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Scientific and management perspectives in wetland groundwater hydrology

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SCIENTIFIC AND MANAGEMENT PERSPECTIVES IN WETLAND GROUNDWATER HYDROLOGY

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To the most wonderful family in the whole world,
Mom, Pat, Claudia, Sonia and Lirlene

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Abstract

Wetland groundwater hydrology was investigated at different spatial scales to assess the usefulness of the information to coastal managers. Specific objectives were to: (1) review studies related to coastal groundwater discharge, evaluating techniques and identifying controls; (2) understand regional trends in groundwater flow along the U.S. East and Gulf coasts; (3) evaluate the applicability of naturally occurring radioisotopes as indicators of groundwater in a shallow deltaic system; and (4) evaluate groundwater-surface water exchange within Barataria Basin, Louisiana. Results of the review confirmed discharge estimates vary over several orders of magnitude, due to differences in precipitation and tidal prisms. In addition, in very few areas such as the Sippewissett Marshes, Massachusetts, groundwater information was extensive enough for management planning. A regional water budget study performed for 29 watersheds along the East and Gulf of Mexico indicate a likely insignificant net annual (30-year average) export of groundwater in the northeast (17 cm). However, a net import was found in the southeast (10 cm), and eastern Gulf coast (14 cm). The mid-Atlantic and western Gulf showed no net groundwater flow. In Barataria Basin estuary, ^{222}Rn increased exponentially from the mouth to 120 km upstream. Significant excess ^{222}Rn activities suggest that an additional source is required to balance the geochemical budget, such as groundwater. Radium-226 activities demonstrated non-conservative mixing, and appear to indicate an additional source at intermediate salinities, likely desorption of ^{226}Ra from suspended and bottom sediments. Further evaluation of groundwater in Barataria was performed using more in depth experiments with radioisotopes. Tracer mass balance estimates of SGD at three sites produced a range in

SGD flux of 1.6 to 9.6 cm/d, with the highest groundwater flux at Kenta Canal. A comparison of results from the water budget and mass balance for Barataria Basin confirm that the water budget was the lower estimate of SGD. Coastal managers can utilize SGD information in planning as better estimates and consistent techniques become available. In Barataria Basin, a further study of stratigraphy, groundwater flows, and SGD-derived nutrients is necessary if Louisiana's coastal planners are to fully understand the hydrology and resulting impacts on its wetlands.

Chapter 1

The Importance of Groundwater Hydrology Research at Global, Regional and Local Scales

1.1 Introduction and Significance

Submarine groundwater discharge (SGD) is defined as the upward flow of water across the sediment water interface. Early recognition of this phenomenon was the result of vigorous, low-salinity upwelling associated with offshore springs and boils in karstic systems (Brooks, 1961), but another slower process of SGD identified as diffuse seepage into surface waters over large intertidal areas is also important (Kohout, 1966; Manheim, 1967). This SGD term also includes cycling of porewater within shallow sediments as well as recycled seawater (Simmons et al., 1990). Thus, the definition here includes flow of both fresh and brackish water. Salinity measurements associated with offshore SGD may be so high they mask the presence of any freshened inputs. Some scientists further expand the definition of SGD to include high salinity flows as well (Simmons et al., 1992). An additional complication of defining SGD relates to the process of movement – large-scale advection as with springs, smaller scale seepage, and dispersed movement. Regardless of the mechanism, resulting subsurface inputs to surface waters are considered SGD in the current body of marine science literature.

Groundwater flow occurs under conditions where a hydraulic gradient exists, sloping from upland areas downstream and the underlying aquifer is permeable enough to allow vertical and/or horizontal water movement (Johannes, 1980). Along coastlines, groundwater typically flows seaward along hydraulic gradients. The groundwater mixes with recirculating seawater near the land-sea interface, and the resulting discharge from the sediments usually reflects the ratio between fresh, new groundwater inputs from land

and recirculated seawater from coastal waters. Should the gradient along the coastline slope toward land, salt water can intrude into the aquifer. Consequently, leakage can occur at the coast, regardless of the direction of the hydraulic gradient. Further offshore SGD may also be present and, at times, affect such a wide area that community assemblages within the vicinity may move further away from the freshened outputs (Brooks, 1961). Thus, occurrence of SGD is widespread, from offshore springs and seeps, to coastal leakage, and within many coastal ecosystems, such as salt marshes and other wetland systems (Valiela et al., 1978).

Little doubt exists that precipitation drives the recharge of aquifers and the offshore flow of SGD. For flow to persist, aquifer recharge rates must be high enough to maintain hydraulic gradients (Kasenow, 2001). In offshore environments, wave pumping and tidal influences also drive saltwater shoreward and downward into sediments changing the SGD constituents. However, additional influences with site-specific implications for affecting hydraulic gradients aquifer pumping for drinking water and surrounding land use.

The study of SGD is important for a number of reasons: (1) its seaward flow may help prevent salt water intrusion into coastal aquifers; (2) its contribution to global, regional, watershed, and local water budgets may be more significant than previously thought; and (3) it may act as an efficient conduit for dissolved constituents, such as nutrients, pesticides and other contaminants, increasing the likelihood of coastal eutrophication (Johannes, 1980). For these reasons, groundwater has been studied in major oceans and seas, embayments, estuaries and associated wetlands, and along coastlines (Zekster et al., 1973; Valiela et al., 1978).

Early techniques developed to study groundwater/surface water interactions included salinity measurements, where the study of mixing processes facilitated the identification and origin of mixed water bodies. Later, benthic flux chambers, previously used for nutrient studies, were incorporated into subsurface fluid research to provide an estimate of total benthic flux (Martens et al., 1980). More recently radioisotopes have been successfully used to estimate the fraction of total benthic flux due to advection (Burnett et al., 1990). As new techniques are developed, the error associated with flux estimates is reduced.

1.2 Dissertation Objectives

As coastal management around the world becomes more complex due to increased coastal population, scientific information regarding neglected aspects of hydrology is required for sound management planning and implementation. Thus, the overall purpose of this research project is to investigate groundwater hydrology in wetlands at different spatial scales, and to assess the utility of this relatively new science for coastal managers around the world. The specific research objectives are to:

(1) Review studies related to SGD to compare flux values, techniques and controls;

(2) Examine regional trends in groundwater flow within watersheds along the United States (U.S.) eastern and Gulf coasts;

(3) Assess the utility of using radioactive tracers as indicators of groundwater flow within shallow water systems; and

(4) Determine the relative significance of groundwater flow within Barataria Basin.

First, I present a review of studies (Chapter 2) with a SGD flux or an SGD-derived nutrient flux, include a case study, and suggest ways that scientists and managers might facilitate the use of such information in planning. In Chapter 3, I present results of a regional study of impacts of SGD and compared results to published values. For a single watershed included in the regional study, Barataria Basin, I explore subsurface/surface interactions based on cycling of two naturally occurring radioisotopes (Chapter 4). Finally, I present values of groundwater flux for a small site within Barataria Basin, based on mass balance estimates (Chapter 5).

1.3. Description of Chapters

1.3.1 Groundwater inputs to coastal ecosystems: A comparative review of available science with management requirements – The objectives of Chapter 2 are to: (1) present a brief summary of the history and evolution of wetland groundwater research; (2) present a review of wetland and open water studies that include an estimate of flux or an estimate of groundwater-derived nitrogen flux; (3) discuss SGD controls tested in the studies; and (4) present a case study to demonstrate the utility of groundwater information for coastal managers. More than 50 studies were selected from peer - reviewed journals to conduct this review, and for comparison, studies were divided into two groups – within coastal wetland ecosystems and in open water systems. Five studies were also included which compared SGD flux using different measurement techniques.

1.3.2 Hydrologic evaluation of U.S. Coastal Watersheds in the Eastern and Gulf Coasts – The research objectives presented in Chapter 3 are: (1) to evaluate water budgets for 29 coastal watersheds; (2) to estimate the net import or export of groundwater within these watersheds; and (3) to compare those results with measured groundwater fluxes from published studies in the same watershed. Using U.S. Geological Survey well water level data, average change in storage was calculated for each watershed using a minimum of five wells distributed throughout the watershed. An annual water budget was then calculated for each watershed to estimate annual net groundwater. Thirty-year averages for precipitation, evapotranspiration, area-weighted stream discharge, estimated change in storage, and estimated net annual groundwater flow were included in a 30-year water budget for each watershed.

1.3.3 Evaluation of ^{222}Rn and ^{226}Ra cycling in a deltaic estuary of the Mississippi River – The objectives of Chapter 4 are: (1) to evaluate spatial and temporal patterns in geochemical tracers; (2) to test the utility of naturally occurring radioactive tracers to measure SGD within a shallow deltaic water system; and (3) to understand the subsurface hydrology of Barataria Basin using tracer systematics. Nine sampling trips were taken over a two-year period, and eight stations were sampled for ^{222}Rn and ^{226}Ra activities both in sediments and in the overlying water column. Results were then evaluated relative to Mississippi River stage, salinity, precipitation, and wind speed to understand the movement of these tracers, including the relative role of atmospheric evasion in removing ^{222}Rn gas at the water surface.

1.3.4 Implications for groundwater/surface water exchange within a deltaic wetland using ^{226}Ra and ^{222}Rn – The objectives of Chapter 5 are: (1) to assess the loss of ^{222}Rn gas across the air-sea interface; (2) to determine whether observed water column ^{222}Rn inventories could be explained by diffusion across the sediment-water interface; (3) to present a preliminary evaluation of groundwater significance in the watershed based on excess ^{222}Rn activities; and (4) to determine whether a subsurface hydraulic connection to the Mississippi River exists in this region of the estuary. Mass balance estimates of groundwater flux were carried out for three sites within Jean Lafitte National Park using data collected over six seasons. SGD was calculated as the difference between the observed water column inventory (corrected for atmospheric evasion) and sediment diffusive flux.

Chapter 2

Groundwater Inputs to Coastal Ecosystems: A Comparative Review of Available Science with Management Information Requirements

2.1 Introduction

2.1.1 Background - Wetlands are now considered to be one of the most important coastal ecosystems on earth, for many reasons, including their potential to support biodiversity. In addition, their role in wastewater treatment and storm surge mitigation is now well established (Mitsch and Gosselink, 2000). Accordingly, greater funding has become available within the last four decades for interdisciplinary research around the world. Science has only recently (in the last two decades) forged an awareness of the importance of groundwater information to effective wetland management.

Groundwater can be a significant source of dissolved constituents to surface wetlands, affecting water quality and changing the chemical signature. In particular, this conduit can be a potentially significant source of nutrients, increasing their availability for productivity (Page et al., 1995; Valiela, 1983). Increases in population around wetlands have already been shown to have an impact on nutrient concentrations and likely contribute to eutrophication (Persky, 1986). Other critical effects of groundwater solute transport include public health (improving the efficiency of pollutant dispersion) and ecosystem stability since freshened inputs change ambient salinity levels.

2.1.2 Aims and Objectives - This review explores the history and evolution of groundwater research in wetlands, briefly examining the following: (1) techniques used; (2) established as well as implied controls on submarine groundwater discharge (SGD); (3) results of studies and evidence of links to those controls; (4) uncertainty associated with results; and, (5) the utility of this new body of information to wetland managers. A case study is presented to highlight available SGD information and its application in management decisions for a specific coastal area.

2.1.3 History of Groundwater and Wetlands Research - It has long been understood that SGD occurs along continental margins (Matson and Sandford, 1913). Various submarine springs, boils, and unusual depressions were identified and found to have lower salinity and chlorinity values associated with them than the surrounding marine waters (Brooks, 1961). These lower salinities were a result of freshened groundwater inputs coming into contact with high salinity marine waters. This mixing zone was identified as an early indicator of SGD (Cooper et al., 1964). Concurrently, the technique of using natural radioactive tracers to identify water masses and understand mixing processes was put forward in the 1960's (Broecker, 1965). It became clear to early researchers that sediment stratigraphy was one factor controlling groundwater flow, having found that anisotropic sediments permitted greater movement of SGD either vertically or horizontally (Manheim, 1967).

From 1960 to 1970, techniques to identify SGD were limited to the use of submersibles, and piezometric head was measured by using drill holes. It seemed relatively simple to identify sites of strong SGD upwelling, but areas of low-velocity advection such as seepage required better techniques for identification than those available up to 1970. No methods had been developed for quantifying the speed and volume of groundwater flux. However, some early suggestions for further research included more precise mathematical modeling of SGD, and studying spatial and temporal salinity/conductivity changes, as well as the development of more sensitive detection methods (Manheim, 1967).

