stations) maintained by the Louisiana Department of Wildlife and Fisheries (LDWF), the United States Geologic Survey

(USGS) and the Louisiana Department of Natural Resources (LA DNR). Multiple stations within a box were averaged to obtain the mean hourly water levels. Precipitation (P) and evaporation (E) data were obtained from the National Climatic Data Center (NCDC). Daily precipitation and evaporation were only available as daily totals. Hourly values of precipitation were obtained by dividing the total daily precipitation by 24. The evaporation was pro-rated over a 24-hour period using the average difference between temperature and dew point, which generated a curve with minimum evaporation at night and maximum during the day (Figure 2.3). The evapotranspiration (ET) was calculated using the Thornthwaite equation, as described in Mitsch and Gosselink (1993). Sea level elevation data (L) at the coastal station Grand Isle were obtained from the National Ocean Service (NOS). Davis Pond discharge (Q) data were obtained from the LA DNR.

The fluxes of nitrate (defined as $\text{N-NO}_3 + \text{N-NO}_2$) and total organic carbon (TOC) were estimated based on simulated water fluxes through the Barataria passes and concentration gradients between the lower Barataria estuary and the coastal Gulf of Mexico (Table 2.2). The nitrate and TOC data were obtained from monthly water quality transects conducted by researchers at Louisiana State University (R. E. Turner, unpublished). Nitrate analysis followed EPA Method 353.2 using a Lachat Series 8000 QuickChem[®] FIA⁺ auto analyzer. TOC was measured by the high temperature catalytic oxidation (HTCO) method using a Shimadzu TOC-5000A. The estimates of water and land areas within the Barataria estuary (courtesy of J. Barras, USGS) were used to define model boxes.

MODEL RESULTS

The 2002 data were used as the reference data set for model calibration. The Davis Pond diversion started operating in July 2002 (Figure 2.3) and so we were able to examine system responses with and without diversion. During 2002, coastal Louisiana experienced frequent frontal passages that increased the amplitude of sea level variations significantly above the mean

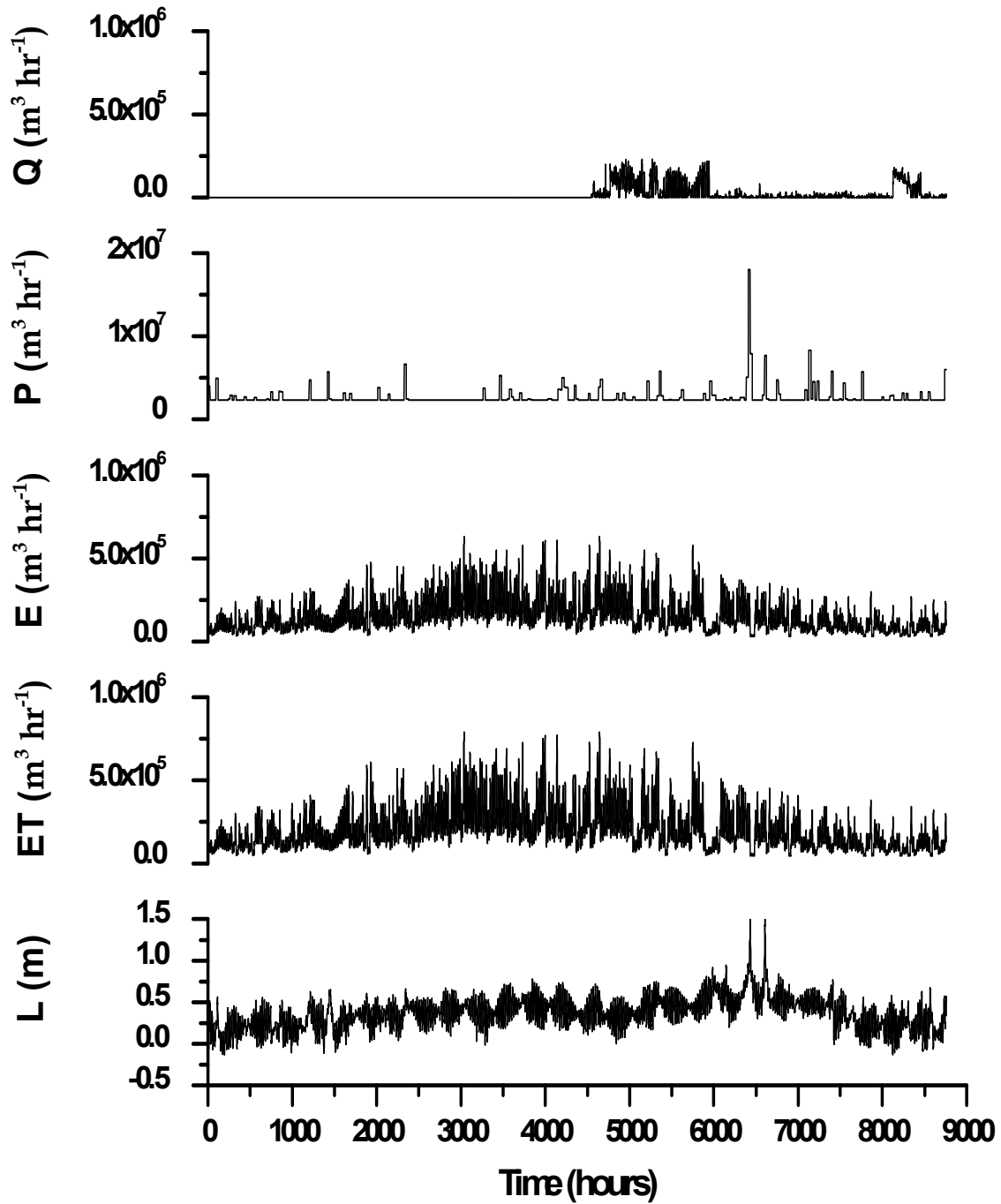


Figure 2.3. Input data set consisting of hourly observations of Davis Pond discharge (Q), precipitation (P), evaporation (E), evapotranspiration (ET) and Sea level elevation (L) during 2002.

tropical diurnal tide range of 0.35 m (Figure 2.3). Also, tropical storm Isidore and hurricane Lili affected the area during September 2002. These storms had similar water level responses but significantly different rainfall amounts that provided a unique opportunity to test model responses to simultaneous variations in the two key forcing functions. Finally, between October 24 and December 7, 2002, the ADCP current measurements were carried out in all four tidal passes (Moffatt and Nichol, 2005). Incidentally, those were the only ADCP data available for 2002, but they provided a benchmark against which the calibrated model was verified.

The results from a correlation analysis revealed that the calibrated model performed well (Figure 2.4), explaining 63 %, 75 % and 82 % of the observed variability in hourly, daily and

Table 2.2. Average concentrations of nitrate and total organic carbon (TOC) in the lower Barataria estuary and in the coastal Gulf of Mexico for 1994 - 2005 (R. E. Turner, unpublished data). The Barataria Bay average was computed from the monthly data ($n = 680$) collected at five stations depicted in Fig 1. The offshore sampling station ($n = 134$) was located approximately 2 km from the entrance into the estuary. Negative sign denotes that a constituent is exported from the estuary, and * denotes a significant difference ($\alpha = 0.05$) based on Tukey's studentized range test.

Constituent	Barataria estuary	Gulf of Mexico	Difference	Standard error
Nitrate (μM)	6.3	10.9	4.6*	0.8
TOC (mg l^{-1})	5.1	4.1	-1.0*	0.2

weekly water level records, respectively. Due to the relative simplicity of our model, the entire dynamics of water in the estuary (e.g., marsh flooding/drying and local wind effects) could not have been fully reproduced. For example, during winter months, the model consistently underestimated water levels in the Barataria Bay. Nevertheless, flux calculations with and

without winter months were within 3 % of each other so this did not significantly affect the overall flux calculations. Interestingly, the model accurately described high amplitude non-tidal variations in sea level that propagated through the estuary with significantly less attenuation compared to tides, causing greater inundation in the upper reaches of the estuary. The modeled fluxes of water (Q) through the Barataria passes ranged from near zero to over $\pm 40,000 \text{ m}^3 \text{ s}^{-1}$ (Figure 2.5). The calculated mean hourly Q value was $6,930 \text{ m}^3 \text{ s}^{-1}$, or about 43 % of the average discharge of the lower Mississippi River (Table 2.3). The agreement between the modeled and observed (Moffatt and Nichol, 2005) Q values was also very good. The average modeled Q values for November and December (the only time period for which data was available) were $8,020$ and $6,643 \text{ m}^3 \text{ s}^{-1}$ and for the ebbing and flooding stages respectively. The average measured q values were $7,307$ and $6,099 \text{ m}^3 \text{ s}^{-1}$ for the ebbing and flooding stages respectively. The mean residual error was 9 %, which was deemed acceptable given the wide range of flow conditions in the tidal passes (Figure 2.5).

The nitrate and TOC were collected as single monthly discrete samples, which did not allow for the determination of flood and ebb concentrations of these constituents. The annual imports (or exports) of nitrate and TOC were calculated by multiplying the average estuary-shelf gradient in these constituents (Table 2.2) by the cumulative annual flux of water during the flooding (or ebbing) stages of the tidal cycle. The results show that the Barataria estuary annually exports $109 \times 10^6 \text{ kg}$ TOC to the coastal Gulf of Mexico, while importing $7 \times 10^6 \text{ kg}$ nitrate (Table 2.3). The overall errors in these export and import terms were estimated by combining the uncertainty in the modeled fluxes of water (residual error) and uncertainty in the measured nitrate and TOC values (standard errors). They range from 9 % in case of Q , 26 % for nitrate, to 29 % for TOC (Table 2.3).

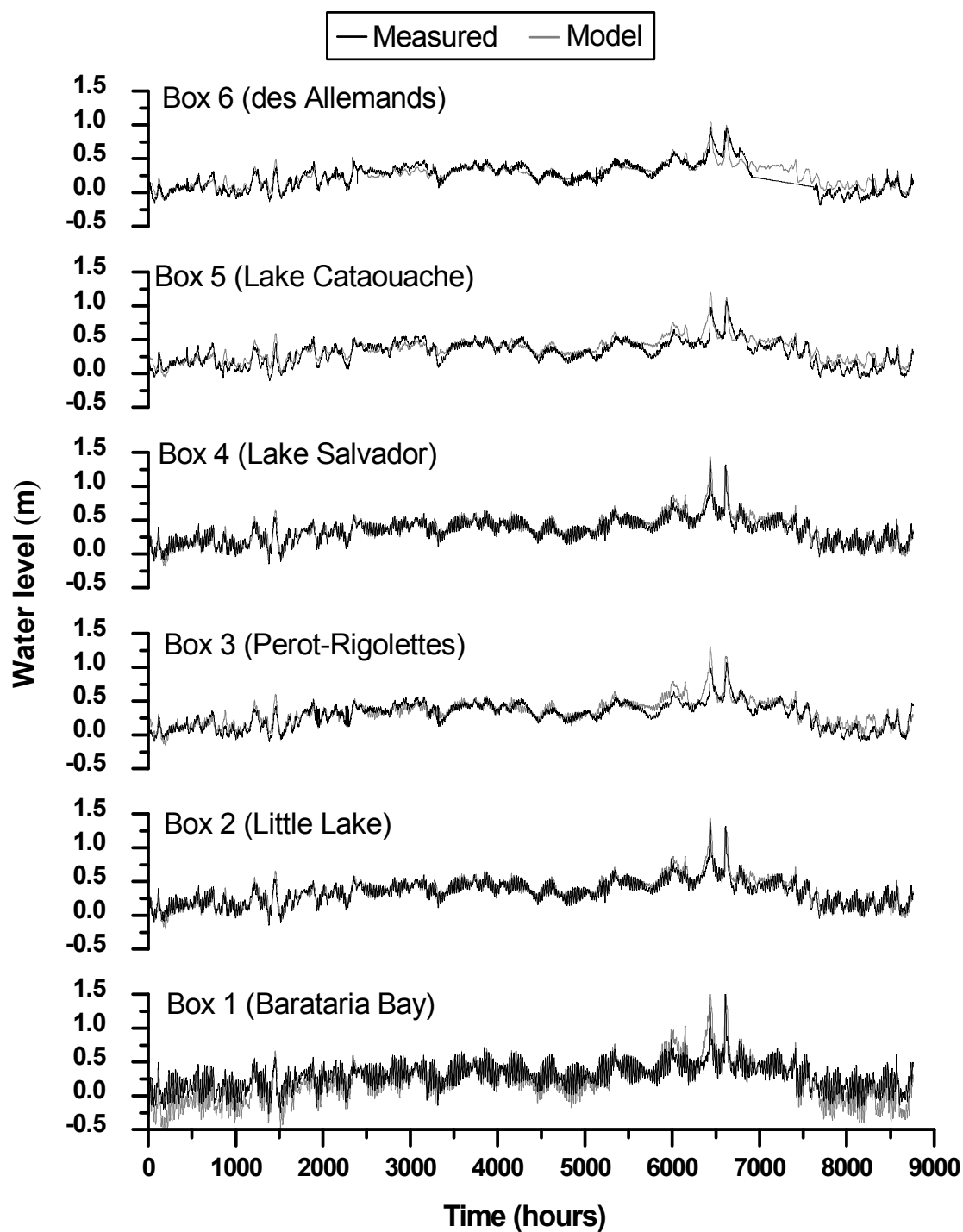


Figure 2.4. Observed (black line) and predicted (grey line) hourly water levels for the six model boxes in the Barataria estuary during 2002. The results are arranged from inland (Box 6) to the coast (Box 1).

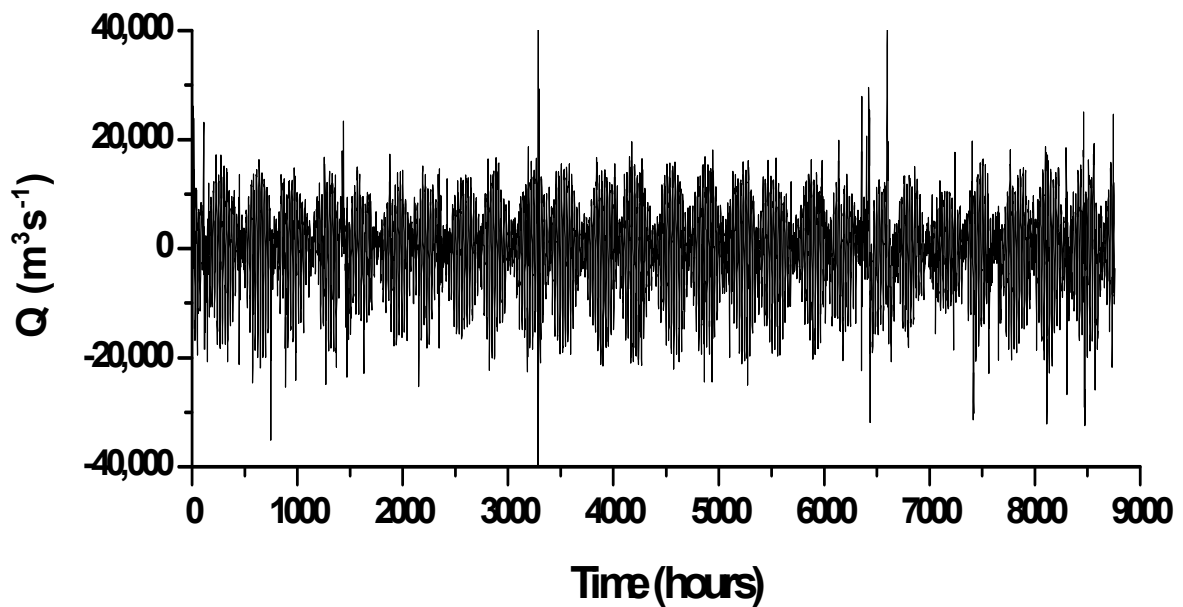


Figure 2.5. Simulated hourly fluxes of water through the Barataria passes during 2002.

Table 2.3. Estimates of fluxes of water (Q), nitrate and total organic carbon (TOC) for the lower Mississippi River (MR) and the Barataria estuary (BE). Error terms associated with constituent flux estimates include uncertainty in the modeled fluxes of water (residual error) and uncertainty in the measured NITRATE and TOC values (standard errors). Negative sign denotes that a constituent is exported from the estuary; a - Turner et al. 2007; b – Bianchi et al. 2007; c - this study.

Constituent	Mississippi River	Barataria estuary	BE : MR (%)
Q (m ³ s ⁻¹)	16,000 ^a	6,930 ± 624 ^c	43.3
Nitrate (10 ⁶ kg N yr ⁻¹)	723.6 ^a	7.0 ± 1.8 ^c	1.0
TOC (10 ⁶ kg yr ⁻¹)	4,000.0 ^b	-109.3 ± 31.7 ^c	2.7

DISCUSSION

A number of different statistical and simulation models have been developed to study physical and ecological processes in estuarine systems. They range from simple tidal prism models (e.g., Dyer and Taylor, 1973; Luketina, 1998; Sheldon and Alber, 2006), box models (Miller and McPherson, 1991; Roson et al., 1997; Hagy et al., 2000; Humborg et al., 2000; Kohlmeier and Ebenhoh, 2007), one dimensional models (Flindt and Kamp-Nielsen, 1997; Hinrichsen and Wulff, 1998), two dimensional finite element (Canu et al., 2003; Ferrarin and Umgiesser, 2005) and finite difference models (Inoue et al., 2008), to three dimensional models (Rajar and Cetina, 1997; Zheng et al, 2004; Solidoro et al, 2005; Lin et al, 2008).

Box models are comparatively simple models and are best suited for decadal change scenarios and for long term hindcasting (Humborg et al., 2000) or forecasting. Box models have been used to estimate mixing and flushing times (Zimmerman, 1976) which in turn can be used to predict fluxes of water and dissolved constituents at the estuary-ocean interface (Helder and Ruurdij, 1982). Miller and McPherson (1991) applied a box model to estimate estuarine residence times in Charlotte Harbour, Florida, using freshwater flow and tidal flushing. Mohrholz and Lass (1998) combined a box model with a numerical model to estimate water exchange between the Oder estuary and Pomeranian Bight. Box models have also been used to calculate net physical transport and residence times in partially stratified estuaries (Roson et al, 1997; Hagy et al, 2000).

Tidal prism models are helpful in determining residence times and concentrations of dissolved and particulate constituents in well mixed estuaries (Luketina, 1998). Simple tidal prism models (e.g., Dyer and Taylor, 1973) have been modified in a number of ways to describe flushing characteristics of different estuarine systems. For example, Wood (1979) and Sanford et al. (1992) modeled the segmentation based on the ebb tide rather than on the flood tide. Brown

and Arellano (1980) included mixing at various branches of the estuary. Models developed by Pritchard (1960) and Guo and Lordi (2000) included incomplete mixing of the flood flow. Smith (1993) further modified Pritchard's (1960) model by including six tidal constituents to account for spring-neap tidal cycles and tidal and non-tidal flushing. A modification proposed by Sheldon and Alber (2006) included partial in-estuary mixing to get a better estimate of turnover times. Kuo et al. (2005) incorporated the tidal prism model in their numerical computation for small highly branched coastal basins.

Most tidal prism models (e.g., Dyer and Taylor, 1973) are statistical models in a sense that they provide water level and salinity estimates only for high and low water. The model described here is a dynamic model which yields hourly water level, volume and salinity estimates for individual model boxes. The model is driven by sea level variations at the open boundary. It takes into account both tidal and non-tidal variations in water level and has only one scaling constant that needs to be estimated by calibration.

The export of wetland-derived materials to the coastal ocean (the "Outwelling" hypothesis, Odum, 1980) has been tested numerous times over the past several decades (e.g. Nixon, 1980; Moran et al., 1991; Dame 1996; Alongi, 1998; Jickells et al., 2000). While a number of studies have shown that estuaries export large amounts of nutrients and carbon (e.g., Dame et al., 1986), few studies have attempted to estimate the importance of estuarine sources for the coastal carbon budgets in river-dominated coastal ecosystems. In case of the Barataria estuary, two additional issues deserve consideration. First, the estuary has been the site of a massive wetland loss, and the carbon from eroded wetlands has not been accounted for. If the estuary acts as an exporter of carbon, it would be important to find out if the magnitude of export relates to the spatial scale of wetland loss. Second, given the vicinity of the Mississippi River

delta, it would be interesting to know how estuarine fluxes compare to riverine nutrient and carbon subsidies.

Our results show that the Barataria estuary annually exports 109×10^6 kg TOC (Table 2.3), or about $57 \text{ gC m}^2 \text{ yr}^{-1}$ when prorated to the total water area of $1.9 \times 10^9 \text{ m}^2$ (Table 1). This estimate is lower than previously reported by Happ et al. (1977), who estimated TOC flux to lie between the extremes of 25 and $540 \text{ gC m}^2 \text{ yr}^{-1}$, with the most probable values around $150 \text{ gC m}^2 \text{ yr}^{-1}$. Assuming that carbon content in wetland soils is 0.026 g cm^{-3} (Gosselink et al., 1984), the TOC export is equivalent to the loss of $4.2 \times 10^6 \text{ m}^3$ of wetlands. On an areal basis, carbon export from the Barataria estuary is equivalent to a loss of 0.5 m of wetland soil horizon over an area of 8.4 km^2 , or equivalent to about 34 % of the observed annual wetland loss between 1978 and 2000 (Barras et al., 2003). Interestingly, the magnitude of TOC export from the Barataria estuary is equal to 2.7 % of the Mississippi River TOC flux (Table 2.3).

The extent to which carbon export from the Barataria estuary may influence processes in the coastal Gulf of Mexico depends largely of the following three factors: 1) the magnitude of TOC flux, 2) lability of estuarine TOC, and, 3) the existence of a favorable current regime that would stimulate cross-shelf transport. Reports based on biochemical assessments suggest that 50-60% of estuarine TOC may be labile (Ittekkot, 1988; Spitzy and Leenheer, 1991). However, studies based on bioassays (e.g. Søndergaard and Middelboe, 1995) suggest that only 14-25% of dissolved organic carbon in riverine and marine samples should be considered labile. In the northern Gulf of Mexico, favorable conditions for cross-shelf transport exist only during autumn and winter, when short-term wind reversals from frontal passages often reverse plume direction. During the rest of the year, a westward flowing coastal current presents a strong obstacle for cross-shelf transport. Nevertheless, assuming that all the TOC exported from the Barataria estuary is delivered to the shelf and evenly distributed over a $16,000 \text{ km}^2$ hypoxic zone (Rabalais

et al., 2007), the loading rate would be $6.8 \text{ gC m}^{-2} \text{ yr}^{-1}$. The primary production rates in the coastal northern Gulf of Mexico range from $160 \text{ gC m}^{-2} \text{ yr}^{-1}$ (Chen et al., 2000) to $300 \text{ gC m}^{-2} \text{ yr}^{-1}$ (Sklar and Turner, 1981), so carbon export from the Barataria estuary could potentially account for 2.3 - 4.3 % of the annual primary production. If all the estuarine carbon were labile and respired within the hypoxic zone, it would create an oxygen demand of $24 \text{ gO}_2 \text{ m}^{-2} \text{ yr}^{-1}$, or about 10 % of measured ($222 \text{ gO}_2 \text{ m}^{-2} \text{ yr}^{-1}$; Rowe et al., 1992) and modeled ($197 \text{ gO}_2 \text{ m}^{-2} \text{ yr}^{-1}$; Justić et al., 1996) values. Further, Rabalais et al. (1991) suggested that around 50% of surface primary production may be reaching the bottom ($\sim 20 \text{ m}$ on average) in the northern Gulf of Mexico. Because of the reasons stated above, these estimates of the possible importance of the estuarine carbon export for offshore hypoxia are probably grossly exaggerated.

The Barataria estuary annually receives $7 \times 10^6 \text{ kg}$ nitrate from the coastal Gulf of Mexico (Table 3). This value corresponds to one percent of the lower Mississippi River nitrate flux (Table 3). The finding that the Barataria estuary imports nitrogen from the coastal ocean is not surprising given the relative vicinity (65 km) of the Mississippi River delta. Walker et al. (2005) have shown that easterly winds, prevalent in autumn, winter, and spring, drive a westward flow of river waters around the delta onto the Louisiana shelf (Figure 2.6). During peak river flow, this westward current exhibits velocities of $0.4 - 0.9 \text{ m s}^{-1}$, is 20 km wide, and transports $1.40 - 1.65 \times 10^5 \text{ m}^3 \text{ s}^{-1}$ of river and shelf water. It usually turns towards the coast between 89.5° W and 90° W , feeding a clockwise gyre in the Louisiana Bight and a westward coastal current (Walker et al., 2005). Under these conditions, a parcel of water originating from the Mississippi River could theoretically reach the mouth of the Barataria estuary in 20 - 45 hours.

CONCLUSIONS

The Barataria is an “inverted” or “river-injected” estuary that receives nitrogen through the tidal passes and releases carbon to the coastal ocean. The mean calculated tidal pass flow of 6

$930 \text{ m}^3 \text{ s}^{-1}$ is equivalent to a 43 % of the lower Mississippi River discharge. The annual TOC export is $109 \times 10^6 \text{ kg}$, or $57 \text{ gC m}^2 \text{ yr}^{-1}$ when prorated to the total water area of the estuary. This carbon export is equivalent to loss of 0.5 m of wetland soil horizon over an area of 8.4 km^2 , and

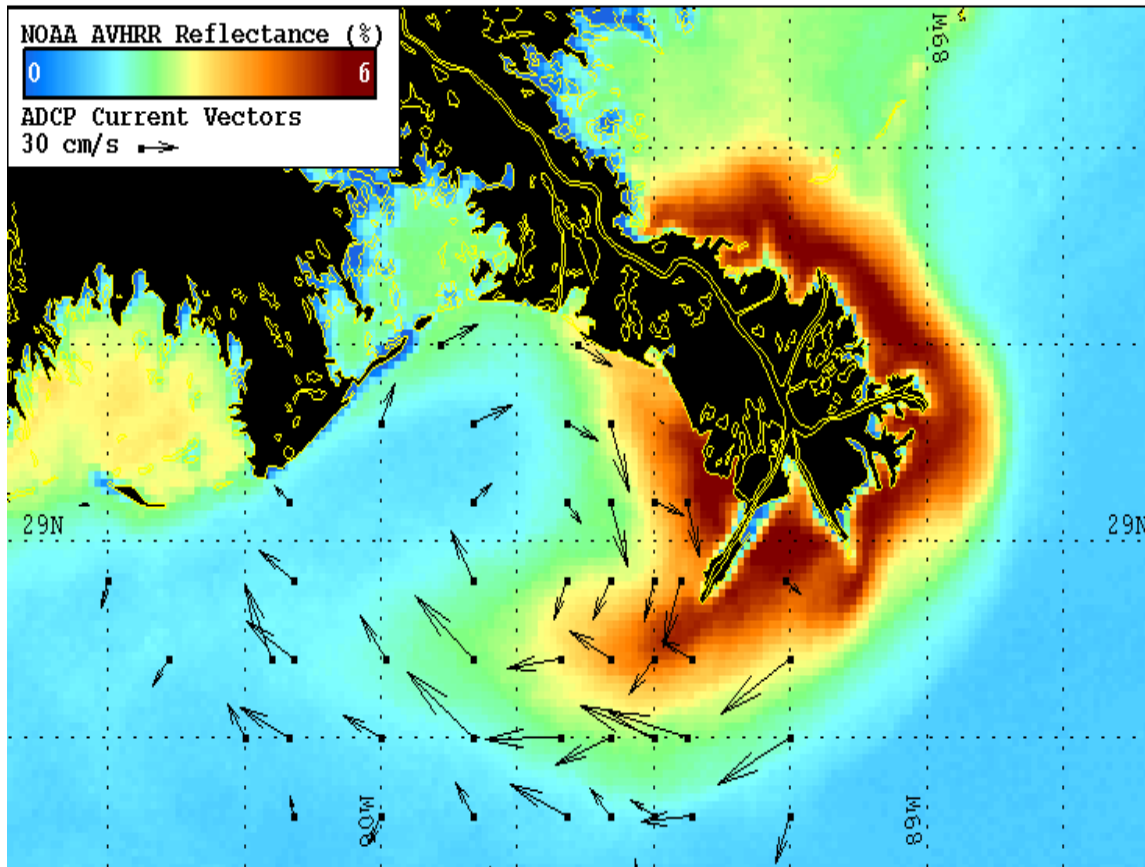


Figure 2.6. Anticyclonic eddy west of the Mississippi River delta (modified from Walker et al. 2005).

equivalent to 34 % of the observed annual wetland loss in the estuary between 1978 and 2000. Compared to the lower Mississippi River, the Barataria estuary appears to be an insignificant source of TOC for the northern Gulf of Mexico (2.7 % of riverine TOC). Assuming that all the TOC exported from the Barataria estuary is delivered to the shelf and evenly distributed over a $16,000 \text{ km}^2$ hypoxic zone (Rabalais et al., 2007), it could potentially account for 2.3 – 4.3 % of the annual primary production. If all the estuarine carbon would be labile and respired within the

hypoxic zone, it would create an oxygen demand of $24 \text{ gO}_2 \text{ m}^{-2} \text{ yr}^{-1}$, or about 10 % of observed oxygen demand. In the coastal Gulf of Mexico, favorable conditions for cross-shelf transport exist only during autumn and winter. The results of this study strongly suggest that carbon export from the Barataria estuary alone has little impact on coastal carbon budgets and development of the Gulf's hypoxia.

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CHAPTER 3

VERIFICATION AND VALIDATION OF BARATARIA BAY MODELS: THE OUTWELLING HYPOTHESIS REVISITED

INTRODUCTION

In recent years, mechanistic models are being increasingly used in simulating estuarine hydrodynamics and water quality. Mechanistic models attempt to simulate processes within the aquatic environment. However, given the high structural and functional complexity of aquatic ecosystems, it is often difficult to construct a mechanistic model that adequately describes the natural system. Also, complete verification and validation of complex mechanistic models is often not possible, because an independent data set is not available. In the absence of such data, perhaps the best way to examine model validity is to compare it with other existing models developed for a given ecosystem, and crosscheck model responses across the same parameter space using identical driving variables.

In the previous chapter I have shown, by using a simple 6-box model, that the Barataria estuary exports carbon to and imports nitrogen from the coastal waters. In this chapter, I will compare the results of three hydrodynamic models of differing complexity that were developed for the Barataria estuary: (1) a low-resolution (6-box or 6-element) model (Das et al., 2010), (2) a medium-resolution (16,749 elements) 2-dimensional (2-D) TABS-2 model (Moffatt and Nichol, 2005), and, (3) a high-resolution (2-D) coupled hydrodynamic-water quality model. The hydrodynamic part of this model was originally developed by Inoue et al. (2008), and it has been refined and coupled to a water quality model within this study. I will use these different models to examine water levels, salinity and estuarine-shelf exchanges in the Barataria estuary. The main objective is to find out whether simulated estuarine-shelf exchanges confirm or negate the “Outwelling” hypothesis (Odum, 1980).

THE “OUTWELLING” HYPOTHESIS

The exchanges of carbon and nutrients occurring at the estuary-ocean interface have been debated for decades (Nixon, 1980; Childers et al., 2000). At the heart of the debate is the “Outwelling” hypothesis which states that marsh-estuarine systems produce more material than can be degraded or stored within the systems. The excess material is exported to the coastal ocean where it supports ocean productivity (Odum, 1980). Recent research on the outwelling hypothesis has still not settled the debate. For example, Moran et al. (1991) studied the dynamics of lignin oxidation products, a proxy for terrestrially derived carbon, in the waters of the continental shelf of the southeastern U.S. They concluded that a contribution of terrestrially derived carbon (as represented by lignin) was highly variable on both weekly and seasonal time scales. Cai et al. (1999) measured dissolved oxygen concentrations and pCO₂ values in five southeastern estuaries in the U.S. They found that the estuarine export of organic matter was very small compared to the amount of dissolved inorganic carbon being exported from the estuaries. Similarly, Pomeroy et al. (2000) studied a decade-long data set (1990-1999) collected in five Georgia estuaries. Their results have shown that the amount of dissolved and particulate organic carbon entering the coastal ocean from these estuaries could explain only a very small fraction (<1%) of the overall primary production in the coastal waters. In contrast, Dame et al. (1986) found that carbon, nitrogen and phosphorous were exported both seasonally and annually from the North Inlet, North Carolina. Jickells et al. (2000) also showed that the Humber Estuary in England exported phosphorous. Cunha et al. (2003) found that carbon is exported to the coastal ocean during the summer when freshwater supply to the Ria de Aveiro estuary in Portugal is low. Also, McGuirk Flynn (2008) showed that the Mullica River-Great Bay estuary in New Jersey was a net exporter of carbon, nitrogen and phosphorous.

IMPORTANCE OF MODELS

A model is a simplified version of a real system which is built to attempt to give us an idea of how the system functions at present (near-real time simulation), how it has functioned in the past (i.e., model hindcasting), and how it may function in the future (i.e., model forecasting). Basically, the model is only an abstraction of reality because it reduces a system to its basic properties, as defined by the user. Ecosystem modeling has come a long way from being just a mathematical exercise to applying ecological knowledge in concert with mathematical and computer techniques (Jorgensen, 1994). Ecosystem modeling is a highly technical exercise that involves, among other considerations, knowledge of ecological principles, various conservation laws (e.g., mass, energy and momentum) and numerous mathematical constraints (e.g., boundary and initial conditions) (Jorgensen, 1994).

An ecological system can be modeled in several ways depending on the purpose of the model (Halfon and Reggiani, 1978; Jorgensen, 1994). The model can have simple or complex numerical equations (i.e., simple or complex code) depending on how complicated a system of interest is perceived to be and what is the purpose for which the model is developed. A model is usually divided into several sub-systems or compartments depending on the ecological hierarchy and complexity of the model code. The choice of compartments depends on model conceptualization and, in the ecological sense, it is often based on the functional groups rather than individual species. In any case, it is possible to develop many different models of varying complexity for the same ecological system (Halfon and Reggiani, 1978). As stated in Chapter 1, simple models (i.e., box models) and complex models (i.e., 2-D and 3-D models) each have their own unique sets of advantages and drawbacks. An advantage of simple models is that their data requirements for inputs and calibration/validation are much less extensive than for complex models. An advantage of complex models is that they can provide understanding of the cause-

effect relationships that are impossible to obtain from simple models. Examples of such cause-effect relationships include processes affecting mass-balance of carbon, oxygen and nutrients and the relative importance of light, temperature and nutrients in limiting primary production. A disadvantage of complex models is their extensive data requirements for inputs, calibration and validation.

An important concern in ecosystem modeling is the time scale at which the model is applied (i.e., the model temporal domain). The time scales can range from a few hours to several decades. However, the caveat is that the long term processes depend on the short term processes. For example, the integrated annual estuarine fluxes of water, nitrogen and carbon depend on hydrodynamic processes occurring over time scales ranging from minutes to days (i.e., wind stirring, currents and tides). One way of overcoming this problem is to reduce the spatial resolution by dividing the system into a number of subunits or boxes. For example, Miller and McPherson (1991) represented Charlotte Harbor, Florida, using a box model to estimate estuarine residence times; Hagy et al., (2000) used a box model to represent the Patuxent River estuary to estimate non-tidal physical circulation and water residence times; Smith (1993) used a segmented tidal prism model of the Indian River Lagoon, Florida, to study flushing characteristics; and Sakov and Parslow (2004) used a box model to calculate water transport in the Gippsland Lakes estuary, Australia.

More complex models with high spatial and temporal resolutions can now be used because of developments in computer technology (Cugier and Hir, 2002), especially the development of computer clusters and parallel processing in the early 1990's (Sterling et al., 1999).

Estuarine models have many uncertainties, and compromises have to be made to achieve a proper balance between the hydrodynamic and biological processes. Careful choices have to be

made while defining the model temporal and spatial domains, choosing the numerical methods, and selecting the boundary conditions and driving variables (Cugier and Hir, 2002). These choices may be relatively incorrect and so model output may not, a priori be equal to measured values. It is a point of this paper to explore the consequences of different choices in these three models on their ability to describe the measured values.

MODEL DESCRIPTIONS

Barataria Box Model

The spatial domain of this low-resolution model (henceforth called BOX) consists of six boxes representing the major water bodies in the Barataria estuary. The model formulation represents a variation of a tidal prism model that calculates volumes, water level and flux variations in response to hydrologic and hydrodynamic forcings. The BOX model simulations were carried out for the period January to December 2002. The first 300 hours were used in model calibration while the rest of the output was used to validate the model. A detailed model description is given in Chapter 2 and in Das et al., (2010).

2-D TABS-2 Model

The TABS-2 model is a medium resolution 2-D model consisting of 16,749 elements and 49,455 nodes. The southern model boundary is in the northern Gulf of Mexico, the western boundary is along Bayou Lafourche, the eastern boundary is along the Mississippi River and the northern boundary is at Lac des Allemands. In order to reduce the computation time, the finite element mesh is denser in the main water bodies, channels and bayous, and sparser over marshes and swamp areas. Details of the model grid are discussed in Moffatt and Nichol (2005).

The TABS-2 model combines the US Army Corps of Engineers finite element hydrodynamics (RMA-2) and salinity (RMA-4) models (Thomas and McAnally, 1990). The

TABS-2 model has been used to study the depth-averaged hydrodynamics, sedimentation and transport processes in shallow water bodies such as rivers, reservoirs, bays and estuaries. The RMA-2 model calculates water surface elevations and horizontal velocity components. It does not take into account vortices and acceleration in the vertical direction, so it cannot be used to solve near-field problems. The RMA-4 model is a constituent transport model that is designed to simulate the advection-diffusion process in the water. It is used to calculate concentrations of both conservative and non-conservative substances such as the horizontal salinity distributions, salinity intrusions, residence times and the extent of the mixing zones. The TABS-2 model used the finite element method to obtain a numerical solution for the governing equations. The TABS-2 modeling system can also simulate wetting and drying of marshes and intertidal areas.

2-D Coupled Hydrology-Hydrodynamic-Water Quality Model

The 2-D model is a high resolution (100m x 100m grid; 1.3 million elements), coupled hydrology-hydrodynamic-water quality model of the Barataria estuary (henceforth called 2-D). The hydrology-hydrodynamic part of the model was originally developed by Inoue et al. (2008). The model was refined and a water-quality component was added within this study.

The main objective behind developing a high-resolution 2-D model of the Barataria estuary was to examine the effects of freshwater diversions on salinity, residence times and water quality in the Barataria estuary. In order to simulate the hydrodynamics in the Barataria estuary, the model requires information about the hydrological cycle (precipitation and evaporation over the land areas) and the hydrodynamic forcing functions that apply to these water bodies. The approach adopted was to deal with the hydrological cycle and the hydrodynamics in the water bodies separately, and then to link them together through local runoff.

Runoff from the drainage basins is the major source of fresh water for the Barataria estuary (Marmer, 1954; Kjerfve, 1973; Light et al., 1973). The hydrology part of the model deals

with estimating runoff from rainfall and evaporation data, and then integrating them into the hydrodynamic model. The hydrology model was based on the unit hydrograph, i.e. the difference between actual local precipitation and evaporation integrated over the model domain that was used to calculate runoff. Groundwater flow is not considered significant because of the shallow depth of the Barataria estuary and the fact that most of the land is covered by wetlands that are saturated with water (Gagliano et al., 1973). Precipitation records were obtained from the U.S. Weather Service site at New Orleans Airport (MSY), because it best represents the dominant upstream region (61% of the total land). Also, rainfall is measured at the MSY at hourly intervals, which is important given the small estuarine catchment area. In order to estimate local runoff, the Barataria estuary was divided into 8 sub-basins. Each of the 8 runoff hydrographs included a specific time-delayed function to mimic marsh drainage.

The hydrodynamic part of 2-D model of the estuary is a depth-integrated two-dimensional model of estuarine circulation. Depth-integrated equations were used because the water column in this shallow estuary is generally well-mixed due to winds and tides (Inoue et al., 1998). The equations of conservation of mass and momentum, including baroclinic pressure gradient, are written in Cartesian coordinates in terms of depth-integrated transport:

$$\frac{\partial U}{\partial t} + \frac{\partial}{\partial x} \frac{U^2}{H} + \frac{\partial}{\partial y} \frac{UV}{H} - fV = -gH \frac{\partial \zeta}{\partial x} - \frac{1}{2} gH^2 \frac{\partial \rho}{\partial x} - g \frac{\frac{U}{H} \left\{ \left(\frac{U}{H} \right)^2 + \left(\frac{V}{H} \right)^2 \right\}^{1/2}}{C^2} + \frac{\tau_x}{\rho} + A \nabla^2 U \quad (1)$$

$$\frac{\partial V}{\partial t} + \frac{\partial}{\partial x} \frac{UV}{H} + \frac{\partial}{\partial y} \frac{V^2}{H} - fU = -gH \frac{\partial \zeta}{\partial y} - \frac{1}{2} gH^2 \frac{\partial \rho}{\partial y} - g \frac{\frac{V}{H} \left\{ \left(\frac{U}{H} \right)^2 + \left(\frac{V}{H} \right)^2 \right\}^{1/2}}{C^2} + \frac{\tau_y}{\rho} + A \nabla^2 V \quad (2)$$

$$\frac{\partial \zeta}{\partial t} + \frac{\partial U}{\partial x} + \frac{\partial V}{\partial y} = 0 \quad (3)$$

$$\frac{\partial HS}{\partial t} + \frac{\partial US}{\partial x} + \frac{\partial VS}{\partial y} = D_s \left(\frac{\partial H}{\partial x} \frac{\partial S}{\partial x} + \frac{\partial H}{\partial y} \frac{\partial S}{\partial y} \right) \quad (4)$$

$$\frac{\partial HT}{\partial t} + \frac{\partial UT}{\partial x} + \frac{\partial VT}{\partial y} = D_T \left(\frac{\partial H}{\partial x} \frac{\partial T}{\partial x} + \frac{\partial H}{\partial y} \frac{\partial T}{\partial y} \right) \quad (5)$$

where,

$$U = \int_{-h}^{\zeta} u \, dz$$

$$V = \int_{-h}^{\zeta} v \, dz$$

$$H = h + \zeta$$

$$S = (1/H) \int_{-h}^{\zeta} s(z) \, dz$$

$$T = (1/H) \int_{-h}^{\zeta} t(z) \, dz$$

In the above equations, t denotes time, x , y , and z are Cartesian coordinates, u and v denote velocity components in the direction of x and y , respectively, z is elevation of the free surface above mean sea level, h is the undisturbed depth of the water, f is the Coriolis parameter (assumed to be a constant), g is the acceleration due to gravity, τ_x and τ_y are the x and y components of wind stress, respectively, ρ is the density of water, $s(z)$ and $t(z)$ are depth-dependent salinity and temperature, respectively, A is the horizontal eddy viscosity, S is depth-averaged salinity, T is depth-averaged temperature, D_S and D_T are the horizontal eddy diffusivities for S and T , respectively, and C is the Chezy coefficient which is depth dependent. The bottom roughness is represented through Manning's n coefficient, such that the Chezy coefficient is evaluated as

$$C = \frac{1}{n} H^{1/6} \quad (6)$$

The model equations are finite-difference equations on a staggered mesh grid C of Arakawa (Mesinger and Arakawa, 1976). The conservation of mass and total energy is done by using the Grammelvedt C scheme (Grammelvedt, 1969). The centered-difference Leapfrog scheme is used for time integration, interspersed at regular intervals with the Euler scheme to remove the computational mode due to central time differencing. The advection scheme of Hsu and Arakawa (1990) based on Takacs scheme (Takacs, 1985) is used because this is an accurate upstream scheme with very little numerical dispersion, which is usually seen in centered-difference schemes.