Wetlands were far more studied by biologists than physical scientists at that time mainly because biologists were concerned about the impact of different parameters, including nutrient levels, on wetland productivity (Odum and Odum, 1955). Improvements were geared in the direction of productivity and the flow of energy through the system (Teal, 1962; Day et al., 1973). Anthropogenic effects on nitrate and phosphate concentrations were being assessed during the 1960s, but up to that point, nutrient inputs were thought to be limited to surface runoff, rainfall, and streamflow with some influx from high concentration porewater diffusion (Berner, 1971; Gardner, 1975). It was not until the 1970's, the same decade in which groundwater research was accelerated, that an underground conduit for nutrients into coastal ecosystems was seriously considered (Valiela et al., 1978). The impact of nutrient loading in coastal waters was further studied through improved understanding of diagenetic processes and their effects on interstitial water chemistry (Bear, 1972; Berner, 1971). Scientists noted that certain solute concentrations were unusually high in waters directly above coastal sediments, which intensified the study of groundwater-sediment interactions (Degens, 1965).

These three fields of study – biology, groundwater hydrology, and sediment diagenesis - converged in the 1970s to produce a suite of scientific methods for identifying and measuring SGD in coastal estuarine systems. In an effort to estimate the global discharge of groundwater, water budget models were used (Zektzer et al., 1973). However, it quickly became clear that independent estimates yielded highly varying results. Regional and local SGD studies continued to be scarce until the late 1970's when techniques were developed to quantify this relatively unknown input to wetland water budgets and biogeochemical mass balances. The application of the seepage meter (Lee, 1977) represented an important advance in groundwater research, and was followed by methods for measuring diffusive flux (Martens et al., 1980). Later studies were carried out to determine the efficacy of utilizing natural radioisotopes for tracing groundwater discharge (Cable et al., 1996a).

2.1.4 Current Priorities in Wetland Management – During the past twenty-five years of intensive groundwater research, wetlands management evolved with new policies and legislation supporting the efforts of managers. Before the 1970s, wetland conservation was simplistically limited to managing for a single designated use such as waterfowl conservation (Ramsar, 2001). Goals and achievements were clear and easily measured. In the more recent phase of evolving wetland policy however, management goals are geared toward conservation, protection, restoration, and wise use. As residential and commercial development increase, along with the resulting groundwater and surface water contamination in coastal ecosystems, wetland management has reached a new level of complexity. To deal adequately with these anticipated or current conditions, ideal management strategies today incorporate some or all of the following basic information: (1) wetland type according to groundwater and surface water hydrology; (2) neighboring land uses; (3) vegetation type and distribution; (4) wildlife habitat; (5) water quality and water regime; and (6) biogeochemistry of sediments (Cicin-Sain and Knecht, 1998).

2.2 Approach

This review of studies related to wetland groundwater flux is limited to research within, or in the vicinity of, coastal wetlands worldwide. Only studies published in peer-reviewed journals during the last forty years are included, up to a maximum of 50 studies. Consequently, this list of studies is by no means exhaustive. Studies with a calculated SGD flux are included in the review, and, where possible, units of measurement are converted for ease of comparison. The SGD controls examined in this study are precipitation, tidal height, substrate type, and technique. Flux is descriptively compared to these controls. In addition, I analyze the utility of this current body of information for coastal managers by presenting a case study where SGD information has been successfully incorporated into management decisions. Finally, I present some suggestions for both scientists and managers that might improve the flow and use of groundwater discharge information amongst researchers and planners.

2.3 Distribution of Groundwater Information

The global extent of groundwater discharge measurements and/or calculations is extremely narrow (Figure 2.1). Eighty-six percent of the studies included in this review have been carried out on the United States (U.S.) Atlantic and Gulf coasts. On the U.S. eastern seaboard, some sites have been studied extensively, such as Florida's coastal ecosystems, the marshes of South Carolina (S.C.), Chesapeake Bay, and the Massachusetts coast. Nine percent of studies within developing nations are included for review here. Of the studies within developing nations, only one study gave a value for SGD. Government reports and conference proceedings yielded additional studies for Asia, particularly India, as well as Europe, but only journal publications are included in this review.

2.4 Controls on SGD Values

2.4.1 Precipitation - According to the hydrologic cycle, precipitation is the main input to any coastal ecosystem, and infiltration takes this inflow to subsurface storage areas (Freeze and Cherry, 1979). One would expect that the greater the precipitation during a specific year, the higher the aquifer water table. Thus, increased pressure head is produced within the system, leading to higher values of SGD. Studies in open water and some coastal areas generally support this theory (Table 2.1).

Some studies present results showing a relationship between SGD, nutrient concentrations, and rainfall (Table 2.2). In Florida, Cable et al. (1997) noted that fluctuations in local precipitation with SGD were very similar over the study period, and in Kenya during drought conditions, reduced seepage values were measured (Kitheka, 1998). In Great South

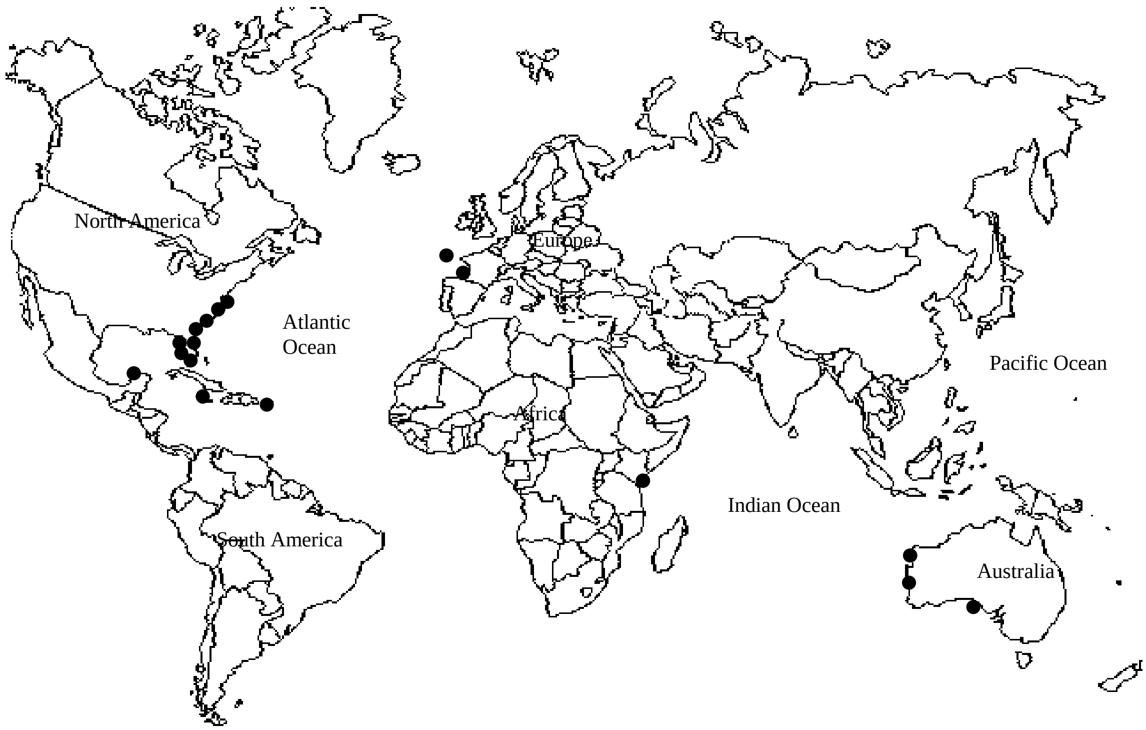

Figure 2.1 World map showing general location of sites mentioned in this review.

Bay, New York, elevated water table heights and sediment nitrate levels varied with increased rainfall patterns, suggesting a higher rate of discharge of groundwater (Capone and Slater, 1990). A similar pattern was observed in the Yucatan Peninsula, but in this case ammonia concentrations increased dramatically during the rainy season (Herrera-Silveira, 1996).

However, not all studies demonstrate such trends with respect to precipitation and SGD, especially smaller coastal ecosystems. Subsurface drainage basins are often much larger than the surface system under observation. Thus, regional precipitation rates may sometimes be more applicable for comparison with levels of SGD and nutrients than local precipitation values. Seasons of low rainfall and resulting low water table height may increase tidal influences on coastal ecosystems, further complicating SGD analysis.

2.4.2 Tidal Influences - Pressure reduction within coastal aquifers often creates conditions where the hydraulic gradient slopes landward from offshore to the continents. This slope causes tidal forcing, and drives high salinity water through nearshore sediments, leading to salt water intrusion (Martin and McCutcheon, 1999). A consequence of such intrusion is that SGD includes some percentage of seawater (as high as 90%) being cycled through the sediments (Giblin and Gaines, 1990; S. Joye, pers. comm.). One study measuring salinity and tidal influences on SGD was carried out in a mangrove-fringed creek in Kenya (Table 2.1). In this area, groundwater discharge was masked by intrusion of hypersaline water during periods of drought (Kitheka, 1998). In Ringfield Marsh, Virginia, researchers found that the groundwater flux peaked with increased hydraulic head, reducing the width of the vadose zone available for tidal influences. Thus, SGD studies which incorporate analyses of salinity and tidal pumping may ultimately be considered to be more rigorous than those that measure flux rates only.

2.5 Comparison of SGD Values across Ecosystems

Many types of coastal ecosystems were studied recently for SGD, including mangroves, salt marshes, and lagoons (Table 2.1). However, SGD studies were first done in open water and along coastlines, especially in areas of submarine springs and boils (Table 2.2). Interestingly, no relationship emerged when SGD values were plotted against sediment type. Soil permeability certainly affects the magnitude and distribution of groundwater flow (Kasenow, 2001). Yet, it has been noted that flow rate in Great South Bay was not affected by the range of different vertical permeabilities measured in the study area, but by local pockets of impermeable sediments (Bokuneiwicz, 1980). Thus, permeability of the dominant sediment type described for each study site may be less important to SGD predictions than the stratigraphy of the study area sediments.

Although seepage meters were the dominant technique (43%) in the studies reviewed in Tables 2.1 and 2.2, benthic flux chambers and radioactive tracers were widely used (36%). It was not clear from these studies whether a single technique consistently produces higher or lower SGD values (Table 2.3). At the Par Pond site, the water budget SGD value was considered to be an upper limit because overland flow estimates were not included (Table 2.3). However, upper limits from tracer estimates were even higher than the water budget results (Corbett et al., 1997). Conversely, at the Little Pond site, measurements agreed very well from four different techniques, an indication that more comparative studies must be performed to determine which techniques provide conservative, average, or higher estimates of SGD. For managers considering the utility of SGD information, appropriate planning adjustments may need to be made for differences in estimates.

Table 2.1 Groundwater studies within coastal ecosystems where SGD is reported in units of length and volume per time. Rainfall, tidal change, and sediment type are given when included in the study report.

Site	Wetland Environment	Annual Rainfall (cm)	Tidal Change (cm)	Major Surface Sediment Type	SGD cm/d	Average Benthic Nitrogen Flux Mmol/m^2/d	Techniques	Authors	Year
SGD rate									
Fire Island, NY	Barrier island	112	25	sand/clay	1.0-7.0	-	Flux chambers	Bokuniewicz & Pavlik	1990
Lot River, France	Well Field			gravel/clay	60	-	Tracers	Bertin & Bourg	1994
Little Pond, MA	Coastal Salt Pond	111	-	sand	0.27	-	Seepage meters	Vanek	1993
Indian River Lagoon, FL	Coastal Lagoon	-	37	sand/clay	48	-	Seepage meters	Belanger & Walker	1990
Mida Creek, Kenya	Mangrove creek	104	260	sand	0.24	-	Salinity/tide study	Kitheka	1998
Florida Keys, FL	Mangroves	100	24	limestone	12-114	55	Flow meters	Lapointe et al.	1990
Clambank Creek, SC	Marsh	-	-		0.8-2.8	6.08 mg/m^2/d	Flux chambers	Whiting & Childers	1989
North Inlet, SC	Marsh	-	-		10	-	Radium tracers	Rama & Moore	1996
North Inlet, SC	Marsh	-	-		2.0-4.0	0.27-9.6 x 10e^5	Radium tracers	Krest et al.	2000
North Inlet, SC	Marsh	-	-		1	-	Water/salt balance	Morris	1995
Hog Island	Marsh	-	250	sand	11.3	0.48	Seepage meters	Osgood	2000
Pritchard Island	Marsh	-	-	sand	11.8	0.53			
Carter Creek, VA	Marsh	-	-	clayey sand	0.1	-	Piezo meters	Harvey & Odum	1990
Eagle Bottom, VA	Marsh	-	-	-	0.02	-			
Great Sippewissett, MA	Marsh	-	-	-	0.9	-	Flow meters	Valiela et al.	1978
Cherrystone Inlet	Coastal Lagoon	108	70	Sandy loam	0.05-8.9	-	Seepage Meters	Reay et al.	1992
Nauset Marsh, SC	Marsh	110	50	Sand/silt	6.5-8.2	24-72	Flux chambers	Portnoy et al.	1998
SGD volume									
Chesapeake Estuary	Marsh	-	30	-	240 l/m/d	-	Dye tracers	Jordan & Correll	1985
Marmion, Australia	Lagoon	80-90	70	limestone reef	4.8x10e^4 m^3/d	266 kg/d	Salinity profile	Johannes & Hearn	1985

A large number of nutrient studies implicate groundwater as a significant conduit for these dissolved solutes and some are included here (Table 2.4). All possible sources and sinks are identified, and the concentration required to balance observed water column inventory is assumed to be the result of groundwater influx. Tables 2.1 and 2.2 show that few studies measure both SGD and nutrient fluxes, even though one set of results might support the other. To effectively compare nutrient and SGD fluxes, it is clear that a basic set of site characteristics should be included in all studies. For example, study area dimensions are important for simplifying unit conversions to make cross comparisons among studies where necessary. Other basic characteristics may include dominant sediment type, tidal range, and regional climate.

2.6 Case Study

Considering the above complex array of parameters governing groundwater discharge at different wetland sites, we might ask, "Can science provide coastal wetland managers with the appropriate groundwater information required to manage their systems effectively?" The following case study is designed to explore this question by assessing five issues related to groundwater which might be important to managers. These issues are: (1) the definition of SGD as used in the literature; (2) the magnitude and direction of groundwater flow; (3) the physical parameters that control SGD magnitude at a given site; (4) accompanying information on dissolved solutes to the system; and (5) the potential impacts of SGD and solutes to ecosystem health. The research associated with each of these six issues is briefly explored, prefaced by a general description of the case study site.

2.6.1 <u>Site Description</u> - Sippewissett Marshes in Cape Cod, Massachusetts have been studied extensively for the past 25 years and provide an excellent example of a site where the results of scientific investigation are available to managers and are being utilized in making

Table 2.2 Groundwater studies in open water or along coasts where SGD is reported in units of length and volume per time.

Site	Major Sediment Type	SGD cm/d	Nitrate ($mmol/m^2/d$)	Techniques	Authors	Year
SGD Rate						
Big Bend, FL	limestone/dolomite	13	-	Tracers, seepage meters	Cable et al.,	1996a
NE Gulf of Mexico, FL	limestone/dolomite	2.0-10.0	-	Tracers	Cable et al.,	1996b
Great South Bay, NY	sand glacial outwash	2.0-8.0	-	Seepage chambers	Bokuniewicz	1980
Waquoit Bay, MA	coarse sands	0.9	0.55	Radium balance	Charette et al.	2001
Florida Bay	-	1.9	110 $mmol/m^2/yr$	Seepage meters, tracers	Corbett et al.	1999
South Atlantic Bight	-	0.5	-	Radium balance	Moore	1996
NE Gulf of Mexico, FL	Limestone/dolomite	-1.44-34.56	0.6 kg/d	Methane, seepage meters	Bugna et al.	1996
Bogue Sound, NC	-	2.8	-	Seepage meters	Lee	1977
Buzzards Bay, MA	Coarse sand	4.8-43.2	-	Seepage meters	Valiela et al.	1990
Florida Keys	karst	0.54-0.89	-	Seepage meters	Simmons	1992
St George Sound	karst	0-23	-	Seepage meters	Cable et al.	1997
SGD volume		-				
Peace River, FL	-	0.5-4.5m^3/s	-	Radium Tracer	Miller et al.	1990
Myakka River, FL	-	0.5-2.9 m^3/s	-	Radium Tracer		
Yucatan Peninsula	-	8.6 x 10e^6 $m^3/yr/m$	-	Water budget	Hanshaw & Back	1980

Table 2.3 Studies comparing the results from different methods used in calculating groundwater discharge. Only those site characteristics reported are included.

Site	Annual Rainfall (cm)	Tidal Range (cm)	Sediment Type	Author	Year
Ringfield Marsh, VA	60-840	100	sandy marsh peat	Tobias et al.	2001
Little Pond, MA	111	-	Sand	Millham & Howes	1994
St George Island, FL	140	50	med/fine sand	Corbett et al.	2000
Town Cove, MA	119	-	-	Giblin & Gaines	1990
Par Pond, SC	-	-	-	Corbett et al.	1997

Site	Seepage meters (cm/d)	Darcy Discharge (cm/d)	Tracer Discharge (cm/d)	Salt Balance Discharge (cm/d)	Water Budget (cm/d)	Inlet Block (cm/d)
Ringfield Marsh, VA	-	- 0.8-8.0	1.6	0.06-2.2	-	-
Little Pond, MA	-	1.2	-	2	1.5	1.4
St George Island, FL	-	-	2.1-6.4	-	0.7-2.9	-
Town Cove, MA	2.4-7.2	-	-	2.9-6.4	0.5-0.7	-
Par Pond, SC	-	-	0.13-1.17	-	0.82	-

Table 2.4 Studies implicating groundwater in nutrient fluxes without measuring SGD directly. Nitrogen was measured as ammonia or nitrate.

Site	Nitrogen in Groundwater	Units	Author/Year
Discovery Bay, Jamaica	0.08	mg/l	D'Elia et al. 1981
Great South Bay, NY	0.0-40.0	μM/l	Capone & Slater, 1990
Celestun Lagoon, Yucatan	11.0-15.0	μM/l	Herrera-Silveira, 1996
	SGD-N flux		
Potomac Estuary	1.0-21.0		Callender & Hammond, 1982
Indian Heights, MA	0.13	mmol/m/yr	Weiskel & Howes, 1991
Newport, Neuse Estuaries, NC	0.0-12	$mmol/m^2/d$	Fisher et al., 1982
Buttermilk Bay, MA	0.76	$mmol/m^2/d$	Valiela et al., 1988

effective management decisions. In this area, effective management is important because of threats to the marsh in the form of dramatic population growth on Cape Cod, leading to marsh development pressures.

The Sippewisset Marshes are located on the shore of Buzzards Bay near a prime urban area, yet they remained in a reasonably healthy condition. They consist of Great Sippewisset Marsh and Little Sippewisset Marsh. The length of the entire marsh system is approximately 2.4 km and the width varies from 0.4 to 1.6 km (U.S. Fish and Wildlife, 2001). *Spartina alterniflora* dominates the low lying areas of the marsh while *Spartina patens* and spike grass dominate the high ground vegetation.

The continued health of these marshes results in part from thorough research for 25 years, including temporal and spatial variations in chemistry, flow patterns, vegetation, geology, and many other aspects of marsh processes, structure and function (U.S. Fish and Wildlife, 2001).

This marsh was developed on glacial till topped by peat forming sediments and sands to an average depth of 1 to 2 meters with some areas having a peat depth of as much as 5 meters. Clays intersperse the peat, while sand is dominant in the surface layers. Organics within Sippewissett sediments average 26% and peat porosity averages 0.75 (Knott et al., 1987). Sippewisset Creek is the main tidal connection.

2.6.2 Groundwater Information Currently Available – The five groundwater management issues previously identified have been largely addressed in research at Sippewisset Marshes and surrounding areas. SGD was reported to be a mixture of freshened inputs of groundwater plus some percentage of recycled seawater, fulfilling requirements for a full definition of subsurface fluid inputs to the system (Giblin and Gaines, 1990). Surface hydrology has been documented as well as the interaction of surface and subsurface fluids, providing independent confirmation, using nutrients, of the origins of advecting fluids (Nuttle, 1986; Nuttle and Portnoy, 1992; Hemond and Goldman, 1984).

Thorough research has also been carried out on the magnitude and direction of groundwater flow in and through the marshes. For example, Hemond and Fifield (1982) conducted an analysis of subsurface flow within the peat of Sippewissett Marsh 15 m from the nearest creek to determine the effect of proximity to creeks on subsurface water movement. They found that at that distance creek influence was minimal, and groundwater flow was vertical due to evapotranspiration, and that horizontal flow of groundwater in the underlying sands at Sippewissett Marshes was seaward due to the piezometric head / tidal height interaction. An estimate of upward groundwater flux to the vegetative zone was 0.005 m/s (Hemond and Fifield, 1982).

Information on hydrologic parameters and controls on SGD were reported by Knott et al. (1987) among others. They measured average hydraulic conductivity and found that the values ranged from of 10^{-1} to 10^{-5} cm/s in Sippewissett Marsh. However, when the peat was compressed, the specific storage of the sediment was changed by as much as 10^{-3} cm^{-1}. Since specific storativity measured change in the pore water storage per unit change in hydraulic potential, this parameter was considered to be critical in determining water movement potential (Hemond and Fifield, 1982). Both hydraulic conductivity and specific storativity information could then be used to assess drainage response times. However, the permeability of peat in some areas of the Marsh were found to be relatively low, suggesting a disparity in the extent of tidal influence driving subsurface flow between peat and the underlying aquifer (Hemond and Fifield, 1982).

When conducting field investigations on dissolved solutes and other constituents, it was shown that in the interior of the marsh where evaporation seemed to drive vertical groundwater movement, the tidal influence on the Marsh nitrogen budget was greater. Sinks of groundwater-derived nutrients such as nitrogen, carbon, and phosphorus were assessed and the relative significance of observed concentrations reported (Valiela and Costa, 1988; Valiela et al., 1990). It was found that the seaward flux of groundwater-derived nitrogen was about 6.1x 10^3 m^3/yr, a significant contribution to the nitrogen budget when compared with the seaward flux of nitrogen derived from surface waters (Valiela et al., 1978).

The above scientific findings have been used in planning and decision- making. Managers are aware that values of SGD include some percentage of recirculated seawater, fulfilling requirements for a complete definition of SGD. In addition, they have information on periods of high and low flow, effects of SGD on nutrient concentrations, and their resulting impacts on the ecosystem. This information is important to wetland managers, since it allows careful planning of cleanup efforts in the event of contamination. Substrate transmissivity and pollutant viscosity determine the potential extent of groundwater contamination (zone of influence).

2.6.3 <u>Managers' Use of Groundwater Information</u> - From these data, managers determined that residential development and marine construction threatened the groundwater and surface water quality of the marsh due to excessive nutrient loading. They concluded that a direct result of construction is likely to be changes in the type of vegetation, which may alter the current subsurface flow balance. In addition, high nutrient loading might be an alteration of the sediment biogeochemistry, limiting plant growth, and leading to erosion within the wetland (U.S. Fish and Wildlife, 2001).

Local managers and some residents applied this body of hydrologic and biogeochemical information, and lobbied the U.S. Fish and Wildlife Service to declare the Great Sippewissett Marshes a National Wildlife Refuge. In addition, the Falmouth Wetlands regulations were amended in 1997 to include the following resource values for Great Sippewissett Marshes: (1) water quality improvement; (2) reduction of pollutant discharge to groundwater; (3) protection and enhancement of existing vegetation cover to maintain water quality and wildlife habitat; (4) prevention of new storm water runoff discharges; and (5) improvement of groundwater recharge (Cape Cod Commission, 2001). As a result of the 1997 amendment, the Buzzards Bay National Estuary Program's plan for addressing these values included: (1) reduction in improving impervious surfaces; (2) increased distance between sources and sinks of pollution; (3) maximizing the naturally vegetated land area; and (4) limiting the area which is filled, dredged, built upon, degraded or otherwise altered to less than 560 m^2 per residential lot. Other significant policies include stringent limitations on building locations, setbacks, and sewage disposal. Great Sippewissett Marsh is a good example of a well-managed salt marsh estuary with highly developed management policies that clearly incorporate the scientific studies available to make informed management discussions. However, it must be noted that many studies have been performed over a number of years to produce such a comprehensive body of usable knowledge.

Other areas of the U.S., such as in Florida, and to a lesser extent, South Carolina, have also amassed a body of knowledge regarding the local groundwater hydrology in their wetlands. Scientists are working to address deficiencies in available information, and as a result, managers in these regions are still attempting to transform the science into policies and practices aimed at conserving wetlands.

2.7 Discussion

Using only the studies included in this review paper, SGD research has increased dramatically from the 1970s to the 1990s, as more techniques are developed to quantify SGD flux. However, few SGD studies were found for many developing nations and industrial nations. The distribution of SGD information around the world remains uneven probably due to availability of funding. Nevertheless, a few developing nations, such as Kenya and India, have begun work on the study of groundwater in small areas, and this will prove to be beneficial in decision-making. Managers within these countries may sometimes to utilize information from completed studies and, after careful site comparison, extrapolate values of nutrient and freshwater flux for their area.