The land boundaries of the model have a no normal flow and no-slip boundary conditions. Wind forcing is considered to be uniform over the model domain. The forcing functions used in this depth-integrated two-dimensional model are tides coming in from the Gulf of Mexico, rainfall and evaporation over the model domain, local runoff, salinity, water temperature, wind, and freshwater diversions from the Mississippi River (Davis Pond, West Pointe a la Hache, Naomi).

The model was run on Tezpur, the fastest supercomputer at Louisiana State University. Tezpur has a 15.3 TFlops Peak Performance and 360 computational nodes, each consisting of two Dual Core Xeon 64-bit processors operating at a core frequency of 2.66 GHz. A typical run of the 2-D model used 32 Tezpur nodes (64 processors) and required 72 hours of computing time to simulate hydrology, hydrodynamics and water quality over a 9-month period (March 20 to December 31, 2002).

Model Forcing Functions

Evaporation - Because evaporation data were available from one station, an empirical formula developed by Roll (1965), suitable for the Gulf coastal regions, was used to estimate

evaporation:

$$E = C_E \rho (q_{\text{sea}} - q_{\text{air}}) U_{10}$$

where E is the evaporation rate, C_E is the transfer coefficient, ρ is the air density, q_{sea} and q_{air} are the specific humidity for the sea and air, respectively, and U_{10} is wind speed at the 10 m reference height. Hourly measurements of the parameters used to calculate evaporation (such as atmospheric pressure, wind speed, sea surface temperature, and dew-point temperature) were obtained from the NOAA monitoring station at Grand Isle (Inoue et al., 2008).

Precipitation and Runoff – I used the precipitation data for the New Orleans airport (MSY). Because the small drainage areas of the Barataria Basin result in relatively rapid runoff, evapotranspiration was not considered. Accordingly, the runoff from any of the 8 drainage sub-basins was calculated using the simple model:

$$\text{Runoff (m}^3 \text{ hr}^{-1}) = (\text{Rainfall-Evaporation (m hr}^{-1}) \times \text{Area (m}^2)$$

The Barataria estuary was divided into 22 watershed management units using a pre-existing watershed chart (a digital map of Louisiana) to estimate the discharge rate via a network of 64 known and 522 unknown and ungauged streams (for details, see Inoue et al., 2008). A persistence time for each stream was estimated by a simple linear interpolation based on the longest stream, Bayou Chevreuil (Figure 3.2), which had a persistence time of 72 hours. The persistence time varied from 3-72 hours in the estuary. The shape of the hydrograph depends on precipitation and basin characteristics (Viessman et al., 1989). Total volume of runoff entering through a sub basin is calculated by multiplying the amount of precipitation by area of the sub basin. So there are 8 hydrographs for runoff data, each having a certain number of hours it takes to enter the basin. A one-sided filter is used to calculate rainfall per unit time per unit area for each sub basin (Inoue et al., 2008):

$$y(t_i) = a_1 x(t_{i-1}) + a_2 x(t_{i-2}) + \dots + a_j x(t_{i-j})$$

where j is the filter width and $a_1 \dots a_j$ are filter coefficients. The filter width for each sub basin is chosen to be equal to the persistence time of the water within the sub basin so that the effective runoff time is less than, or equal to, the persistence time.

Water Level, Wind and Salinity - Sea level elevation data at the coastal station Grand Isle were obtained from the National Ocean Survey (NOS, <http://www.tidesonline.nos.noaa.gov/>). Davis Pond discharge data were obtained from the Louisiana Department of Natural Resources (LADNR, http://sonris-www.dnr.state.la.us/www_root/sonris_portal_1.htm).

Water Level - Hourly water level data were obtained from recording gages (41 stations) maintained by the Louisiana Department of Wildlife and Fisheries (LDWF), the United States Geological Survey (USGS, <http://la.water.usgs.gov/>) and the LA DNR. To allow for a comparison of BOX, TABS-2 and 2-D model outputs with the observed water levels, multiple stations within the six major water bodies in the Barataria estuary (Table 3.3) were averaged to obtain the mean hourly water levels (Figure 3.1). Hourly water levels at a coastal station at Grand Isle obtained from the National Ocean Service (NOS) were used as a forcing function in BOX and 2-D models.

Wind - Wind speed was converted to wind stress using an empirical relationship derived for the Barataria Estuary (Park, 2002; Inoue et al., 2008):

$$\tau = \rho_a U_*^2 \text{ and } U_* = 0.037 U_{10} - 0.03$$

where τ is the wind stress and ρ_a is the air density and U_* is the shear velocity.

Salinity - Hourly salinity records for the station at Grand Isle (the mouth of the Barataria Bay) were used as a forcing function in both BOX and 2-D models. The BOX, TABS-2 and 2-D model salinity outputs were compared with the observed salinity values at several Barataria transect stations (Turner et al., unpublished data, Figure 3.2).

Model calibration, verification and validation - The model was calibrated and initially verified by simulating a typical dry summer condition, from July 7 to August 5, 1999 (Inoue et al., 2008). A dry season was selected to reduce the noise-to-signal ratio that might interfere with the effects of freshwater diversions. Subsequently, within this effort, the model was calibrated and validated using the data from 2002. Unlike the BOX model that was run for an entire year, because of the limited input data set for January and February, the 2-D model was run only from March to December 2002.

As explained in Chapter 2, the 2002 data provided a good reference data set for model calibration and validation. The Davis Pond diversion started operating in July 2002 and I was able to examine system responses with and without river diversion. Further, during 2002, coastal Louisiana experienced frequent frontal passages that increased the amplitude of sea level variations significantly above the mean tropical diurnal tide range of 0.35 m. Also, tropical storm Isidore and hurricane Lili affected the area during September and October 2002. These storms had similar water level responses, but significantly different rainfall amounts that provided a unique opportunity to test model responses to simultaneous variations in the two key forcing functions. Finally, between October 24 and December 3, 2002, ADCP current measurements were carried out in all four tidal passes (Moffatt and Nichol, 2005). This data set was used as a benchmark to verify the calibrated model.

Estimates of Fluxes at the Estuary-Ocean Interface - The direction of flow (u and v) at each of the grid points at the four passes at the mouth of the Barataria Bay was determined to compute flux of water at the estuary-ocean interface. Because the passes are not in a straight line, the appropriate direction of each component had to be determined for every model cell in the four tidal passes. This was done by plotting the boundary on a graph paper and determining the

direction of flow in each cell. After the model simulation was completed, the values of u and v were extracted as separate files and then used to compute the total flux.

The model outputs for flux were compared with measured data obtained from Moffat and Nichol (2005) (Figure 3.15). The data were collected using an Acoustic Doppler Current Profiler (ADCP) located at the bottom of the Barataria Pass where it measured currents throughout the water column. For this particular comparison, I used the data collected in the middle of the water column (at a depth of 15m) between October 24 and December 3, 2002.

CONTINUOUS DATA STATIONS: WATER VARIABLES

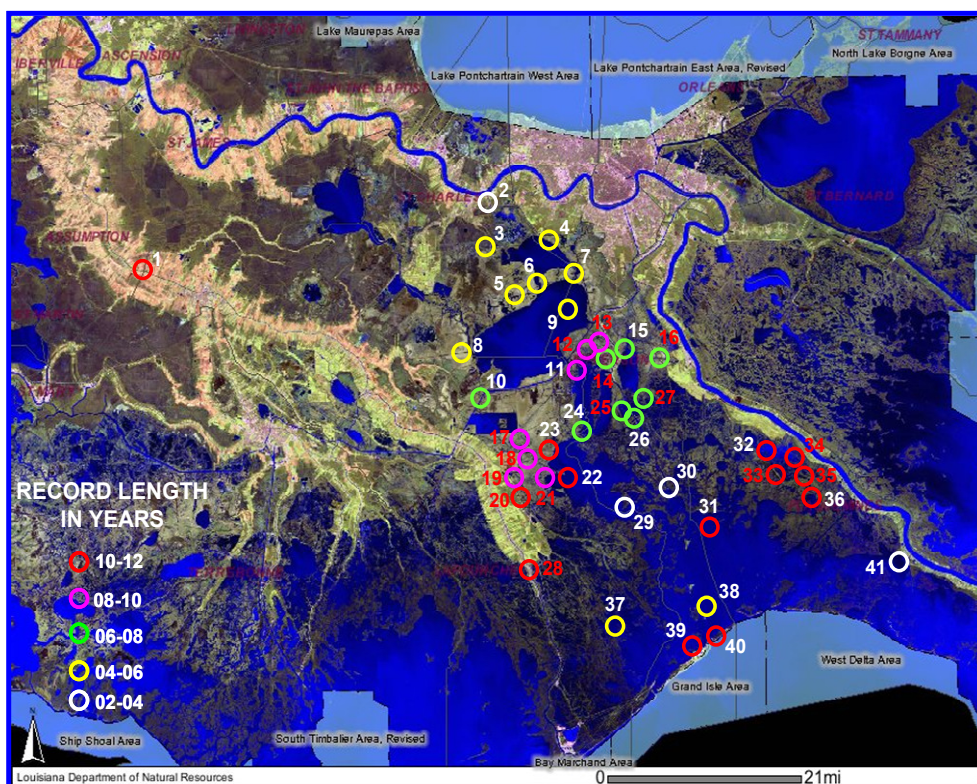


Figure 3.1. Locations of USGS and LADNR water quality monitoring stations in the Barataria estuary.

As calculated in Chapter 2, the annual imports (or exports) of nitrate, total organic carbon (TOC), particulate organic carbon (POC), dissolved organic carbon (DOC), and chlorophyll a (Chl a) were calculated by multiplying the average estuary-shelf gradient in these constituents by

the cumulative annual flux of water during the flooding (or ebbing) stages of the tidal cycle (obtained from the 2-D model) (Table 3.4).

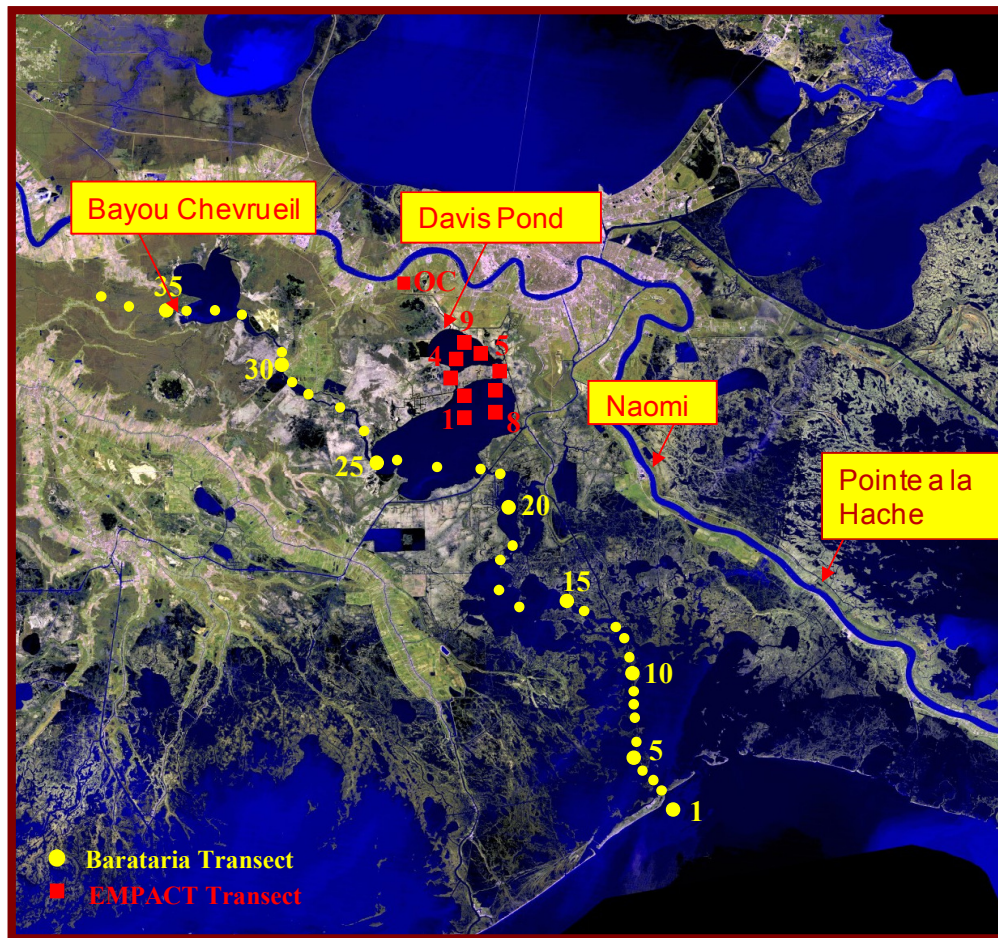


Figure 3.2. Barataria transect and EMPACT transect sampling stations in the Barataria estuary (Turner et al., unpublished data).

MODEL RESULTS

A comparison of the three models described is given in Table 3.1. The TABS-2 model (Moffatt and Nichol, 2005) had a very limited temporal coverage (one week each in September and December, 2002). The BOX model was run from January to December 2002, and the 2D model was run from March to December 2002.

Comparison of Measured and Simulated Salinity Values

The salinity values obtained from the TABS-2 model were compared to the USGS and

LADNR continuous monitoring stations (Moffatt and Nichol, 2005). The salinities obtained from the BOX and the 2-D models were compared with the discrete measurements carried out along the Barataria estuary transect (Turner et al., unpublished, Figure 3.2). In case of Lake Cataouatche, which was not occupied by the transect stations, model results were compared with the average salinity values obtained from the USGS and LADNR continuous sampling stations. Thus, the model cells from which the time series were extracted from the 2-D model simulation results, for comparison with the measured values, had the same locations as the transect stations, except in case of Cataouatche, where the measured values represent an average of several USGS and LADNR stations (Figure 3.4).

A comparison between observed and predicted salinity values (Figure 3.3 – 3.8) shows that, in general, the relative model error decreases from the upper to the lower parts of the estuary. Nevertheless, because of the low salinities in the upper parts of the estuary, the differences between the observed and the predicted values are still relatively small. Another feature that was consistently observed in all 2-D model results is the poor model performance during the first three months of simulation (March through May), which was likely due to the model spin-up effect (see Discussion). During the rest of the study period (June through December), the 2-D model provided a good estimate of salinities across the Barataria estuary and outperformed the BOX model in all model runs. The TABS-2 model also provided good salinity estimates, although the performance of this model was difficult to assess due to the very limited model temporal domain.

The R^2 values (coefficient of determination) for the comparison between the measured and simulated salinities for BOX and 2-D models are given in Table 3.2. As explained previously, because of systematic model error due to the spin-up effect during the first three months of simulation (March to May 2002; Figures 3.1. to 3.6) the R^2 values (between BOX and

Table 3.1. Comparison of the BOX, TABS-2 and 2-D models.

Attribute	MODEL		
	BOX	TABS-2	2-D
General description	Simple, low resolution	Complex, finite element grid, medium resolution	Complex, finite difference grid, high resolution
Spatial scale	1-D, 6 elements	2-D depth averaged, 16,749 elements	2-D depth averaged with 1.3 million elements
Integration step	0.1 hour	1 hour	3 seconds
Temporal coverage	Annual	Weekly	Annual
Calibration periods	12 days; January 1-12, 2002	16 days; September 15-21, December 2-10, 2002	30 days; July 7 to August 5, 1999
Initiation of model	Computation is carried out only in water cells; marsh cells are used to estimate runoff	The entire model domain is initially submerged	There is always a clear delineation between land and water grids
Runoff considerations	Runoff into Lac des Allemands is included	There is no runoff into Lac des Allemands	Runoff into Lac des Allemands is included
Initial conditions and simulation intervals	Initial conditions set for January 1 st 2002; subsequently, the model was run for one year	Initial conditions set for September 15 th , 2002; subsequently, the model was run for one week Initial conditions set again for December 2 nd , 2002; subsequently, the model was run again for one week.	Initial conditions set for March 20 th , 2002; subsequently, the model was run until 12-31-2002

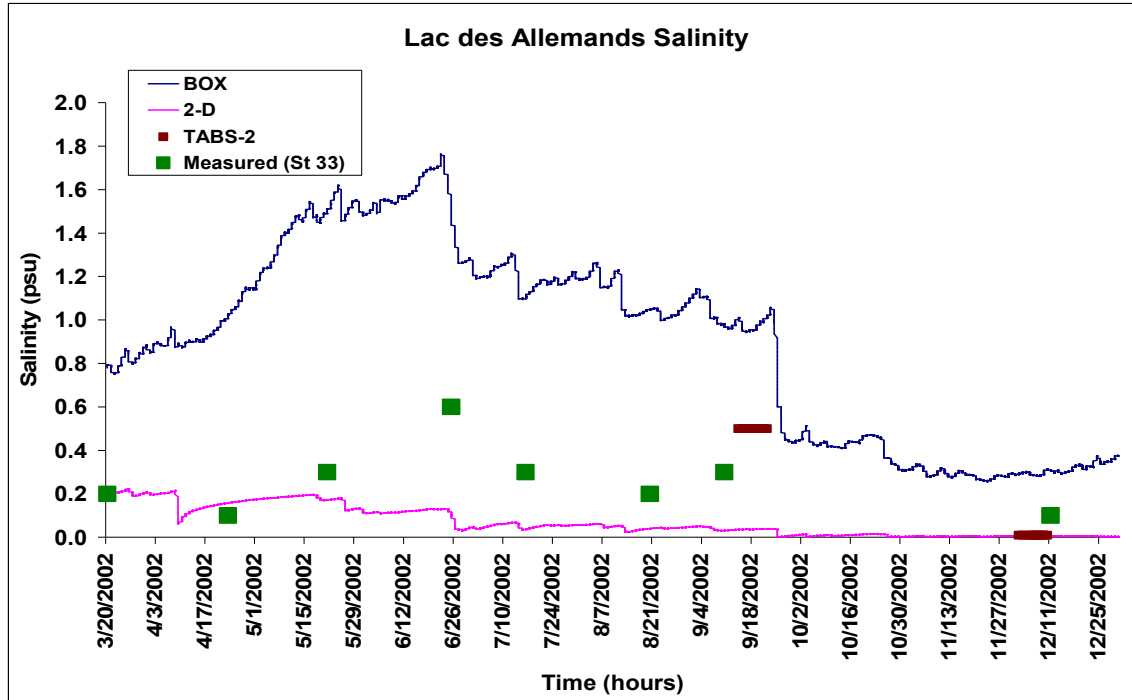


Figure 3.3. Time series of measured (Station 33) and simulated salinities in Lac des Allemands.

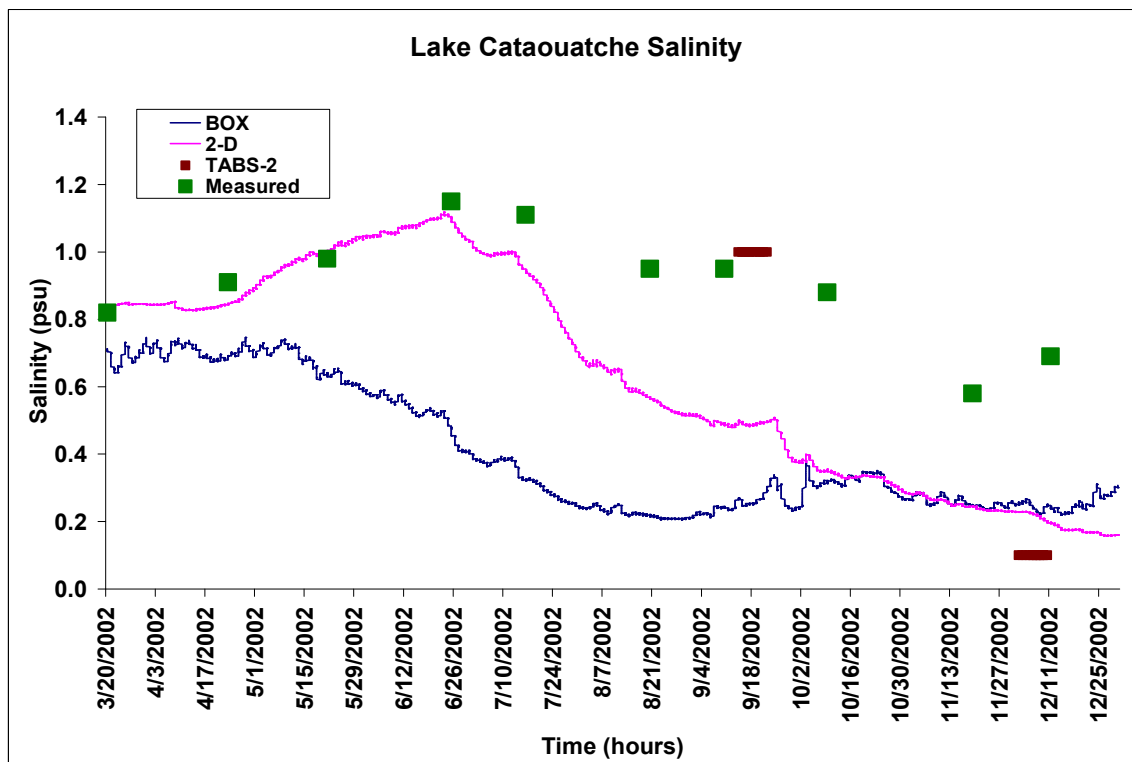


Figure 3.4. Time series of measured (average of USGS and LADNR sampling stations) and simulated salinities in Lake Cataouatche.

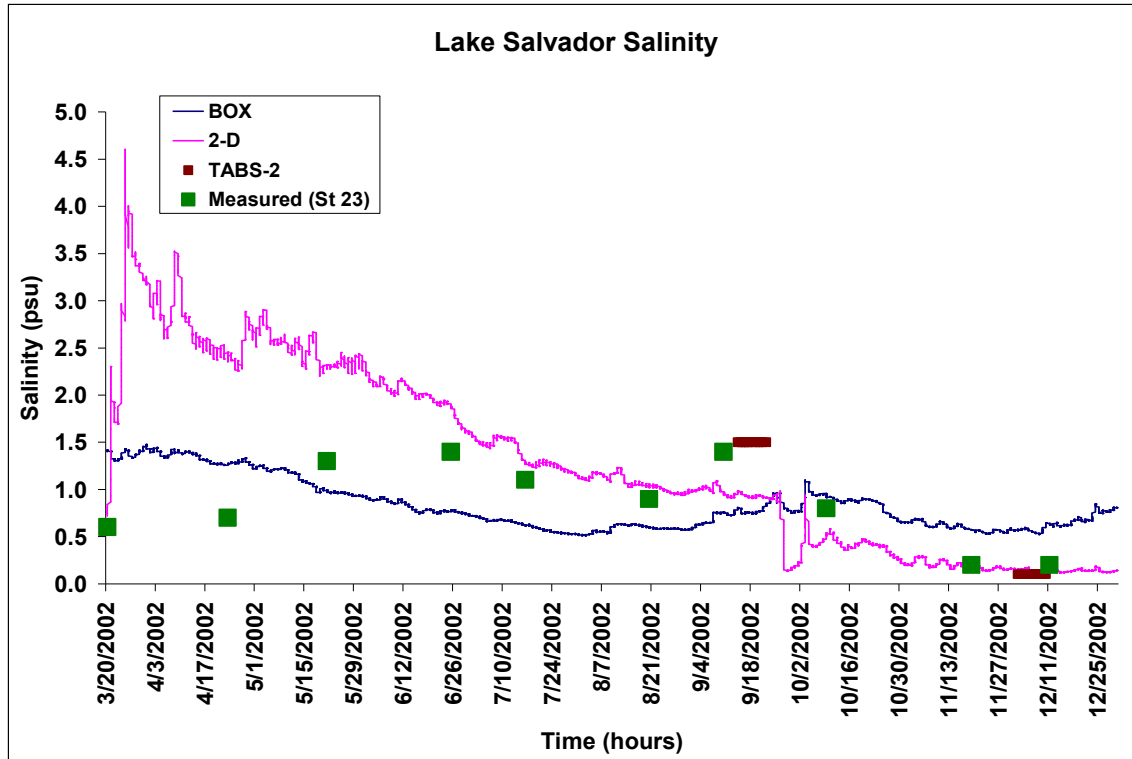


Figure 3.5. Time series of measured (Station 23) and simulated salinities in Lake Salvador.

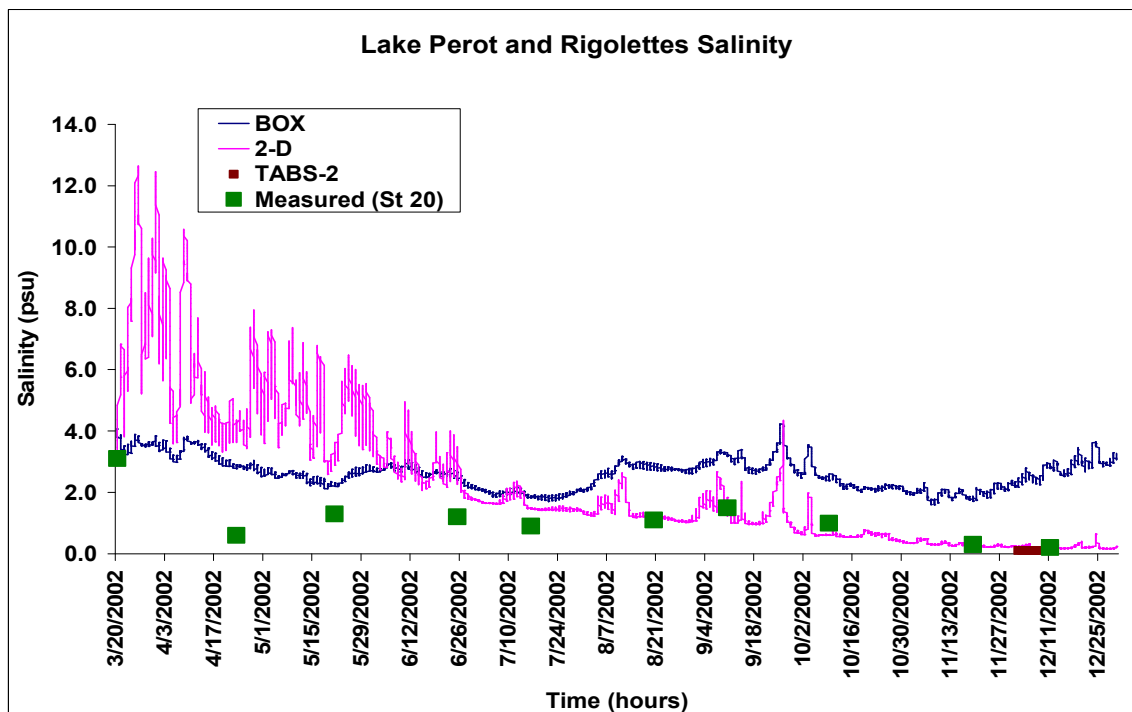


Figure 3.6. Time series of measured (Station 20) and simulated salinities in Lake Perot and Rigolettes.

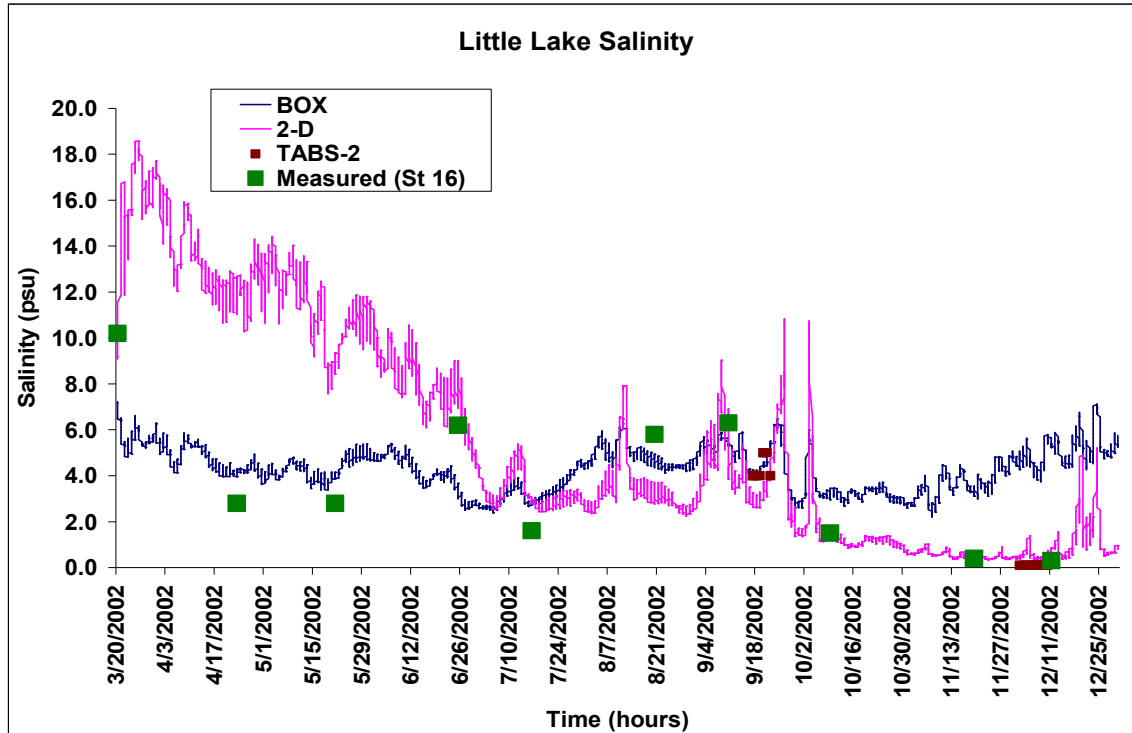


Figure 3.7. Time series of measured (Station 16) and simulated salinities in Little Lake.

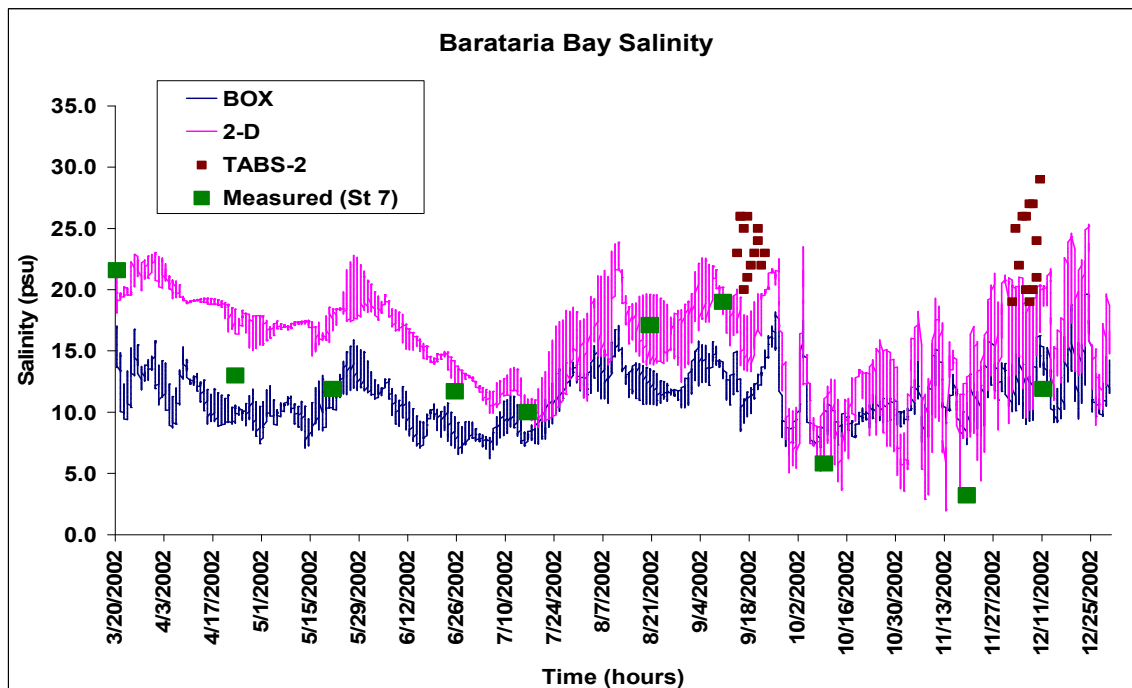


Figure 3.8. Time series of measured (Station 7) and simulated salinities in Barataria Bay.

measured and 2-D and measured) have been calculated only for the period from June to December 2002. The results show that the BOX model correctly predicts salinity only in the Barataria Bay and Lac des Allemands (Table 3.2). In contrast, the 2-D model predicts salinity quite well in all the major water bodies in the estuary. R^2 values could not be computed for the TABS-2 model because simulation periods did not coincide with data collection carried out in this study.

Table 3.2. Comparison of R^2 values for measured and simulated BOX and 2-D salinities in the major water bodies of the Barataria estuary, based on model output for June-December, 2002. The asterisk denotes an R^2 value that is significant at $p < 0.05$.

	BOX	2-D
Lac des Allemands	0.83*	0.94*
Lake Cataouatche	0.42	0.84*
Lake Salvador	0.17	0.77*
Lake Perot and Rigolettes	0.26	0.65*
Little Lake	0.16	0.75*
Barataria Bay	0.70*	0.74*

Comparison of Measured and Simulated Water Levels

A comparison between observed and predicted water levels (Figures 3.9 - 3.14) shows that both the BOX and the 2-D models reproduced the observed water level variations reasonably well. However, the 2-D model systematically over-predicted the observed water levels in all major water bodies in the estuary, which was likely due to the absence of wetting and drying capabilities in the version of the 2-D model that was used in the study (see Discussion).

Nevertheless, the 2-D model results showed the same dynamic features as the measured data, as indicated by the high R^2 values (Table 3.3).

The box model values for water levels are usually lower than the measured values. Interestingly, both BOX and 2-D models accurately predicted high amplitude non-tidal variations in sea level caused by tropical storm Isidore and hurricane Lili that propagated through the estuary with significantly less attenuation compared to tides, causing greater inundation in the upper reaches of the estuary (Figure 3.9-3.14). Because of the limited temporal domain, it was difficult to examine the accuracy of the TABS-2 model. It appeared that the model accurately predicted water levels in Lac des Allemands (Figure 3.9), Bayou Perot and Rigolettes (Figure 3.12), while underestimating water levels in for Lake Cataouatche (Figure 3.10), Lake Salvador (Figure 3.11) and Little Lake (Figure 3.11).

The R^2 values for the comparison between the measured and simulated water levels for BOX and 2-D models are given in Table 3.3. The 2-D model outperformed the box model five out of six major water bodies. Also, the predictions of the 2-D model are more accurate in the lower part of the estuary, compared to the upper part of the estuary.

Comparison of Measured and Simulated Water Fluxes

A comparison of simulated flux of water through the tidal passes predicted by the BOX and the 2-D models showed a good agreement between the two models (Figure 3.15). The BOX model results show greater overall variability when compared with the 2-D results. The average annual fluxes of water during the 2002 predicted by the BOX and the 2-D models were $6,944 \text{ m}^3 \text{ s}^{-1}$ and $6,951 \text{ m}^3 \text{ s}^{-1}$, respectively. There is a small difference, less than 7%, when the modeled fluxes are compared monthly, which is not a significant difference ($p > 0.05$) based on a paired t-test (Figure 3.16).

A comparison of the measured and modeled fluxes revealed that both models provided an

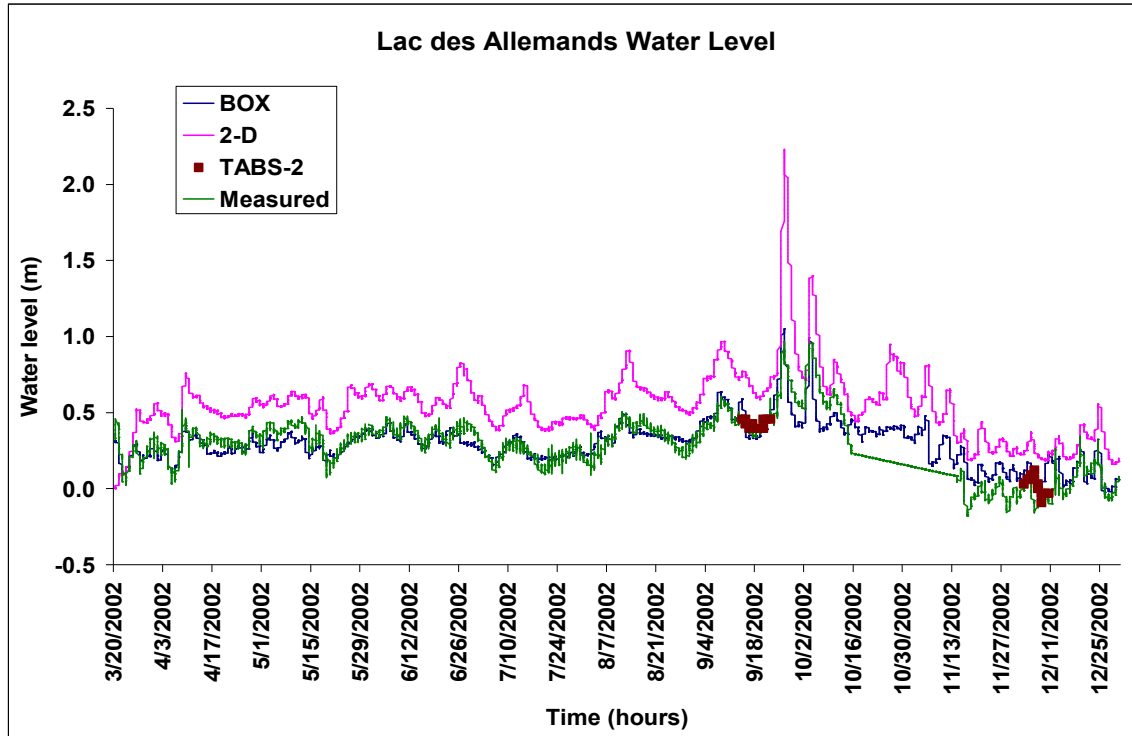


Figure 3.9. Time series of measured and simulated water levels in Lac des Allemands.

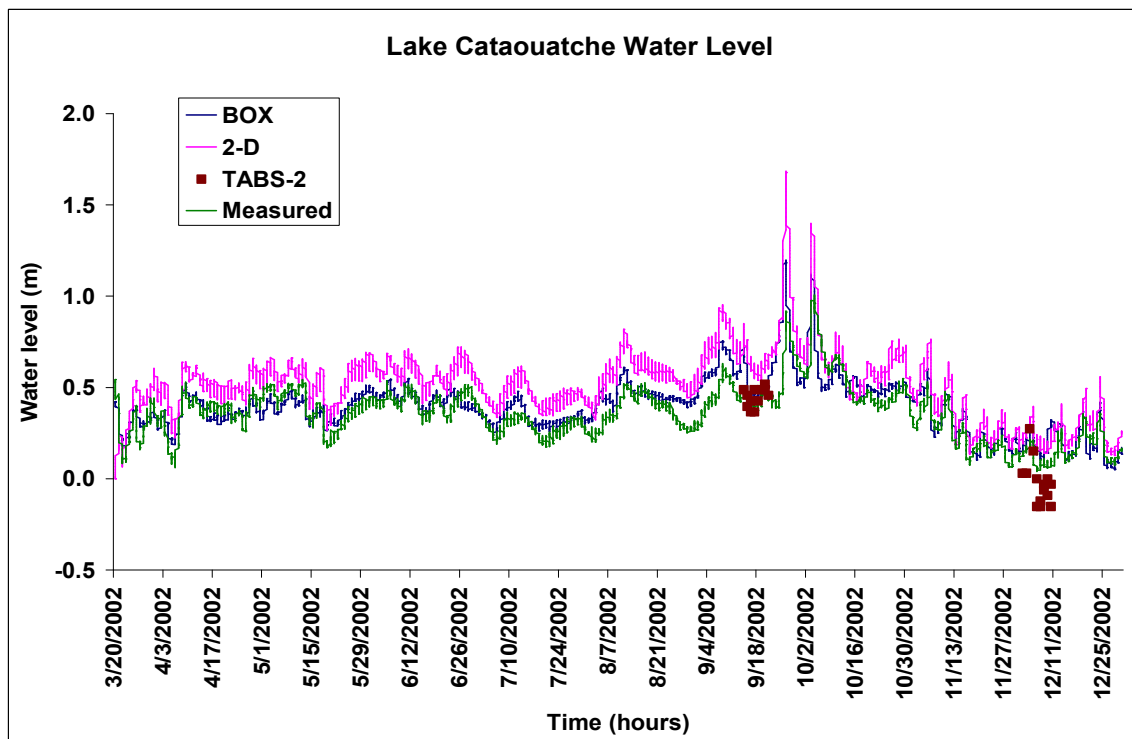


Figure 3.10. Time series of measured and simulated water levels in Lake Cataouatche.