Some basic site-specific hydrologic information is necessary even in cases when extrapolation is used, however. A number of important hydrologic parameters can be easily measured such that synthesis of the information results in an understanding of the magnitude, direction, velocity and constituents of all surface and subsurface fluids. These parameters include hydraulic gradient, which is easily measured using a simple technique such as piezometers. Because the hydraulic gradient controls flow direction, managers will have an idea of the potential direction of any SGD flow. From this review, it is clear that even a simple investigation of substrate type and stratigraphy using sediment cores can contribute significantly to an overall understanding of SGD potential at a specific site. These parameters along with climate data such as precipitation and evaporation, give managers some of the tools necessary to choose a comparable site for extrapolation, or to begin developing a picture of subsurface hydrologic flows in their own area.

From the studies included in this review paper, two controls on SGD levels have been established: rainfall and tidal pumping. As regional precipitation increases and tidal height decreases, SGD levels tend to increase. However, other controls, such as substrate type and land use still require additional research and corroboration. Sediments with higher hydraulic conductivity are known to facilitate SGD flow. However, the hydraulic conductivity identified for each site is associated with the dominant sediment type, and is not necessarily representative of the sediment mix. Similarly, a study of dominant land use as it relates to groundwater has similar problems. Nonetheless, such adjunct information may be useful to managers and planners.

Deficiencies exist within the literature, which must be addressed by scientists. Sites similar on the surface may have very different subsurface hydrologic regimes, but once these differences are clearly understood, managers can develop plans based on credible information to achieve their management goals. Background information, such as study area dimensions, precipitation, tidal range, land use, hydraulic conductivity, and stratigraphy, provide a framework for managers evaluating the SGD value presented in a scientific paper for possible extrapolation.

Groundwater discharge information is now available to managers for planning and policy making. Scientists have made it clear that the principle factor governing wetlands is hydrology, and as such, managers must also consider any anthropogenic alterations of hydrology. These alterations may include diversions, canals, aquifer drawdown for public supply, dams, or containment levees. Even in cases where maintenance of hydrology is the sole focus of management and policy, the task of managers is still complicated within the modern landscape. Generally though, management goals are not limited to hydrology alone. Nutrient inputs from groundwater must also be considered, and may be significant enough to undermine results of conservation policies by destroying the health of coastal ecosystems. To this end, well-rounded management decisions are based on all parameters affecting the magnitude, velocity, and quality of surface and subsurface fluids.

2.8 Summary and Conclusions

An examination of the 50 studies explored in this review has revealed the following:

1) Basic climate and ocean information presented with SGD results facilitates data manipulation and unit conversion so that meaningful comparisons may be made among studies at different sites.
2) In spite of major technological advances in the study of groundwater, research is still concentrated in specific areas, usually in industrial nations.
3) Results of nutrient budgets often implicate groundwater as a nutrient source without actually quantifying SGD to support the implication.
4) Rainfall and tidal influences were shown to affect the magnitude and direction of SGD. However, effects of substrate type on SGD could not be corroborated by this review.
5) A comparison of techniques for measuring SGD did not produce any significant differences in results obtained from each technique.

Uncertainty still surrounds estimates of SGD, but it is clear that, given appropriate physical conditions, groundwater discharge is as significant a conduit for solutes as its surface discharge. In essence, uncertainty in groundwater estimates can no longer be a reason for ignoring subsurface fluid flows within coastal areas.

Chapter 3

Hydrologic Evaluation of U.S. Coastal Watersheds in the Eastern and Gulf Coasts

3.1 Introduction

Hydrologic evaluations of a specific region require use of the law of conservation of mass, particularly with regard to water budgets. Regions are often delineated as watersheds based on the surface water expression, but without consideration of the aquifer's areal extent. Once water enters a watershed as precipitation or groundwater discharge to surface waters, some fraction is intercepted within that area by plant leaves and roots (Dingman, 1993). Some water will evaporate or infiltrate the sediments and recharge the aquifer, and the balance drains into streams within the watershed. Water budget analyses provide a tool for determining water volumes entering, leaving, or being stored in ecological systems (Watson & Burnett, 1995).

Depending upon the scale of study for a water budget analysis, a variety of parameters are included to document sources and sinks. These parameters may include groundwater input/output, surface water (channelized flow) input/output, precipitation, sewage effluent as an input, industrial/commercial/residential outfalls, surface runoff, transpiration, open water evaporation, biological utilization, and change in storage. Parameters are often simplistically limited to only three parameters: precipitation, surface discharge and evapotranspiration. Careful evaluation of sources and sinks must be performed before calculating a water budget if one is to be consistent with the conservation of mass. Despite some shortcomings to be discussed later, water budgets have proven useful in improving our understanding of the hydrology of many different ecosystems around the globe.

Water budgets are utilized extensively for a variety of purposes and may be applied on many scales in time and space. In an effort to understand how planets develop over time, water budgets were applied on a planet-scale to compare and estimate water content between Venus and Earth (Lecuyer et al., 2000). They ultimately estimated that the Earth held an excess of surface water equal to approximately 1.2 x 10^{21} kg relative to Venus, presumably giving Earth an advantage to develop and sustain life. Budgets have also been used at the poles to provide a hydrologic synopsis in the sub-arctic regions of Canada (Petrone and Rouse, 2000). Vorosmarty et al. (2000) used a global water budget to consider the direct human impacts on water supply and the resulting potential effects on climate change. These models were also used to assess the impact of a catastrophic flood and storm event (Smith et al., 1996). Another successful utilization of water budgets on a major sea-scale is a comparison of temperature, water use and salinity changes in the Mediterranean Sea over the long term (Bethoux and Gentili, 1999).

The most common usage of water budgets is on much smaller spatial and temporal scales, such as within watersheds and lakes. Many of these budgets have allowed scientists to contrast water movement through systems that differ geologically, such as peatlands (Drexler et al., 1999), Alaskan tundra (Harazono et al., 1998), Florida mangroves (Twilley and Chen, 1998), and semi-arid regions where long term soil and water records were used to measure infiltration rates (Townsend et al., 1995). Water budgets in lakes are facilitated by their spatial isolation from other water bodies, whereas in coastal watersheds, major connections to the ocean, upland areas, and adjacent sites must be considered (Sanford et al., 1995).

Water budget deficits can occur due to a lack of information in certain parameters, such as runoff, evapotranspiration, or groundwater fluxes. However, with the need for accurate estimates and better methods, more attempts are being made to quantify more elusive components, such as groundwater imports/exports. At present, improved techniques and a large network of groundwater wells in US watersheds allow us to include this groundwater parameter in watershed evaluations using empirical data, and to do this with some degree of success (USGS, 2000).

Comparisons of calculated (water budget) and measured (field evaluations) fluxes often yield different results. However, calculated fluxes can be utilized to shed light on important influences that may not be previously considered. To assess the groundwater inputs to Town Cove on Cape Cod, Massachusetts, the results of three research methods were compared. They used a water budget, collected empirical data using flux chambers and seepage meters, and carried out a water and salt balance at the Cove mouth (Giblin and Gaines, 1990). The water budget model produced the lowest estimate of groundwater inputs, while the water and salt balance gave the highest estimates. The seepage meter results were closer to, but less than, the water and salt balance. Thus the particular method applied may determine the magnitude of groundwater discharge obtained. One reason Giblin and Gaines suggested for the differences between measured and calculated results was that re-circulated seawater might be inflating the measured fluxes at the mouth of the cove. Tidal influences may also confuse estimates of this component of the discharge. Other reasons for differences might include errors, both in field evaluations using nutrient loading to calculated groundwater discharge, and in water budget calculations. Giblin and Gaines also noted that groundwater nitrate concentrations exhibited spatial and temporal variation over several orders of magnitude, and nutrient loss may have occurred during travel. In addition, depending on the season, measured groundwater fluxes varied drastically (Giblin and Gaines, 1990).

Uncertainty exists in estimates of each component used in a water budget due to temporal and spatial variations in the measured parameters and measurement techniques. It is still not clear how accurately precipitation and discharge can be measured, considering that a single rain gauge may not accurately depict the precipitation over an entire region (Winter, 1981). Precipitation error appears from a lack of spatially well-distributed rain gauges and regular gauge maintenance. Discharge gauges, on the other hand, can be well placed but improper calibration and maintenance will result in significant measurement errors. Water budget results are dependent on an accurate estimation of error.

In order to assess the utility of water budgets on a regional scale, this study estimates the relative groundwater fraction of water budgets for coastal watersheds, many of which are included in the National Estuary Program (NEP). Three objectives of this study were: (1) to evaluate water budgets for 29 coastal watersheds of the eastern and gulf coasts of the US; (2) to estimate the net import or export of groundwater within these watersheds; and (3) to compare those results with observed groundwater fluxes for some of the same areas based on available literature.

3.2 Site Description

The study areas encompassed 29 coastal watersheds from Maine to Texas divided into four regions: the Northeast, mid-Atlantic, Southeast and Gulf Regions (Figure 3.1). Geologically, this area consists of some alluvial basins such as the Mississippi River Valley, the Piedmont and Blue Ridge Region, Southeast Coastal Plain, Northeast and Superior Uplands, and the Atlantic and Gulf Coastal Plains (Watson & Burnett, 1995). These areas show a large variation in geology and, as a result, the quantity of available groundwater could potentially vary widely in these regions.

Precipitation - The coastal watersheds from Maine through Florida have an average precipitation of 100-150 cm per year, whereas rainfall in the Gulf watersheds of Mississippi and

Louisiana usually is 150-250 cm/yr (NCDC, 2001). The only watersheds in the study area showing lower precipitation values are the coastal bays and estuaries of western Texas.

Geology - The northeast region, which includes the states of Maine, New Hampshire, Massachusetts, Rhode Island, Connecticut, and New York, is geologically characterized by glacial drift covering fractured crystalline rocks. Fluvial deposits mix with glacial drift in major stream valleys and it is common to find fractured bedrock in this area (Watson & Burnett, 1995). This area thus shows a relatively high potential for aquifer recharge.

With the exception of Florida, the Gulf Coast and the southeast Atlantic watersheds are geologically similar, and have with largely unconsolidated and semi-consolidated rock beneath layers of sand, silt, and clay beds (Dingman, 1993). The southeast coastal watersheds, which include those in the states of North Carolina, South Carolina and Georgia, are influenced by the flow of water and precipitation from the Appalachian Mountains. Large volumes of surface waters are available from the mountain precipitation flowing across the coastal plain. The states influenced by this flow include those in the mid-Atlantic region - Maryland, New Jersey, Virginia and Delaware (Watson & Burnett, 1995).

Major alluvial valleys, such as the lower Mississippi and Mobile Rivers, occur along the northern Gulf of Mexico where clays and fine sands overlie loosely consolidated rock. Sand and gravel make up the major water-bearing sediments of underlying aquifers. Because the area is humid, recharge to the aquifers is adequate for balancing discharge and use. The geology of Texas is also characterized by unconsolidated and semi- consolidated rock, with large sand/gravel and sandstone beds (Watson and Burnett, 1995).

Florida is geologically unique in many ways. The Floridian aquifer is the largest in the southeastern United States and extends from Southern Alabama across Georgia into South

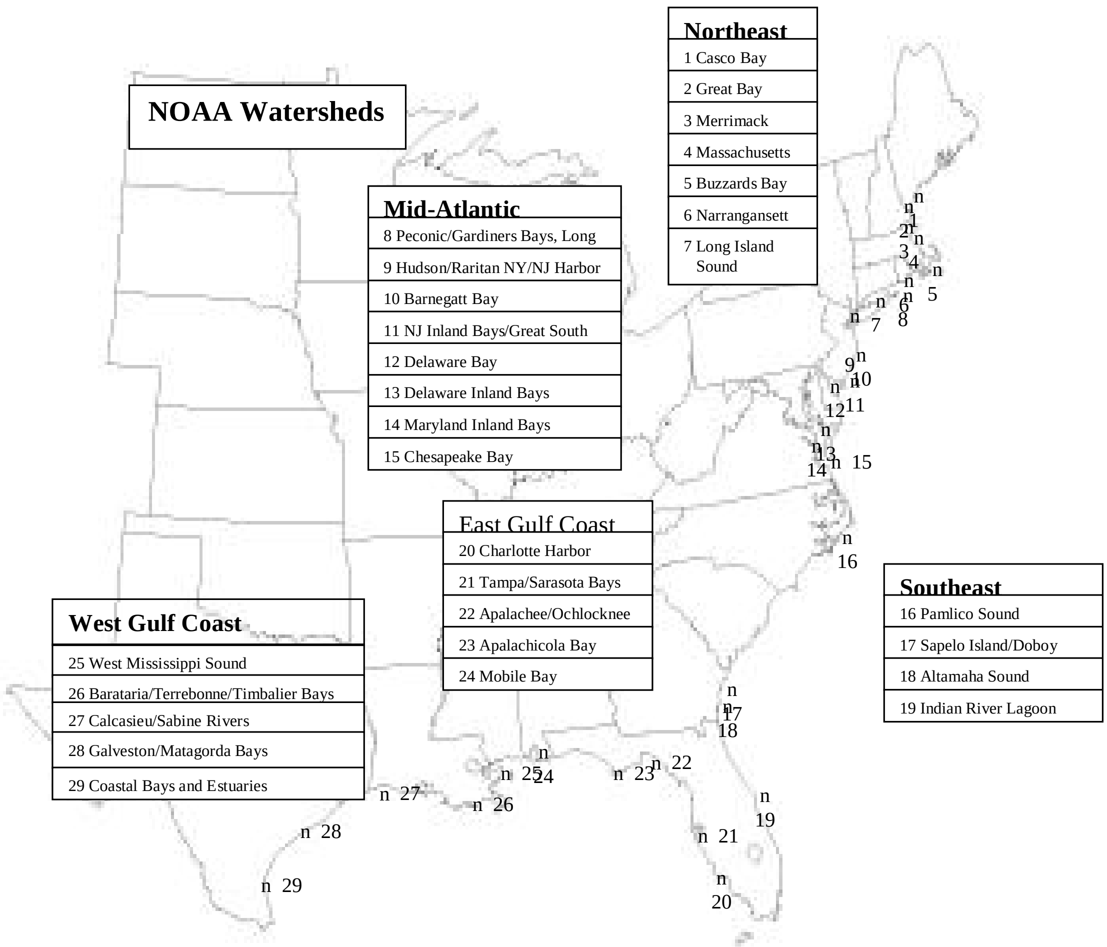

Figure 3.1 Site map of the U.S. Eastern and Gulf coasts showing watersheds studied.

Carolina and encompasses all of Florida. This aquifer consists of limestone and dolomite overlain by clastics. The southern tip of Florida receives as much precipitation as the Gulf coastal watersheds. Given the high conductivity of this aquifer, it is easily recharged, particularly in areas where limestone is exposed at the surface, which keeps the water level high (Scott et al., 1990).

Land Use - Land use and population densities differ widely among these watersheds. In the northeast, forests are the dominant land use and agriculture utilizes only a minor portion of the land. This is in contrast to the mid-Atlantic, where agriculture dominates land uses. In the southeast region, many upland forests persist and these, along with agriculture, constitute the major use of land. Land use in the Gulf region has no evident pattern. Louisiana watersheds dominated by river-sustained wetlands are Barataria Basin, Terrebonne Basin, and West Mississippi Sound. Agriculture dominates the western Florida watersheds of Apalachee Bay, Apalachicola Bay, along with Mobile Bay, Alabama and West Mississippi Sound. Among the Texas watersheds, upland forests are found in more than 50 percent of the Calcasieu/Sabine watershed, whereas in the Galveston/Matagorda region and the Texas coastal bays, a major percentage of the available land is used for rangeland and agriculture (Castro et al., 2000).

The regions can be clearly defined by population densities, with the northeast having the highest and the southeast having the lowest densities (Table 3.1). Once again, the region with the greatest disparity in population densities is the Gulf region. Tampa/Sarasota Bays, Barataria/Terrebonne/Timbalier Bays and Galveston/Matagorda Bays all have larger than the average population densities for the Gulf coast watersheds. Since land use, geology, and population are more diverse along the Gulf of Mexico, the Gulf watersheds are sub-divided into east and west sub-regions, with those in Florida making up the east and those from Alabama to Texas encompassing the west.

3.3 Methods

The water budgets for long periods (decades) assume no change in storage, no geologic changes, and no dramatic climate changes. In this study, water budgets were calculated for 29 watersheds included in the National Estuary Program (NEP) of the Environmental Protection Agency (EPA), and a change in storage was calculated. This approach was taken for two reasons: (1) to obtain a more accurate net groundwater result; and (2) to account for probable changes in population and subsequent water consumption over the 30-year time frame of the budget.

The general water budget equation is composed of various water sources and sinks, with the net result equal to zero:

$$\text{Inputs} - \text{Outputs} \pm \Delta S = 0 \qquad (1)$$

In the case of a watershed, the water budget may be simplified to the following:

$$(P \pm p) + (GW_I \pm gw_I) - (PET \pm pet) - Q - (GW_O \pm gw_o) \pm \Delta S = 0 \qquad (2)$$

where: P is precipitation (cm), GW_I is groundwater inflows (cm), PET is potential evapotranspiration (cm), Q is area-weighted stream discharge (cm), GW_O is groundwater outflows (cm), and ΔS is change in storage (cm). This equation must include error terms for each parameter, which are represented by lower case symbols. This equation was applied to estimate net GW over a 30-year period for each watershed.

Data compiled or calculated for the water budgets are found in Table 3.2. Precipitation information was gathered from NADP gauge stations within each watershed. Stream discharge was gathered from USGS stream gauging stations within each watershed, using the available data for the gauge which was situated further downstream.

Because the measurement of actual evapotranspiration may include the use of complex variables with large error, potential evapotranspiration (PET) was estimated using Thornthwaite's model (Thornthwaite and Mather 1957). This model assumes that PET is independent of vegetation density and maturity, and is mainly dependent on climate. For the purpose of these watershed budgets, long-term (30-year) climatic averages were used to calculate the PET. Those averages are mean monthly air temperature and hours of daylight (Watson and Burnett 1995). The Thornthwaite equation for calculating PET is as follows:

$$PE_m = 16\ N_m\ (10\ T_m\ /\ I)^a\ \text{mm} \qquad (3)$$

where PE_m = monthly potential evapotranspiration (mm)

N_m = adjustment factor for the monthly hours of daylight based on the latitude

Where N_m = hours of daylight/ 12 to normalize all N_m to 12–hour days;

T = average monthly temperature in degrees C;

I = annual heat index, calculated as $I = \Sigma\ I_m = [T_m/5]^{1.5}$ calculated for each month;

$a = 6.7 \times 10^{-7}\ I^3 - 7.7 \times 10^{-5}\ I^2 + 1.8 \times 10^{-2}\ I + 0.49$

and annual PET = ΣPE_m for the year.

Scientific opinions vary on the utility of Thornthwaite's model for measuring PET. This approximation of PET in short-term field experiments usually represents the upper limit of actual evapotranspiration (ET). However, one study measuring groundwater recharge in the Azul aquifer in Argentina by different methods showed that potential evapotranspiration (PET) was underestimated using Thornthwaite's model, leading to excessive recharge values (Varni et al., 1999). The use of a three-dimensional model such as MODFLOW was suggested as an

Table 3.1 Land use areas, population densities, and annual temperature for watersheds in the study area.

No	Watershed Name	Watershed Area (10^4 ha)	Wetland Area (10^4 ha)	Water Surface Area (10^3 ha)	Mean Annual Temp. (^oC)	Population Density (# per ha)	State
Northeast							
1	Casco Bay	21.88	2.66	42.686	7.4	0.93	ME
2	Great Bay	24.91	6.48	4.746	7.3	0.94	NH
3	Merrimack River	124.58	13.22	1.548	10.7	1.60	MA
4	Massachusetts Bay	20.89	1.51	95.331	10.7	8.05	MA
5	Buzzards Bay	10.21	5.06	63.901	10.7	3.22	MA
6	Naragansett Bay	40.18	6.84	41.563	10.2	3.66	RI
7	Long Island Sound	407.74	64.96	330.089	10.9	1.78	CT
Mid-Atlantic							
8	Peconic/Gardiners Bays	10.14	0.97	51.216	12.6	3.82	NY
9	Hudson/Raritan Bays	361.14	41.47	79.903	12.6	3.60	NY
10	Barnegatt Bay	13.65	16.97	18.207	12.6	2.83	NJ
11	NJ Inland Bays/Great South Bay	32.15	76.40	27.81	11.7	2.33	NJ
12	Delaware Bay	307.92	103.78	206.948	12.3	2.26	DE
13	Delaware Inland Bays	5.07	2.64	9.024	12.3	0.49	DE
14	Maryland Inland Bays	2.96	4.18	5.403	12.8	0.34	MD
15	Chesapeake Bay	1607.95	165.18	1126.194	14.1	0.84	VA
Southeast							
16	Pamlico Sound	250.90	257.58	558.854	16.7	0.53	NC
17	Sapelo Island/Doboy	19.73	59.50	18.75	24.7	0.23	GA
18	Altamaha Sound	367.11	188.66	3.909	22.4	0.46	GA
19	Indian River Lagoon	24.41	20.81	86.634	21.4	1.22	FL
East Gulf Coast							
20	Charlotte Harbor	76.10	98.46	50.249	28.9	0.59	FL
21	Tampa/Sarasota Bays	56.11	47.85	102.59	24.0	2.66	FL
22	Apalachee/Ochlocknee Bay	142.15	195.83	177.284	26.7	0.24	FL
23	Apalachicola Bay	482.16	274.60	59.303	25.7	0.51	FL
24	Mobile Bay	1126.65	211.99	107.866	25.2	0.33	AL
West Gulf Coast							
25	West Mississippi Sound	384.07	238.57	420.341	23.8	0.53	MS
26	Barataria/Terrebonne/ Timbalier Bays	104.09	471.06	211.456	25.3	0.62	LA
27	Calcasieu/Sabine Rivers	614.78	288.53	52.455	20.8	0.25	TX
28	Galveston/Matagorda Bay	1753.03	143.03	257.136	23.1	0.50	TX
29	Coastal Bays & Estuaries	675.67	43.68	270.852	26.2	0.19	TX

Table 3.2 Parameters included in the 30-Year Average water budget by region.

No	NOAA Watershed	PPTN (cm)	PET (cm)	Q (cm)	DS (cm)	GW_{annual}	Budget (cm)	Net GW
	Northeast							
1	Casco Bay	113±7	59±4	63	44±50	–200±17	147±53	150
2	Great Bay	92±6	59±4	44	5±5	10±12	–26±14	-30
3	Merrimack River	105±13	68±8	57	12±20	–35±14	4±29	0
4	Massachusetts Bay	105±7	68±4	45	14±19	–144±164	122±165	0
5	Buzzards Bay	105±6	67±4	123	6±12	–100±30	9±33	0
6	Narragansett Bay	116±8	66±5	49	3±5	–24±24	22±26	0
7	Long Island Sound	106±30	68±19	57	1±5	–3±39	–18±53	0
	Regional Average	106±7	65±4	63	12±15	–71±79	37±69	17
	Mid-Atlantic							
8	Peconic/Gardiners Bays, Long Island	120±7	75±4	93	1±11	14±8	–63±15	-60
9	Hudson/Raritan NY/NJ Harbor	112±29	76±19	62	4±10	–8±34	–22±49	0
10	Barnegatt Bay	112±6	75±4	48	–10±20	–44±63	42±66	0
11	NJ Inland Bays/Great South Bays	102±7	70±5	45	–7±10	–47±45	41±47	0
12	Delaware Bay	104±23	73±17	61	831±1879	–44±113	–818±1882	0
13	Delaware Inland Bays	104±5	73±4	62	–5±7	–64±123	37±123	0
14	Maryland Inland Bays	104±5	75±4	98	5±17	–139±25	64±31	60
15	Chesapeake Bay	109±105	79±76	40	–59±153	–53±140	103±245	0
	Regional Average	108±6	75±2	64	95±298	–48±45	–77±304	0
	Southeast							
16	Pamlico Sound	142±28	87±17	38	7±7	8±33	3±47	0
17	Sapelo Island/Doboy Sound	125±8	171±10	43	–6±2	–113±12	30±17	30
18	Altamaha Sound	128±33	129±34	38	–10±13	–26±44	–4±65	0
19	Indian River Lagoon	122±8	111±7	24	100±42	–43±13	–69±45	-70
	Regional Average	129±9	124±35	36	23±52	–44±51	–10±42	-10
	East Gulf Coast							
20	Charlotte Harbor	136±13	258±24	38	11±20	–89±98	–82±104	0
21	Tampa/Sarasota Bays	113±9	126±10	46	–1±8	–29±47	–30±50	0
22	Apalachee/Ochlocknee Bays	167±22	219±20	33	–8±7	–4±32	–73±49	-70
23	Apalachicola Bay	163±53	190±62	50	39±139	–58±78	–58±179	0
24	Mobile Bay	162±113	182±126	52	5±39	–123±250	47±304	0
	Regional Average	148±23	195±49	44	9±18	–61±48	–39±52	-14
	West Gulf Coast							
25	West Mississippi Sound	150±40	161±43	50	–11±12	–39±71	–11±93	0
26	Barataria/Terrebonne/Timbalier Bays	157±13	179±15	94	–57±52	–95±164	37±173	0
27	Calcasieu/Sabine Rivers	126±50	118±47	34	–9±50	–145±118	129±146	0
28	Galveston/Matagorda Bays	97±102	135±42	11	–18±347	–223±215	192±444	0
29	Coastal Bays and Estuaries	73±32	168±73	14	479±645	–536±324	–51±726	0
	Regional Average	121±33	152±48	40	77±273	–208±98	59±266	0

alternative for evaluating PET. In the Argentina case, this model was found to be more accurate (Varni and Usinoff, 1999). However, Thornthwaite's model usually produces more reasonably accurate results over annual to decadal periods, such as in long-term studies like this one, because assumptions are smoothed out over 30 years (Thornthwaite and Mather, 1957).