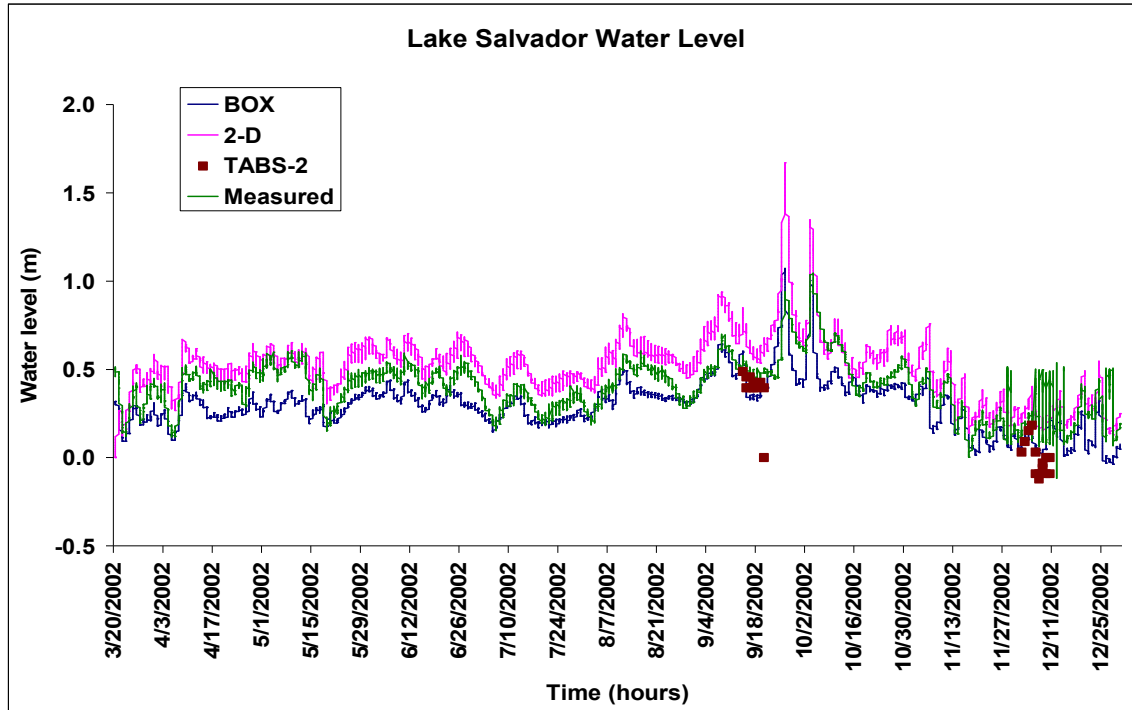


Figure 3.11. Time series of measured and simulated water levels in Lake Salvador.

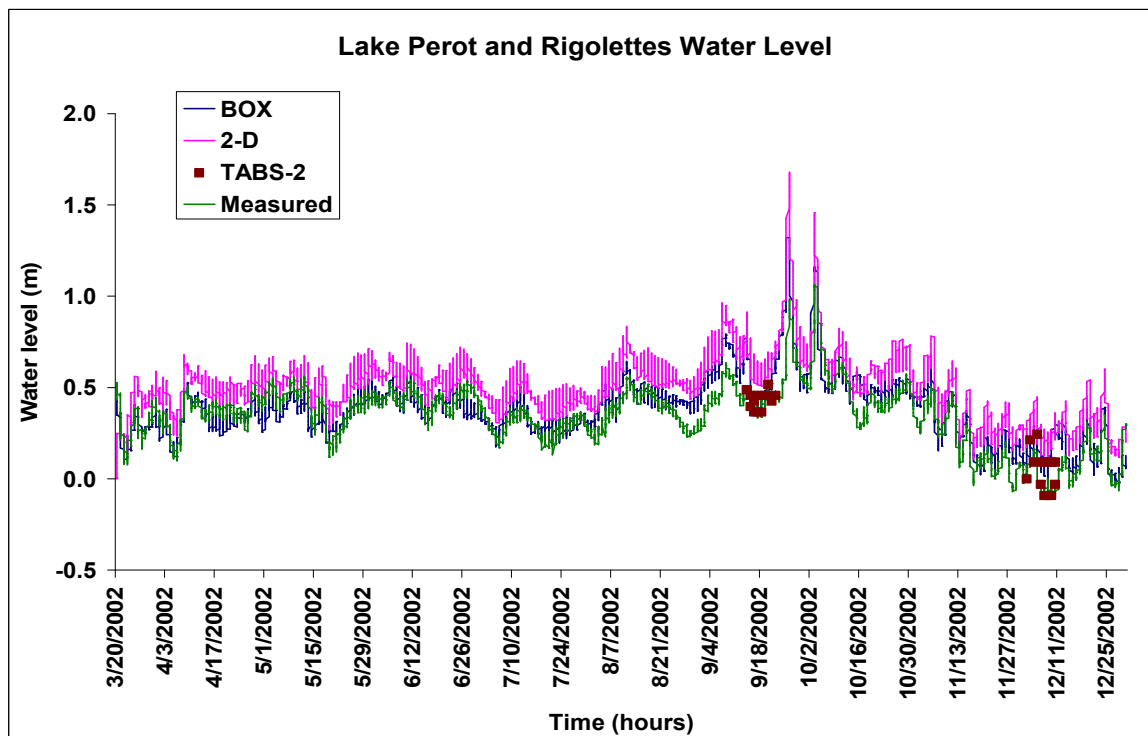


Figure 3.12. Time series of measured and simulated water levels in Lake Perot and Lake Rigolettes.

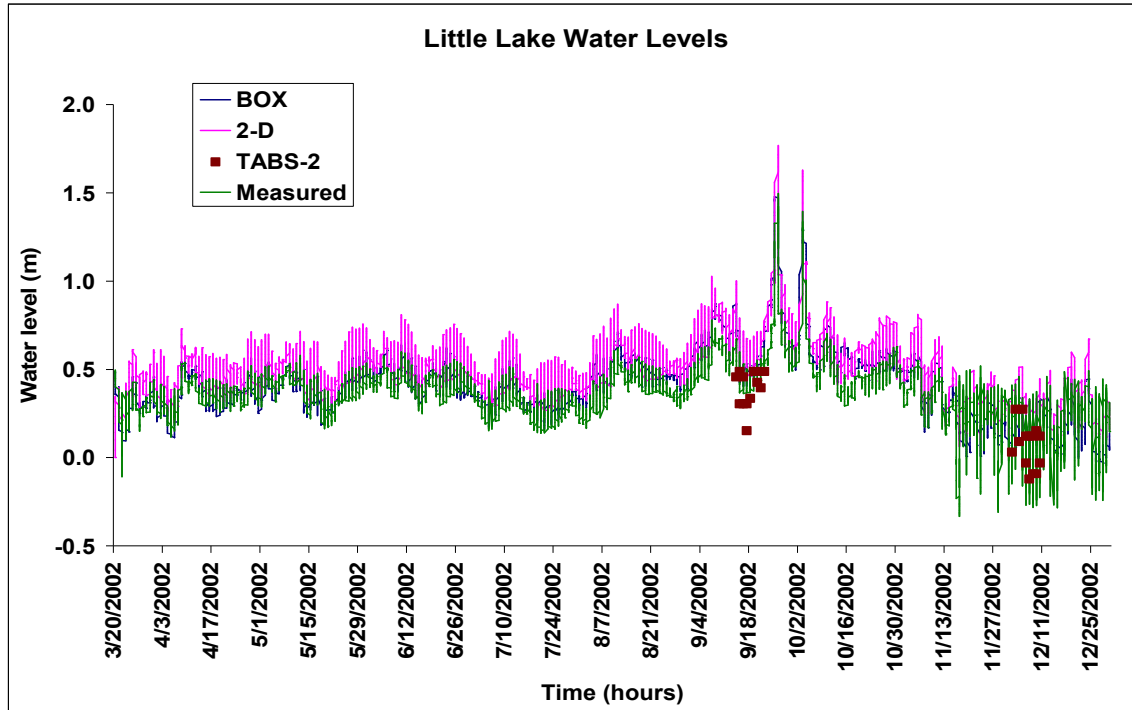


Figure 3.13. Time series of measured and simulated water levels in Little Lake.

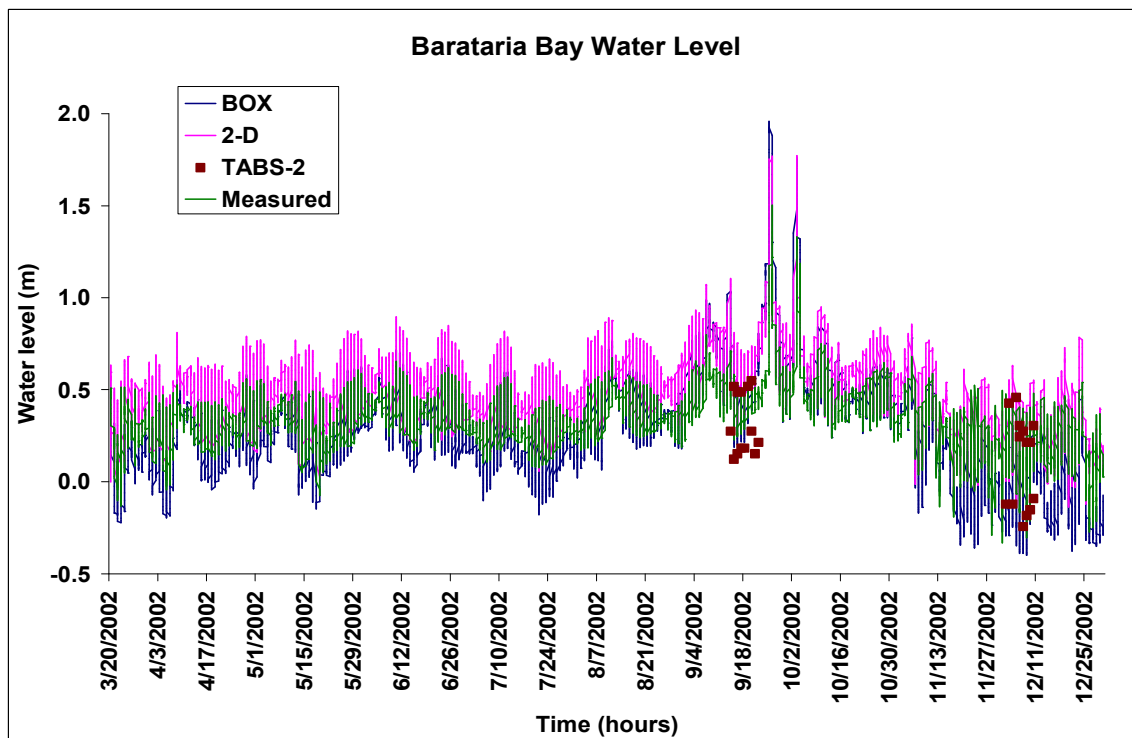


Figure 3.14. Time series of measured and simulated water levels in Barataria Bay.

Table 3.3. Comparison of R^2 values for measured and simulated BOX and 2-D water level estimates in the major water bodies of the Barataria estuary, based on model output for March-December, 2002. The asterisk denotes an R^2 value that is significant at $p < 0.05$.

	BOX	2-D
Lac des Allemands	0.71*	0.65*
Lake Cataouatche	0.71*	0.80*
Lake Salvador	0.67	0.74*
Lake Perot and Rigolettes	0.71*	0.81*
Little Lake	0.71*	0.83*
Barataria Bay	0.72*	0.81*

accurate representation of the flow through the tidal passes (Figure 3.15). Nevertheless, the BOX model results showed a tendency towards overestimating the flux during the periods of maximum tidal excursions.

The average measured flux values for October 24 – December 3, 2002, from the Moffatt and Nichol (2005) data was $6,609 \text{ m}^3\text{s}^{-1}$. The average fluxes predicted by the BOX and the 2-D models for the same period were $7,053 \text{ m}^3\text{s}^{-1}$ and $6,775 \text{ m}^3\text{s}^{-1}$, respectively. The differences between the measured and predicted flux averages (6.7% and 2.5%, for the BOX and the 2-D model, respectively) were deemed acceptable given the wide range of flow conditions in the tidal passes (Figure 3.17). The results of the flux of the different constituents show that the Barataria estuary annually exports $109 \times 10^6 \text{ kg}$ TOC ($6.6 \times 10^6 \text{ POC}$ and $102 \times 10^6 \text{ DOC}$) and 0.3×10^6 chlorophyll *a* to the coastal Gulf of Mexico, while importing $7 \times 10^6 \text{ kg}$ nitrate.

DISCUSSION

Model calibration and validation are critical to the process of model development. Validation is done to ascertain whether the model is useful in providing accurate information of

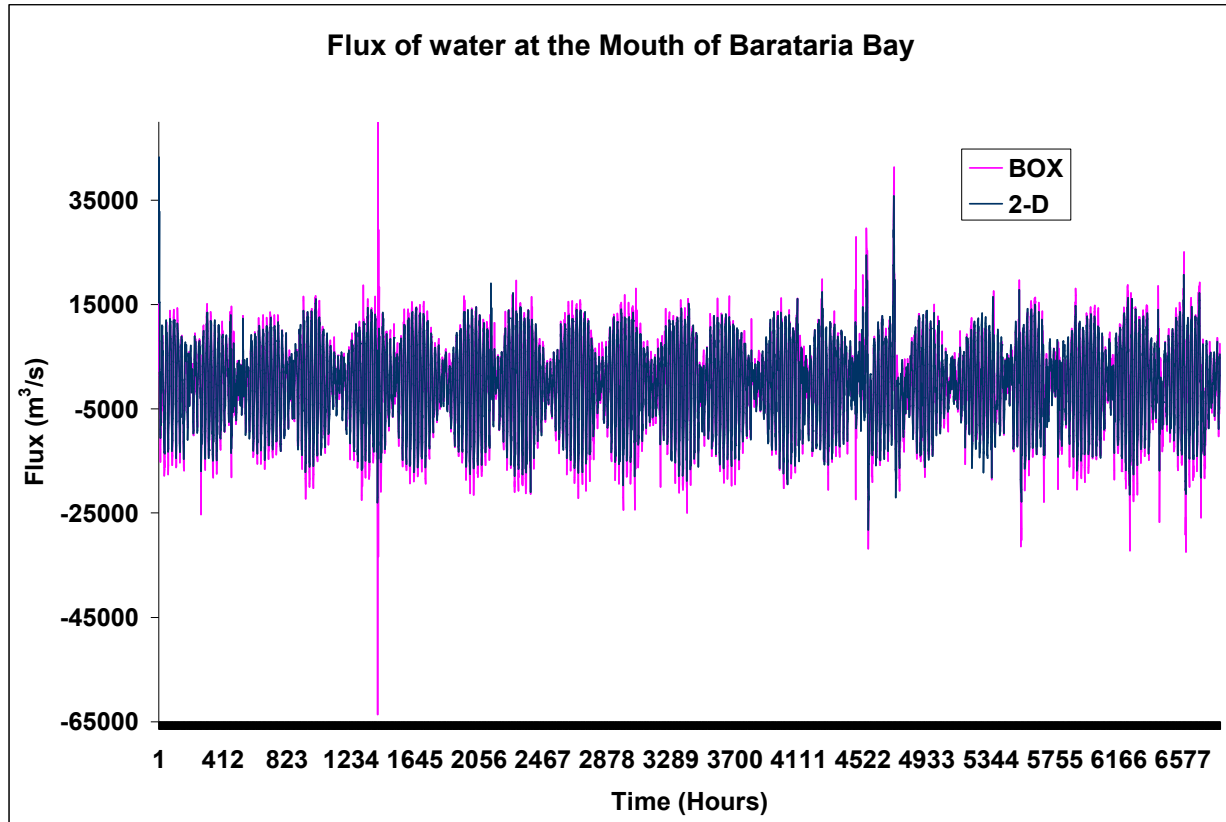


Figure 3.15. Simulated flux of water, using the BOX model and the 2-D model, at the ocean-estuary interface in the Barataria Bay, from March 20, 2002 to December 29, 2002.

the system being modeled. The model skill, or the “goodness of fit”, can be judged by two methods – visual and statistical (Jain and Sudheer, 2008). Two common methods of statistically determining the goodness of fit for models are the coefficient of determination (R^2) and the Nash-Sutcliffe efficiency index (Nash and Sutcliffe, 1970). Both methods have their advantages and limitations. The coefficient of determination is generally an appropriate estimate of the goodness of fit for linear models, but can be a poor measure if the model is biased (McCuen et al., 2006). Model bias occurs due to systematic error variation. A model with a positive bias is one where values are over-predicted, and a model with a negative bias has values that are under-predicted. Similarly, the Nash-Sutcliffe efficiency index is very sensitive to model bias (McCuen et al., 2006). For example, the index can be zero for a negative or positive bias of 40%. The Nash-Sutcliffe efficiency index is also influenced by sample size, presence of data outliers,

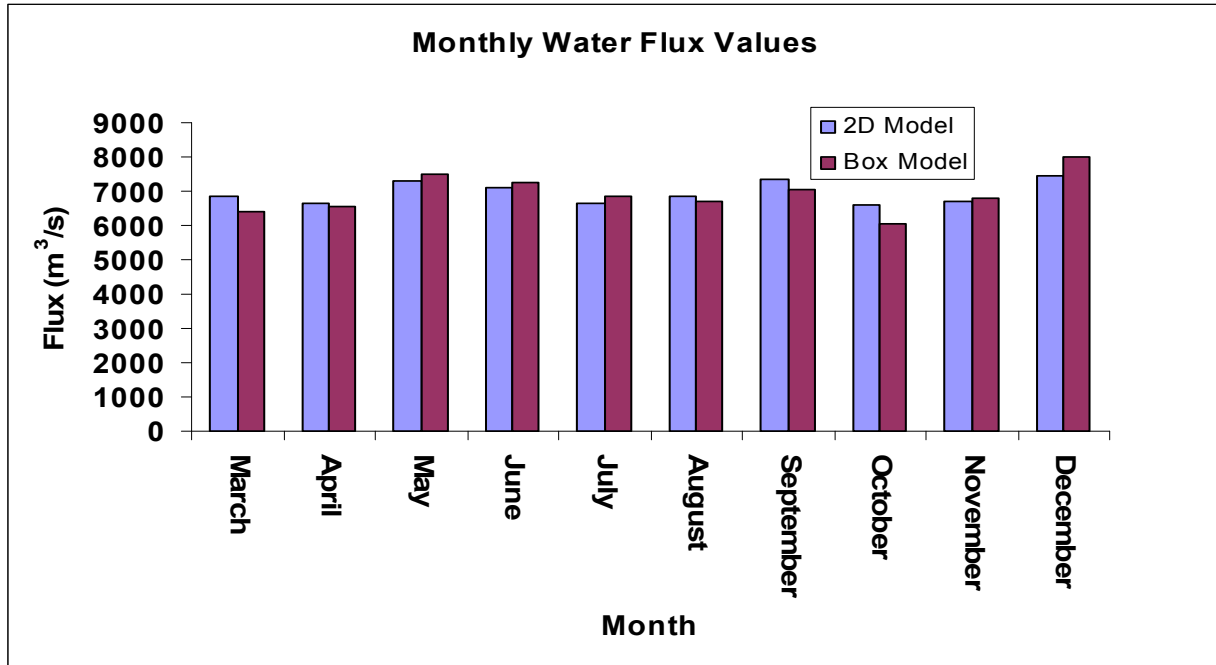


Figure 3.16. Average simulated monthly water flux values at the ocean-estuary interface in Barataria estuary. The flux differences between the two models are not significant based on the paired sample t-test ($p < 0.05$).

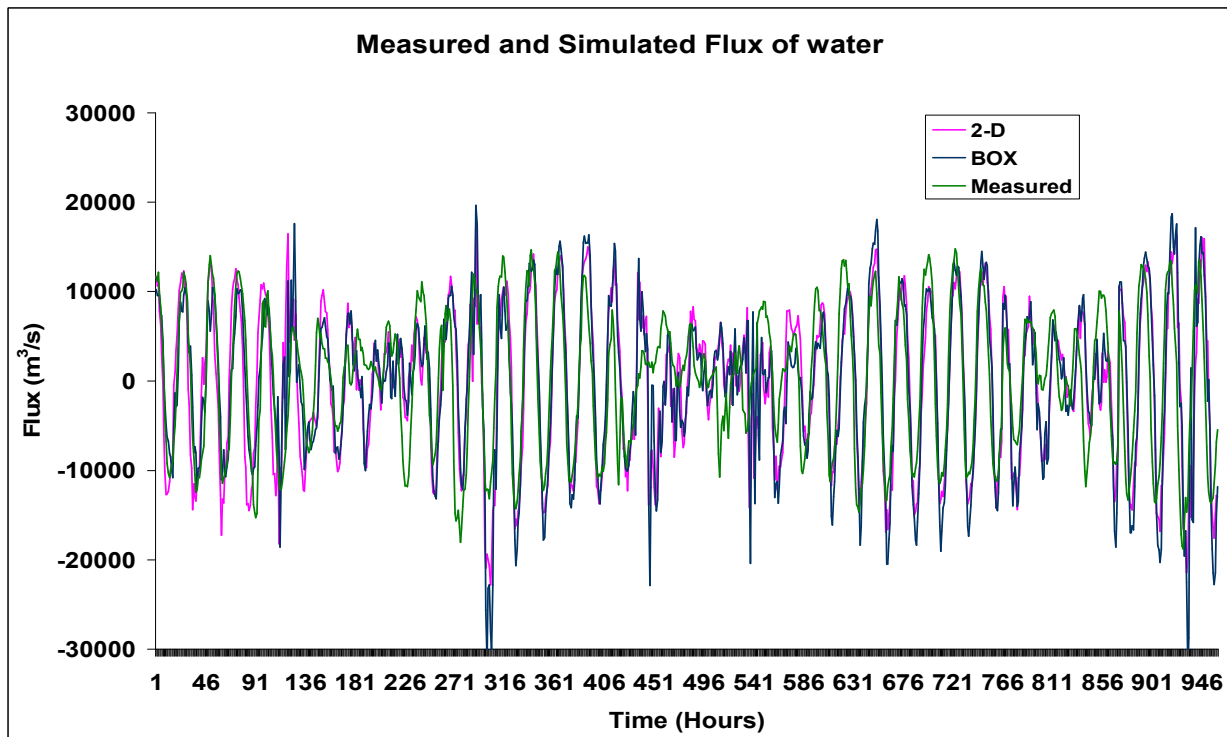


Figure 3.17. Measured and simulated flux of water from October 24, 2002 to December 3, 2002, at the ocean-estuary interface in the Barataria estuary.

Table 3.4. Estimates of fluxes of water (Q), nitrate, total organic carbon (TOC), particulate organic carbon (POC), dissolved organic carbon (DOC), and chlorophyll *a* (Chl *a*) for the lower Mississippi River (MR) and the Barataria estuary (BE). The Barataria estuary fluxes are calculated from the 2-D model. Negative sign denotes that a constituent is exported from the estuary; a – Turner et al. 2007; b – Bianchi et al. 2007; c – based on the POC:DOC ratio of 12% in the Mississippi River from April 2009 to May 2010 (Turner et al., unpublished data); d – based on the Chl *a* value of 3.9 mg m⁻³ Apr 2009 to May 2010 (Turner et al., unpublished data); e - this study.

Constituent	Mississippi River	Barataria estuary	BE : MR (%)
Q (m ³ s ⁻¹)	16,000 ^a	6,951 ^e	43.7
Nitrate (10 ⁶ kg N yr ⁻¹)	724 ^a	7 ^e	1.0
TOC (10 ⁶ kg yr ⁻¹)	4,000 ^b	- 109 ^e	2.7
POC (10 ⁶ kg yr ⁻¹)	480 ^c	- 7.5 ^e	1.5
DOC (10 ⁶ kg yr ⁻¹)	3,520 ^c	- 94 ^e	2.7
Chl <i>a</i> (10 ⁶ kg yr ⁻¹)	2 ^d	- 0.3 ^e	18.4

magnitude bias, and the time-offset bias, and these limitations may lead to rejection of a good model (McCuen et al., 2006). Also, the index can indicate a high goodness of fit in case of poor models (Jain and Sudheer, 2008).

I initially used both the coefficient of determination and the Nash-Sutcliffe efficiency index to examine the accuracy of the various models used in this study. However, I found out that the Nash-Sutcliffe efficiency index values were highly variable and often contradictory to my visual judgment. Thus, I decided to use only the coefficient of determination to evaluate model fit.

The 2-D model is superior to the BOX model for simulated salinity and water levels in most model runs. The performance of the TABS-2 model was difficult to assess due to the

limited model temporal domain. Nevertheless, I noted that the 2-D model has consistently showed poor performance during the first three months of simulation (March through May). This is likely due to the model spin-up, i.e., the response of the model to imposed initial conditions at the beginning of the model run (e.g. Figures 3.18). To initiate the model, salinities identical to those observed at the transect stations were imposed as longitudinal bands across the main axis of the estuary. The Barataria estuary is large and has a very complex morphology, so the number of monitoring stations used to initialize the model (Figure 3.1 and 3.2) was not sufficient to fully describe the spatial salinity distribution at the beginning of the model run. Given the high water residence times in the Barataria estuary (Chapter 5), the 3-month model spin-up time is not surprising.

Also, the 2-D model overestimated water levels in all major water bodies of the Barataria estuary. This was likely due to the absence of wetting and drying capabilities in the version of the 2-D model that was used in the study. Thus, a fraction of the tidal prism volume that would normally flood the marshes is retained within the model domain leading to unrealistically high water levels.

The flux of a material could be measured by direct and indirect processes. Direct processes are based on using direct water flux measurements and multiplying the concentration of the material with the flow of water (Kjerfve, 1975, Kjerfve and Proehl, 1979). This process is data and labor intensive. Indirect methods are based on obtaining a material mass-balance for the entire estuary, and then estimating the flux to obtain satisfactory budget closure (Kjerfve et al., 1991; Rahm and Wulff, 1992; Vorosmarty and Loder, 1994; Dame and Allen, 1996; Childers et al., 2000). This approach requires less effort but could give incorrect estimates. In this study, I used a modification of the direct method, in which the simulated flux of water was multiplied by the measured concentrations to obtain the constituent flux.

As far as the flux computations are concerned, it appeared that both the BOX and the 2-D models provided accurate estimates of the exchanges of water between the estuary and the coastal ocean. Thus, the hypothesis that the Barataria estuary exports carbon (outwelling) and imports nitrogen which was originally proposed using a BOX model (Chapter 2; Das et al., 2010) has now being reaffirmed by a more complex mechanistic 2-D model (Table 3.4). As stated in Chapter 2, the outwelling of the total organic carbon (TOC) is small compared to the Mississippi River TOC load (2.7%) and likely has little impact on coastal carbon budgets and development of the Gulf's hypoxia.

Although the BOX and 2-D models are useful to estimate salinities, water levels and fluxes in the Barataria estuary, there are numerous ways to improve these models. For example, I used rainfall measurements from a single monitoring station at the New Orleans airport. Those records cannot accurately capture variability in the actual amounts of rain falling over different portions of the Barataria Basin, especially during local storms that are frequent in summer. Further, the Gulf Intracoastal Water Way (GIWW) can bring in substantial amounts of water from the Fourleague Bay into the Barataria estuary causing an increase in freshwater flow from upstream to downstream (into the Bay). This happens because of a difference in freshwater levels and saline water levels (coming in from the Gulf). There are no long-term measured records of the water inflow from the GIWW, so it has not been incorporated into the BOX model. The 2-D model assumes a constant inflow from the GIWW of $50 \text{ m}^3\text{s}^{-1}$ (E. Swenson, personal communication) which is comparable to the average measured flow between 1996 and 1999 ($60 \text{ m}^3\text{s}^{-1}$; Swarzenski, 2003). In addition, local wind variations in space can also cause differences in water levels between the upstream and downstream sections in the estuary, which cannot be captured using the wind data from a single monitoring station at Grand Isle.

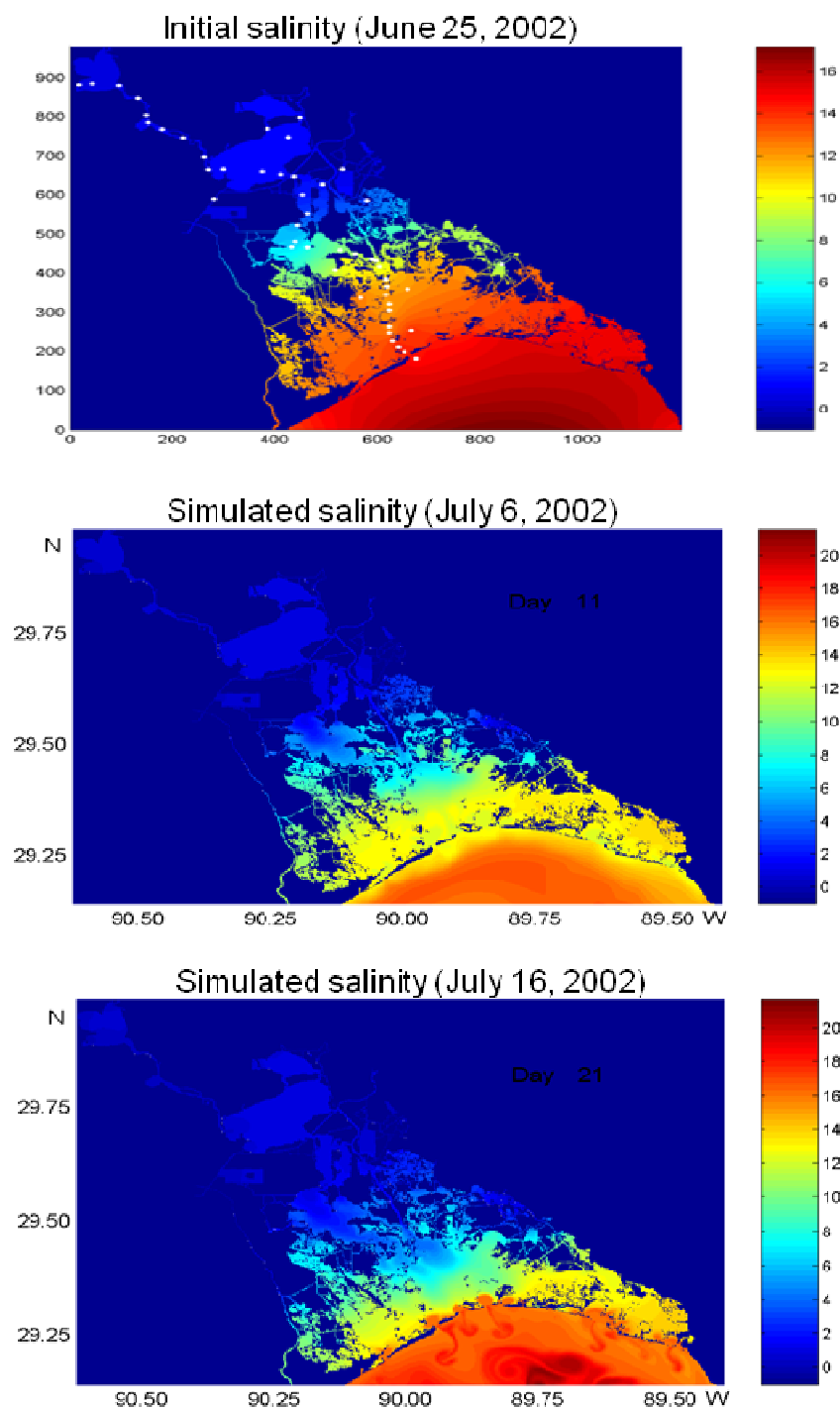


Figure 3.18. Upper panel – the model domain with sampling stations and superimposed initial salinity values observed on June 25, 2002. Middle panel – simulated salinity values on July 6, 2002. Lower panel – simulated salinity values on July 16, 2002.

Habib et al. (2008) applied a monthly salinity box model to the Barataria estuary and concluded that paucity of rain gauges (about 1.3 gauges /1000 km² in the U.S.) add to the high degree of uncertainty in predicting salinity. This is especially true for estuaries such as the Barataria estuary (only 5 rain gauges in the Barataria estuary), where rainfall and the associated runoff determine the freshwater budget of the estuary. Rainfall in this region is highly variable in time and space, and using just one rain gauge for the whole estuary will result in incorrect simulations. The uncertainty is greater inland (because the influence of the surrounding drainage basin is greater in the upper basin) than at the mouth of the estuary where salinity values are influenced by the highly variable Mississippi River discharges. Habib et al. (2008) also pointed out the influence of rainfall uncertainty on parameter estimations during model calibration, a fact that I also observed for the salinity comparisons carried out in this study.

In my model simulations, the salinity measurements from the salinity monitoring station at Grand Terre were used to force the models at the mouth of the Bay. This is the only station with continuous salinity measurements in the vicinity of the Barataria Bay passes. The continuous salinity measurements have a resolution of 1 psu, which is too coarse to force a complex model like the 2-D model. Also, the location of this particular station at Grand Isle is also problematic. The station is located in a sheltered area of the island and generally has higher salinity compared to that in the open water areas at the mouth of the Barataria estuary (and so is not a good representation of salinity at the mouth of the Bay), which is artificially increasing the salt content in the model domain during lengthy model simulations.

Ecosystem management often requires models that can predict both long term as well as short term changes in a system of interest. This study has shown that both simple and complex numerical models can be useful in this regard. Simple models provide only a coarse spatial representation, but they are very fast and therefore suitable for assessing changes on annual and

decadal time scales (Humborg, 2000). Complex models, on the other hand, are good for predicting changes over short term periods because they include detailed description of short term processes and therefore are better suited for the short term forecasts. Complex models are also useful for predicting changes over large spatial scales. However, these antecedent models require ample time and large computational capabilities, which make them generally unsuitable for long term simulations. For example, each of the model runs discussed in this section required about 72 hours of supercomputer time. This puts severe limitations on the number of model runs that can be carried out during the calibration and validation process.

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CHAPTER 4

IMPACT OF PULSED RIVERINE INFLOWS ON SALINITY IN A DELTAIC ESTUARY

INTRODUCTION

Estuaries maintain their transitional nature because they receive both fresh water and seawater. A change in the amount of fresh water entering an estuary, such as the one resulting from freshwater diversions, affects salinity, temperature, stratification, nutrient concentrations and turbidity, all of which profoundly influence estuarine biological communities (Rozas et al., 2005; Nuttle et al., 2008). For example, by altering the ambient water temperatures, freshwater diversions directly affect temperature-sensitive biological and geochemical processes in an estuary (Reddy and Patrick, 1984). The water column can become less stratified with increasing freshwater flow, as in the case of the Hudson River estuary (Howarth et al., 2000). It can also become more stratified, as in the case of the northern Gulf of Mexico, where Mississippi River inflow increases the vertical density gradients (Justić et al., 1996). Stratification can foster phytoplankton blooms by increasing residence times (Howarth et al., 2000; Rabalais et al., 2007). It can also slow or prevent oxygen transport to the bottom waters and cause hypoxic or anoxic conditions (Justić et al., 1996; Rabalais et al., 2007).

Riverine inflows typically have higher nutrient concentrations and different nutrient ratios compared to estuarine and coastal waters, both of which are important in controlling estuarine eutrophication (Ren et al., 2009). On the other hand, river water coming in through the diversions is more turbid, which can be limiting for phytoplankton growth (Cloern 1987; Harding, 1994; Valdes-Weaver et al., 2006). Increased freshwater inflow decreases estuarine water residence times, that, in turn, affects the overall nutrient uptake within the estuary proper (Nixon et al., 1996; Dettman 2001). Suspended sediments help sustain vertical accretion, and

nutrients bound to these sediments may further promote vertical accretion via formation of organic soil from wetland plant production (DeLaune et al., 1983).

MISSISSIPPI RIVER DIVERSIONS

Viosca (1927) was one of the first to point out that leveeing of the Mississippi River will have detrimental effects to the ecology and economy of Louisiana with respect to agriculture and fisheries. Other processes, such as channelization and spoil bank construction have also affected estuarine hydrology and salinity regimes and contributed to wetland loss (Turner and Streever, 2002).

The idea of diverting the Mississippi River water back into the estuaries and coastal wetlands was proposed over three decades ago (Chatry et al., 1983). The primary motivation at the time was to prevent increased salinities in the oyster producing regions of the estuary. The use of fresh water diversions was later expanded to include marsh nourishment via riverine sediment and nutrient additions (Gagliano et al., 1971; Day and Templet, 1989), and nutrient control in the Mississippi River (Mitsch et al., 2001). Some scientists and managers agree that controlled diversions of the river water back into coastal wetlands are important mechanisms that could reverse coastal land loss (e.g., Boesch et al., 1994). There are concerns, however, that diversions may increase nutrient inputs and thus create eutrophication problems in estuaries and wetlands adjacent to the diversion sites. For example, the opening of the Bonnet Carré spillway during the spring of 1997 was associated with a major bloom of cyanobacteria in Lake Pontchartrain (Dortch et al., 1999). High levels of hepatotoxins that were measured during the peak of the bloom resulted in a health advisory for recreational use of the lake. On a larger scale, high nutrient concentrations in the Mississippi and Atchafalaya Rivers have led to eutrophication in stratified coastal waters of the northern Gulf of Mexico, where a seasonally severe hypoxic zone has persisted for over 23 years (Justić et al., 2002; Rabalais et al., 2002, 2007). The

concerns of increased nutrient inputs are not limited to the water column. High nutrient concentrations could also adversely affect marsh plants by causing faster soil decomposition (Bragazza et al., 2006; Swarzenski et al., 2008), lower soil strength (Swarzenski et al., 2008; Turner et al., 2009) and lowered accumulation of belowground biomass (Darby and Turner, 2008a, b).

There are two major controlled freshwater diversions in Louisiana, Caernarvon and Davis Pond. The Davis Pond diversion structure (Chapter 1, Figure 1.2) is located in St. Charles Parish, Louisiana, on the west bank of the Mississippi river. With a maximum capacity of $300 \text{ m}^3\text{s}^{-1}$ (10,500 cfs) it is the largest fresh water diversion in the world (LaCoast, 2002). The primary purpose of Davis Pond diversion was salinity control in the Barataria estuary (Allison and Meselhe, 2010). Salinity intrusions from the Gulf of Mexico have pushed the brackish, intermediate and fresh water zones further up the estuary. Low freshwater inflow conditions can push back the 5 psu isohaline over 20 km in the estuary (Swenson, 2003). The fresh water discharge from Davis Pond is expected to move the pre-project 5 psu isohaline about 20 km southward and the pre-project 15 psu isohaline 12 km southward (Figure 4.1). According to the study by the US Army Corps of Engineers (USACE, 2004), this should increase the production of seafood (oysters, shrimps, crabs, etc.) as well as provide a more favorable habitat for other animals and birds.

Mississippi River diversions are considered to be an important part of future restoration plans for coastal Louisiana (Reed and Wilson, 2004). With the creation of the Coastal Wetlands Planning, Protection and Restoration Act (CWPPRA) in 1997, a plan was developed to address problems along the Louisiana coast with the help of people in academia, industry, and local, state and federal agencies (Reed and Wilson, 2004). This produced the “Coast 2050: Toward a Sustainable Louisiana” report in 1998 (LCWCRTF, 1998) as part of a restoration strategy that

encompasses both large and small scale projects that deal with coastal restoration. This report differed from previous efforts at restoration of coastal Louisiana by focusing on meeting strategic goals rather than listing projects, and focusing on problems on a regional scale (Reed and Wilson, 2004). The aim of this project was not a return to previous landscape conditions, but rather to sustain a coast that has social, cultural and economical features desired within a natural system (Reed and Wilson, 2004). The LCWCRTF (1998) plan had three broad goals to address the land loss: (1) to ensure vertical accretion, (2) to maintain a salinity gradient in estuaries such that a maximum diversity of habitats can be maintained, and, (3) to maintain linkages and interfaces between habitats to ensure flow of materials and energy (Reed and Wilson, 2004). This plan further evolved into the Louisiana Coastal Area (LCA) reconnaissance report. The strategies presented in that report formed the basis of the final LCA Plan (LCA, 2004). In this plan, the long-term and short-term recommendations for coastal restoration in the Barataria estuary (Table 4.1) are two diversions with a maximum discharge rate of $28 \text{ m}^3\text{s}^{-1}$ (1000 cfs) from the Mississippi River at Lac des Allemands and Edgard into Lac des Allemands, the Davis Pond diversion operating at half its maximum capacity ($142 \text{ m}^3\text{s}^{-1}$ or 5000 cfs) and another diversion with a maximum discharge rate of $142 \text{ m}^3\text{s}^{-1}$ (5000 cfs) at Myrtle Grove (Figure 4.2).

In this chapter, I discuss the impacts of freshwater inflows on salinity for a variety of observed and hypothetical discharge scenarios. I used the 2-D model in these simulations because, as demonstrated in Chapter 3, it outperformed the BOX model in the case of both salinity and water level predictions.

MODEL SIMULATION SCENARIOS

The different diversion scenarios were chosen such that they are representative of viable

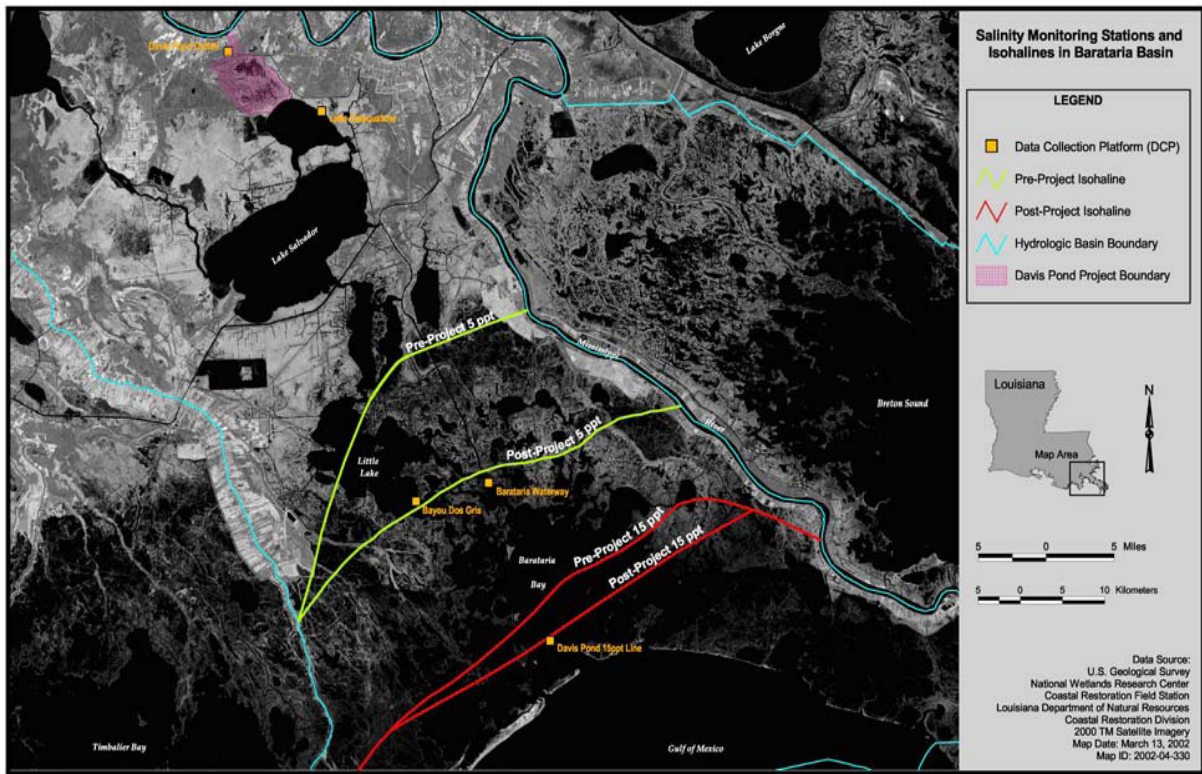


Figure 4.1. Barataria estuary showing pre- and post-Davis Pond construction isohalines (Source: Davis Pond Freshwater Diversion Annual Report, 2005).

Table 4.1. The diversions proposed for the Barataria estuary in the LCA plan (LCA, 2004).

Diversion	Discharge rate (m^3s^{-1})
Lac des Allemands	28
Edgard	28
Davis Pond	142
Myrtle Grove	142

management options and actual discharges. The different scenarios are described in Table 4.2.

Where applicable, simulated salinity values for different model scenarios are compared with the measured salinities from the Barataria transect (Figure 4.2, Turner et al., unpublished data). The

purpose of the different hypothetical scenarios (Table 4.2) is to examine how the introduction of different amounts of fresh water affects salinity in the Barataria estuary. The standard simulation uses the actual Davis Pond discharge data for 2002, when the diversion started operating. The simulation without any diversion should provide an idea about background salinities in the Barataria estuary without any freshwater input, other than the siphons at Naomi and West Pointe a la Hache, and the Gulf Intracoastal Waterway. Davis Pond discharge running at maximum capacity should give an idea about the lowest salinities that can occur due to the Davis Pond diversion. The 2008 Davis Pond scenario is representative of a fully operational Davis Pond diversion. Finally, the “Coast 2050” (Table 4.2) scenario should provide an idea about conditions that may be encountered in the future with several new diversions that have been recommended by the LCA final report (LCA, 2004).

The offshore salinity data used to force the 2-D model were obtained from the station at Grand Terre at the mouth of the Barataria Bay (Figure 3.3, Chapter 3). The 2-D model was initialized on March 20, 2002, exactly on the day the March monthly Barataria transect was occupied, to ensure that initial conditions match the measured salinity values. Simulated spatial salinity plots (May-December, Figures 4.4 to 4.11) and simulated salinity profiles along the Barataria transect (April-December, Figure 4.12-4.20) were compared for the same days when the sampling was conducted along the Barataria transect.

MODEL RESULTS

Because the Davis Pond diversion started operating in July 2002 (Fig. 4.3), there was no difference in the salinity distributions between the “Standard” run and the “Davis off” scenario during the April-August period. During 2008, the Davis Pond discharge was much higher compared to 2002 (Figure 4.3), so simulated salinity values for the “Davis 2008” scenario were

Table 4.2. Simulated freshwater diversion scenarios.

Scenarios	Description
Scenario 1 Standard	The 2002 Davis Pond discharge (Figure 4.3)
Scenario 2 Davis off	The Davis Pond discharge is set to zero
Scenario 3 Davis max	Davis Pond diversion discharge set to the maximum capacity ($300 \text{ m}^3\text{s}^{-1}$) whenever the Mississippi River stage in Baton Rouge exceeded 4m (Figure 4.3)
Scenario 4 Davis 2008	The 2008 Davis Pond discharge (Figure 4.3)
Scenario 5 Coast 2050	The Davis Pond diversion is set to discharge at half its maximum capacity ($142 \text{ m}^3\text{s}^{-1}$). There are two additional diversions into Lac des Allemands (at Lac des Allemands and Edgard, each $28 \text{ m}^3\text{s}^{-1}$), and one diversion at Myrtle Grove ($142 \text{ m}^3\text{s}^{-1}$). All these diversions are operational whenever the Mississippi River stage in Baton Rouge is above 4m (Figure 4.3)

consistently lower compared to the “Standard” and “Davis off” scenarios. For the “Coast 2050” scenario, the predicted salinity values (Figures 4.12-4.20) were lower compared to the “Davis 2008” scenario, except for August and September (Figures 4.16 and 4.17). This is because the Davis Pond discharge during August and September 2008 was higher than the recommended discharge for “Coast 2050” (Figure 4.3).

The predicted salinity values are the highest for the “Davis off” scenario and the lowest for the “Davis max” scenario (Figure 4.4-4.20). Interestingly, the predicted salinity differences between the “Standard” and “Davis max” scenarios being as high as 10 psu in some months (Figures 4.13 - 4.14 and 4.17 - 4.18). Salinity distributions for the “Davis 2008” scenario and the “Coast 2050” scenario are very similar (Figures 4.12 to 4.20, Table 4.3).

A Duncan's multiple range test was performed to compare the impacts of different discharge scenarios on salinity at stations along the Barataria transect (Table 4.3). This test controls the comparisonwise Type 1 error rate and so is considered more powerful in finding differences among treatments (or scenarios, in this case) compared to other tests that control experimentwise Type 1 error rates (Kemp, 1975). The Duncan grouping for each transect station is denoted by letters. The numbers represent the mean annual salinity for a particular transect station (Table 4.3). The results show that salinities in the middle and lower sections of the Barataria estuary are most impacted by the investigated freshwater diversion scenarios. In contrast, salinities at stations 1-7, station 24, and 27-34 are not significantly affected. There is a 3-5 psu difference in mean annual salinities at these transect stations for the different scenarios. Figure 4.21 shows the differences in mean annual salinities between the different model scenarios with salinities from the "Davis off" scenario. "Davis max" salinity differences are significantly different from "Davis off" from stations 8 to 23. "Davis 2008" salinities are consistently lower than "Davis off" but significantly different only at stations 21, 22 and 23. Similarly, salinities for "Coast 2050" are consistently lower than the "Davis off" salinities but the difference is significant only at stations 17, 18, 19, 21, 22 and 23. There are no significant differences in the salinities between the "Standard" and "Davis off" scenarios.

DISCUSSION

Implications of the Different Fresh Water Diversion Scenarios

The differences between the observed and "Standard" simulation salinities during the first three months of simulation (March through May) are likely due to the model spin-up effect. The model was initialized by imposing measured salinities as longitudinal bands across the estuary, where the number of monitoring stations was not sufficient to adequately describe complex salinity distributions in the estuary. The higher simulated values of salinity along the

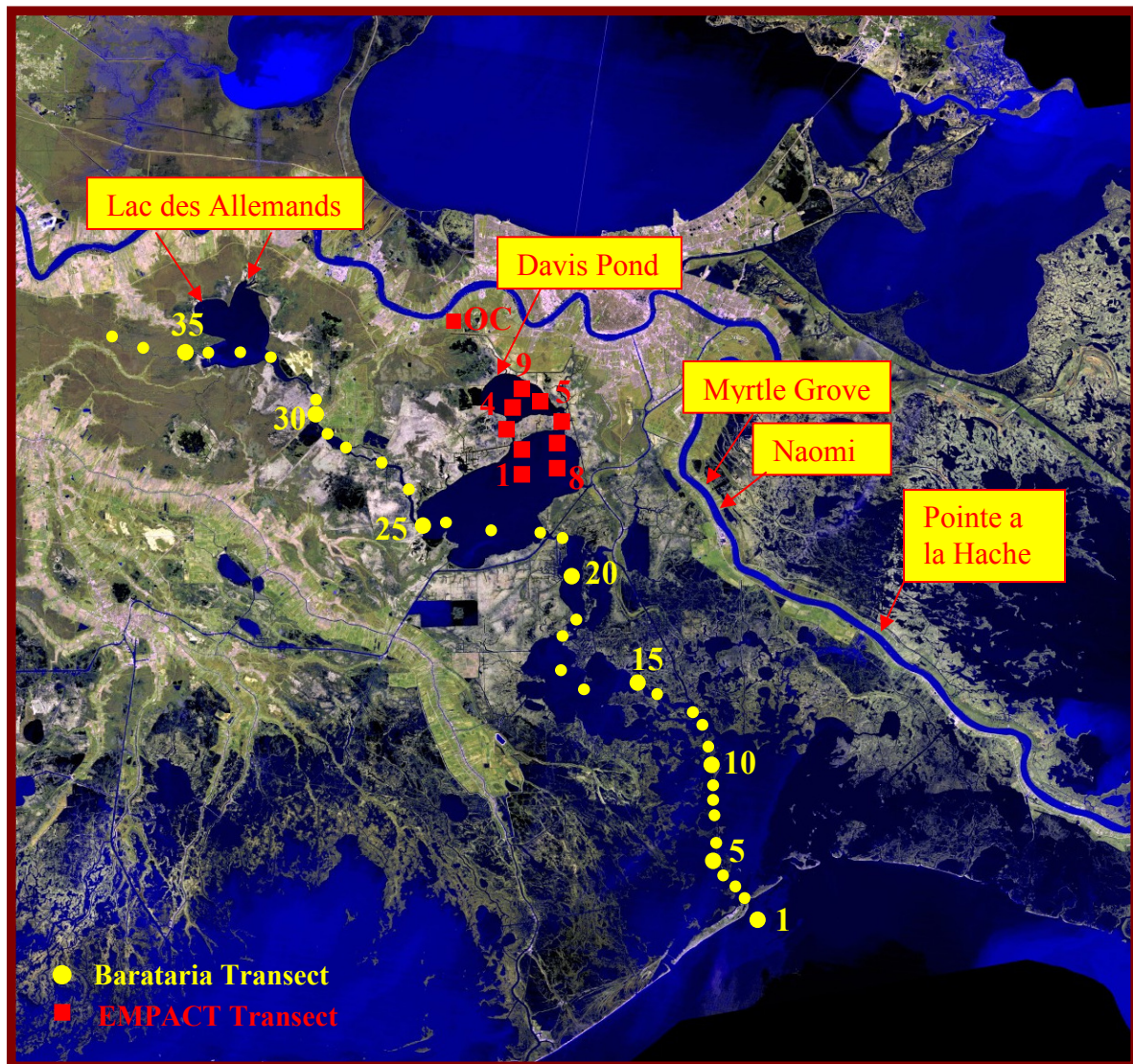


Figure 4.2. Barataria transect sampling stations in the Barataria estuary (Turner et al., unpublished data), and locations of present and potential diversions.

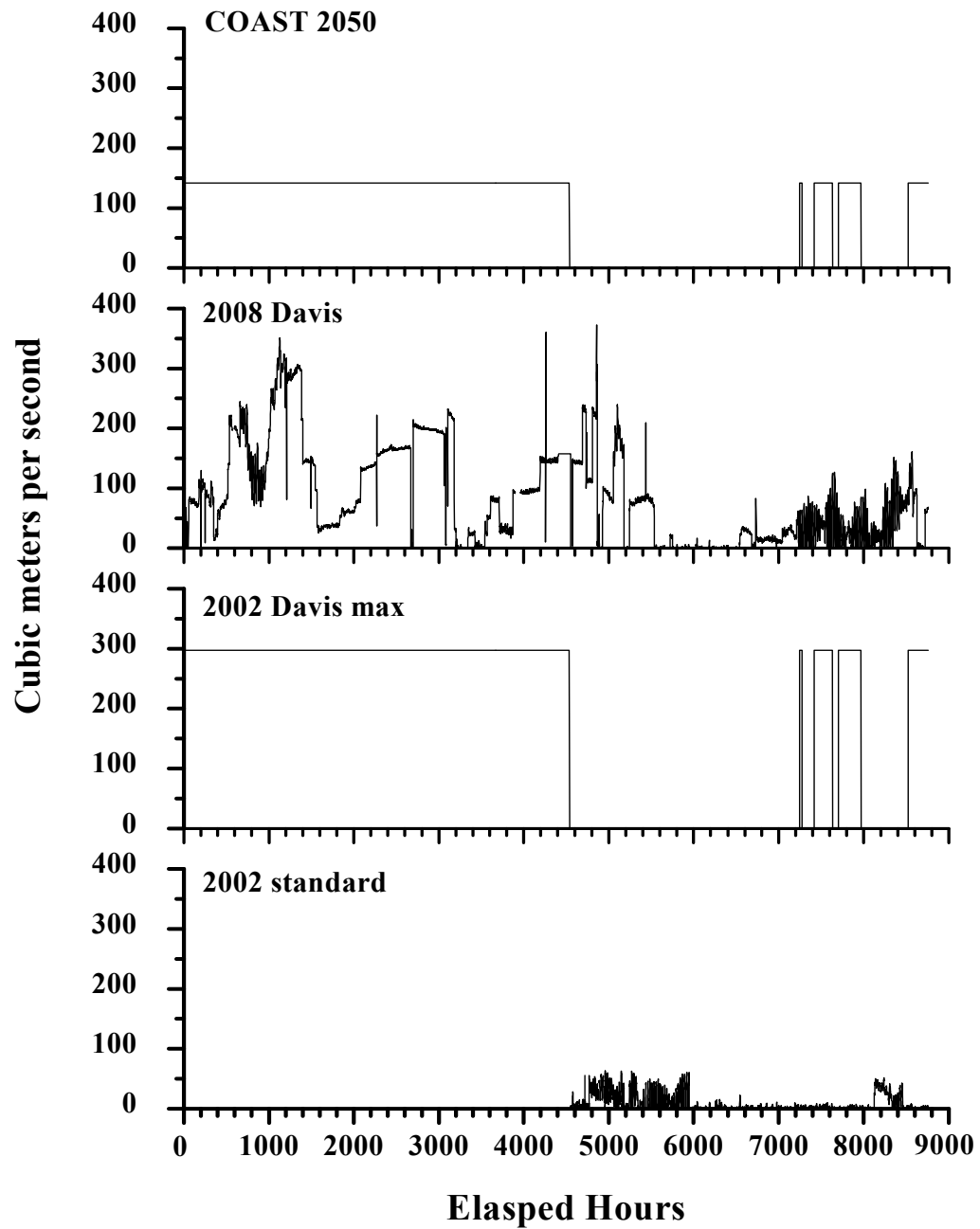
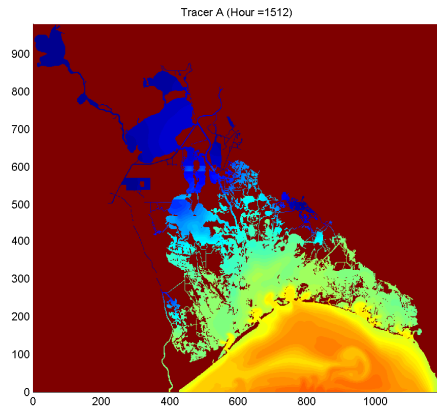
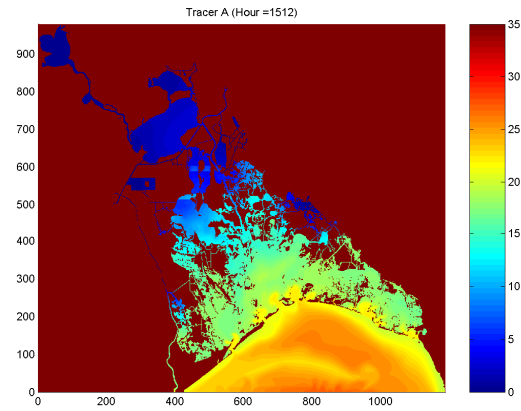


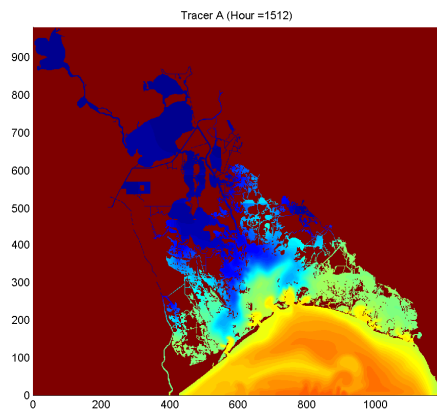
Figure 4.3. Annual Davis Pond discharge for the different diversion scenarios.



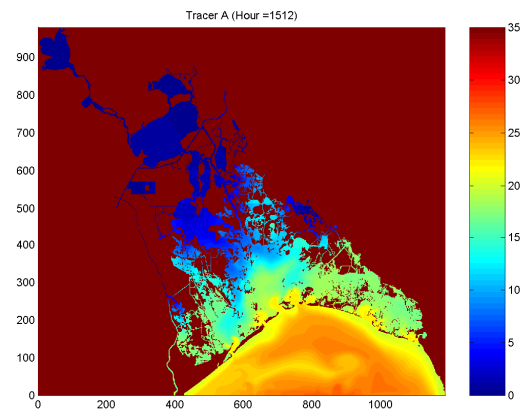
(a) Standard



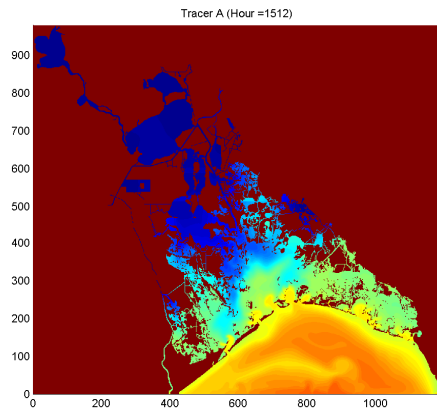
(b) Davis off



(c) Davis Max

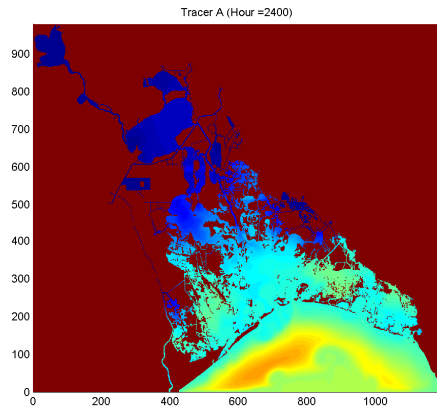


(d) Davis 2008

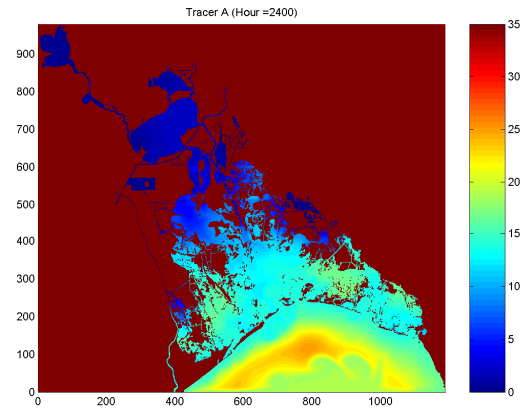


(e) Coast 2050

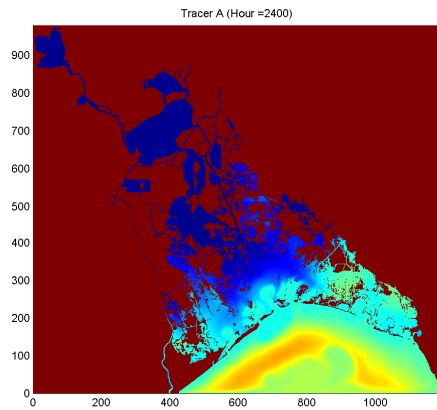
Figure 4.4. Spatial plots of simulated salinities for different diversion scenarios (Table 4.2) for May 21, 2002.



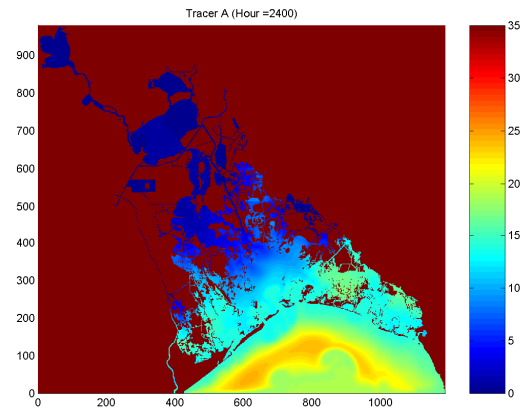
(a) Standard



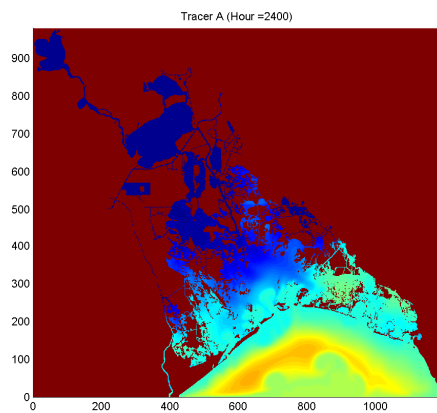
(b) Davis off



(c) Davis Max

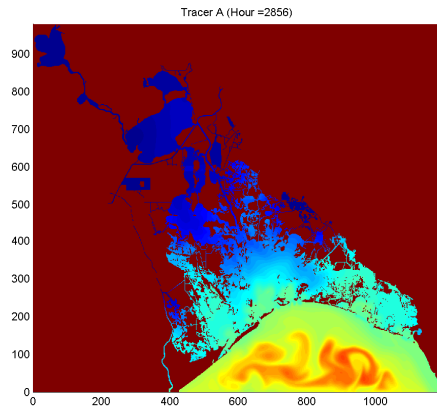


(d) Davis 2008

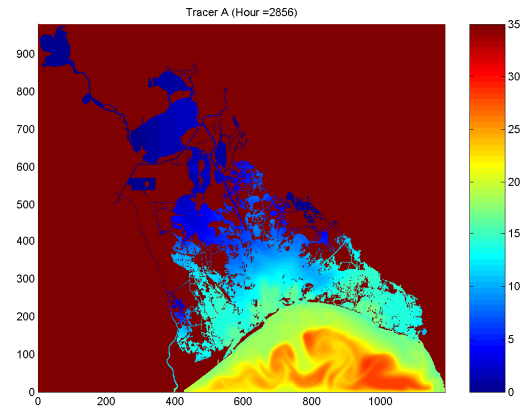


(e) Coast 2050

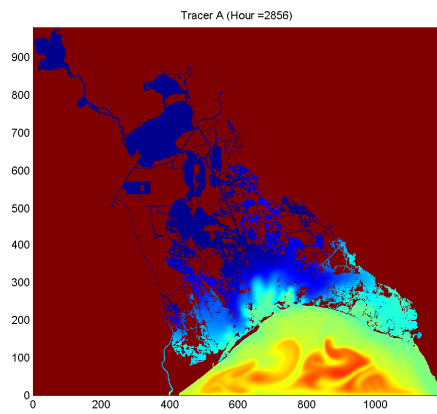
Figure 4.5. Spatial plots of simulated salinities for different diversion scenarios (Table 4.2) for June 25, 2002.



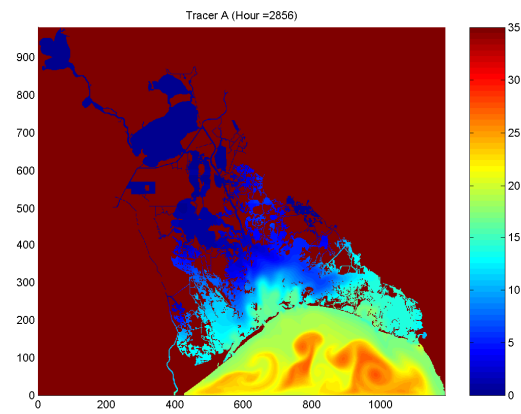
(a) Standard



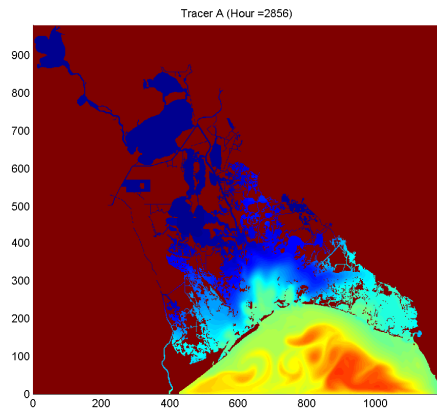
(b) Davis off



(c) Davis Max

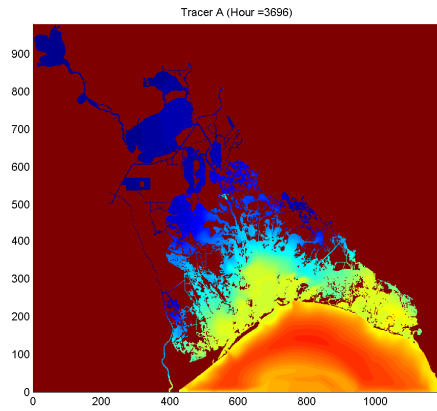


(d) Davis 2008

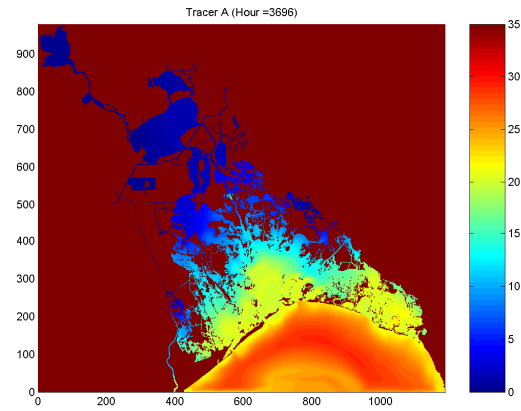


(e) Coast 2050

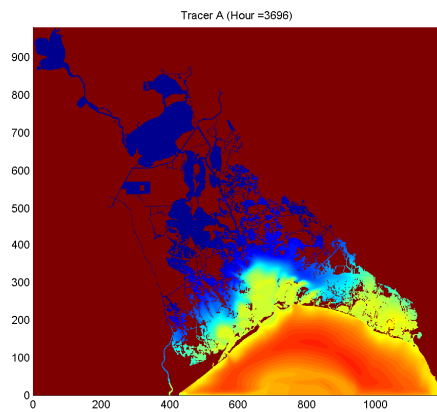
Figure 4.6. Spatial plots of simulated salinities for different diversion scenarios (Table 4.2) for July 16, 2002.



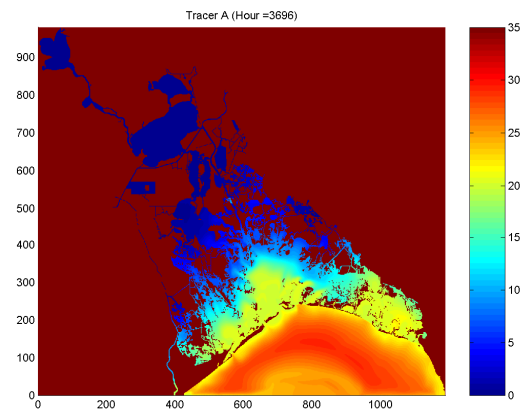
(a) Standard



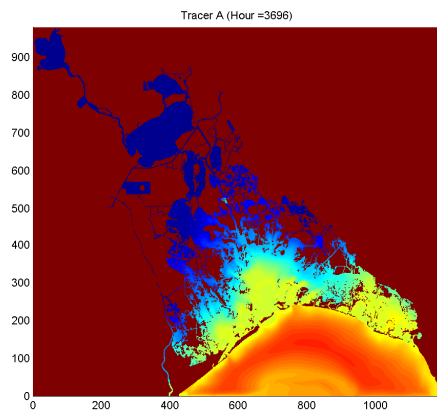
(b) Davis off



(c) Davis Max

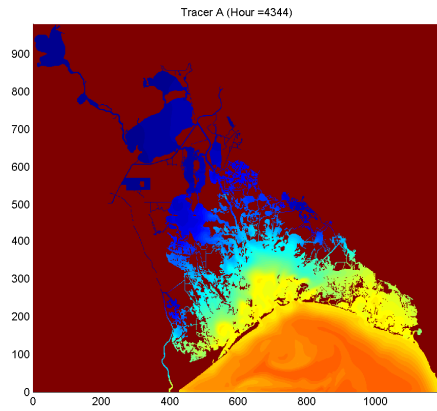


(d) Davis 2008

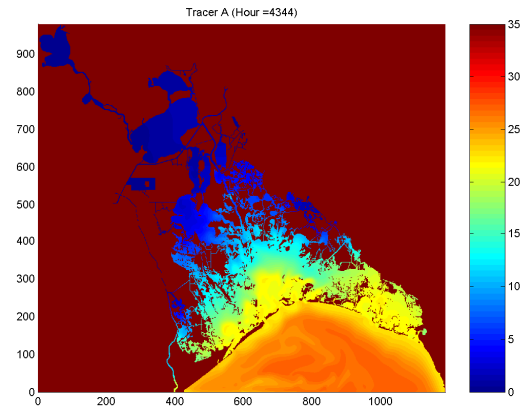


(e) Coast 2050

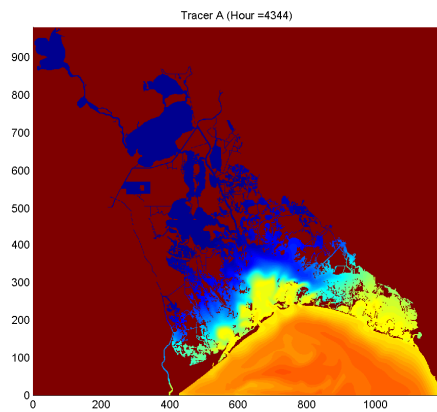
Figure 4.7. Spatial plots of simulated salinities for different diversion scenarios (Table 4.2) for August 20, 2002.



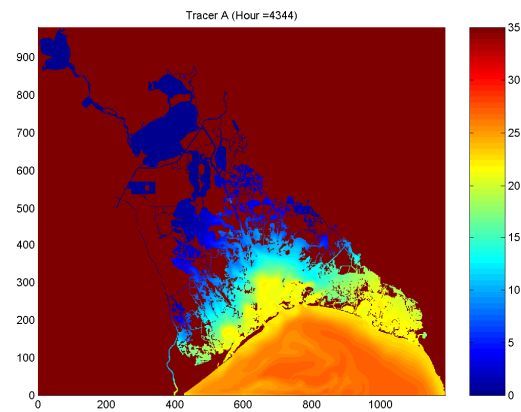
(a) Standard



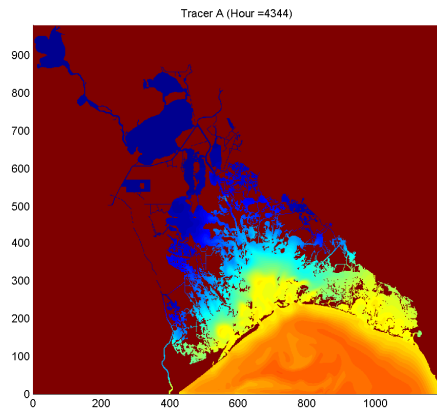
(b) Davis off



(c) Davis Max

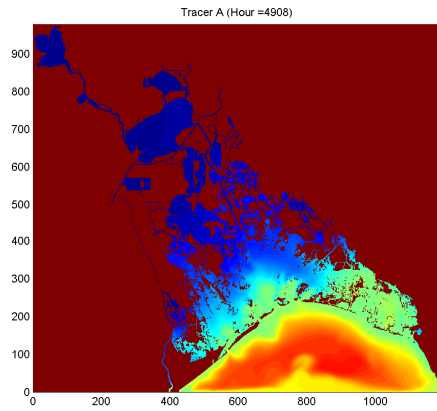


(d) Davis 2008

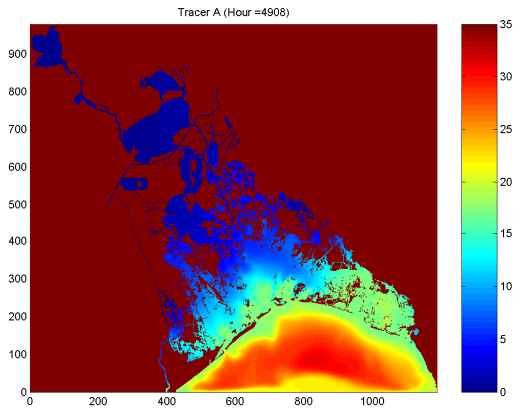


(e) Coast 2050

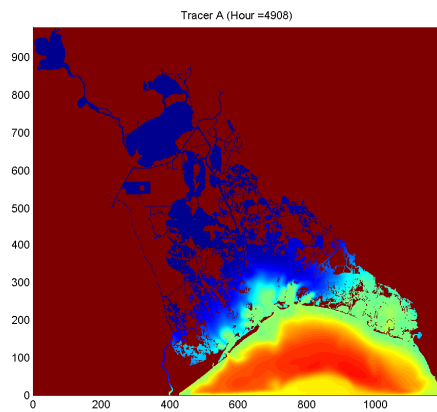
Figure 4.8. Spatial plots of simulated salinities for different diversion scenarios (Table 4.2) for September 16, 2002.



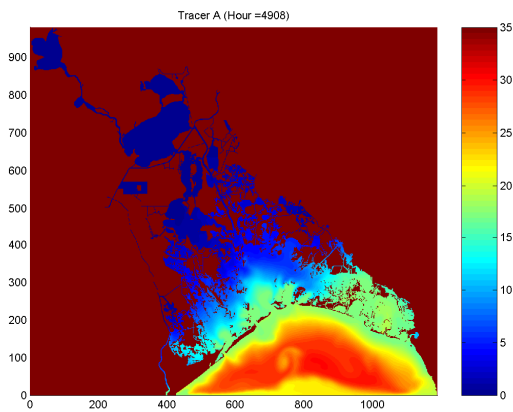
(a) Standard



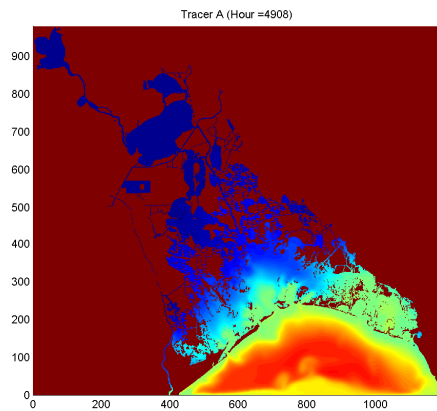
(b) Davis off



(c) Davis Max

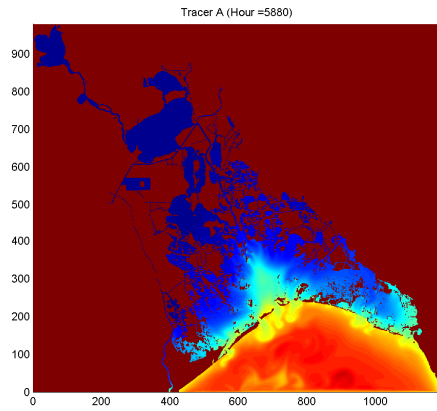


(d) Davis 2008

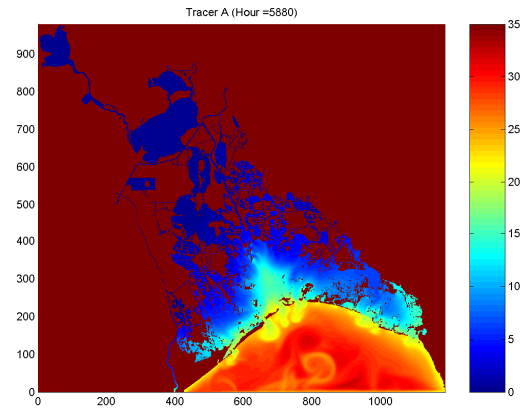


(e) Coast 2050

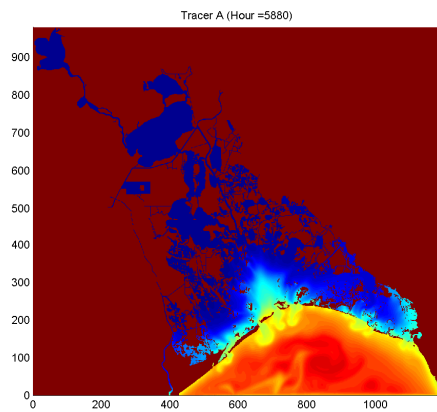
Figure 4.9. Spatial plots of simulated salinities for different diversion scenarios (Table 4.2) for October 9, 2002.



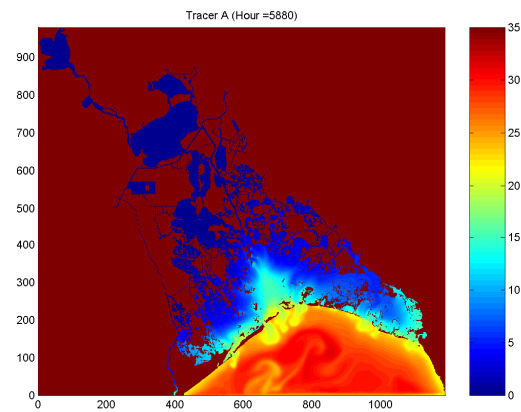
(a) Standard



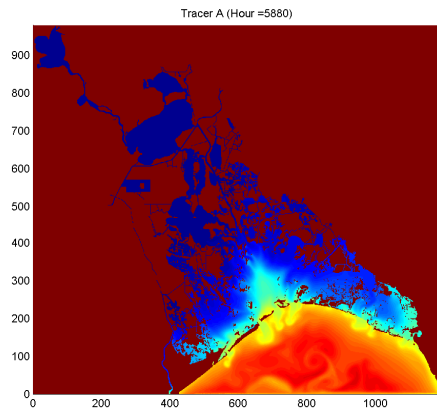
(b) Davis off



(c) Davis Max

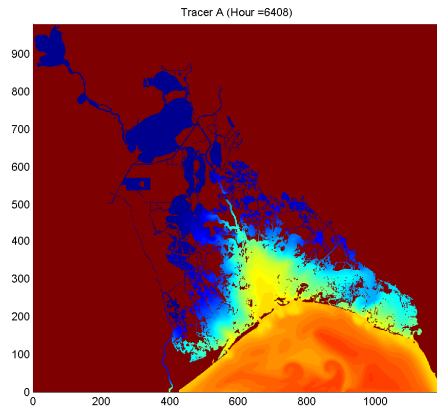


(d) Davis 2008

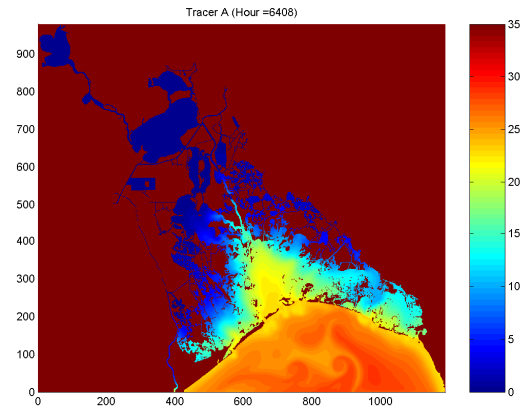


(e) Coast 2050

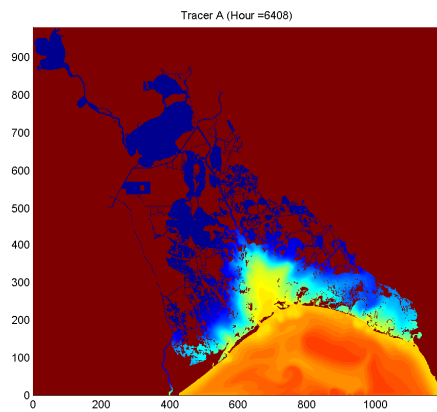
Figure 4.10. Spatial plots of simulated salinities for different diversion scenarios (Table 4.2) for November 19, 2002.