Change in storage (ΔS) is an important term to estimate because it accounts for the net loss or gain in water in the region of interest. A negative ΔS is added to the budget to balance the net loss, whereas a positive ΔS must be subtracted. In order to calculate the change in storage, well water level data was obtained from the US Geological Survey (USGS, 2000). A minimum of five groundwater wells was chosen for each watershed. Wells were chosen based on: (1) their location within the watershed to allow coverage of all areas within the watershed; and (2) the period and frequency of well water level measurements taken (5-30 years). Raw well data used in the calculation of ΔS, as well as other parameters applied to the analysis for change in storage, are recorded in Appendix A, Tables 1-4.

The general geology of each watershed and, where possible, the specific geology through which the wells penetrated was collated from the USGS database. Aquifer geologic features were then compared with a general list of specific yield (Sy) values and geology (Kasenow, 2001). I calculated the difference between the highest and the lowest recorded water levels for each well over the period of record. The result was then multiplied by the estimated aquifer specific yield, defined as the ratio of water volume drained from the aquifer under gravity to total aquifer volume. All components used in the water budget were thirty-year averages.

Watershed change in storage was calculated using the following equation:

$$\sum \Delta S = \sum [(WL_H - WL_L)/ t_{H\text{-}L}]\ Sy\ A \qquad (4)$$

where WL_H and WL_L are the highest and lowest recorded water levels (m) in each groundwater well during the period of record, respectively; Sy is the specific yield of the aquifer – defined as the ratio of volume of water drained under gravity to aquifer total volume; A is the area of the watershed (m^2); and t_{H-L} is the length of time between the recorded high and low water levels (s); and $\sum \Delta S$ refers to ΔS integrated over the watershed area (m^3/s).

It was necessary to evaluate groundwater inputs, GW_i to the watershed before estimating the 30-year net GW. Even though watershed delineation is usually based on topography, aquifer contributions may occur from outside this surface watershed. Subsurface watershed inputs via groundwater movement are necessary to obtain a better estimate of coastal net groundwater flow. Hence, for each coastal watershed, a representative annual water budget was performed, using the following equation:

$$(P \pm p)_{yr} - (PET \pm pet)_{yr} - Q_{yr} \pm (\Delta S \pm \Delta s)_{yr} = Residuals_{yr} \qquad (5)$$

The annual budgets could not be estimated for the same year for all watersheds because of the timing of the well water level measurements. However, all the wells for a specific watershed were analyzed for the same year. Wherever possible, typical years were chosen. If a specific year seemed atypical for precipitation and/or discharge, an average of two or three years was used instead. For the northeast, all annual budgets were carried out for 1994 to 1995. The range of years for the mid-Atlantic was from 1995 to 1997, except for Peconic/Gardiners Bays where data for 1985 was used. In the southeast region, Pamlico Sound and Sapelo/Doboy Sound were analyzed for 1982 to 1984. Altamaha Sound and Indian River Lagoon were analyzed for 1995 and 1999, respectively. Annual budgets were done for years ranging from 1990 to 1999, except for Barataria Bay and the Mobile Bay systems, which were analyzed for 1981.

Precipitation, hours of daylight, and monthly temperature for the year chosen were available from the National Climatic Data Center NCDC Network. Stream discharge for the annual budget was collated from the USGS. Change in storage was then calculated for a single year (Tables A.5 to A.8).

This annual budget analysis produced a residual, assumed to be mainly groundwater flux as all other flows were included. A negative residual suggests that it is a necessary to have a net import of groundwater into the watershed in order to balance the budget. Conversely, a positive residual would mean that excess water is in the watershed and must be exported to balance the budget.

Having obtained an annual groundwater flux, the overall long-term water budgets were calculated, using the following equation:

$$(P \pm p) \pm R_{yr} - (PET \pm pet) - Q \pm (\Delta S \pm s) = \pm (net\ GW \pm gw)_{residual} \qquad (6)$$

where R_{yr} is the annual residual calculated from the annual water budget. All other parameters in this equation are thirty-year averages. Regional averages were then calculated for the northeast, mid-Atlantic, southeast and gulf regions (Table 3.2).

Taking the average 30-year average monthly rainfall, errors were determined based on Winter (1981), where sampling errors for precipitation are based on the density of rain gauges within the watershed. He estimated an error based on a 80 km^2 gauge density. Since the rainfall was estimated from a single gauge within each watershed, the error for 80 km^2 gauge density was integrated over the size of each watershed to obtain a single gauge precipitation error. The two errors (single gauge precipitation and Winter's sampling error) were then multiplied to give the total precipitation error for each watershed. This was the same error applied to PET, since this parameter is based on the same NCDC database as precipitation. Errors for change in storage were calculated as the standard deviation (n>5). Stream discharge did not have errors associated with the numbers because gauge error is highly variable between different types of gauges and information on gauge types used was not available. Stream discharge error would most likely be based on internal precipitation, not accuracy.

3.4 Results

3.4.1 Annual Budget Residuals - The aim of the annual budget was to obtain an estimate of the groundwater input value for each watershed which could then be used in the long-term water budget. In this way, the accuracy of the long-term budget was improved. First, the following annual values were estimated: precipitation, evapotranspiration, discharge and change in storage (Figure 3.2 to 3.4). Net groundwater forms the bulk of the annual residuals calculated, along with any errors that may not be reflected in the individual error estimates (Table 3.2). Twenty-six of the 29 watersheds showed a net import of groundwater for the annual budget (Figure 3.5). The only watersheds showing a net export were Great Bay in the northeast, Peconic/ Gardiners/Long Island in the mid-Atlantic, and Pamlico Sound in the southeast, with relatively low values when compared with the net import values obtained for the other watersheds (Table 3.3).

All regional averages reflected a net import of groundwater into all watersheds on an annual basis, with the west Gulf Coast region showing the highest average import (208 cm) and the lowest import in the southeast (44 cm). Net groundwater import of less than 10 cm was observed in the following watersheds: Apalachee/Ochlocknee Bays, Hudson/Raritan and Long Island Sound. These same trends were observed when the results were integrated over the entire watershed.

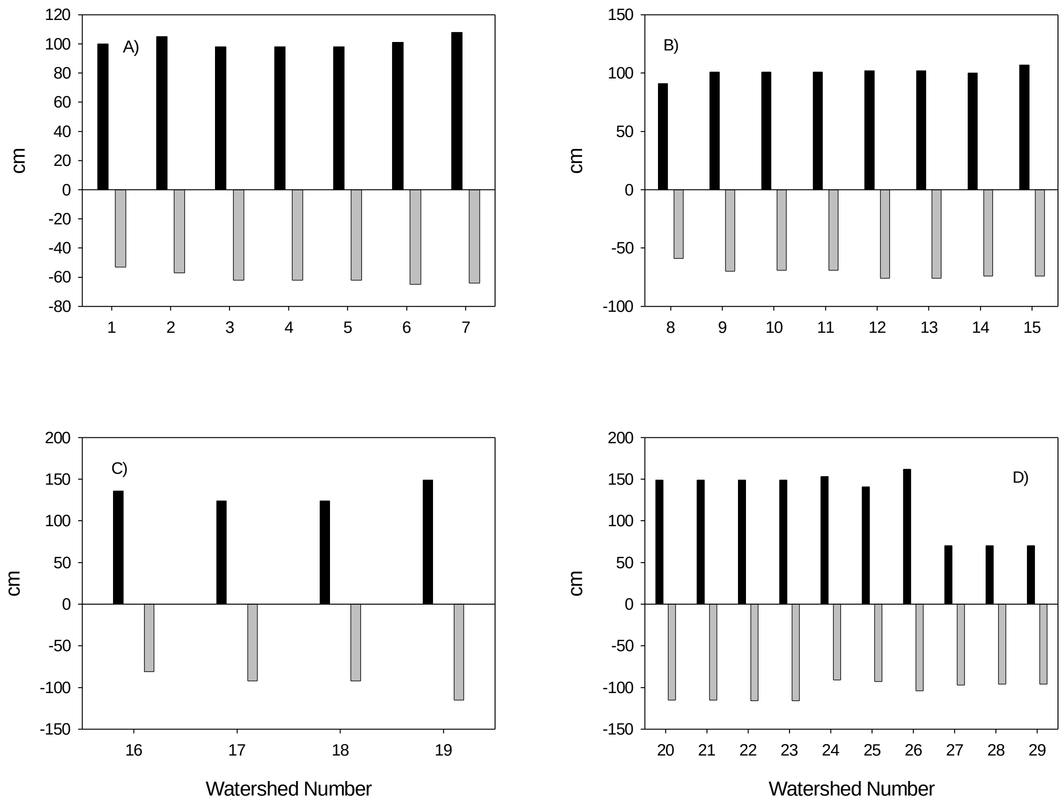

Figure 3.2 Precipitation (black bar) and evapotranspiration (grey bar) results for the watersheds as numbered by region where A) Northeast, B) Mid-Atlantic, C) Southeast, and D) Gulf Coast. (Refer to Table 3.1 for watershed reference numbers).

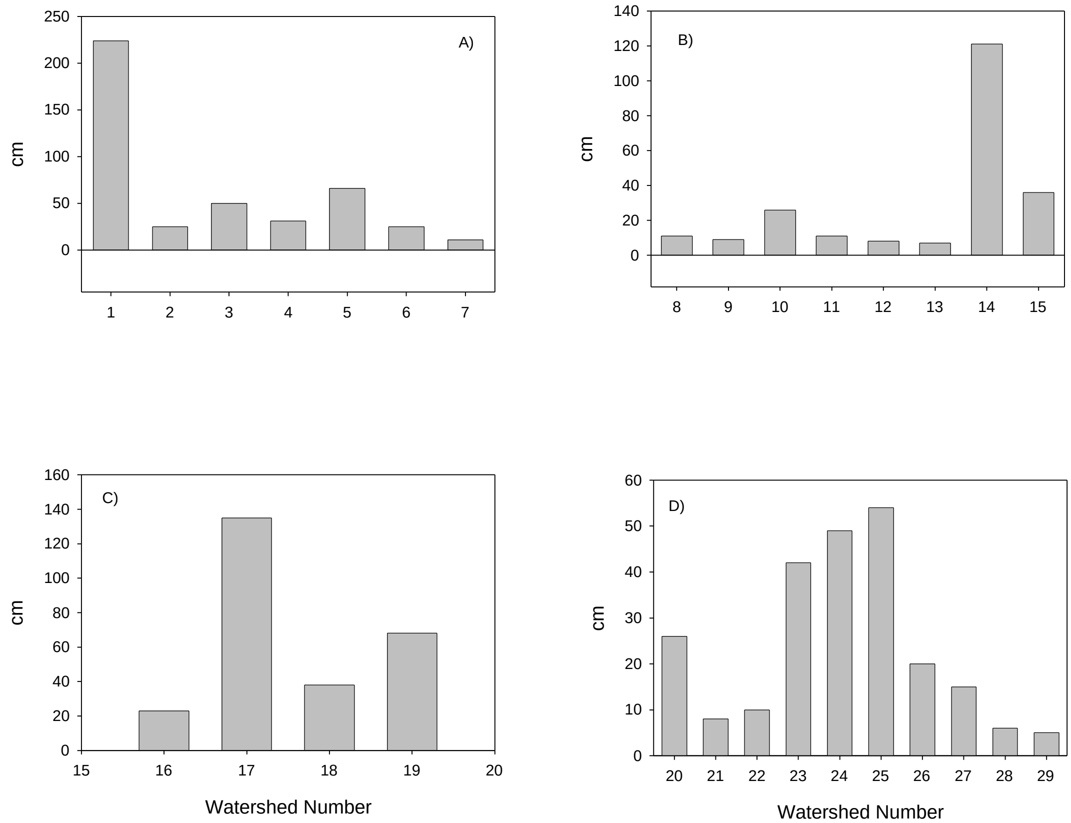

Figure 3.3 Annual stream discharge by region. A) Northeast, B) Mid-Atlantic, C) Southeast, and D) Gulf Coast. (Refer to Table 3.1 for watershed reference numbers).