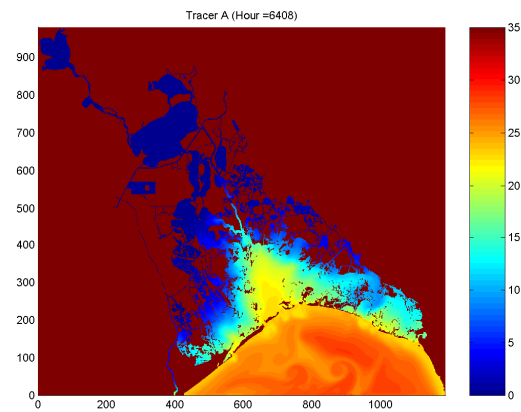
(a) Standard



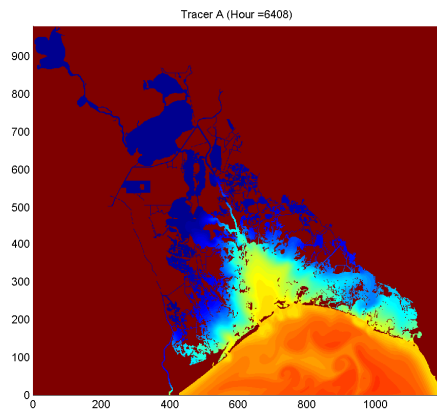
(b) Davis off



(c) Davis Max



(d) Davis 2008



(e) Coast 2050

Figure 4.11. Spatial plots of simulated salinities for different diversion scenarios (Table 4.2) for December 12, 2002.

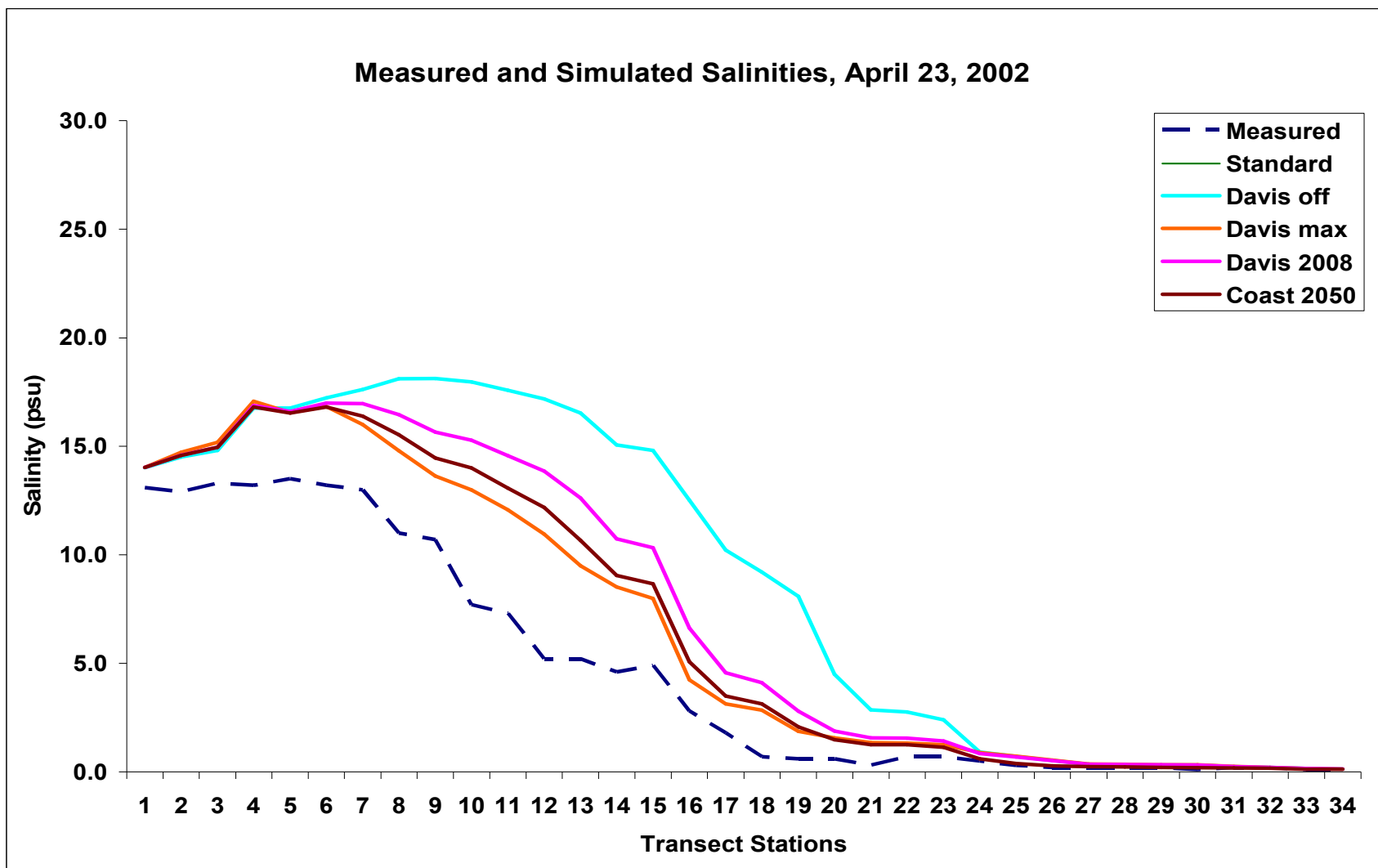


Figure 4.12. Measured and simulated salinities for different Davis Pond freshwater diversion scenarios (Table 4.2) for April 23, 2002. “Measured” denotes observed salinity values along the Barataria transect (Turner et al. unpublished data). Please note that “Davis off” scenario cannot be distinguished from the “Standard” scenario.

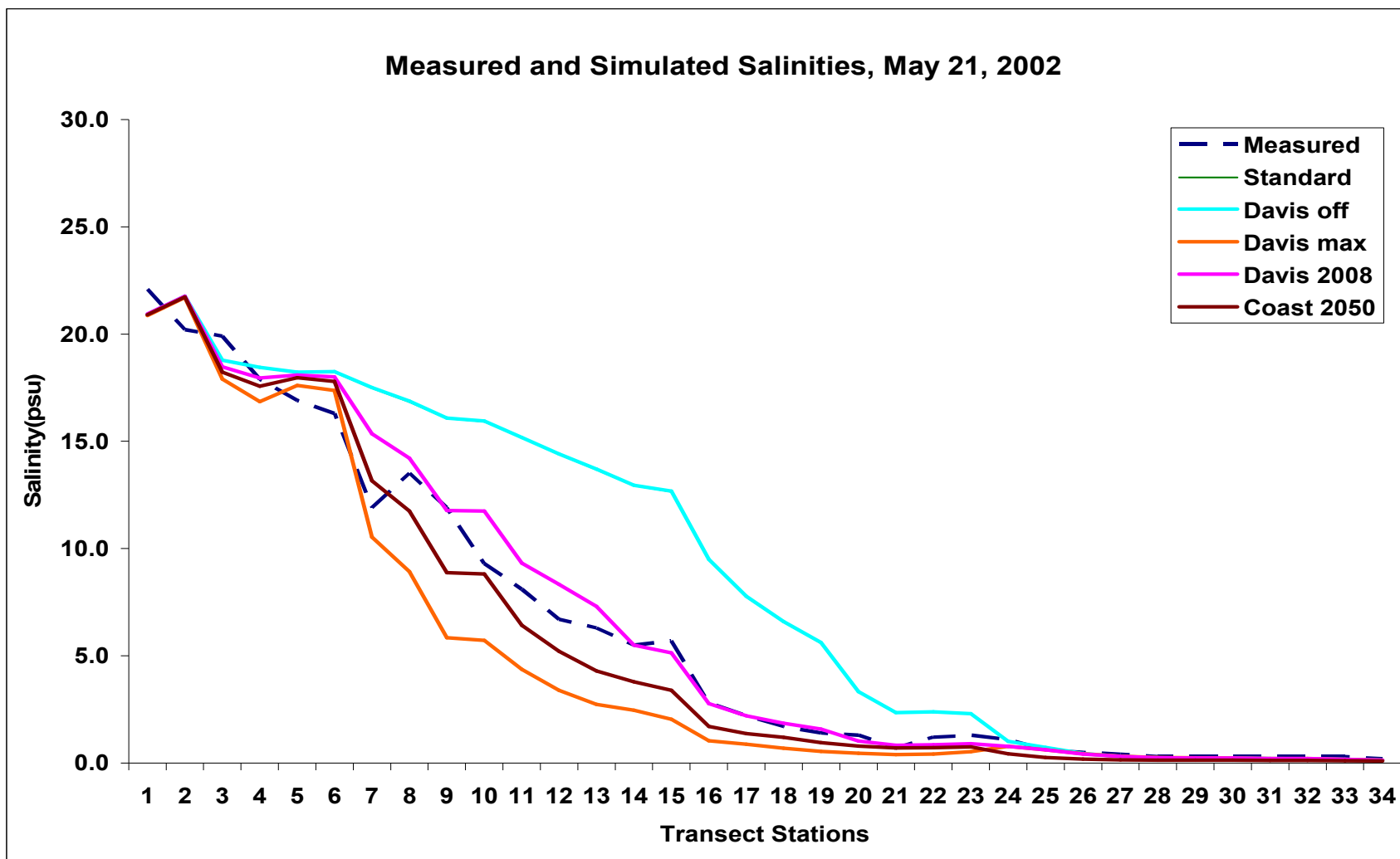


Figure 4.13. Measured and simulated salinities for different Davis Pond freshwater diversion scenarios (Table 4.2) for May 21, 2002. “Measured” denotes observed salinity values along the Barataria transect (Turner et al. unpublished data). Please note that “Davis off” scenario cannot be distinguished from the “Standard” scenario.

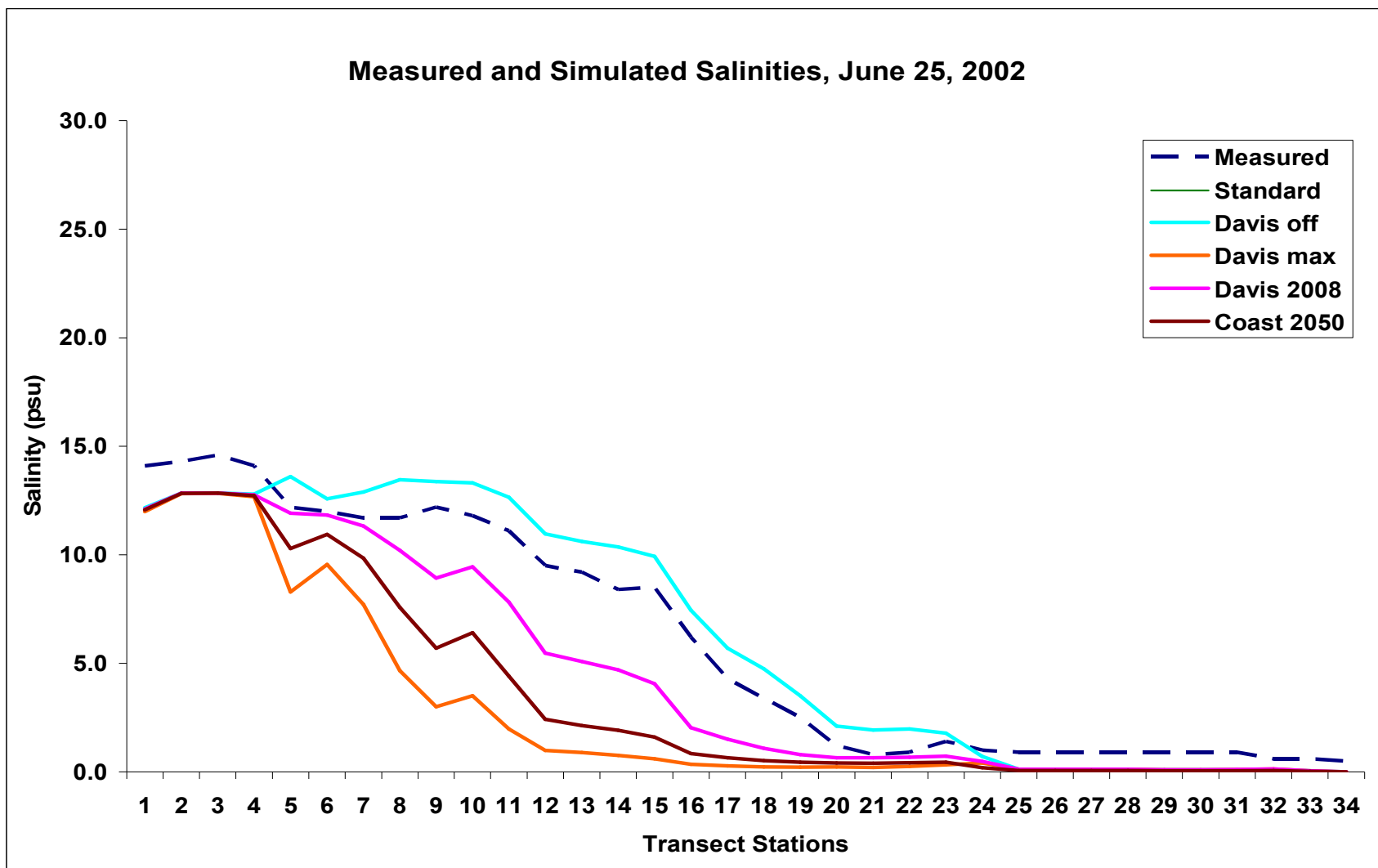


Figure 4.14. Measured and simulated salinities for different Davis Pond freshwater diversion scenarios (Table 4.2) for June 25, 2002. “Measured” denotes observed salinity values along the Barataria transect (Turner et al. unpublished data). Please note that “Davis off” scenario cannot be distinguished from the “Standard” scenario.

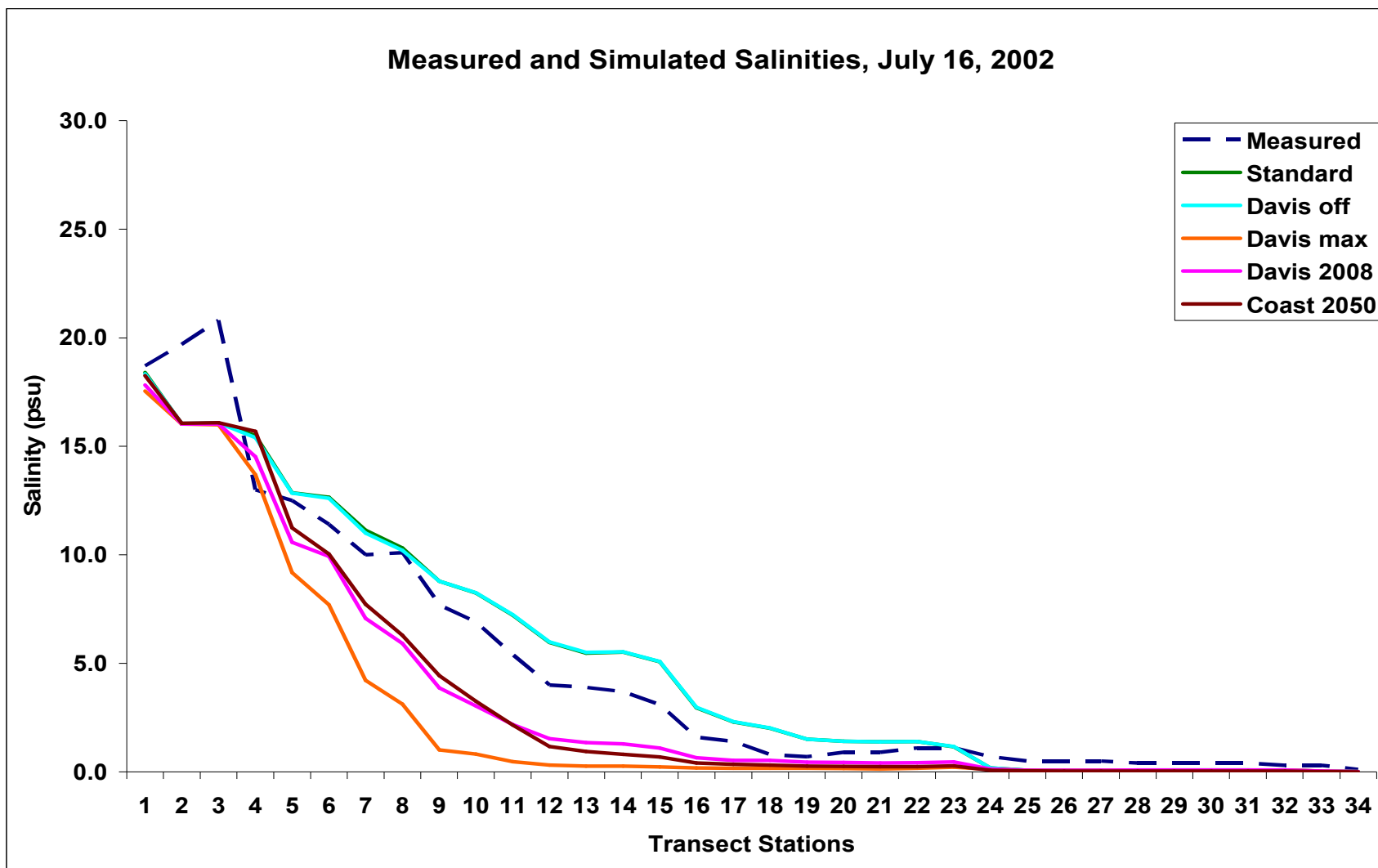


Figure 4.15. Measured and simulated salinities for different Davis Pond freshwater diversion scenarios (Table 4.2) for July 16, 2002. “Measured” denotes observed salinity values along the Barataria transect (Turner et al. unpublished data). Please note that “Davis off” scenario cannot be distinguished from the “Standard” scenario.

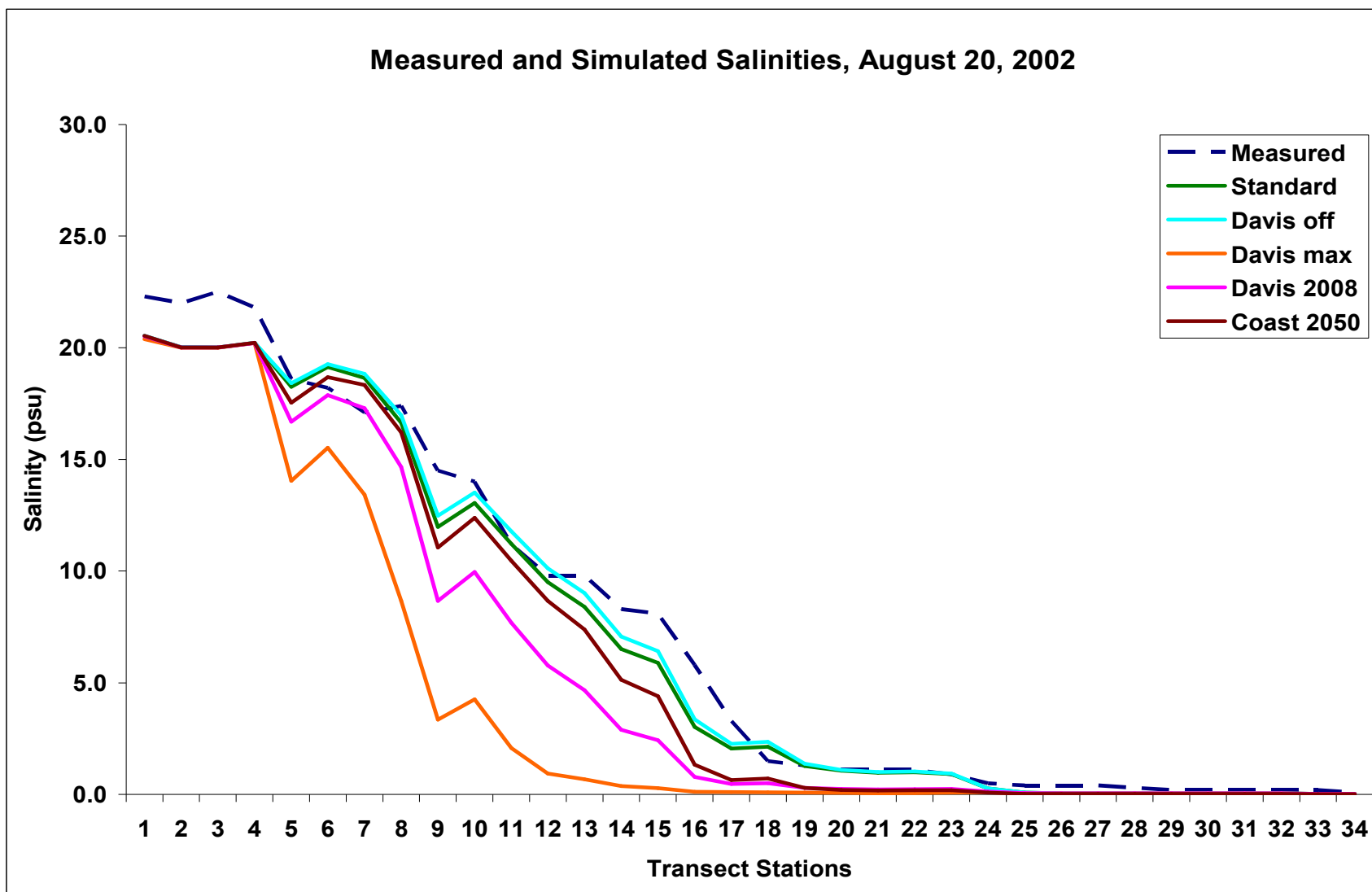


Figure 4.16. Measured and simulated salinities for different Davis Pond freshwater diversion scenarios (Table 4.2) for August 20, 2002. “Measured” denotes observed salinity values along the Barataria transect (Turner et al. unpublished data).

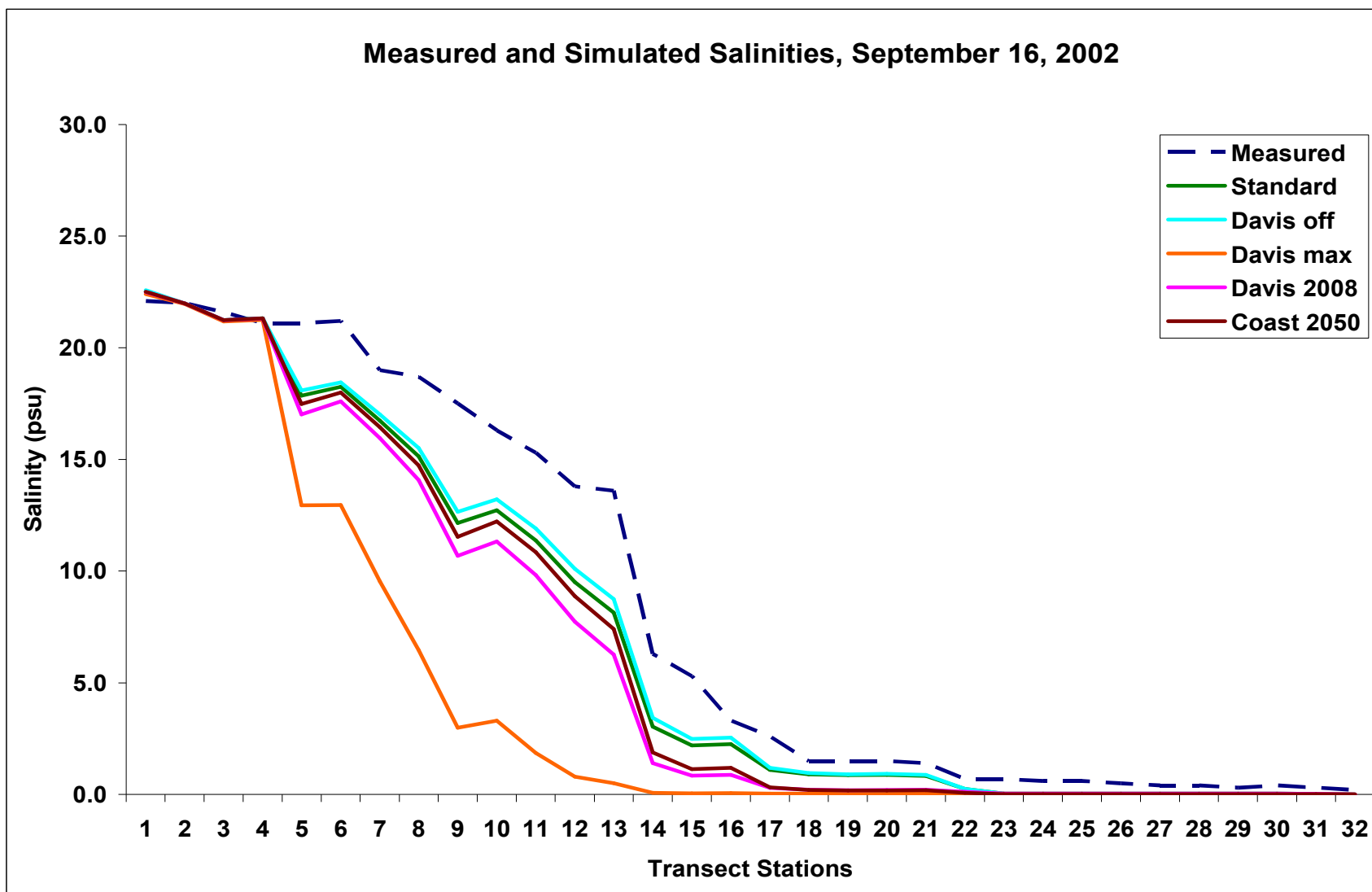


Figure 4.17. Measured and simulated salinities for different Davis Pond freshwater diversion scenarios (Table 4.2) for September 16, 2002. “Measured” denotes observed salinity values along the Barataria transect (Turner et al. unpublished data).

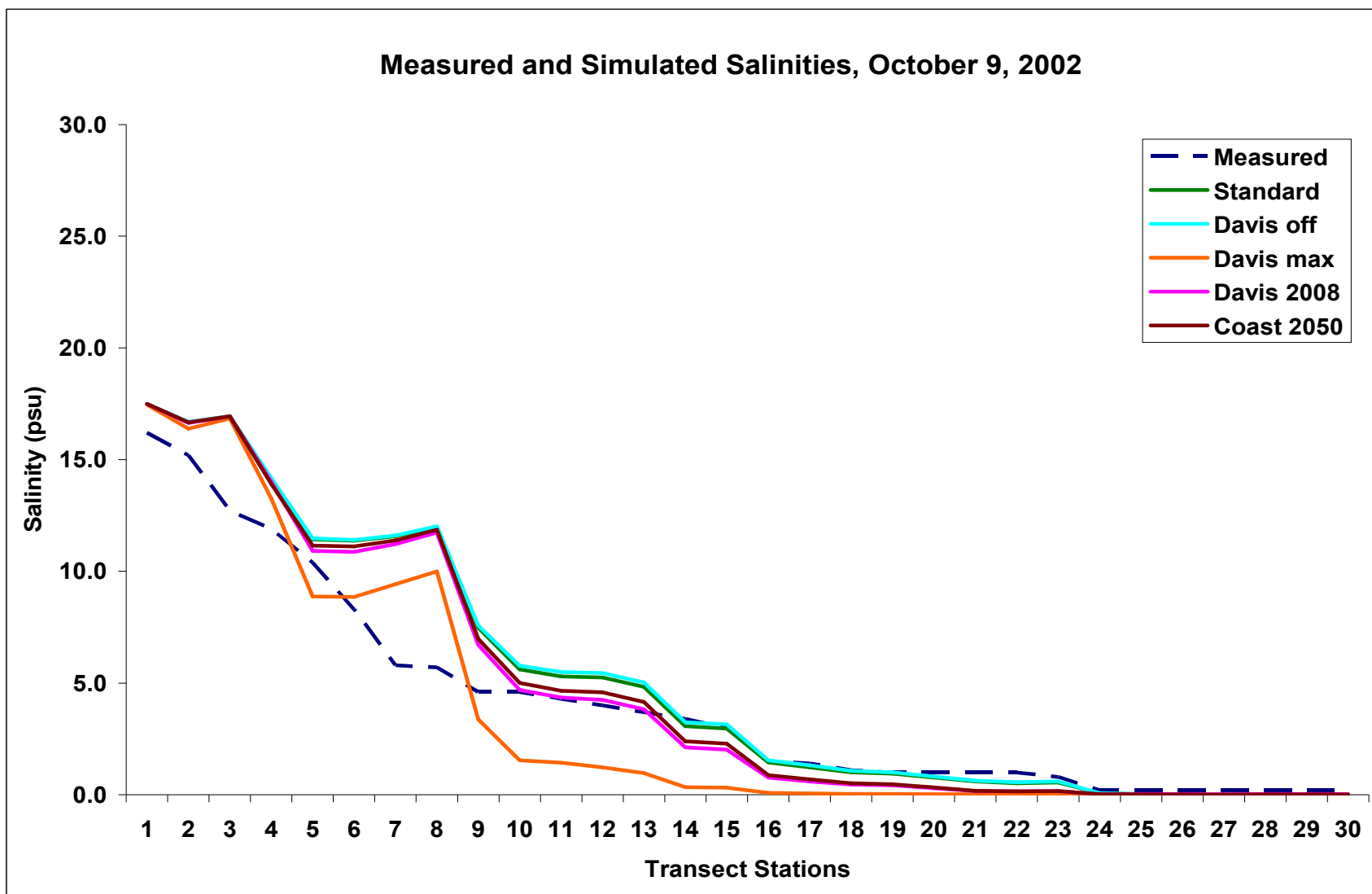


Figure 4.18. Measured and simulated salinities for different Davis Pond freshwater diversion scenarios (Table 4.2) for October 9, 2002. “Measured” denotes observed salinity values along the Barataria transect (Turner et al. unpublished data).

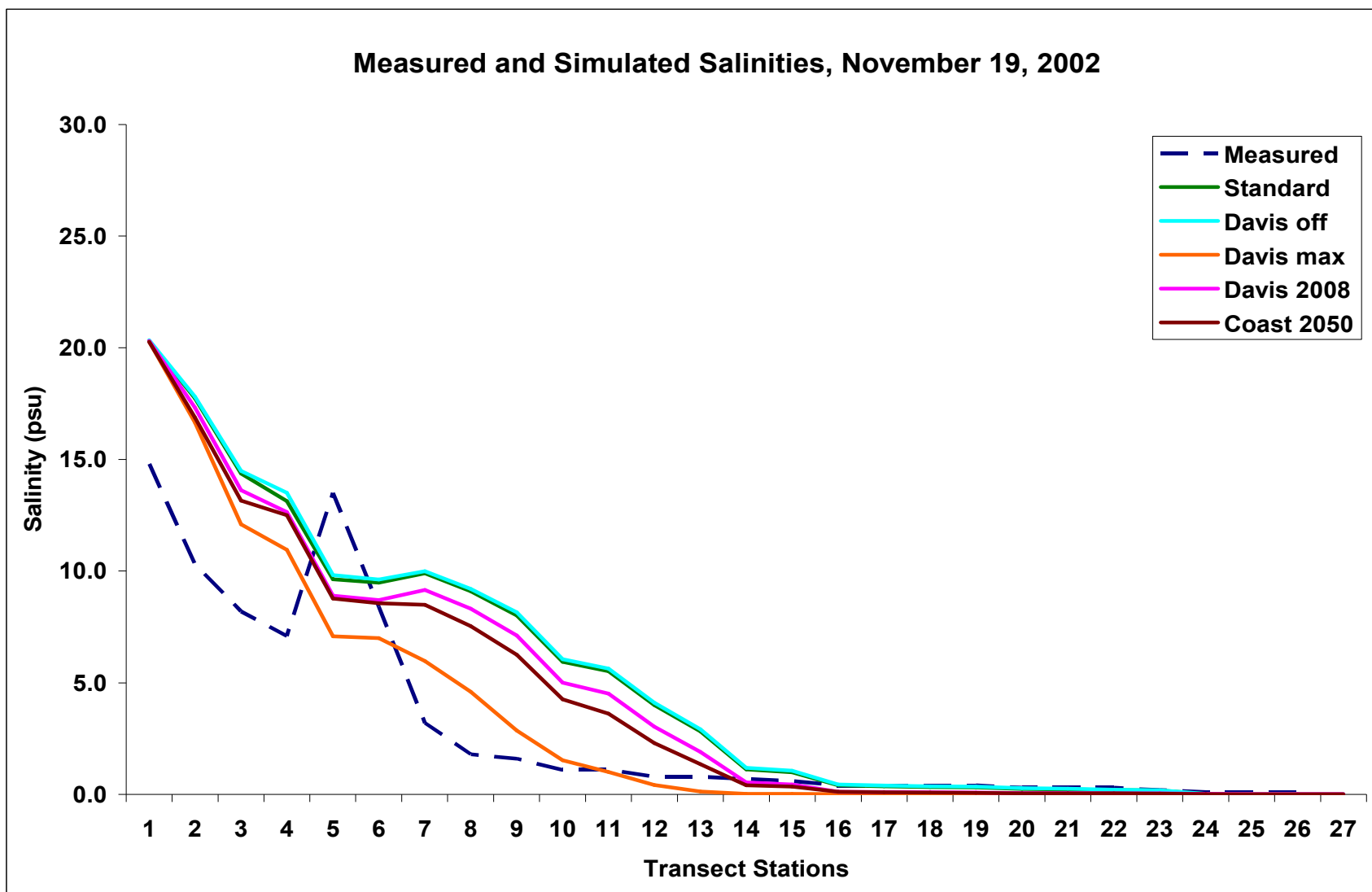


Figure 4.19. Measured and simulated salinities for different Davis Pond freshwater diversion scenarios (Table 4.2) for November 19, 2002. “Measured” denotes observed salinity values along the Barataria transect (Turner et al. unpublished data).

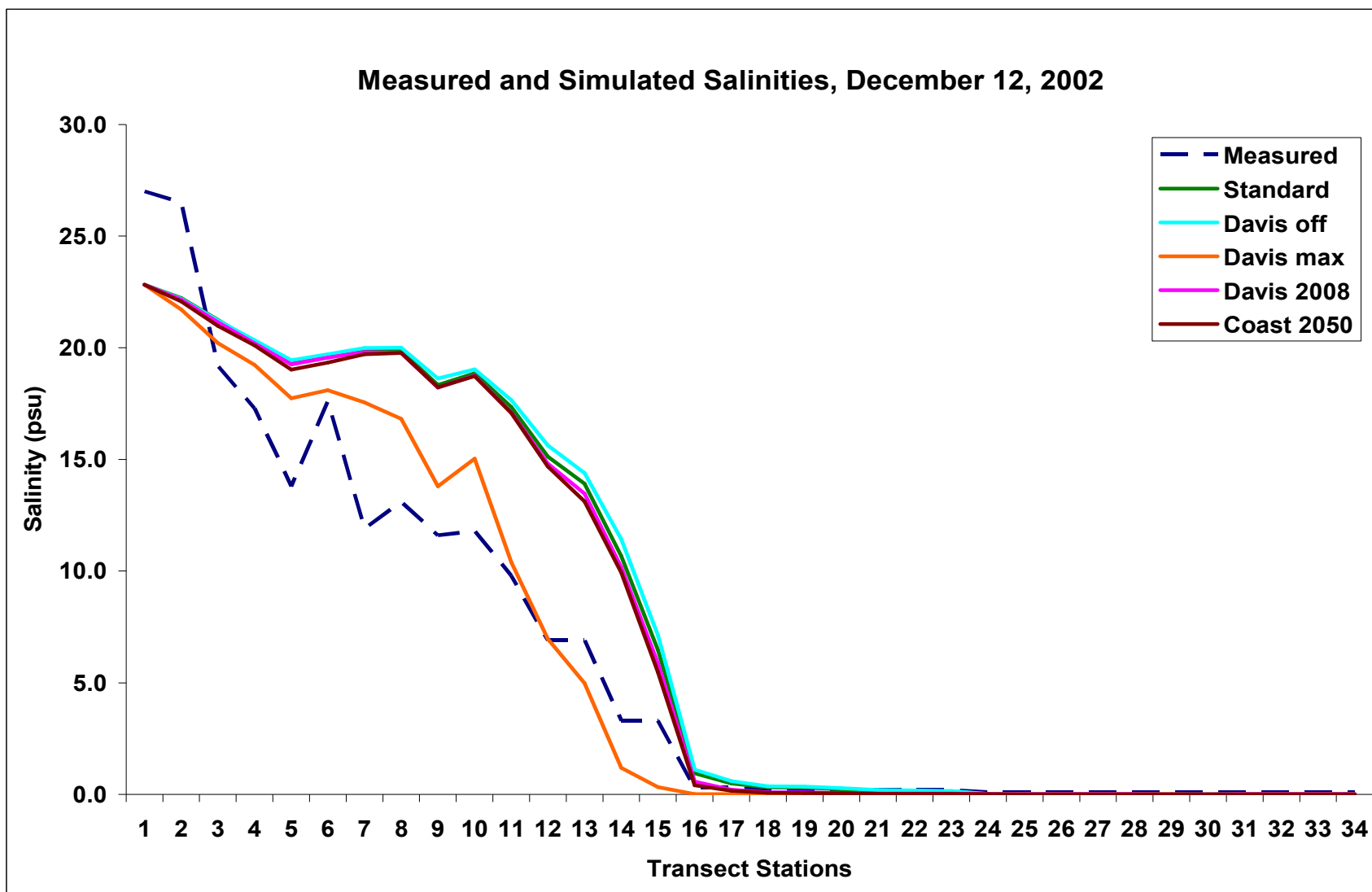


Figure 4.20. Measured and simulated salinities for different Davis Pond freshwater diversion scenarios (Table 4.2) for December 12, 2002. “Measured” denotes observed salinity values along the Barataria transect (Turner et al. unpublished data).

Table 4.3. Comparison of measured and simulated mean salinity values for the March-December period at each of the 34 stations along the Barataria transect based on Duncan's Multiple Range Test. Letters A, B and C denote statistically significant differences ($\alpha = 0.05$) between different model scenarios. Multiple letters (AB, or ABC) indicate overlap between A, B and C clusters for a particular station.

Station	Measured	Standard	Davis off	Davis max	Davis 2008	Coast 2050
1	A 18.7	A 18.7	A 18.7	A 18.7	A 18.7	A 18.7
2	A 18.0	A 18.1	A 18.1	A 17.9	A 18.0	A 18.0
3	A 17.0	A 17.0	A 17.4	A 17.0	A 17.3	A 17.2
4	A 15.5	A 17.1	A 17.1	A 16.3	A 16.9	A 16.9
5	A 15.3	A 15.8	A 15.9	A 13.2	A 15.0	A 15.0
6	A 14.8	A 15.9	A 16.0	A 13.4	A 15.2	A 15.2
7	A 12.5	A 15.6	A 15.7	A 11.5	A 14.5	A 14.2
8	A B 12.4	A 15.2	A 15.3	B 9.9	A B 13.6	A B 13.2
9	A B 11.2	A 13.4	A 13.5	B 6.9	A B 11.1	A B 10.7
10	A B 10.0	A 12.9	A 13.1	B 6.6	A B 10.7	A B 10.3
11	A B 9.1	A 12.1	A 12.3	B 5.3	A B 9.5	A B 9.0
12	A B 7.8	A 10.9	A 11.1	B 4.3	A B 8.2	A B 7.7
13	A B 7.6	A 10.1	A 10.3	B 3.7	A B 7.3	A B 7.6
14	A B 5.9	A 8.9	A 9.1	B 3.2	A B 5.9	A B 5.4
15	A B 5.5	A 7.8	A 8.4	B 2.6	A B 4.8	A B 4.3
16	A B 3.7	A 5.0	A 5.1	B 1.5	A B 2.4	A B 2.1
17	A B C 2.5	A B 3.8	A 3.8	C 1.0	A B C 1.6	B C 1.4
18	A B C 1.8	A B 3.3	A 3.4	C 0.9	A B C 1.4	B C 1.2

(Table 4.3 continued)

Station	Measured	Standard	Davis off	Davis max	Davis 2008	Coast 2050
19	A B 1.4	A 2.6	A 2.6	B 0.6	A B 1.0	B 0.8
20	A B 1.1	A 1.7	A 1.7	B 0.5	A B 0.7	A B 0.6
21	A B 0.8	A 1.2	A 1.3	B 0.3	B 0.5	B 0.4
22	A B 0.9	A 1.2	A 1.2	B 0.3	B 0.5	B 0.4
23	A B 0.8	A 1.0	A 1.1	B 0.3	B 0.3	B 0.4
24	A 0.5	A 0.4	A 0.4	A 0.3	A 0.3	A 0.2
25	A 0.4	A B 0.2	A B 0.2	A B 0.2	A B 0.2	B 0.1
26	A 0.4	A B 0.1	A B 0.1	A B 0.1	A B 0.1	B 0.1
27	A 0.4	B 0.1	B 0.1	B 0.1	B 0.1	B 0.1
28	A 0.3	B 0.1	B 0.1	B 0.1	B 0.1	B 0.1
29	A 0.3	B 0.1	B 0.1	B 0.1	B 0.1	B 0.1
30	A 0.3	B 0.1	B 0.1	B 0.1	B 0.1	B 0.1
31	A 0.3	B 0.1	B 0.1	B 0.1	B 0.1	B 0.09
32	A 0.3	B 0.1	B 0.1	B 0.1	B 0.1	B 0.08
33	A 0.2	B 0.08	B 0.08	B 0.08	B 0.08	B 0.06
34	A 0.1	B 0.05	B 0.05	B 0.05	B 0.05	B 0.04

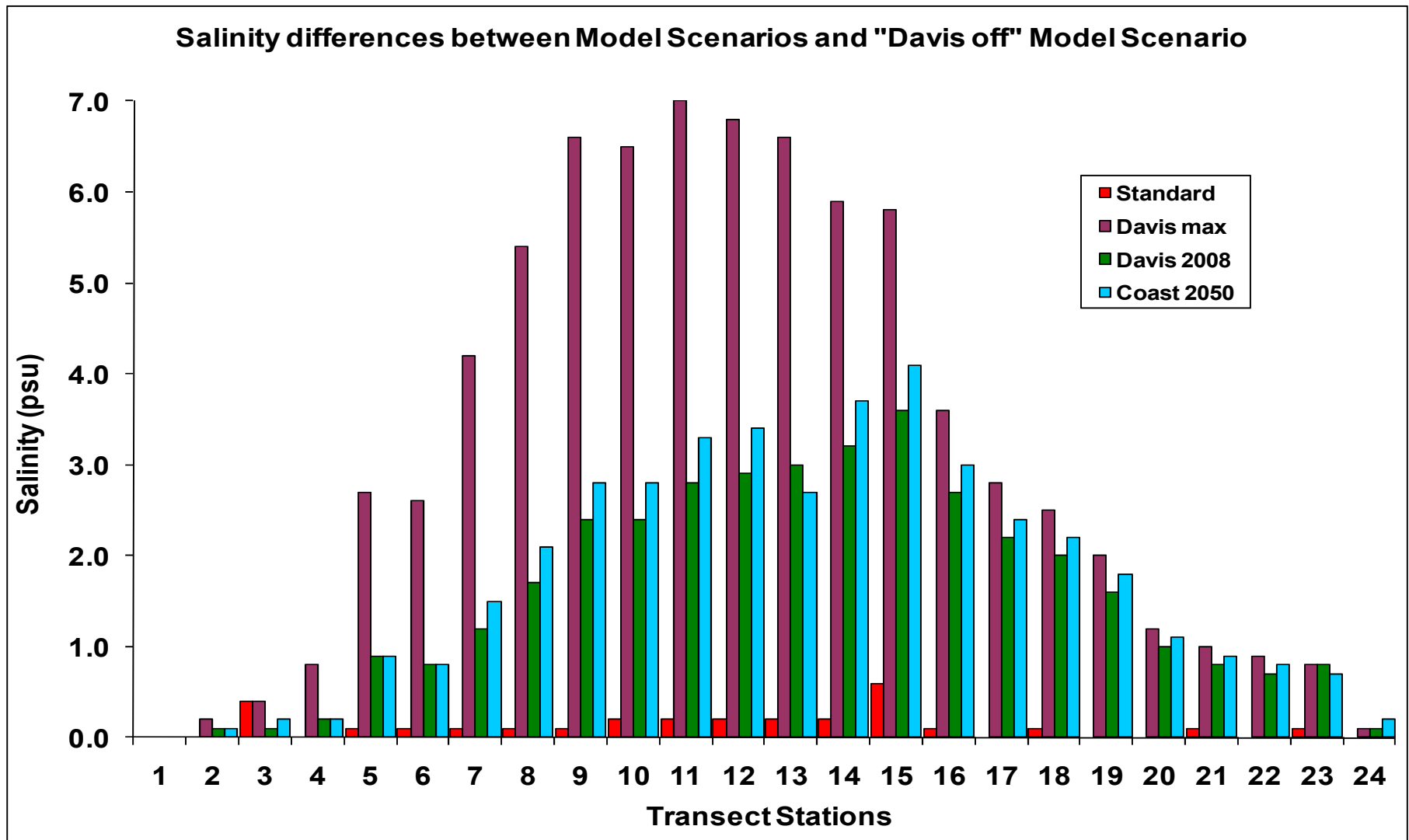


Figure 4.21. Salinity differences between "Standard", "Davis max", "Davis 2008" and "Coast 2050" scenario with "Davis off" scenario (Davis Pond turned off) along the Barataria estuary transect. The asterisk denotes a significant difference based on Duncan's Multiple Range Test (Table 4.3).

estuary could also be due to the fact that the 2-D model does not account for wetting and drying of the marshes.