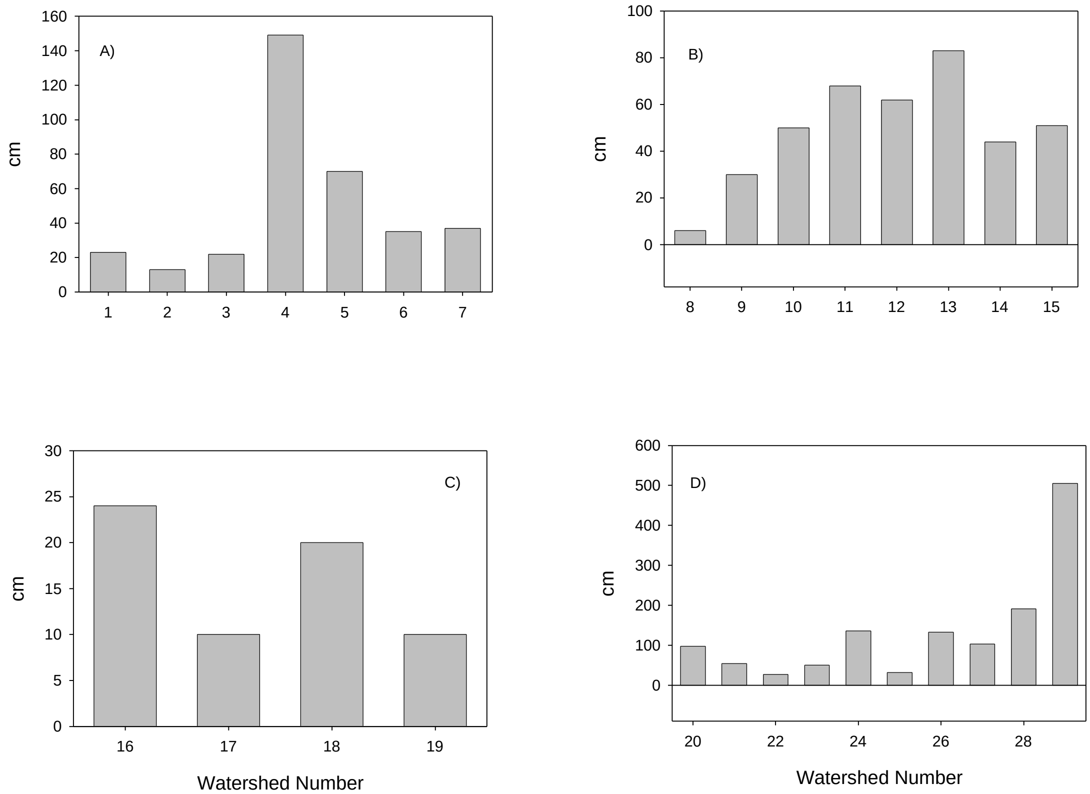

Figure 3-4 Change in storage by region. A) Northeast, B) Mid-Atlantic, C) Southeast, and D) Gulf Coast. (Refer to watershed reference numbers in Table 3.1).

3.4.2 Thirty-year Average Water Budget - The net residual (R_A) term calculated from the annual water budget was incorporated in the 30-year average water budget and considered to include groundwater imported from outside the watershed. Thus, any residuals of this 30-year water budget represented a net flux based on an annual average (R_{30}) import as well as long-term precipitation, PET, discharge and change in storage terms.

Precipitation and PET do not appear to vary much from the northeast to the southeast although the trend is an increase in both parameters from north to south (Figure 3.6). PET for the region is 61% of total annual rainfall. The regional average precipitation in the mid-Atlantic is 108 cm, and 69% is lost from the region through PET. In the southeast region, precipitation increases by more than 18%, but PET is dramatically higher in this region, averaging 124 cm. Of all the regions, the Gulf Coast receives the highest rainfall, with the majority in the East Gulf Coast region (a range of 113 to 167 cm), in contrast to the West Gulf Coast (121 cm). In the West Gulf Coast the watersheds with the highest rainfall are West Mississippi Sound, Barataria/ Terrebonne/ Timbalier Bays and Calcasieu/Sabine Rivers, which receive 150 cm, 157 cm and 126 cm respectively. The highest precipitation (167 cm) was recorded in the Apalachee/ Ochlocknee Bays in the eastern Gulf. PET values varied widely in the Gulf Coast region, but were generally higher than any other region (range: 118 to 258 cm). In general, PET seemed to be independent of the amount of rainfall received by the watershed due to higher annual temperatures and longer days than other regions. The highest PET was recorded in Charlotte Harbor, a watershed with relatively low precipitation. In all cases for the Gulf Coast region, PET values exceeded the total rainfall due in part to high temperatures and larger daylight hours in the watersheds.

Table 3.3 Annual Water Budget Parameters

No.	NOAA Watershed	PPTN (cm)	PET (cm)	Q (cm)	DS (cm)	GW (cm)
	Northeast					
1	Casco Bay	100±6	53±3	224	23±15	-200±17
2	Great Bay	105±7	57±4	25	13±9	10±12
3	Merrimack River	98±12	62±7	50	22±3	-35±14
4	Massachusetts Bay	98±6	62±4	31	149±163	-144±164
5	Buzzards Bay	98±5	62±3	66	70±29	-100±3
6	Narragansett Bay	101±7	65±5	25	35±22	-24±24
7	Long Island Sound	108±31	64±20	11	37±15	-3±39
	Mid-Atlantic					
8	Peconic/Gardiners Bays, Long Island	91±5	59±3	11	6±5	14±8
9	Hudson/Raritan NY/NJ Harbor	101±26	70±18	9	30±12	-8±34
10	Barnegatt Bay	101±6	69±4	26	50±62	-44±63
11	NJ Inland Bays/Great South Bays	101±7	69±5	11	68±44	-47±45
12	Delaware Bay	102±23	76±17	8	62±109	-44±113
13	Delaware Inland Bays	102±5	76±4	7	83±122	-64±123
14	Maryland Inland Bays	100±5	74±4	121	44±24	-139±25
15	Chesapeake Bay	107±104	74±77	36	51±62	-53±140
	Southeast					
16	Pamlico Sound	136±26	81±21	23	24±13	8±33
17	Sapelo Island/Doboy Sound	124±8	92±7	135	10±7	-113±12
18	Altamaha Sound	124±32	92±30	38	20±17	-26±44
19	Indian River Lagoon	149±10	115±11	68	10±4	-43±13
	East Gulf Coast					
20	Charlotte Harbor	149±14	115±16	26	98±97	-89±98
21	Tampa/Sarasota Bays	149±12	115±14	8	54±45	-29±47
22	Apalachee/Ochlocknee Bays	149±20	116±23	10	27±20	-4±32
23	Apalachicola Bay	149±48	116±56	42	50±48	-58±78
24	Mobile Bay	153±106	91±96	49	136±217	-123±250
	West Gulf Coast					
25	West Mississippi Sound	141±38	93±35	54	32±54	-39±71
26	Barataria/Terrebonne/Timbalier Bays	162±14	104±14	20	133±163	-95±164
27	Calcasieu/Sabine Rivers	70±28	97±27	15	103±108	-145±118
28	Galveston/Matagorda Bays	70±73	96±70	6	191±175	-223±215
29	Coastal Bays and Estuaries	70±30	96±29	5	505±320	-536±324

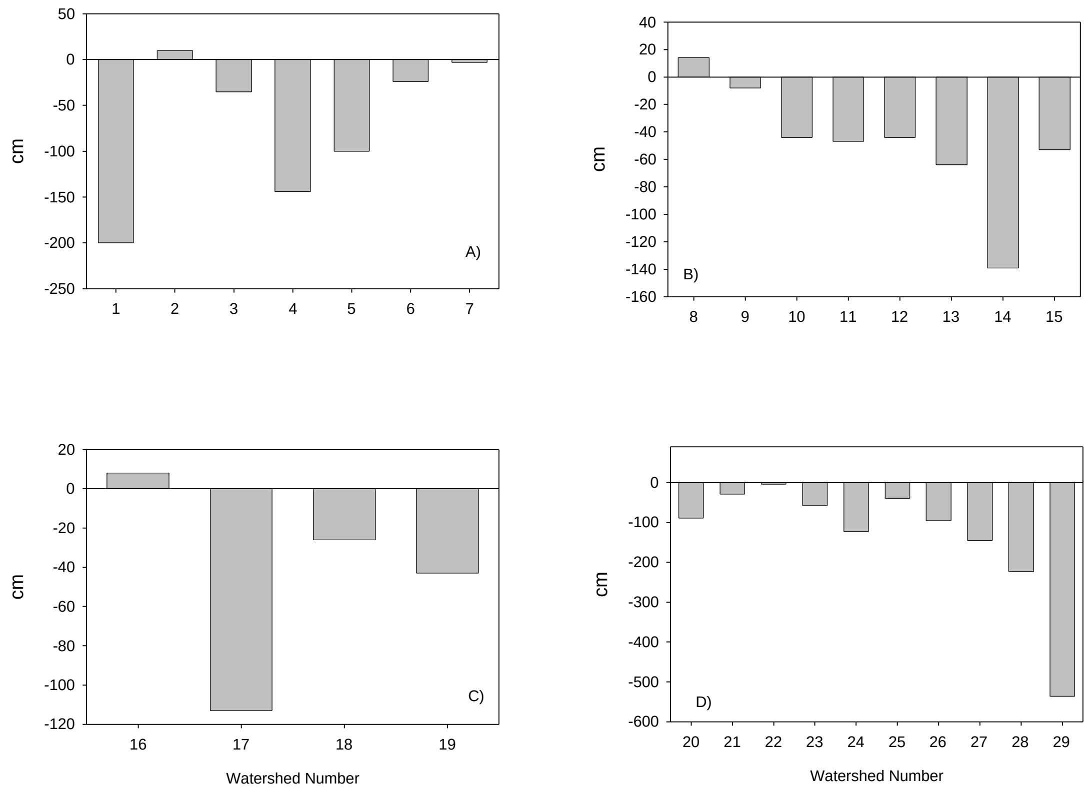

Figure 3.5. Annual net residuals for 29 coastal watersheds. A) Northeast, B) Mid-Atlantic, C) Southeast, and D) Gulf Coast. (Refer to Table 3.1 for watershed numbers).

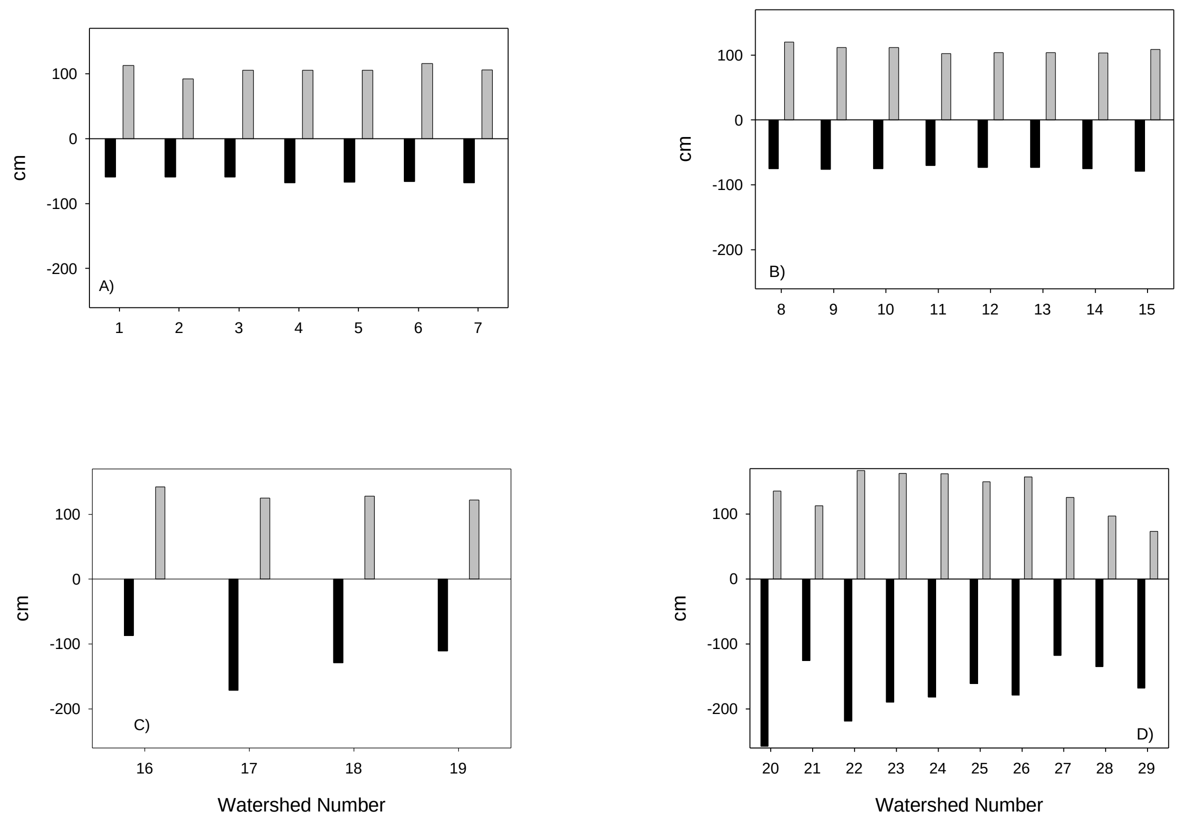

Figure 3.6 Thirty-year average precipitation (grey bar) and evapotranspiration (black bar) for 29 coastal watersheds: A) Northeast, B) Mid-Atlantic, C) Southeast, and D) Gulf Coast. (Refer to Table 3.1 for watershed reference numbers).