It is interesting to note that the effects of different diversion scenarios on the salinity regime are apparent only in the middle and lower sections of the Barataria estuary (stations 8-26). This is because the upper parts of the estuary are fresh most of the time and the excess fresh water from river diversions has little impact on salinity in that area. This presents a challenge when designing a system of monitoring stations in the Barataria estuary, both for assessing changes in the system as a whole and for model validation. For example, the ten EMPACT stations that were established in Lake Cataouatche (Figure 4.2) to monitor the effects of Davis Pond on phytoplankton communities allow only for monitoring of near-field impacts and cannot capture the far-field salinity changes that were observed in this study.

The Davis Pond discharge, even when the diversion is running at maximum capacity, has little impact on salinities in the lower Barataria Bay (stations 1-7; Table 4.3). This is likely because of strong marine influence in this region adjacent to the Gulf of Mexico. It should also be pointed out that the “Davis 2008” and “Coast 2050” scenarios are very similar as far as the Davis Pond discharge is concerned (Table 4.3), so the simulated salinity distributions for 2008 could be used to assess the effectiveness of the “Coast 2050” scenario.

Effects of Changing Salinity Regimes on Estuarine Ecosystems

This study did not specifically investigate how differences in salinities between different diversion scenarios may affect plant and animal communities. However, salinity gradients play an important role in determining the type of organisms that can be found in an estuarine system. Quinlan and Philips (2007), for example, looked at phytoplankton assemblages and abundances along a nutrient, light and salinity gradient of the Suwannee River estuary, Florida, and found that shifts in phytoplankton taxa are best correlated with salinity. The salinity gradients often

determine the composition of species within an algal bloom (Chan and Hamilton, 2001). Many cyanobacterial genera that cause harmful algal blooms, (e.g., *Microcystis*), have a wide range of salt tolerance (Tonk et al., 2007), as evidenced in the *Microcystis* blooms in brackish waters of the Patos Lagoon Estuary, Brazil (Matthiensen et al., 2000), Swan River estuary, Australia (Robson and Hamilton, 2003), San Francisco Bay (Lehman et al., 2005), and Breton Sound estuary (Czubakowski, 2010).

In general, along a salinity gradient in estuaries, one would see a predominance of cyanobacteria and chlorophytes in brackish and fast flowing waters (Nakanishi and Monsi, 1965; Muylaert and Sabbe, 1999; Reynolds, 2006; Valdes-Weaver et al., 2006), dinoflagellates and diatoms in mid-to-high salinities (>10 psu) (Kies, 1997; Pinckney et al., 1999), and the lowest diversity in species at around 5 psu which is considered a lethal limit for most phytoplankton taxa in estuaries (Rijstenbil, 1988). The presence of toxic cyanobacterial genera, such as *Anabaena*, *Microcystis*, and *Cylindrospermopsis*, have been recorded repeatedly in Louisiana estuaries (Dortch et al., 1999; Dortch et al., 2001; Rabalais 2005; Garcia et al., 2010). It is possible that lower salinities resulting from freshwater diversions could extend the range of some potentially toxic cyanobacterial species in the Barataria estuary. Salinity also has an effect on estuarine sediments. Fresh water sediments typically contain more exchangeable ammonium than marine sediments. This is because ammonium in the marine environment has to compete with other cations for negatively charged binding sites (Rysgaard et al., 1999). The adsorption and flocculation of metals also increases with salinity (Elder, 1988). So, lower salinities in the estuary would allow sediments to have more ammonium but lower levels of adsorbed metals. Also, salinity is one of the factors that affects phosphorous sorption and desorption. The phosphate sorption capacity of sediments is more in sediments in lower salinity waters compared to sediments in higher salinity waters (Prastka et al., 1998; Sundareshwar and Morris, 1999).

Changing estuarine salinity gradients also affect higher trophic levels. For example, the Caernarvon fresh water diversion into the Breton Sound estuary led to a decline in brown shrimp, oyster and spotted sea trout landings, and an increase in white shrimp catch, blue crabs, juvenile menhaden, and red drum species (Chesney et al., 2000). Another before-after-control-impact study of the Caernarvon diversion by de Mutsert (2010) showed that nekton species richness, abundance and the proportion of smaller individuals increased relative to the control in Breton Sound. Such results might also be expected to be seen in the Barataria estuary due to the impacts of Davis Pond and other planned freshwater diversions.

Implications for Management

Because the occurrence of cyanobacterial blooms is likely to be higher in lower salinity waters, diversions of freshwater can be timed to occur during the seasons when algal bloom frequency is low. This may help control the spread of potentially toxic cyanobacterial species that are currently present in the Lac des Allemands and Lake Cataouatche into the middle section of the Barataria estuary. Also, considering that the model simulations show that salinity changes are mainly seen in the middle of the estuary (i.e., where salinity gradients are the steepest) for all the diversion scenarios, management decisions that concern salinity regime should be concentrated in these areas. For example, different macrophyte species have different salinity tolerance ranges, and so discharge of fresh water to control salinity can be set to the salinity tolerance range of the species that are considered to be more important in the estuary.

As stated before, the difference in simulated salinity values between the “Standard” and “Davis max” scenarios are frequently as high as 10 psu in the middle of the estuary. The difference in simulated salinities between the “Standard” and other model scenarios (“Davis 2008”, “Coast 2050” and “Davis Max”) is about 5 psu in the middle of the estuary. These

differences can also be biologically significant depending on the salinity tolerance of different species, and could also cause a shift in community composition within the affected region.

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CHAPTER 5

IMPACT OF FRESH WATER INFLOWS ON ESTUARINE WATER QUALITY

INTRODUCTION

Phytoplankton are responsible for about half of the total primary production in most coastal ecosystems (Cloern, 2001; Harding et al., 2002). They play a major role in nutrient cycling, and therefore, play an important part in biogeochemical cycling and water quality (Paerl and Justić, in press). Excess nutrients can cause rapid phytoplankton growth, resulting in phytoplankton blooms that can be noxious or toxic (harmful algal blooms) for animals throughout the food chain (ECOHAB, 1995; Carmichael, 2001). Both noxious and toxic blooms, which are nitrogen fixing, can also be a source of “new” nitrogen for further primary production, and consequently, eutrophication in water bodies. For example, cyanobacterial blooms are seen during the summer months in the Baltic Sea (Elmgren and Larsson, 2001), Pamlico Sound, North Carolina (Paerl, 1983; Paerl et al., 2001), and Lake Pontchartrain, Louisiana, where the nitrogen fixers were dependent on “new” nitrogen other than what they fixed (Dortch et al., 1999).

The presence of toxic cyanobacterial genera, such as *Anabaena*, *Microcystis*, and *Cylindrospermopsis*, have been recorded repeatedly in Louisiana estuaries (Dortch et al., 1999; Dortch et al., 2001; Rabalais, 2005; Garcia et al., 2010). Cyanobacterial blooms can cause odor and taste problems in water, making it unfit for consumption, aquaculture and recreation, and some blooms can also produce toxic alkaloids and peptides (Carmichael, 2001; Stewart and Falconer, 2008). These blooms are often not affected by changing salinities (Moisander and Paerl, 2000; Moisander et al., 2002a) but are affected by other physical changes like turbulence, vertical mixing, and short residence times of water that are usually seen in many coastal ecosystems (Paerl and Zehr, 2000; Moisander et al., 2002b).

Phytoplankton growth rates are variable, ranging from near zero to several $\text{g C m}^{-2}\text{d}^{-1}$. Faster growth rates allow them to adapt quickly to physical, chemical and biological changes in the water (Paerl and Justić, in press). The changes in phytoplankton communities that occur due to such changes cause a bottom-up alteration within the food web and could influence plant and animal species, including commercial fish species. This is because the change in phytoplankton community affects material flux and oxygen balances in the water column and sediments (Paerl and Justić, in press).

Estuaries receive nutrients from watershed-based nutrient sources via freshwater discharge and atmospheric deposition. The freshwater inflow rates, nutrient loads, turbidity, temperature and estuarine residence times determine the temporal scale for nutrient utilization by phytoplankton. All this influences the seasonal and spatial distribution of phytoplankton. High freshwater discharge should reduce residence times and should therefore restrict phytoplankton growth and biomass downstream (away from the discharge source) (Malone, 1977; Cloern et al., 1983) and low freshwater discharge would encourage phytoplankton growth and biomass to increase upstream (near the discharge source) (Ren et al., 2009; Day et al., 2009).

FACTORS AFFECTING PHYTOPLANKTON GROWTH

Phytoplankton productivity, biomass, composition, and distribution are controlled by light, nutrients, temperature, and grazing.

Light

Phytoplankton use part of the light spectrum that is called photosynthetically active radiation (400-700 nm). The amount of primary production depends on the instantaneous flux of light and the amount of light available during the day (Jassby and Platt, 1976; Fee, 1980). The amount of light available to the phytoplankton is controlled by turbidity, which in turn depends on the amount of suspended sediments, chlorophyll and other phytopigments, and

dissolved organic matter (CDOM) (Gallegos et al., 1990; Gallegos, 1992). Phytoplankton production is directly related to the ratio of the photic zone depth to the water column depth as has been shown in San Francisco Bay (Cole and Cloern, 1984), the Chesapeake Bay (Harding et al., 1987), the Neuse River estuary (Mallin and Paerl, 1992) and in Tokyo Bay (Bouman et al., 2010). Lalli and Parsons (1993) and Flores et al. (2005) showed that light influenced nutrient uptake and assimilation.

Nutrients

Freshwater nutrient inputs are generally the largest source of nutrients for estuaries. For example, in the Breton Sound estuary, the Mississippi River discharge through the Caernarvon diversion accounts for 67-83% of total nitrate input, while only 10-17% is from atmospheric deposition (Hyfield et al., 2008; Day et al., 2009). Nevertheless, atmospheric deposition is an important source of nutrients in other estuaries. For example, the Chesapeake Bay receives 20-30 % (Fisher and Openheimer, 1991), Sarasota Bay, Florida receives 26%, Tampa Bay, Florida receives 28%, and the Neuse River-Pamlico Sound receives 38% of nitrogen as atmospheric deposition (Paerl, 1997).

Estuaries also receive nutrients from its sediments through biogeochemical cycling. The mineralization of organic matter (mainly through microbes) from sediments is a major source of recycled nutrients in the water column (Pratihary et al., 2009). These nutrients can enter the water column through molecular diffusion (Li and Gregory, 1974), resuspension of sediments (Hammond et al., 1977), advection of porewater (Marinelli et al., 1998), and through macrobenthic activities (Kristensen, 1985). Benthic fluxes of nutrients can supply a large percentage of phytoplankton nitrogen demand (75%, Billen, 1978; 0-190%, Boynton et al., 1980; 25%, Nixon, 1981; 35%, Callender and Hammond, 1982; 3-34%, Hopkinson, 1987).

Denitrification is an important process by which nitrogen is released back into the environment as gaseous N_2 from reactive nitrates (NO_3^-), thus reducing the impacts of external nitrogen sources on estuarine eutrophication (DeLaune et al., 2005; Seitzinger et al., 2006; Dodla et al., 2008; Rivera-Monroy et al., 2010). Denitrification rates in wetlands are affected by the availability of NO_3^- and organic carbon (Cornwell et al., 1999; Greenan et al., 2006), while the residence times determine the proportion of nitrogen inputs that are denitrified (Seitzinger et al., 2006).

The effects of nutrients are often described as the “bottom-up” control of phytoplankton biomass and community composition. Nitrogen is a common limiting nutrient in estuarine and coastal waters, (Ryther and Dunstan, 1971; Nixon, 1995; Granéli et al., 1999; Elmgren and Larsson, 2001). Nitrogen and phosphorus co-limitation is also seen, especially during periods of high freshwater runoff (Boynton and Kemp, 1985; Malone et al., 1996). Most phytoplankton groups have specific nutrient requirements. Diatoms, for example, need silica in a particular minimum ratio to nitrogen and to phosphorus. Excess nitrogen and phosphorus loading from rivers can reduce the supply ratio of silica in estuarine waters and lead to changes in the phytoplankton community composition, which, in turn, can affect trophic interactions further along the food web (Turner et al., 1998; Turner, 2001). This has been seen in the northern Gulf of Mexico region where excess nitrogen and phosphorus enters the waters through the Mississippi River, and thereby changes the relative ratio of silica to nitrogen and silica to phosphorus (Justić et al., 1995; Turner, 2001). Sedimentological evidence has also shown that *Pseudonitzschia* (which is lightly silicified and thus requires less silicon) abundance has increased since the 1950s, corresponding to an increase in riverine nitrogen flux (Parsons et al., 2002).

Temperature

The effect of temperature on phytoplankton is seen as a bell-curve; growth rates increase up to a certain temperature, after which increase in temperature inhibits the growth rate (Eppley, 1972). It has been observed that phytoplankton growth in nutrient-limited conditions decreases with increase in temperature under laboratory conditions (Rhee and Gotham, 1981; Raven and Geider, 1988) and under natural conditions (Tadonleke, 2010). Tadonleke's (2010) experiments suggest that, in the long term, phytoplankton production responses to warming waters would be different in nutrient-rich and nutrient-poor environments. Different phytoplankton species show better adaptation to certain ranges of temperatures than other species. Many dinoflagellate species prefer warmer temperatures (usually 25-30 °C, Paerl and Justić, in press). This adaptation may also play an important role in the occurrence and expansion of cyanobacterial blooms due to regional and global warming (Paerl and Huisman, 2008, 2009). Diatoms, on the other hand, prefer cooler temperate regions and this could be influenced by secondary factors such as nutrient availability, light and stratification (Finkel et al., 2010). Temperature effects on higher trophic species also influences the changes in phytoplankton communities through grazing pressure (Eppley, 1972; Moisan et al., 2002; Edwards and Richardson, 2004).

Grazing

The effects of zooplankton grazing are often described as “top-down” control of phytoplankton biomass and community composition. During the spring and summer, higher temperatures are favorable for zooplankton growth and thus increase grazing rates. Nevertheless, phytoplankton growth rates are generally higher than zooplankton growth rates, so they cannot be grazed down completely (Paerl and Justić, in press). There is still a lot of uncertainty about how important grazing pressure is on phytoplankton growth. Steeman-Neilsen (1958) recognized that the level at which phytoplankton abundance remained at steady state due to grazing was

limited by environmental conditions such as light, temperature and nutrients. Cushing (1959) used a simple prey-predator model to show that grazing affected both the timing and magnitude of phytoplankton biomass. Sommer and Lewandowska (2010) conducted mesocosm experiments to show that both increased temperatures and overwintering zooplankton need to be taken into account when explaining the structure and extent of spring blooms. In Fourleague Bay, Louisiana, microzooplankton grazing rates of 43-165% of the daily phytoplankton production has been recorded, but such rates were negligible for mesozooplankton (Dagg, 1995). Grazing by macrofauna (e.g., suspension feeding bivalves), in estuaries, can reduce phytoplankton biomass. Some suspension feeding bivalves can filter the entire water column above them in 1-4 days (Cohen et al., 1984; Nichols, 1985; Doering et al., 1986).

Mississippi River diversions are considered to be an important part of future restoration plans for coastal Louisiana (Reed and Wilson, 2004). The diverted river water could affect nutrients, temperature, turbidity, and residence times of water in the receiving estuary. These parameters influence primary productivity. There are concerns that diversions may increase nutrient inputs and thus create eutrophication problems in estuaries and wetlands adjacent to the diversion sites. For example, the opening of the Bonnet Carré spillway during the spring of 1997 was associated with a major bloom of cyanobacteria in Lake Pontchartrain (Dortch et al., 1999). Lane et al. (2007) found that diverted Mississippi River water, from the Caernarvon diversion, into the Breton Sound estuary was generally cooler than the water in the estuary, but the river water temperatures were equilibrated with the rest of the estuary within several kilometers. They also found that suspended sediments from the diverted river water could reach 10 to 15 km into the estuary during pulsed events in the spring, and highest chlorophyll *a* values were observed in the middle of the estuary during low or no discharge from the river diversion.

Modeling Estuarine Water Quality

This chapter seeks to answer the following questions:

1. How are the diversions affecting phytoplankton productivity and standing stock?
2. Will the estuary become increasingly eutrophic, as evidenced in the increased phytoplankton biomass, if freshwater diversions continue at the present rate, or if larger diversions are implemented?

To understand estuarine ecosystem processes, it is important to understand the interactions between biology and physics. The approach adopted for this study was to develop a simple nutrient-phytoplankton-zooplankton (NPZ) model and couple it to the 2-D hydrology-hydrodynamic model that was described in Chapter 3. The expectation was that this fully coupled biological-physical model will be able to simulate the complex patterns of phytoplankton biomass dynamics (represented by the concentration of chlorophyll *a*) in the Barataria estuary. Recognizing that estuarine residence times are an important factor affecting both nutrients and phytoplankton, I have conducted a number of simulated tracer experiments to determine the residence times of various water bodies in the Barataria estuary under different diversion scenarios (i.e., “Standard”, the realistic 2002 forcing, “Davis 2008” and “Coast 2050”, Table 5.1). The same tracer experiments were also used to estimate the travel time for a phytoplankton bloom originating in the upper estuary to reach the oyster beds in the middle and lower estuary.

METHODS

NPZ Model

A simple NPZ (nutrient-phytoplankton-zooplankton) model is used in this study to describe phytoplankton dynamics (Figure 5.1). A general equation for phytoplankton growth was formulated as:

$$\text{Growth} = f(\text{Temperature, Nutrients, Light}),$$

$$\text{or Growth} = G_{\max} N_{\lim} L_{\lim} \quad (1)$$

where, Growth is the phytoplankton growth rate, G_{\max} is the temperature-dependent maximum growth rate, N_{\lim} is the nutrient limitation term, and L_{\lim} is the light limitation term. Based on the field studies in the Barataria estuary (Ren et al., 2009), I assumed that dissolved inorganic nitrogen (DIN) is the stronger limiting nutrient for phytoplankton growth. The maximum phytoplankton growth rate (G_{\max}) is described as an exponential function of temperature (Kremer & Nixon, 1978):

$$G_{\max} = 0.59 e^{0.0633 T} \quad (2)$$

where, G_{\max} (1/day) is the instantaneous rate coefficient and T is temperature ($^{\circ}\text{C}$). The average temperature coefficient (Q_{10}) calculated from this expression is 1.88. Hourly temperature averages were calculated based on 6 recording gauges located in the Barataria estuary (Chapter 3, Figure 3.1). The gauges are maintained by the Louisiana Department of Wildlife and Fisheries (LDWF), the United States Geologic Survey (USGS) and the Louisiana Department of Natural Resources (LA DNR). These stations are located in Barataria Bay.

The nutrient limitation term was evaluated using a normalized Monod's (1942) equation (Figure 5.3):

$$N_{\lim} = \left(\frac{N}{K_s + N} \right). \quad (3)$$

where, the N is ambient steady-state nutrient concentration and K_s is a characteristic half saturation constant. K_s is defined as the concentration at which the growth rate equals one-half the maximum growth rate. The effect of smaller values of K_s is to steepen the rate of ascent to G_{\max} (Figure 5.2).

The light limitation term was evaluated using the normalized Steele's equation (Steele, 1962):

$$L_{lim} = \frac{I}{I_{opt}} e^{[1 - (I / I_{opt})]} \quad (4)$$

where, I is the incident solar radiation and I_{opt} is the optimum radiation with respect to the growth rate. At the optimum light the growth rate equals G_{max} . The light limitation term was further modified to include the normalized Michaelis-Menten type effects of suspended sediments and self-shading of phytoplankton.

Table 5.1. Simulated freshwater diversion scenarios.

Scenarios	Description
Scenario 1 Standard	The 2002 Davis Pond discharge (Figure 4.3, Chapter 4).
Scenario 2 Davis 2008	The 2008 Davis Pond discharge (Figure 4.3, Chapter 4).
Scenario 3 Coast 2050	The Davis Pond diversion is set to discharge at half its maximum capacity ($142 \text{ m}^3 \text{ s}^{-1}$). There are two additional diversions into Lac des Allemands (at Lac des Allemands and Edgard, each $28 \text{ m}^3 \text{ s}^{-1}$), and one diversion at Myrtle Grove ($142 \text{ m}^3 \text{ s}^{-1}$). All these diversions are operational whenever the Mississippi River stage in Baton Rouge is above 4m (Figure 4.3, Chapter 4).

In the model, denitrification (D_N) is controlled by the ambient nitrogen concentration (N_{lim}). Both denitrification (D_N) and benthic nutrient regeneration (R_N) are controlled by the ambient temperature using the following temperature coefficient (C_T):

$$C_T = T / (20 + T) \quad (5)$$

so that

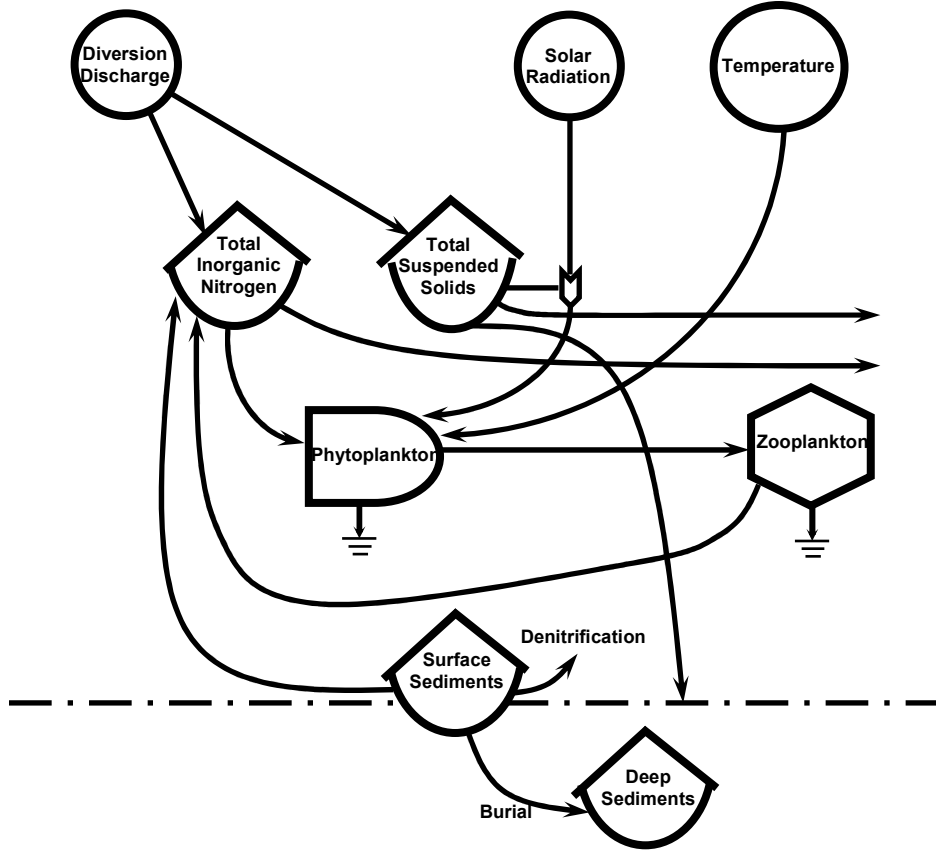


Figure 5.1. A nutrient-phytoplankton-zooplankton (NPZ) model for of the Barataria estuary that was used in this study.

$$D_N = D_{\max} N_{\lim} C_T \quad (6)$$

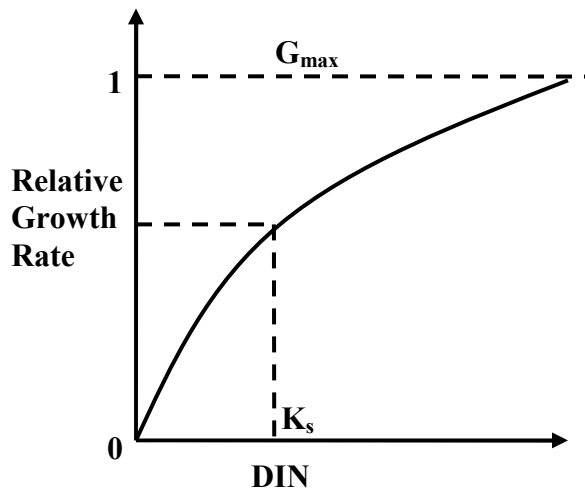
and

$$R_N = R_{\max} C_T \quad (7)$$

where, D_{\max} and R_{\max} and the maximum denitrification and nitrogen regeneration rates, respectively.

Field studies in the Breton Sound estuary (Day et al., 2009) have shown that denitrification was low when the temperature was below 10 °C. At temperatures above 10°C, denitrification was well correlated with temperature. Denitrification was also controlled by

(a)



(b)

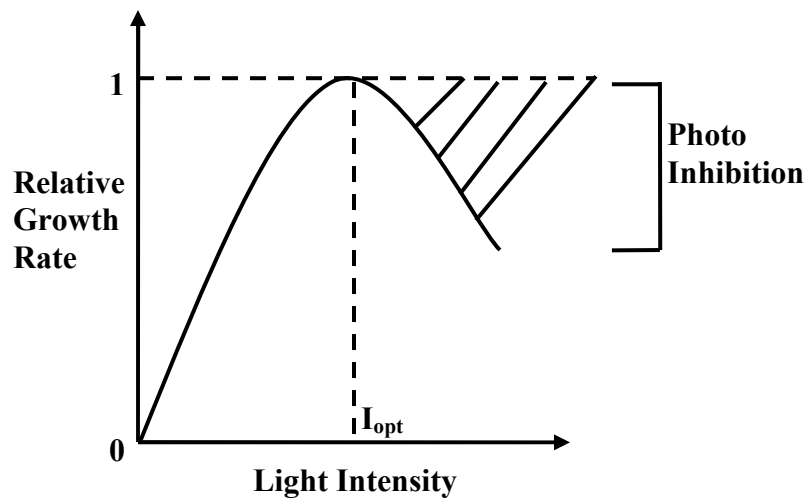


Figure 5.2. Phytoplankton growth in the model is simultaneously limited by (a) DIN concentrations and (b) ambient light intensity.

the ambient DIN concentrations (DeLaune, 2003; Miao, 2006; Day et al., 2009). Based on the laboratory denitrification studies on sediments collected in the Barataria estuary (Lindau et al., 2009), the maximum denitrification rate in the model (D_{\max}) was set at $4 \text{ mg N m}^{-2} \text{ h}^{-1}$. For Lake Cataouatche sediments, Lindau et al. (2009) found out that denitrification potential ranged from 1.8 to $3.9 \text{ mg N m}^{-2} \text{ h}^{-1}$ near the Davis Pond diversion structure, and from 1.25 to $2.36 \text{ mg N m}^{-2} \text{ h}^{-1}$ at other sites within the lake. Also, Miao et al. (2006) reported denitrification values ranging from 0.27 to $3.38 \text{ mg N m}^{-2} \text{ h}^{-1}$ depending on the amount of nitrate present.

The maximum benthic regeneration rate of inorganic nitrogen (R_{\max}) was set to $8.4 \text{ mg m}^{-2} \text{ h}^{-1}$, which was the mid-range value reported by Childers et al. (1999) for the Barataria marshes (0.9 - $19 \text{ mg N m}^{-2} \text{ h}^{-1}$).

Zooplankton grazing (G_z) was evaluated as a first order decay equation

$$G_z = k_G * \text{Chl } a \quad (8)$$

where, k_G was a grazing constant (0.015 per hour). The grazing rate used in the model is consistent with the zooplankton grazing rates reported for the northern Gulf of Mexico (Strom and Strom, 1996) and correspond to the low range of microzooplakton grazing rates reported for Fourleague Bay in Louisiana (Dagg, 1995).

Water Residence Times

The residence time refers to the rate of replacement or flushing of the water in the estuary. The residence times are affected by the magnitude of freshwater inflow, advection, diffusion and exchange processes at the estuary-ocean interface. The approach used here was to fill a given body of water with an imaginary tracer as an instantaneous pulse at a concentration of 100 units per grid cell. The residence time is then evaluated as the time it takes the tracer concentration to decline to e^{-1} times initial concentration ($= 63\%$ removal). This has been described by Miller and McPherson (1991) as the pulse residence time. Residence time is

calculated for the major water bodies in the estuary (Lac des Allemands, Lake Cataouatche, Lake Salvador, Perot and Rigolettes, Little Lake and Barataria Bay). To avoid the errors due to the spin-up time (Chapter 3), the tracer was injected 90 days after the start of the simulation. To increase the accuracy of my model estimates, the residence times were calculated for several grid points within each of the major water bodies (Figure 5.3).

Model Calibration

The model was calibrated using DIN and chlorophyll *a* measurements collected along the Barataria transect (Turner et al., unpublished data) as well as those collected at EMPACT stations (Turner et al., unpublished data and Dortch et al., unpublished data).

RESULTS

Chlorophyll *a* Concentrations

A comparison of the measured (Turner et al., unpublished data) and simulated chlorophyll *a* values along the Barataria transect shows that simulated values for the “Standard”, “Davis 2008” and “Coast 2050” model scenarios are generally higher than the measured values (Figures 5.4-5.12). The higher chlorophyll *a* values at stations 25-34, compared with the rest of the transect, are consistent with the observations. Further, there is little difference between the chlorophyll *a* values for the simulated “Standard”, “Davis 2008” and “COAST 2050” scenarios along the Barataria transect. In contrast, the spatial patterns in chlorophyll *a* concentrations are markedly different between the three investigated model scenarios and show increased chlorophyll *a* values in the proximity of freshwater diversion sites (Figures 5.13). For example, a comparison of different model scenarios for EMPACT stations in Lake Cataouatche (Figure 5.14) shows that simulated values for “Davis 2008” and “Coast 2050” scenarios are significantly higher ($p < 0.05$) when compared with the “Standard” scenario.

Residence Times

“Standard” Diversion Scenario

The model results showed large differences in water residence times at different stations within the Barataria estuary. In Lac des Allemands, for example, the two grid points in the proximity of Bayou Becnel and Bayou Fortier that are affected by the discharges from the sugarcane fields have a residence time of 7 days, (points 1 and 2, Figure 5.15 and 5.16). In contrast, the two grid points located near the middle of the lake (points 3 and 4, Figure 5.15) show a much longer residence time of 50 and 70 days, respectively. The residence times at the four grid points in Lake Cataouatche (Figures 5.17 and 5.18) show a more uniform residence

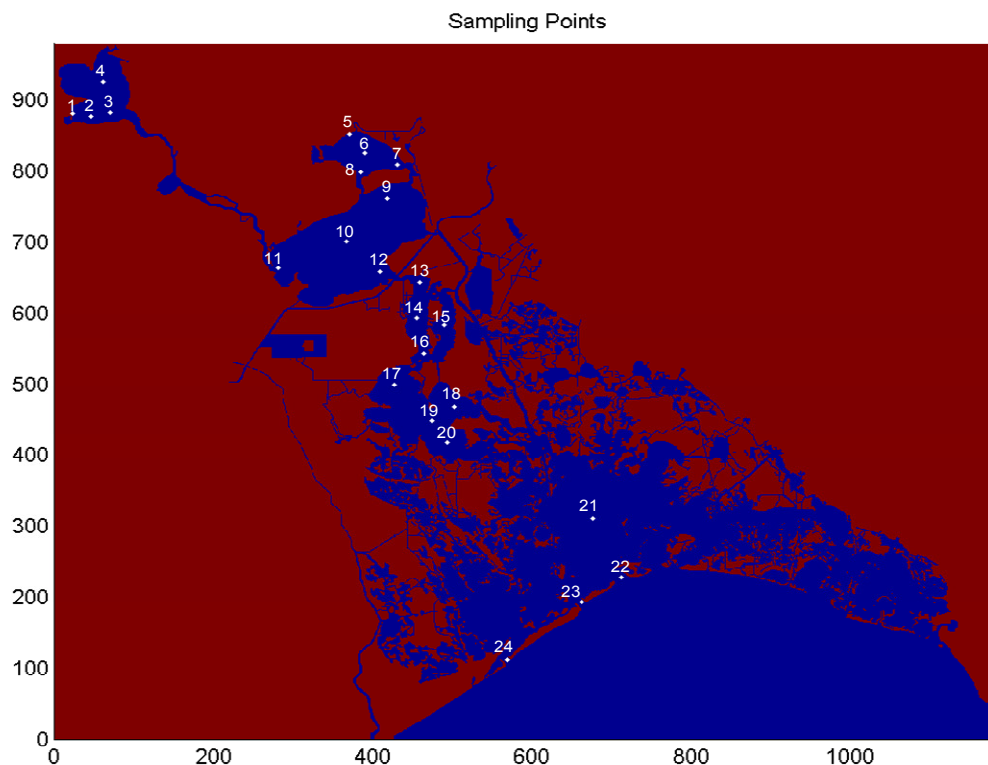


Figure 5.3. Model domain of the Barataria estuary showing the grid points (represented by numbered white dots) at which residence times were calculated.

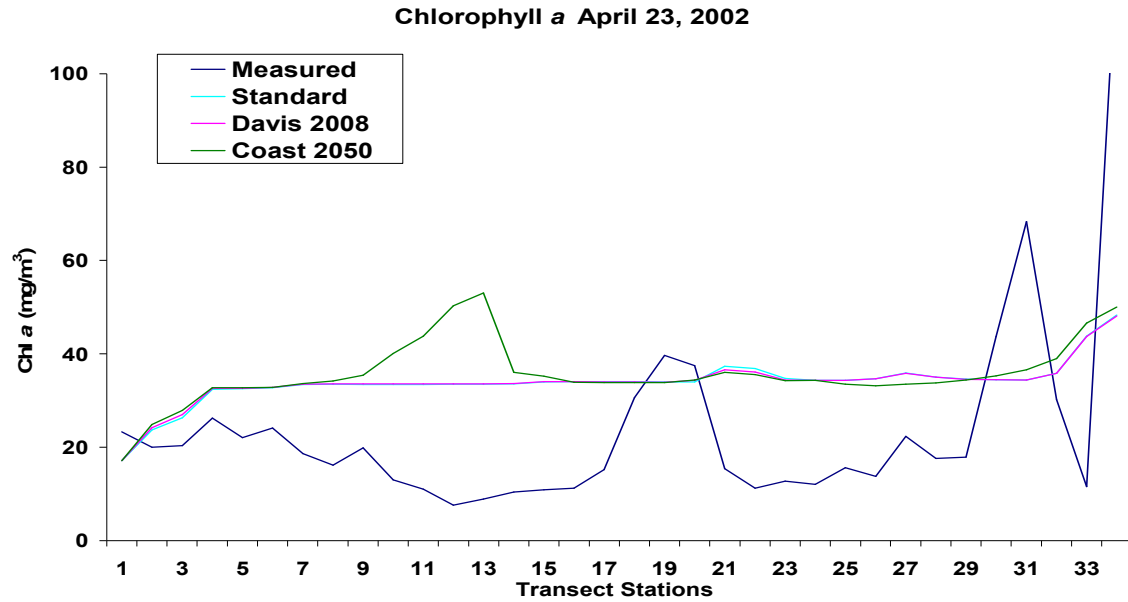


Figure 5.4. Measured and simulated chlorophyll *a* values for different Davis Pond fresh water diversion scenarios (Chapter 4, Table 4.2) for April 23, 2002. “Measured” denotes observed chlorophyll *a* values along the Barataria transect from south to north (Turner et al. unpublished data).

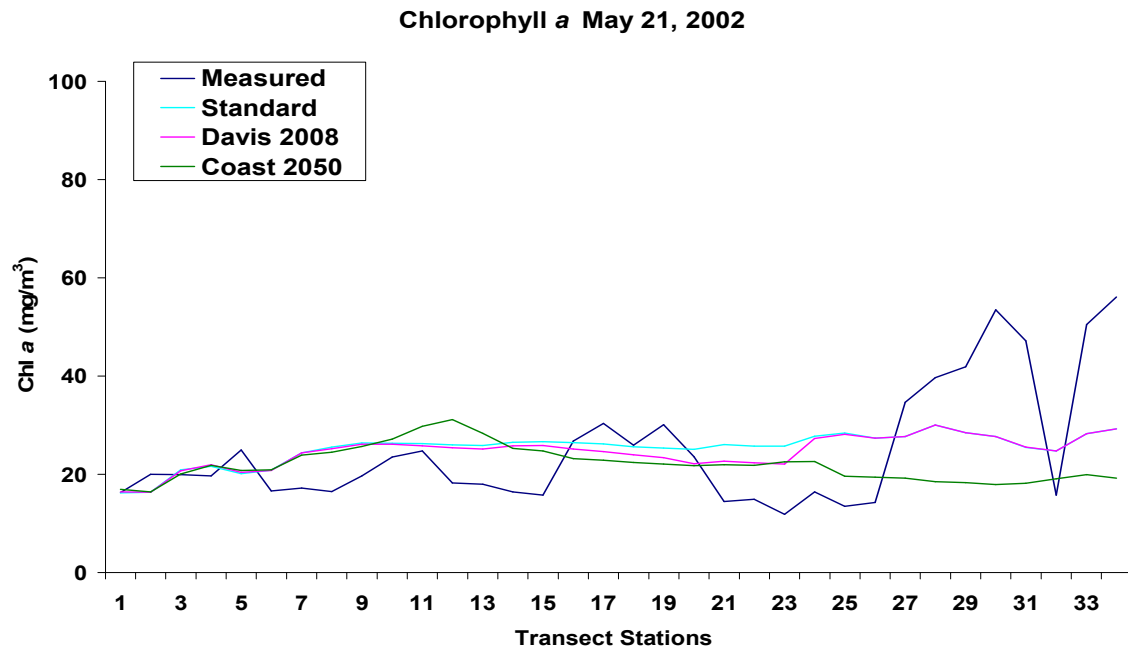


Figure 5.5. Measured and simulated chlorophyll *a* values for different Davis Pond fresh water diversion scenarios (Chapter 4, Table 4.2) for May 21, 2002. “Measured” denotes observed chlorophyll *a* values along the Barataria transect from south to north (Turner et al. unpublished data).

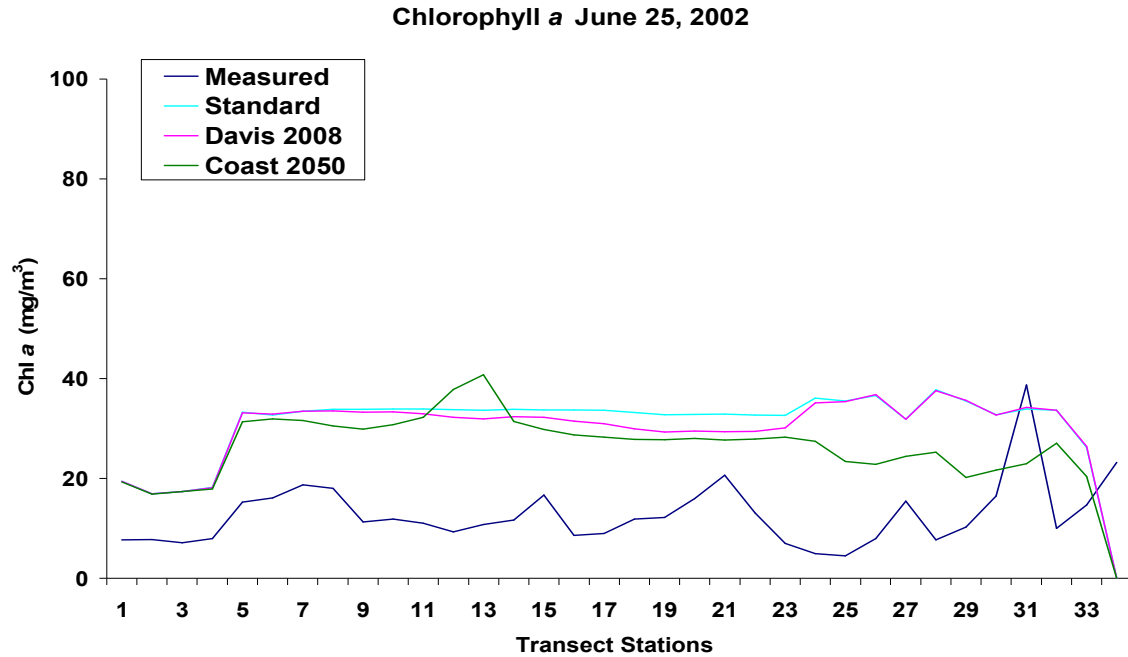


Figure 5.6. Measured and simulated chlorophyll *a* values for different Davis Pond fresh water diversion scenarios (Chapter 4, Table 4.2) for June 25, 2002. “Measured” denotes observed chlorophyll *a* values along the Barataria transect from south to north (Turner et al. unpublished data).

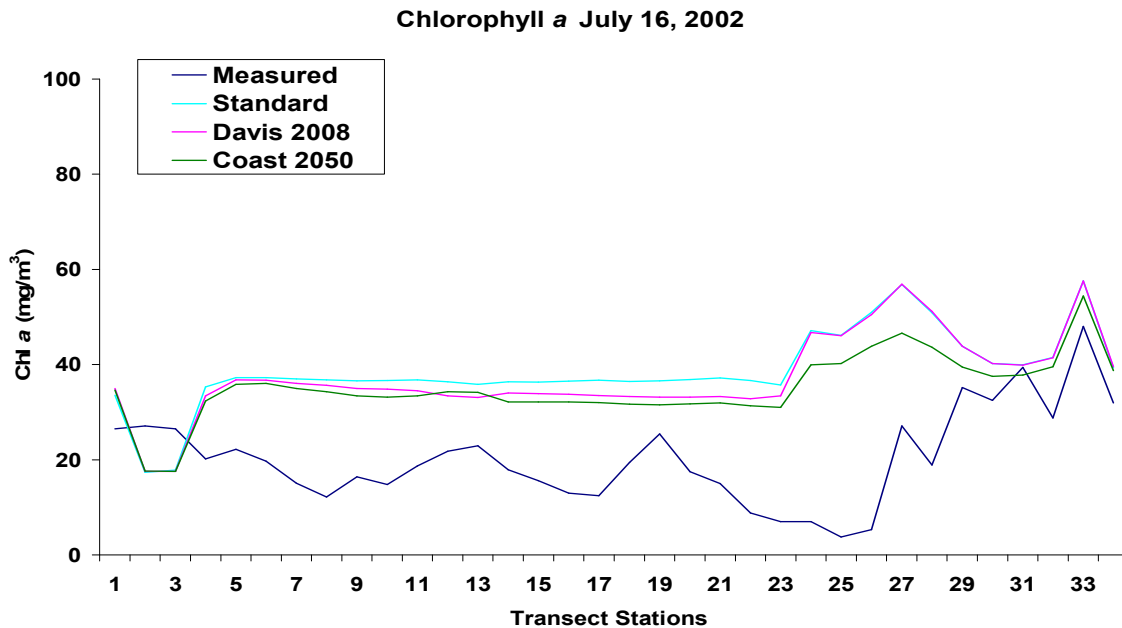


Figure 5.7. Measured and simulated chlorophyll *a* values for different Davis Pond fresh water diversion scenarios (Chapter 4, Table 4.2) for July 16, 2002. “Measured” denotes observed chlorophyll *a* values along the Barataria transect from south to north (Turner et al. unpublished data).

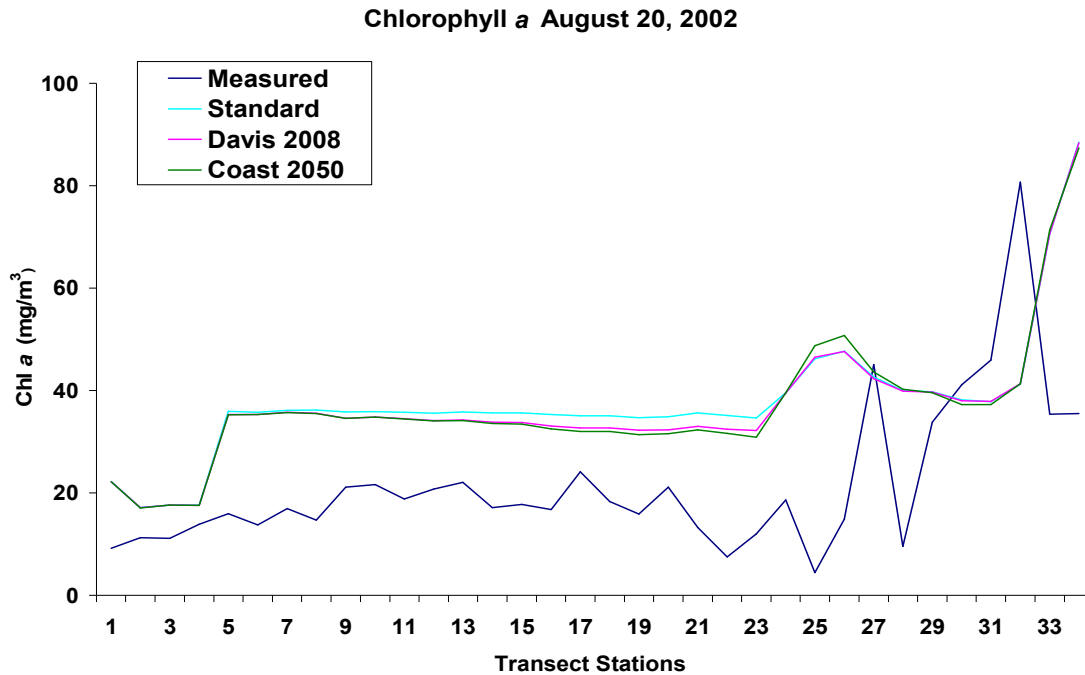


Figure 5.8. Measured and simulated chlorophyll *a* values for different Davis Pond fresh water diversion scenarios (Chapter 4, Table 4.2) for August 20, 2002. “Measured” denotes observed chlorophyll *a* values along the Barataria transect from south to north (Turner et al. unpublished data).

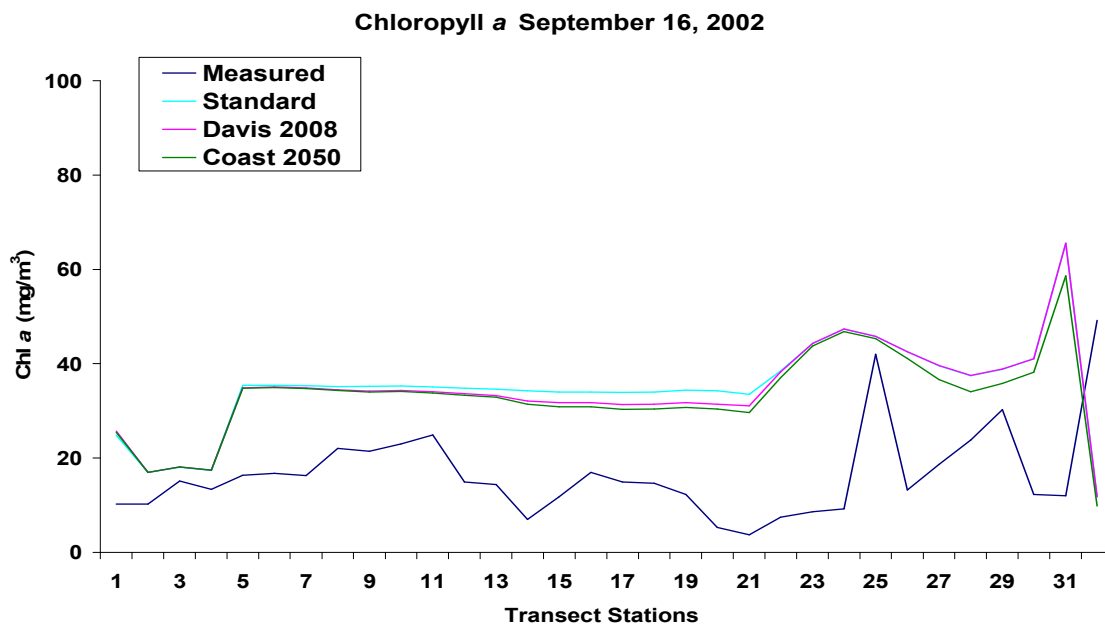


Figure 5.9. Measured and simulated chlorophyll *a* values for different Davis Pond fresh water diversion scenarios (Chapter 4, Table 4.2) for September 16, 2002. “Measured” denotes observed chlorophyll *a* values along the Barataria transect from south to north (Turner et al. unpublished data).

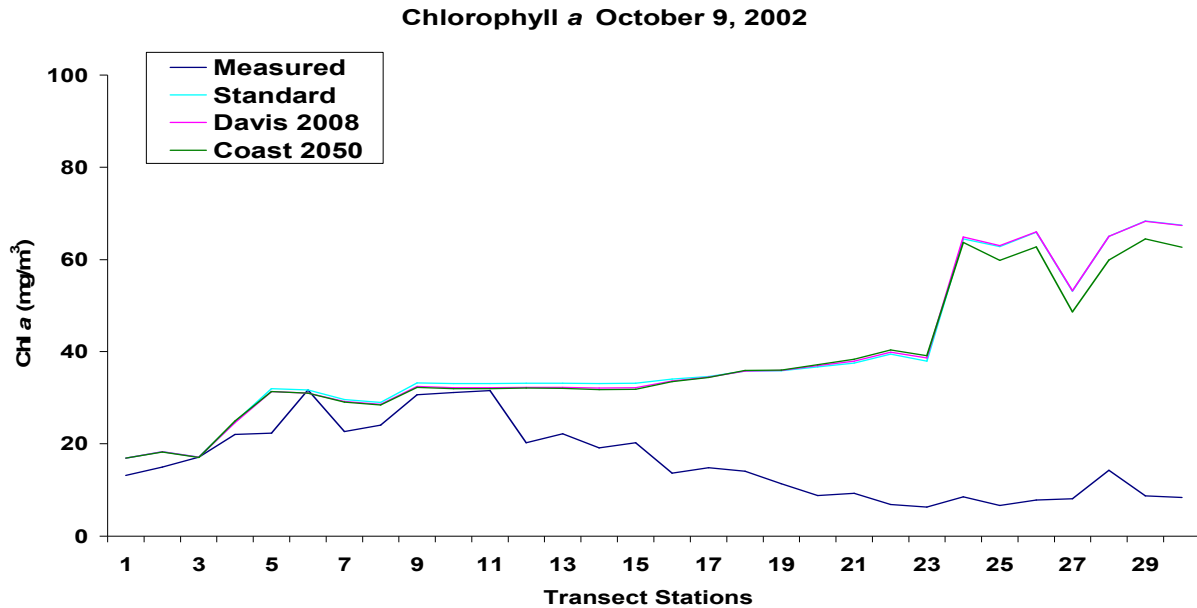


Figure 5.10. Measured and simulated chlorophyll *a* values for different Davis Pond fresh water diversion scenarios (Chapter 4, Table 4.2) for October 9, 2002. “Measured” denotes observed chlorophyll *a* values along the Barataria transect from south to north (Turner et al. unpublished data).

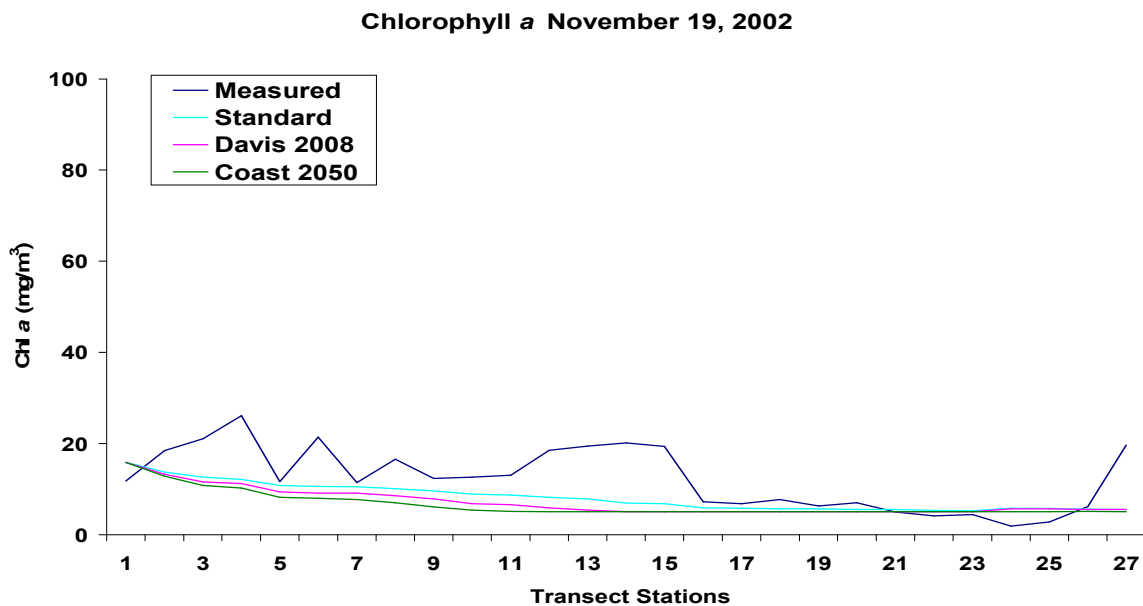


Figure 5.11. Measured and simulated chlorophyll *a* values for different Davis Pond fresh water diversion scenarios (Chapter 4, Table 4.2) for November 19, 2002. “Measured” denotes observed chlorophyll *a* values along the Barataria transect from south to north (Turner et al. unpublished data).

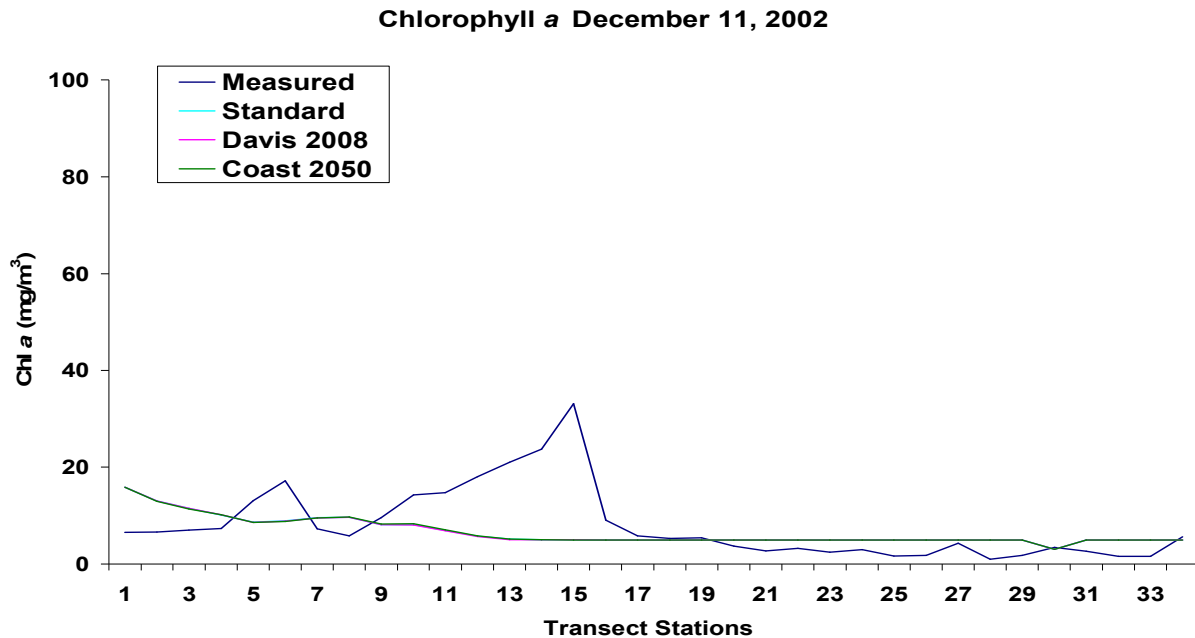


Figure 5.12. Measured and simulated chlorophyll *a* values for different Davis Pond fresh water diversion scenarios (Chapter 4, Table 4.2) for December 11, 2002. “Measured” denotes observed chlorophyll *a* values along the Barataria transect from south to north (Turner et al. unpublished data).

time of 100-120 days. The residence times in Lake Salvador average about 100 days (Figures 5.19 and 5.20). The exception is the area where Bayou des Allemands enters Lake Salvador (point 11, Figure 5.19) where the residence time is about 8 days due to the strong flushing from Lac des Allemands. Bayou Perot and Bayou Rigolettes have much shorter residence times ranging from 8-15 days (Figures 5.21 and 5.22), with residence times increasing from north to south (from Lake Salvador towards Little Lake). Residence times in Little Lake (Figures 5.23 and 5.24) range from 10-30 days. The station near Bayou Perot (Station 1, Figure 5.23) has the shortest residence time. Residence times in Barataria Bay (Figures 5.25 and 5.26) range from 1-4 days. Interestingly, the tracer concentration reaches 37% in about 4 days, but the tracer is periodically pushed back into the estuary by tidal forcing so that its concentration remains above 37% for 20-24 days.

“Davis 2008” Diversion Scenario

The calculated residence times for Lac des Allemands for the “Davis 2008” scenario are very similar those in the “Standard” scenario, except at point 4 where residence time increases by 20 days (Table 5.2). However, the “Davis 2008” residence times in Lake Cataouatche are about an order of magnitude lower compared to the “Standard” scenario. The residence times for Lake Salvador also show a decrease of 40-85 days with respect to the “Standard” scenario. All the stations in Bayou Perot show a slightly lower residence time compared to the “Standard” scenario (Table 5.1). All the grid points in Little Lake show a lower residence time compared to the “Standard” scenario by a few days. Residence times in the Barataria Bay are similar for the three diversion scenarios, except at grid point 24 (Caminada Pass, Figure 5.3) where the “Davis 2008” scenario show a longer residence time (6 days) than the “Standard” scenario (1 day, Table 5.2).

“Coast 2050” Diversion Scenario

The calculated residence across the entire model domain are very similar the “Davis 2008” and “Coast 2050” and differ only by a few days between the two scenarios. The calculated residence times for Lac des Allemands for the “Coast 2050” scenario are very similar those in the “Standard” scenario, except at point 4 where residence time decreases by 10 days (Table 5.2). However, the “Coast 2050” residence times in Lake Cataouatche are about an order of magnitude lower compared to the “Standard” scenario. The residence times for Lake Salvador also show a decrease of 40-85 days from the “Standard” scenario, except at grid point 11 (which is near Bayou des Allemands, Figure 5.3). All the stations in Bayou Perot show a slightly lower residence time compared to the “Standard” scenario (Table 5.2). All the grid points in Little Lake show a lower residence time compared to the “Standard” and “Davis 2008” scenarios by a few days. The Barataria Bay shows no change in residence times between the three diversion

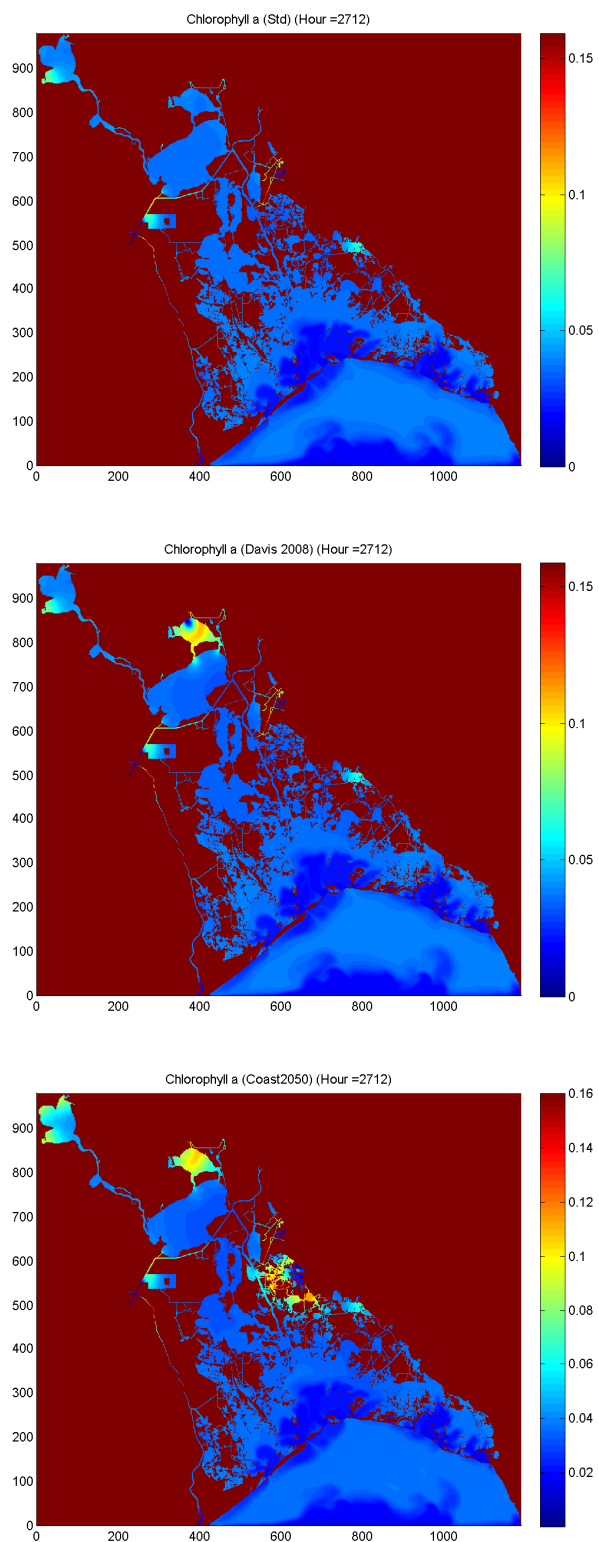


Figure 5.13. Contour plots of chlorophyll *a* concentrations for the “Standard” (upper panel), “Davis 2008” (middle panel) and “Coast 2050” (lower panel) for hour 2712.

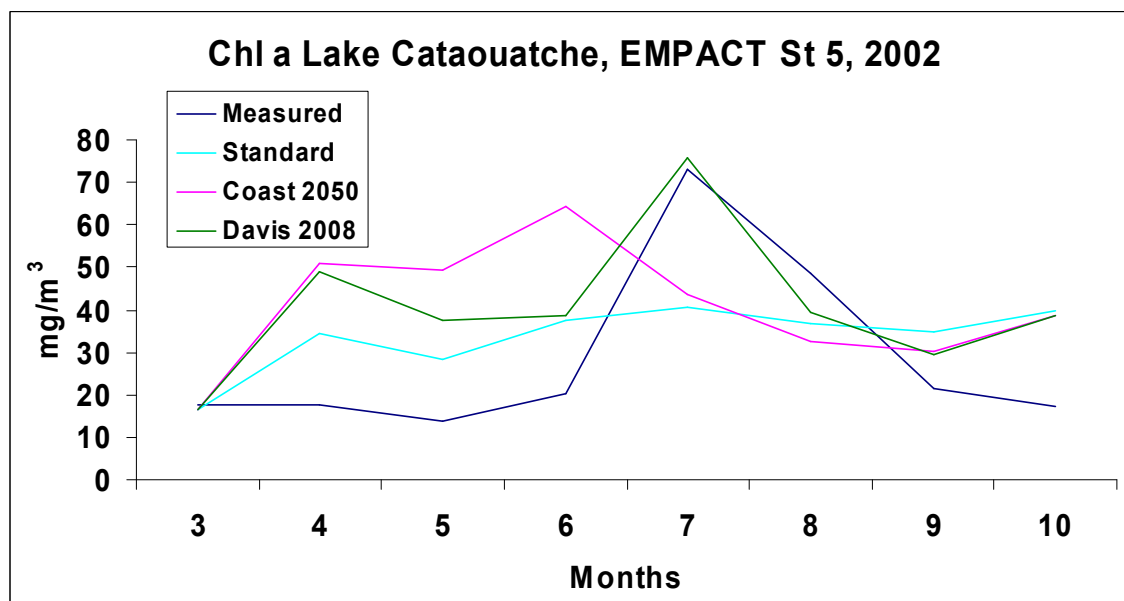
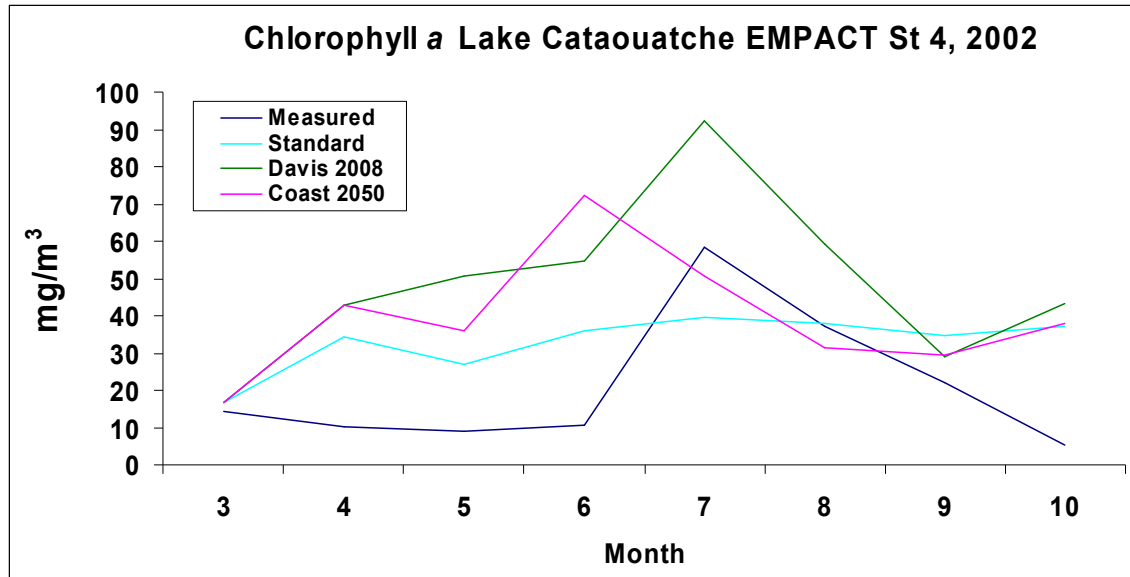


Figure 5.14. Measured and simulated chlorophyll a values for different Davis Pond fresh water diversion scenarios (Chapter 4, Table 4.2) for Lake Cataouatche from March to October, 2002. “Measured” denotes observed salinity values at the EMPACT station number 4 (upper panel) and EMPACT station number 5 (lower panel) (Turner et al., unpublished data).

scenarios, except at grid point 24 (Caminada Pass, Figure 5.3) where the “Coast 2050” and “Davis 2008” scenarios show a longer residence time (6 days) than the “Standard” scenario (1 day, Table 5.2).

The model results indicated that the travel time of the tracer (based on tracer concentration of 1%) from Lac des Allemands to the oyster beds in the middle and lower estuary (Figure 5.27) was 88, 46, and 30 days for the “Standard”, “Davis 2008” and “Coast 2050” scenarios, respectively (Figures 5.28 and 5.29).

DISCUSSION

Predicting Water Quality Changes in Response to Varying Freshwater Inflows

The nutrient-phytoplankton-zooplankton (NPZ) model is one of the simplest models that can be used to describe plankton dynamics (Franks, 2002). These types of models have few parameters and hence can be easily parameterized with existing data than more complicated models. They also have few state variables (N, P, and Z) and these can also be easily initialized and compared with observed values (Franks, 2002). These types of models are useful in providing values that are fairly representative of actual ecosystem dynamics and are good predictors of phytoplankton biomass, rates and derived quantities (Franks, 2002). NPZ models have been coupled to 1-D physical models (Evans and Parslow, 1985; Denman and Gargett, 1995; Edwards, et al., 2000b), 2-D physical models (Evans et al., 1977; Franks et al., 1986b; Klien, 1987) and 3-D physical models (Yoshimori and Kishi, 1994; Chen et al., 1997; Franks and Chen, 2000). The coupling can be done through advection-diffusion equations, depth-dependant irradiance (including self-shading), or temperature dependant biological dynamics (Franks, 2002). NPZ models, as part of coupled physical-biological models, have been used to answer a variety of questions. It has been used in qualitative studies to answer hypothetical question like, how different transfer functions affect model behavior (Franks et al., 1986a; Ruan,

2001), how different parameters affect model behavior (Ruan, 1993; Edwards, et al., 2000a), how patchiness of plankton distribution is affected by the interaction of vertical migration with vertical shear (Evans et al., 1977; Evans, 1978), what factors control the size of phytoplankton (Steele and Frost, 1977), and annual plankton cycles (Evans and Parslow, 1985; Denman and Gargett, 1995). Many quantitative studies have also been done, like simulating phytoplankton biomass and nutrient uptake during a bloom (Droop, 1983; Franks et al, 1986a; Marra and Ho, 1993), it has also been used to study changes in phytoplankton dynamics due to closed circulation and tidal forcing (Lewis et al., 1994; Franks and Chen, 1996, 2000), and mesoscale eddies (Yoshimori and Kishi, 1994). Most NPZ models have been used in hindcasting data and in general have not been used as a predictive tool (Franks, 2002).

The model results for all the simulation scenarios over-predicted the chlorophyll *a* values when compared to the measured values. This is not surprising given the fact that the model represents a very simplified view of phytoplankton dynamics in the Barataria Bay. There are several possible reasons for the discrepancy between the observed values and the model results for the “Standard” scenario. First, the model does not take into account atmospheric nitrogen deposition and nitrogen fixation. Second, the model does not include DIN uptake by submerged aquatics and marsh plants, and therefore overestimates the overall DIN pool that is available to phytoplankton. Third, the model does not take into account wetting and drying of adjacent marshes. For example, the model grid does not cover the 3,760 ha Davis Pond diversion ponding area to the north of Lake Cataouatche (Chapter 1). DeLaune et al. (2005) found that this ponding area could remove all the nitrates from the Mississippi River when the flow was about $35 \text{ m}^3 \text{ s}^{-1}$, but could not remove all the nitrates when the flow was greater than $100 \text{ m}^3 \text{ s}^{-1}$. However, Gardner (2008) suggested that when the diversion discharge is less than $205 \text{ m}^3 \text{ s}^{-1}$, wetlands in the ponding area may be able to remove most of the nitrate in the diverted Mississippi water.

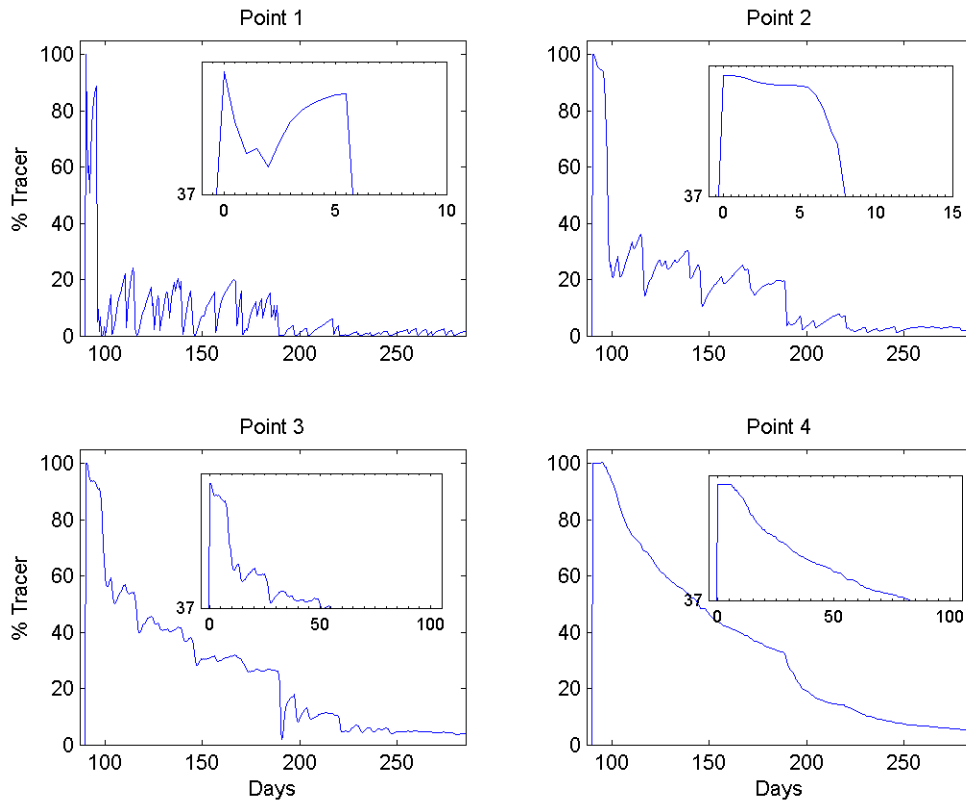


Figure 5.15. Residence times at four points in Lac des Allemands for the “Standard” scenario. Insert figures show the time it takes the tracer concentration to reach the value of 37 %.

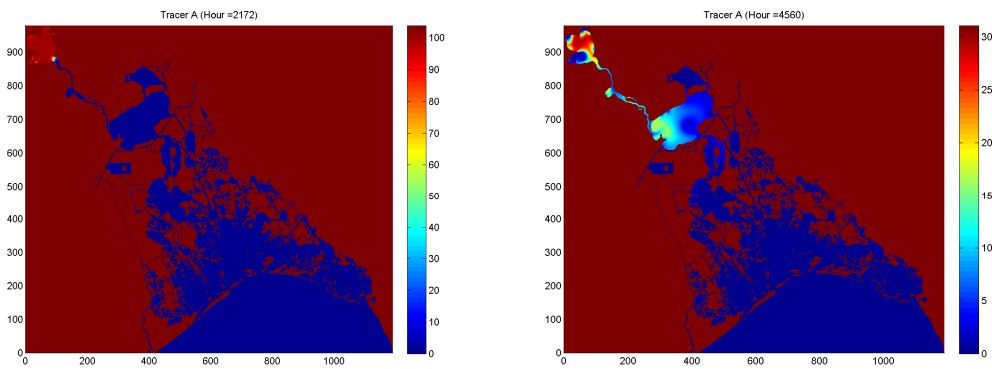


Figure 5.16. Contour plots for tracer simulation for the “Standard” scenario showing the initial concentration of particles at the beginning of the simulation (left) and the approximate time the tracer concentration reaches 37% in Lac des Allemands.

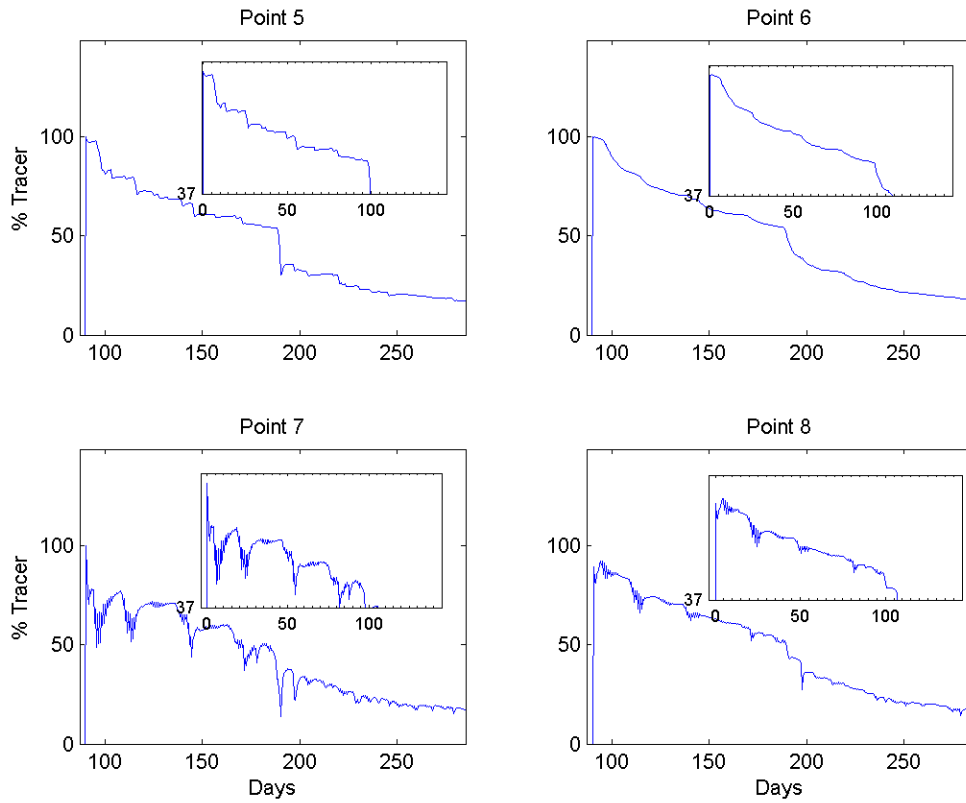


Figure 5.17. Residence times at four positions in Lake Cataouatche for the “Standard” scenario. Insert figures show the time it takes the tracer concentration to reach the value of 37 %.

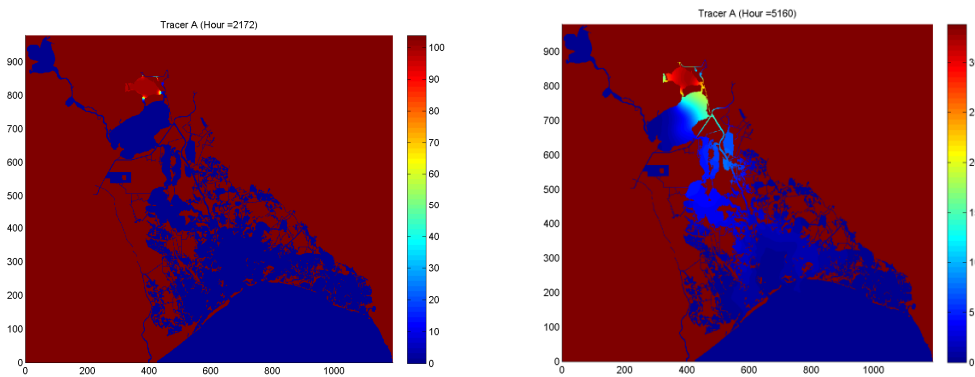


Figure 5.18. Contour plots for tracer simulation for the “Standard” scenario showing the initial concentration of particles at the beginning of the simulation (left) and the approximate time the tracer concentration reaches 37% in Lake Cataouatche.

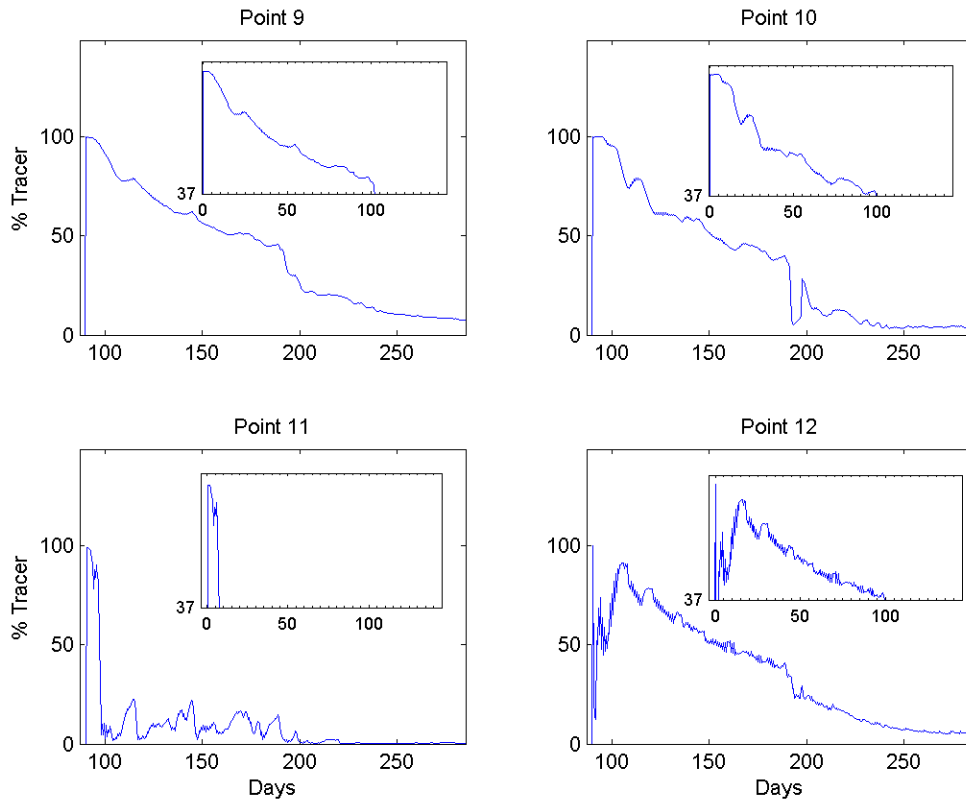


Figure 5.19. Residence times at four positions in Lake Salvador for the “Standard” scenario. Insert figures show the time it takes the tracer concentration to reach the value of 37 %.

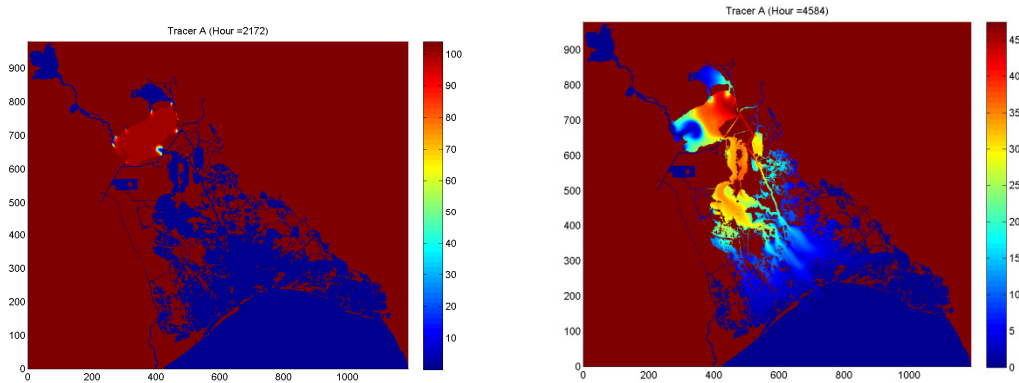


Figure 5.20. Contour plots for tracer simulation for the “Standard” scenario showing the initial concentration of particles at the beginning of the simulation (left) and the approximate time the tracer concentration reaches 37% in Lake Salvador.

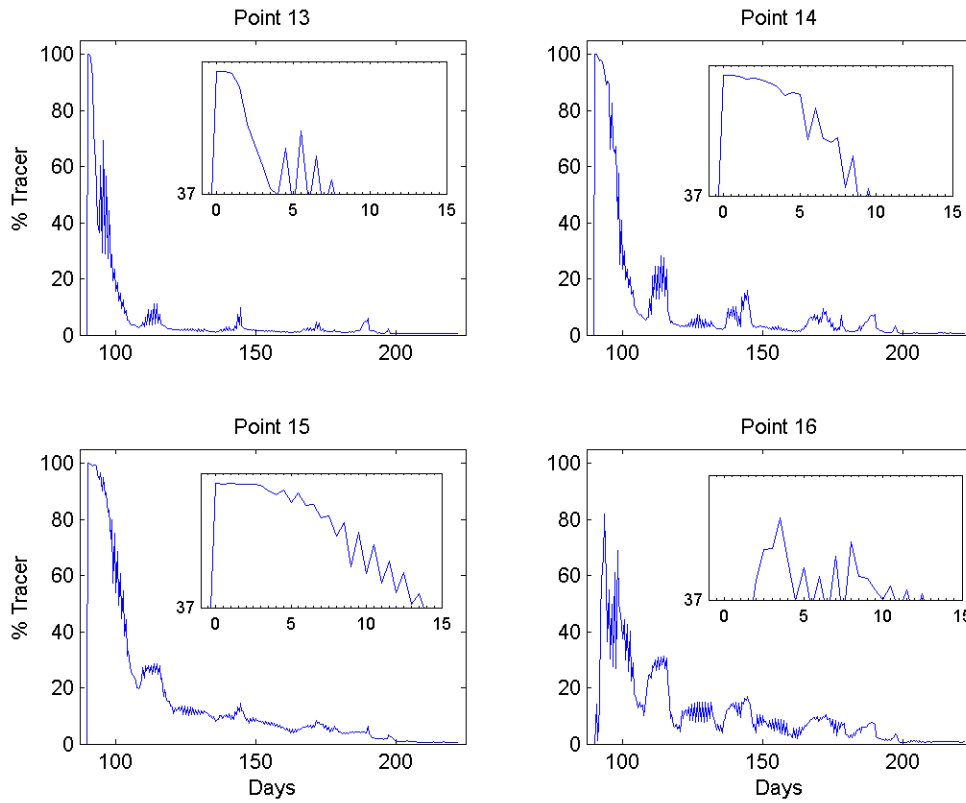


Figure 5.21. Residence times at four positions in Bayou Perot and Rigolettes for the “Standard” scenario. Insert figures show the time it takes the tracer concentration to reach the value of 37 %.

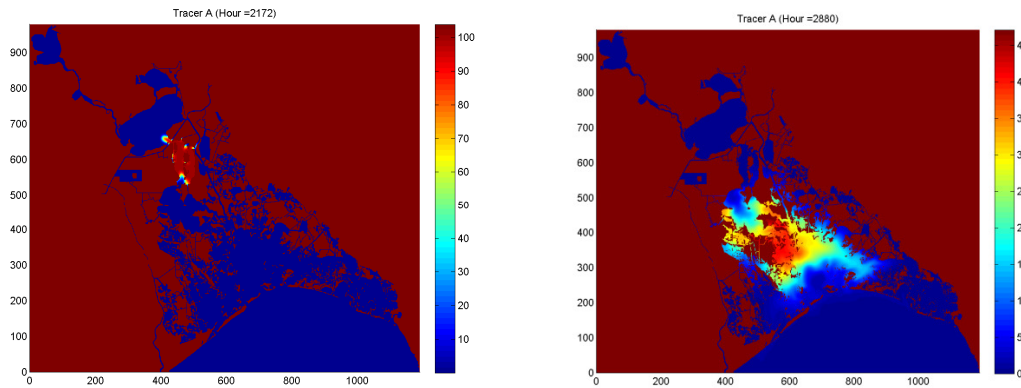


Figure 5.22. Contour plots for tracer simulation for the “Standard” scenario showing the initial concentration of particles at the beginning of the simulation (left) and the approximate time the tracer concentration reaches 37% Bayou Perot and Rigolettes.

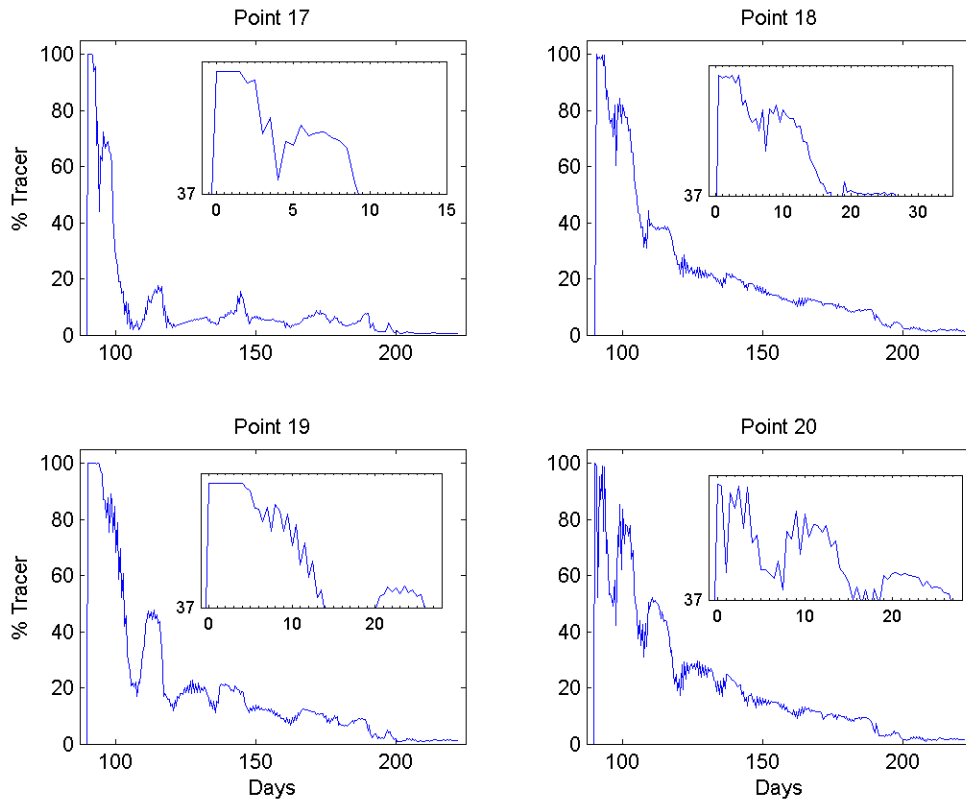


Figure 5.23. Residence times at four positions in Little Lake for the “Standard” scenario. Insert figures show the time it takes the tracer concentration to reach the value of 37 %.

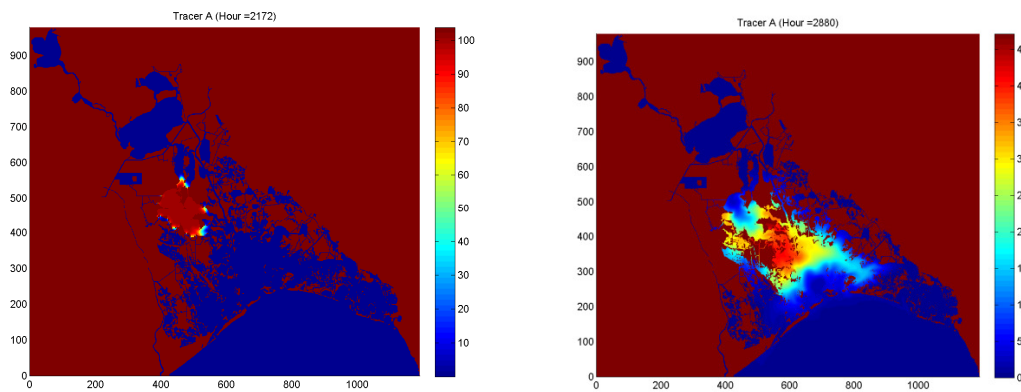


Figure 5.24. Contour plots for tracer simulation for the “Standard” scenario showing the initial concentration of particles at the beginning of the simulation (left) and the approximate time the tracer concentration reaches 37% in Little Lake.

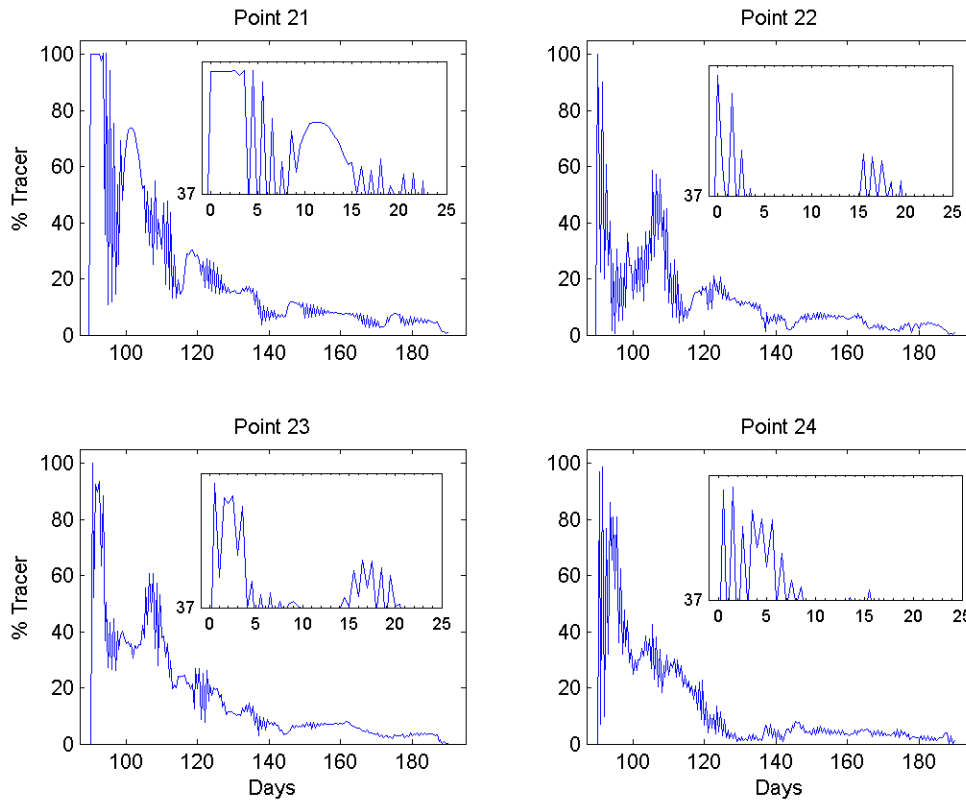


Figure 5.25. Residence times at four positions in Barataria Bay for the “Standard” scenario. Insert figures show the time it takes the tracer concentration to reach the value of 37 %.

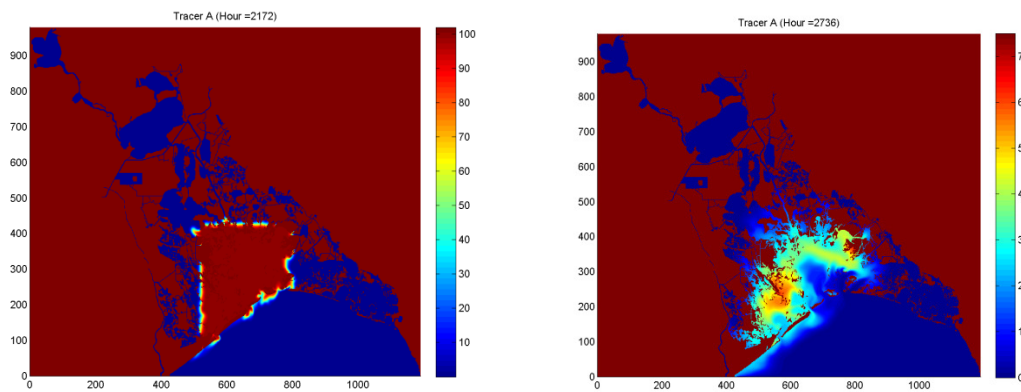


Figure 5.26. Contour plots for tracer simulation for the “Standard” scenario showing the initial concentration of particles at the beginning of the simulation (left) and the approximate time the tracer concentration reaches 37% in Barataria Bay.

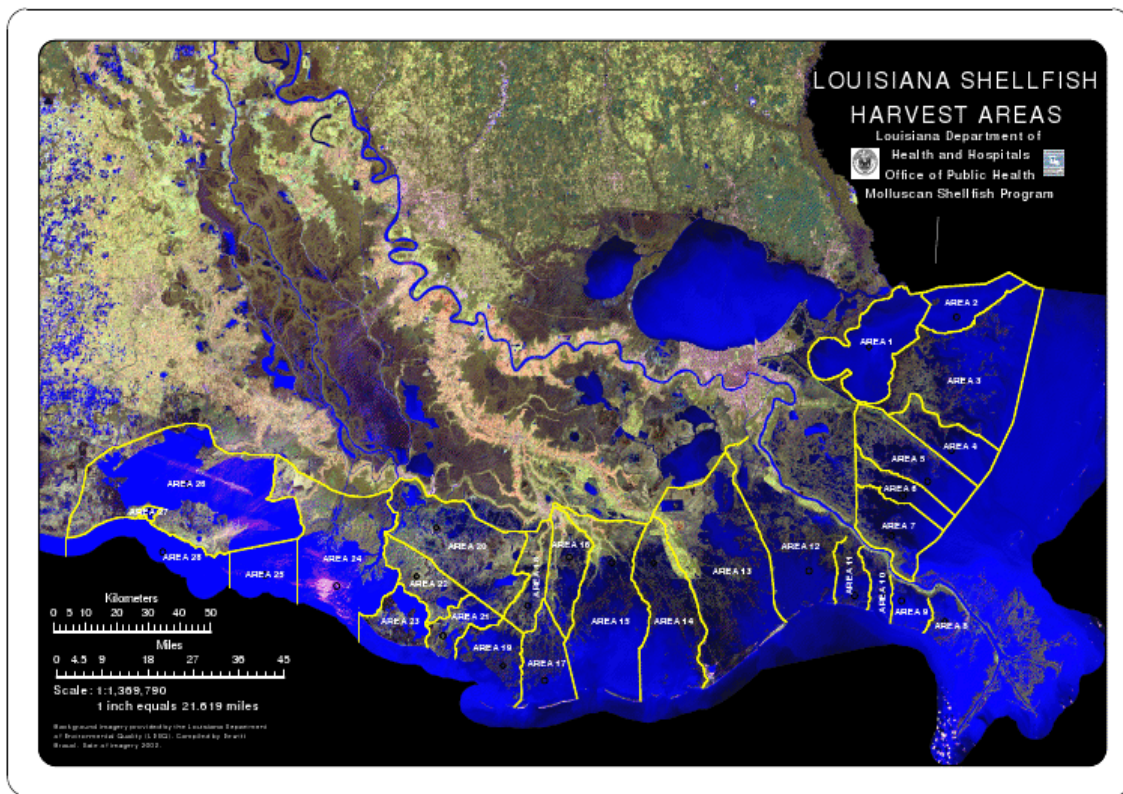


Figure 5.27. Position of the oyster leases in the Barataria estuary (Source: LDWF).

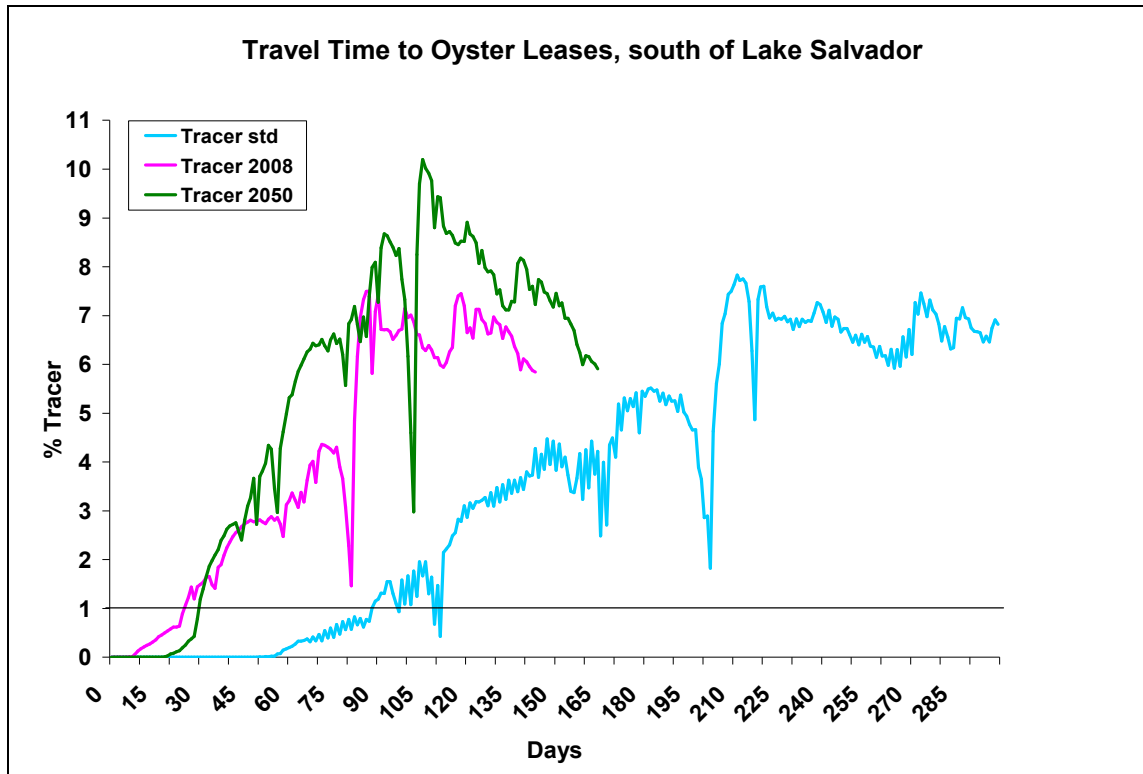


Figure 5.28. Tracer travel time from Lac des Allemands to oyster beds at the junction of Lake Salvador and Bayou Perot (Figure 5.27) for the different diversion scenarios.

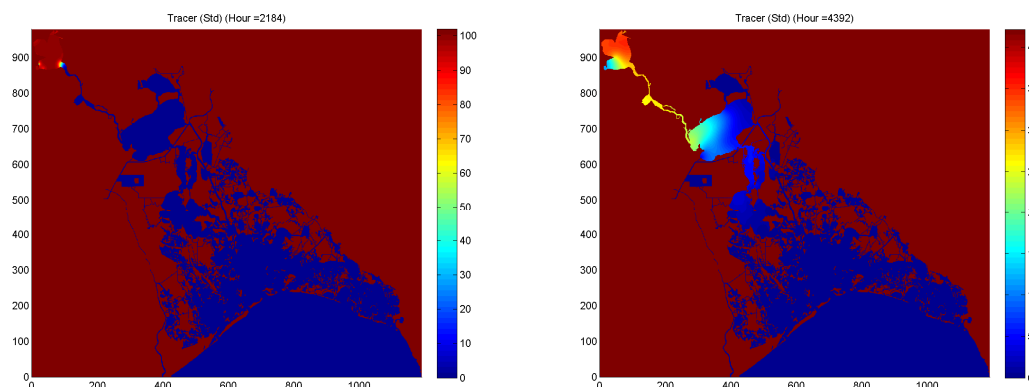


Figure 5.29 Tracer travel from Lac des Allemands to reach oyster beds at the junction of Lake Salvador and Bayou Perot for the “Standard” scenario. The time difference between the two figures is 2106 hours or 88 days.

Table 5.2. Residence times (days) at different points in the Barataria estuary (Figure 5.3) for the “Standard”, “Coast 2050”, and “Davis 2008” scenarios.

	Point	“Standard”	“Davis 2008”	“Coast 2050”
Lac des Allemands	1	7	6	6
	2	7	8	8
	3	50	55	50
	4	70	90	60
Lake Cataouatche	5	100	1	1
	6	120	8	6
	7	100	15	14
	8	110	12	11
Lake Salvador	9	100	15	15
	10	100	60	65
	11	8	7	4
	12	100	65	65
Bayou Perot and Rigolettes	13	8	3	3
	14	9	7	7
	15	14	12	9
	16	13	9	9
Little Lake	17	9	4	3
	18	18	14	13
	19	24	11	10
	20	28	14	12
Barataria Bay	21	4	5	4
	22	1	1	1
	23	4	4	4
	24	8	6	6

Fourth, the NPZ model assumes that the maximum denitrification potential is the same across the entire model domain. In reality, denitrification and nutrient regeneration rates in the bottom sediments vary across the estuary depending on local sediment characteristics and nutrient conditions (DeLaune et al., 2003, 2005; Lindau et al., 2009; Rivera-Monroy et al., 2010). Fifth, macrobenthic grazing of phytoplankton may also be an important parameter controlling

phytoplankton biomass in the upper sections of the Barataria estuary. A recent study by Wong et al. (2010) found that the clearance times of *Rangia cuneata* in Lac des Allemands, Lake Cataouatche and Lake Salvador are high enough to influence phytoplankton biomass in these lakes, especially during times of low nutrient loading, immediately after high fresh water inflows and in shallow zones of the lakes.

Comparison of simulated chlorophyll *a* values along the Barataria transect (Figures 5.4 - 5.12) showed that phytoplankton biomass remains fairly constant when the estuary is subjected to higher freshwater discharges. This was an unexpected result given the fact that riverine nutrient concentrations are at least one order of magnitude higher than those in the estuary. On the other hand, the contour plots of the different simulation scenarios (Figure 5.14) clearly show elevated chlorophyll *a* concentrations near the diversion sites for the “Coast 2050” and “Davis 2008” scenarios compared to the “Standard” run. It appears that riverine nutrients are predominantly being utilized to fuel phytoplankton blooms in the vicinity of freshwater diversions and they are used up before the water reaches the Barataria transect stations. The slight decrease in chlorophyll *a* values along the Barataria transect for the “Davis 2008” and “Coast 2050” scenarios compared to the “Standard” is likely due to the decreased residence times resulting from higher freshwater inflows. This illustrates the complex spatial patterns in phytoplankton response to distributed freshwater and nutrient inflows, reflecting possibly the near-field control of nutrients and far-field control of residence times on phytoplankton standing stock.

Because of the high complexity of the 2-D model grid, the simulated spatial patterns in chlorophyll *a* distribution could not have been fully verified against the observed values. Nevertheless, the chlorophyll *a* values for Lake Cataouatche for the “Standard” scenario show low values during the spring and high values during the fall, consistent with the pattern observed

in the measured chlorophyll *a* concentrations (Figure 5.13). Ren et al. (2008) found increased chlorophyll *a* levels and cyanobacteria dominated phytoplankton communities in Lake Cataouatche and Lake Salvador in the spring and summer of 2003 and 2004. They concluded that even limited outflows from Davis Pond are affecting the salinity and nutrient regimes in these water bodies which are in turn affecting the phytoplankton community structure in these lakes.

The fact that stations positioned along the Barataria transect do not capture the changes in phytoplankton biomass that is occurring near the freshwater diversion sites, brings up the issue of the difficulty in establishing a monitoring framework in a complex estuary like the Barataria Bay, and also illustrates the problem of providing adequate calibration data for large ecosystem models. Also, in our simulation experiments, the “Coast 2050” scenario was introduced only over a single annual cycle without prior conditioning to a higher freshwater and nutrient inflow. In reality, if the “Coast 2050” scenario is to be introduced, the Barataria estuary will be subjected to a strong diversion influence over many months and years, so there will likely be cumulative diversion effects on phytoplankton biomass. Unfortunately, due to a large computational time (72 hours) that is required to simulate a single annual cycle of phytoplankton biomass, such a multi-annual scenario could not have been explored in this study.

The Importance of Residence Times

Residence time in an estuary is defined as the time it takes for any parcel of water to leave the estuary through its outlet to the sea. Residence times of substances in estuaries are important indices for predicting how physical, chemical, and biological processes that depend on water contact time would behave or change under different types of hydrologic and hydrodynamic scenarios. Before complex computer models were developed, the residence time was calculated either using the freshwater fraction method or the tidal prism method. These

simple residence time calculations do not take into account all the factors that can influence the residence time of a substance such as stratification, multiple inflow and outflow points, channelization, mixing, stirring, and dispersion, and the complex geomorphology of estuaries that can trap a substance at any time through its passage through the estuary (Nuttall et al., 2008). Such complexity can be addressed to some extent by numerical models. Residence time calculations have been carried out with models of differing complexity, including box-models, particle models and concentration models. Box-models are the simplest to set-up, requiring little information on the system and small computational resources. The residence time of the pollutants in each box can be computed based on the knowledge of the geometry of the system, the hydrodynamics and the pollution loads (Hagy et al. 2000). Calculations of residence times based on this method are limited by the capability of box models to solve the underlying hydrodynamics (due to their inability to resolve smaller spatial scales), which are generally complex in coastal systems. Particle models are by far the most popular approach to compute residence time. Residence times are computed based on the release of large numbers of particles, scattered throughout the domain of interest, at several release times within the tidal cycle and for different tidal amplitudes. For example, Park (1998) used neutrally-buoyant particle tracers (they are subject to the hydrodynamics of the system, not the hydrology) to calculate residence times in the different water bodies in the Barataria estuary. A certain number of particle tracers were released and tracked over a 10 day simulation period. The residence times were calculated by fitting equations of particles removed/particles deployed against time for each water body studied, and then extrapolating the time it takes for the fraction of removed particles to reach the value of $1 - e^{-1}$ (approximately 63%).

Concentration models (transport models) have also been used for residence time calculations, which is the approach adopted in this study. For these models, residence time can

be defined as the time necessary to reduce the initial pollutant concentration by $(1 - e^{-1})$ or by some other arbitrarily chosen fraction (Mashriqui and Justić, in preparation). The accuracy of the residence time calculations can be limited by numerical errors (mass imbalances, numerical diffusion) and the use of the concentration method cannot be used for several simultaneous sources of the same contaminant. Because of complex geomorphology of the Barataria estuary, complex 2-dimensional hydrodynamic model was a *conditio sine qua non* to examine detailed transport processes and estimate water residence times. Tracers were released at a particular concentration throughout a particular water body and allowed to be influenced by both the hydrology and hydrodynamics of the system. This is a more practical approach to calculating residence times and is directly applicable to constituents such as phytoplankton, which are influenced by both hydrology and hydrodynamics. Also, as seen in Figures 5.15-5.26, even points that are located within the same water body and close to each other can have different residence times due to differences in small scale hydrodynamics resulting from differences in geomorphology and patterns of freshwater inflow.

Studies have shown that the occurrence of phytoplankton blooms may be influenced by estuarine residence times. Boyer et al. (1997) statistically analyzed a 6-year multiparameter dataset of the water quality of the Florida Bay-Whitewater Bay area and found that chlorophyll *a* abundance was mainly a function of differences in freshwater inputs and residence times. Howarth et al. (2000) showed that primary production and eutrophication increased during low freshwater discharges from the watershed into the Hudson River estuary which caused longer residence times and greater stratification. Ho et al. (2010) found that phytoplankton production and biomass varied with the Pearl River discharge in tropical Hong Kong waters, being low in highly flushed parts of the waters and highest in parts with high stratification and low flushing. Phlips et al. (2010) examined 10 years of water quality data for the Indian River Lagoon, Florida,

and found that the northern part of the lagoon that had the longest residence times, had higher occurrence of algal blooms and most of these blooms were potentially toxic dinoflagellate and diatom blooms. Residence times also affect denitrification rates, as shown by Dettman (2001). He modeled the effects of residence times on denitrification in estuaries and showed that loading rate of nitrogen, water residence time and estuarine volume were controlling factors in determining nitrogen concentration in the water column.

The impact of freshwater diversions on higher trophic levels was outside the scope of this study. Nevertheless, because the oyster leases in the Barataria estuary start just below Lake Salvador and extend to the mouth of the estuary (Figure 5.27), there is the question about whether these oyster beds can be affected by the cyanobacterial blooms that routinely occur in Lac des Allemands (Dortch et al., 1999; Dortch et al., 2001; Rabalais 2005; Garcia et al., 2010) and Lake Cataouache (Ren et al., 2009). Tracer experiments indicated that it takes about 88, 46, and 30 days for a tracer simulating an algal bloom in Lac des Allemands to travel to the oyster lease areas for the “Standard”, “Davis 2008” and “Coast 2050” scenarios, respectively (Figures 5.28 and 5.29). So, the higher freshwater discharge into Lac des Allemands and Lake Cataouache will likely decrease the travel time and increase the fraction of algal biomass exported to oyster lease areas to the south of Lake Salvador (Figure 5.28).

Coupled Hydrodynamic-Water Quality Models as Coastal Restoration Tools

A prominent characteristic of the shallow estuaries, typical in the northern Gulf of Mexico, is their complex geomorphology. Consequently, any particle or tracer being advected through these estuarine systems is likely to be trapped at some point, giving rise to significant enhancement of mixing, stirring and dispersion (Inoue and Wiseman, 2000). For that reason, high-resolution models are required to examine detailed transport processes and effectively address the effects of pulsed riverine inflows on estuarine hydrodynamics, salinity and water

quality. This study, focused on the Barataria estuary, has shown that the 2-D model was able to simulate the complex patterns of phytoplankton biomass distributions, by taking into account the key controlling factors that included temperature, residence times, internal and external DIN sources and sinks, and zooplankton grazing. Also, model simulations clearly demonstrated the importance of residence times for the overall functioning of the estuary. Model simulations pointed out the differences in spatial patterns in phytoplankton response to distributed freshwater and nutrient inflows, reflecting possibly the near-field control of nutrients and far-field control of residence times on phytoplankton standing stock. The model reiterates the fact that there are significant tradeoffs in using freshwater diversions in coastal restoration efforts, namely tradeoffs between hydrologic restoration and water quality effects.

The coupled hydrology-hydrodynamic-water quality model that was used in this study can serve as a versatile tool to estimate salinity, residence times and concentrations of various constituents on a very fine scale (~100 m). The model can provide managers with reasonable forecasts of future conditions and different tradeoffs involved in selecting a particular freshwater diversion scenario. Increasing the complexity of the NPZ model and refining the forcing functions would improve model calibration and allow the model to be used in examining the impacts of changing anthropogenic and climatic conditions such as the number, location and size of freshwater diversions, nutrient loading, riverine discharge, rainfall, storminess and sea level rise.

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CHAPTER 6

SUMMARY

River diversions in Louisiana are a major part of a greater effort to restore wetlands. The Davis Pond diversion is one such diversion that is being used to reintroduce fresh water, nutrients and sediment from the Mississippi River into the Barataria estuary. Freshwater diversions are thought to stimulate marsh growth by providing sediments and nutrients, and to supply iron that can precipitate toxic sulfides accumulating in marsh soils (DeLaune et al, 2003). The potential problems associated with freshwater diversions, are increased eutrophication, the potential for more harmful algal blooms (Turner et al., 2004), bioaccumulation of heavy metals, and possible weakening of marsh substrate (Darby and Turner 2008a, b; Swarzenski et al., 2008; Day et al., 2009).

My dissertation research concerns the effects of freshwater diversions on hydrodynamics, salinity and water quality of the Barataria estuary in Louisiana. Using two simulation models of differing complexity, a simple box model (BOX) and a complex 2-dimensional model (2-D), I attempted to answer the following four questions:

1. How are the diversions affecting estuarine-shelf exchanges of water, nutrients and carbon?
2. How are freshwater diversions from the Mississippi River affecting the salinity and the residence times of the various water bodies in the Barataria estuary?
3. How are the diversions affecting phytoplankton productivity and standing stock?
4. Will the estuary become increasingly eutrophic if Davis Pond discharge is increased or additional diversions are implemented?

The simple BOX model is a variation of a tidal prism model that calculates volumes and water level variations in response to hydrodynamic and hydrologic forcings. The model domain

was divided into six boxes that correspond to major water bodies in the Barataria estuary: Barataria Bay (Box 1), Little Lake (Box 2), Bayou Perot-Rigolettes (Box 3), Lake Salvador (Box 4), Lake Cataouatche (Box 5) and Lac des Allemands (Box 6). This box model was used to calculate the fluxes of water, nitrogen and carbon through the Barataria passes, and to estimate the importance of estuarine derived nitrogen and carbon for the overall carbon budget and development of hypoxia in the northern Gulf of Mexico. The model results showed that the Barataria estuary receives nitrogen through the tidal passes and releases carbon (“outwelling”, Odum, 1980) to the coastal ocean. The mean calculated tidal water discharge of $6,930 \text{ m}^3 \text{ s}^{-1}$ is equivalent to about 43 % of the lower Mississippi River discharge. The annual TOC export is 109 million kg, or $57 \text{ gC m}^2 \text{ yr}^{-1}$ when prorated to the total water area of the estuary. This carbon export is equivalent to loss of 0.5 m of wetland soil horizon over an area of 8.4 km^2 , and accounts for about 34 % of the observed annual wetland loss in the estuary between 1978 and 2000. Compared to the lower Mississippi River, the Barataria estuary appears to be a very small source of TOC for the northern Gulf of Mexico (2.7 % of riverine TOC), and is unlikely to have a significant influence on coastal carbon budgets and development of the Gulf’s hypoxia.

The high resolution two-dimensional (2-D) coupled hydrology-hydrodynamic water quality model was developed to describe the spatial patterns in salinity, water residence times and phytoplankton biomass in response to different freshwater diversion scenarios. The hydrodynamic part of this model was originally developed by Inoue et al. (2008), and it has been refined and coupled to a water quality model within this study. The 2-D model results supported the findings obtained by using a simpler BOX model, i.e., the “outwelling” of carbon and “inwelling” of nutrients, and reaffirmed the hypothesis that the outwelling of the total organic carbon (TOC) is small compared to the Mississippi River TOC load and likely has little impact on coastal carbon budgets and development of the Gulf’s hypoxia.

The 2-D model was used to examine how the reintroduction of different amounts of fresh water from freshwater diversions affects hydrodynamics, salinity and water quality in the Barataria estuary. The selected diversion scenarios were based on observed diversion discharges and proposed management options (Chapter 4, Table 4.2). The simulated salinity values for different model scenarios were compared with the measured salinities from the Barataria transect (Turner et al., unpublished data). The “Standard” scenario uses the actual Davis Pond discharge data for 2002, when the diversion started operating. The simulation without any diversion “Davis off” was used to provide an idea about background salinities in the Barataria estuary without any freshwater input, other than the siphons and the Intracoastal Waterway. The “Davis Max” scenario was used to examine the extent of Davis Pond influence under conditions of maximum theoretical discharge. The “Davis 2008” scenario was used to examine conditions during an actual high-flow year. Finally, the “Coast 2050” scenario was used to examine conditions that may be encountered in the future with several new diversions that have been recommended by the LCA final report (LCA, 2004).

The effects of different diversions on salinity are most apparent in the middle and lower sections of the Barataria estuary (stations 8-26, Chapter 4, Table 4.3). The upper parts of the estuary are almost always fresh and the excess fresh water from Davis Pond and other proposed diversions will have little impact on the salinity in that area. Also, the Davis Pond discharge, even when the diversion is running at maximum capacity, has little impact on salinities in the lower Barataria Bay (stations 1-7; Chapter 4, Table 4.3). This is likely because of strong marine influence in this region adjacent to the Gulf of Mexico. The “Davis 2008” and “Coast 2050” are very similar as far as the Davis Pond discharge is concerned (Chapter 4, Table 4.3), so the observed system response during 2008 could be used to assess the effectiveness of the “Coast 2050” scenario.

A simple nutrient-phytoplankton-zooplankton (NPZ) model was coupled to the 2-D hydrology-hydrodynamic model to simulate temporal and spatial patterns of phytoplankton distribution for different diversion scenarios. Toxic cyanobacterial blooms are frequently observed in the Barataria estuary (Dortch et al., 1999; Dortch et al., 2001; Rabalais, 2005) and there are concerns that excess nutrients from diverted Mississippi River can further enhance the growth of cyanobacteria.

The 2-D model was able to reproduce the complex patterns of phytoplankton biomass distributions, by taking into account multiple controlling factors that included temperature, residence times, internal and external DIN sources and sinks, and zooplankton grazing. Nevertheless, the model overestimated the observed chlorophyll *a* values most of the time, which is in part due to the absence of benthic grazers that are likely important in controlling phytoplankton biomass in the upper sections of the estuary. Model simulations clearly demonstrated the importance of residence times for the overall functioning of the estuary. Model simulations also pointed out the differences in spatial patterns in phytoplankton response to distributed freshwater and nutrient inflows, reflecting the near-field control of nutrients and far-field control of residence times on phytoplankton standing stock. The model reiterates the fact that there are significant tradeoffs in using freshwater diversions in coastal restoration efforts, namely tradeoffs between hydrologic restoration and water quality effects.

A prominent characteristic of shallow estuaries, typical in the northern Gulf of Mexico, is their extremely complex geomorphology. Consequently, any particle or tracer being advected through these estuarine systems is likely to be trapped at some point, giving rise to significant enhancement of mixing, stirring and dispersion. The high-resolution of the 2-D model made it possible to examine detailed transport processes and effectively address the effects of pulsed riverine inflows on estuarine hydrodynamics, salinity and water quality. The coupled hydrology-

hydrodynamic-water quality model that was used in this study can serve as a versatile tool to estimate salinity, residence times and concentrations of various constituents on a fine scale (~100 m). The model can provide managers with reasonable forecasts of future conditions and different tradeoffs involved in selecting a particular freshwater diversion scenario. Increasing the complexity of the NPZ model(e.g., by including benthic grazers) and refining the forcing functions would allow the model to be used in examining the impacts of changing anthropogenic and climatic conditions such as the number, location and size of freshwater diversions, nutrient loading, riverine discharge, rainfall, storminess and sea level rise.

Based on the overall model results, the following answers to the four research questions can be proposed:

1. How are the diversions affecting estuarine-shelf exchanges of water, nutrients and carbon?

Estuarine-shelf exchanges in the Barataria estuary are driven primarily by water level variations at the open boundary. The average modeled annual flux of water through the tidal passes is $\sim 7,000 \text{ m}^3 \text{ s}^{-1}$, which is over twenty times higher than the maximum Davis Pond discharge. Thus, freshwater diversions have little impact on the overall estuarine-shelf exchanges in the Barataria estuary.

2. How are freshwater diversions from the Mississippi River affecting the salinity and the residence times of the various waterbodies in the Barataria estuary?

The effects on salinity due to different diversion discharges are apparent only in the middle sections of the Barataria estuary. Discharges from the Davis Pond, even when the diversion is running at maximum capacity, have little impact on salinities in the lower Barataria Bay. The residence time decreases as freshwater discharge increases leading to a shorter travel time of dissolved constituents in the estuary.

3. How are the diversions affecting phytoplankton productivity and standing stock?

Riverine nutrients are likely utilized to fuel phytoplankton blooms in the vicinity of freshwater diversion sites and their far-field impact on phytoplankton biomass seems to be limited. The far-field control on phytoplankton standing stock is exerted primarily through diversion influence on water residence times in the estuary.

4. Will the estuary become increasingly eutrophic if Davis Pond discharge is increased or additional diversions are implemented?

The higher freshwater discharge into Lac des Allemands (if diversions into this lake are implemented) and Lake Cataoutche will likely increase the potential for algal blooms in the vicinity of diversion sites and increase the fraction of algal biomass exported to oyster lease areas to the south of Lake Salvador.

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VITA

Anindita Das is a citizen of India. She stays in the city of Calcutta, West Bengal, India. She obtained her basic education from various schools in different parts of India. She graduated high school from Loreto House, Calcutta, India. She then acquired her undergraduate and graduate degree in zoology from the University of Calcutta, India. She completed a diploma in environmental management from the Indian Institute of Social Welfare and Business Management, Calcutta, India. She started her master's program in the planning and management concentration in the Department of Environmental Studies at Louisiana State University in 2001 and earned her Master of Science degree in 2003. She will earn a Doctor of Philosophy degree from the Department of Oceanography and Coastal Sciences at LSU in December, 2010.