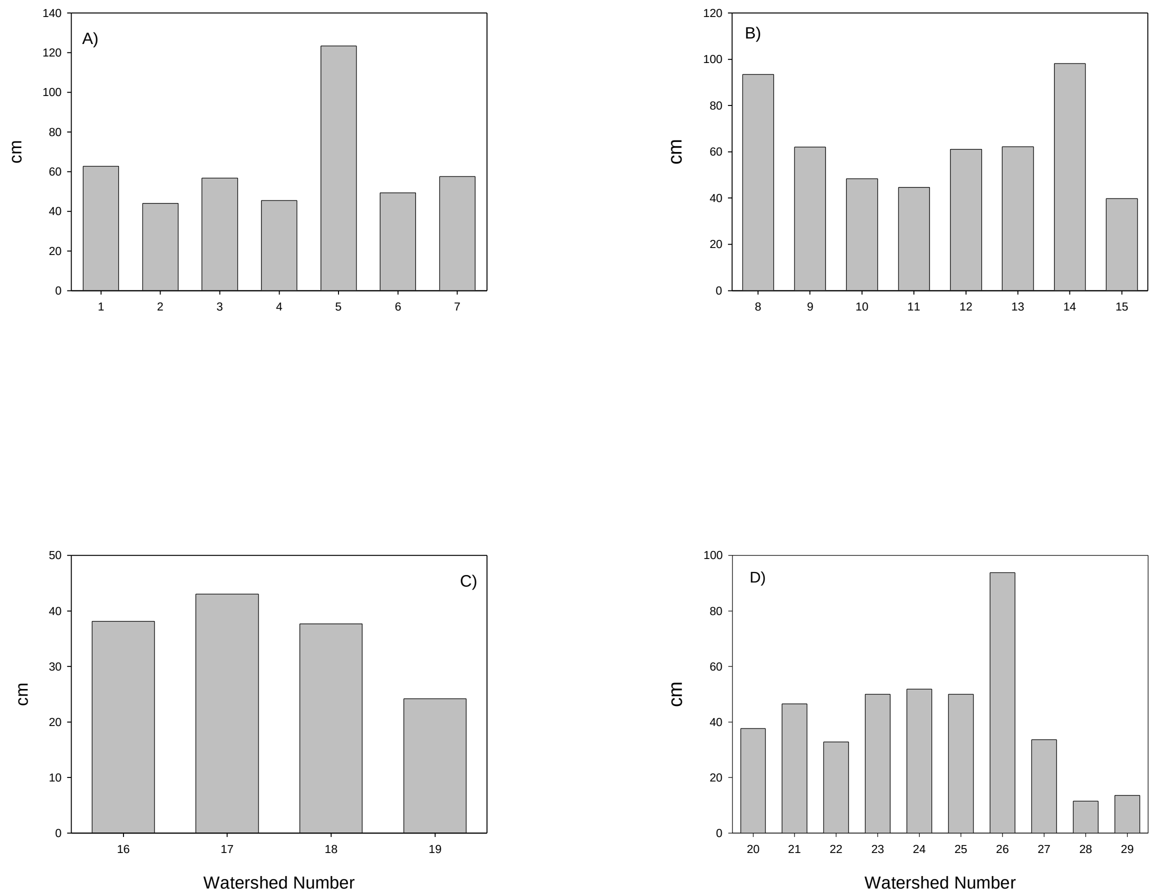

Figure 3.7 Thirty-year average stream discharge for watersheds by region. A) Northeast. B) Mid-Atlantic. C) Southeast. D) Gulf Coast. Please refer to Table 3.1 for watershed reference numbers.

River discharge showed some variation within each region (Figure 3.7). In the northeast (average regional Q= 63 cm), Buzzards Bay lost the most water due to stream discharge (123 cm) and Great Bay the least (44 cm). Area-weighted stream discharge showed that Long Island Sound and Merrimack River had the highest discharge (Table 3.3). These two watersheds are the largest in the northeast region.

In the mid-Atlantic region, area-weighted average stream discharge (63 cm) was comparable to the northeast. Maryland Inland Bays lost 98 cm due to discharge and the smallest loss was Chesapeake Bay (40 cm). When not adjusted for watershed area, Chesapeake showed the highest stream discharge (2026 m^3/s), because it is the largest watershed in the mid-Atlantic region. Rivers associated with the Chesapeake Bay include the Susquehanna, Rappahanock and Potomac, all very large rivers in this region.

Low stream discharge variation occurred in the southeast region (range: 24-43 cm) were Indian River Lagoon was the lowest of all southeast watersheds at 24 cm. Area-weighted stream discharge separated Pamlico Sound and Altamaha Sound from Indian River Lagoon and Sapelo/Doboy Sound. The former two showed discharge an order of magnitude higher than the latter two watersheds.

In the east Gulf coast, stream discharge varied little among watersheds, except when adjusted for watershed area. Mobile Bay was by far the largest watershed with the greatest discharge by area (1850 m^3/s). By contrast, the west Gulf coast watersheds were highly variable (range: 11-94 cm). Higher discharge values were observed for Barataria/ Terrebonne/Timbalier Bays. The Atchafalaya River and Bayou Lafourche are the two major streams associated with this watershed. Low discharge characterized the watersheds of west Texas – the Galveston/Matagorda system and the Corpus Christi/Laguna Madre system known as the Texas

Coastal Bays and Estuaries. In these two watersheds, precipitation was extremely low, and evapotranspiration high. Limited runoff contributed to low stream discharge in these watersheds. Values for change in storage (ΔS) were calculated over the 30-year period of record and integrated over the watershed (Figure 3.8). Fifty five percent of the watersheds studied increased storage over the periods of recorded well water level measurements. The greatest increase occurred in Delaware Bay (831 cm), making the regional average for the mid-Atlantic the highest of all the regions. However, the greatest reduction in storage was also in the mid-Atlantic region. Chesapeake Bay had a reduced storage of 59 cm.

Little change in storage occurred for the northeast and the east Gulf Coast regions (average ΔS = 9 cm). However, higher values were observed in the west Gulf Coast and the southeast, with the largest variation in the former region (range = 479 cm to –57 cm). This region also showed high variation in both precipitation and stream discharge.

3.4.3 Thirty-year Average Net Groundwater For the 30-year average water budget, only seven of the watersheds showed a residual (Figure 3.9). As with the annual budget, errors associated with this 30-year budget were sometimes greater than the residuals themselves. In such cases, the net flux was assumed to be zero. Of those seven, four were found to have a further net import of groundwater beyond the annual import calculated: Great Bay, Indian River Lagoon, Apalachee/Ochlocknee Bays and Peconic/Gardiners/Long Island. Three watersheds had a net thirty-year average export: Casco Bay, Maryland Inland Bays and Sapelo Island/Doboy Sound. In the west Gulf Coast, there was no net thirty-year average flux found for any of the watersheds.

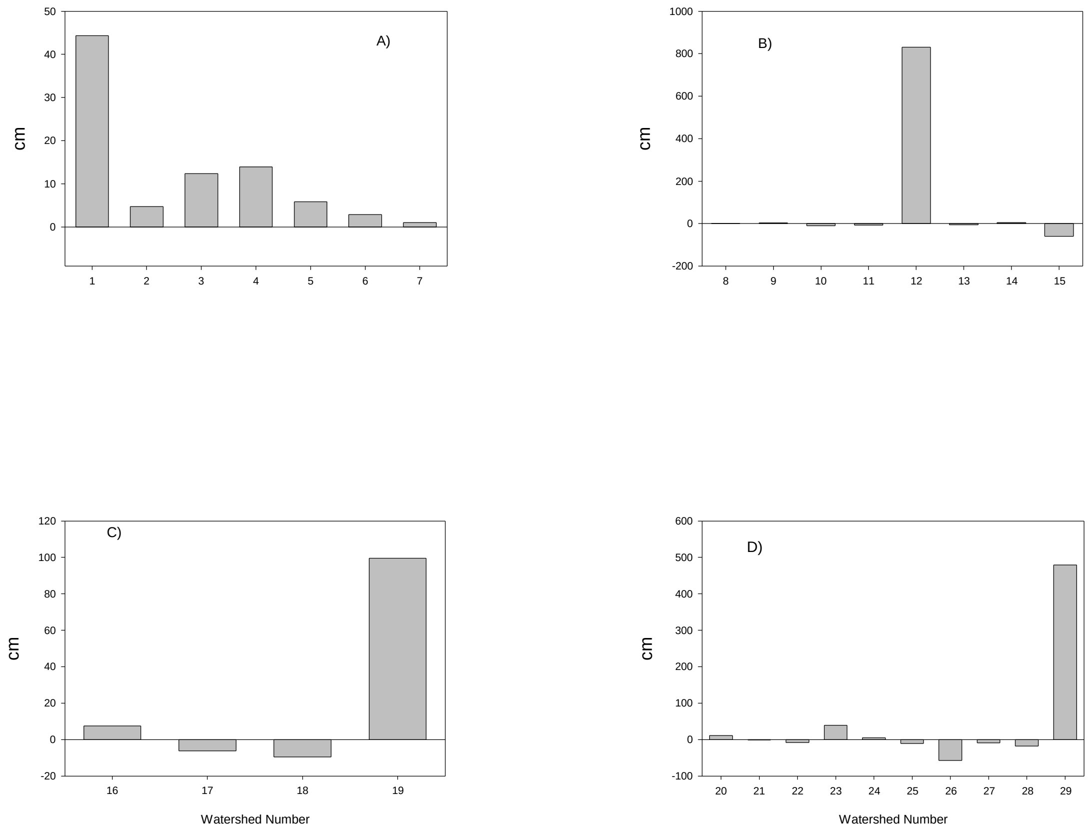

Figure 3.8 Thirty-year average change in storage by region. A) Northeast. B) Mid-Atlantic. C) Southeast. D) Gulf Coast. Please refer to Table 3.1 for watershed reference numbers.

3.5 Discussion

Annual net groundwater varied within and among regions. Groundwater flows in the northeast and mid-Atlantic were very close to zero, whereas the Gulf coast showed a large import (Table 3.2). The annual budget was used to provide an estimated annual groundwater import value for each watershed. Because the result obtained from the 30-year average budget contains other minor parameters besides net groundwater, this calculation of annual groundwater import was justified to improve the accuracy of the 30-year budget results. One limitation of the annual groundwater import values is that the year chosen may have been atypical in terms of one or more of the budget terms. Because the year was chosen solely on the basis of available data, we cannot be certain that the results were always representative. The years studied ranged from 1982 to 1999 for the annual budget. However, more than 50 percent of the annual watershed budgets were carried out for the years 1994 to 1996.

The results show clearly that groundwater import is significant in many of these watersheds, especially in the west Gulf Coast, and specifically in Louisiana and Texas (Tables 3.2 to 3.3). This result is not surprising for Texas because precipitation is low and PET is high in that area. A hydraulic gradient should be expected to set up to cause inflow of groundwater into the aquifers.

Three watersheds were found to have excess water for subsurface export: Pamlico Sound, Peconic/Gardiners Bays, and Great Bay. These export values were small and did not shift any of the regional groundwater averages from net import to net export. All regions showed some level of annual groundwater import, suggesting that groundwater influx into these watersheds may require more intense study.

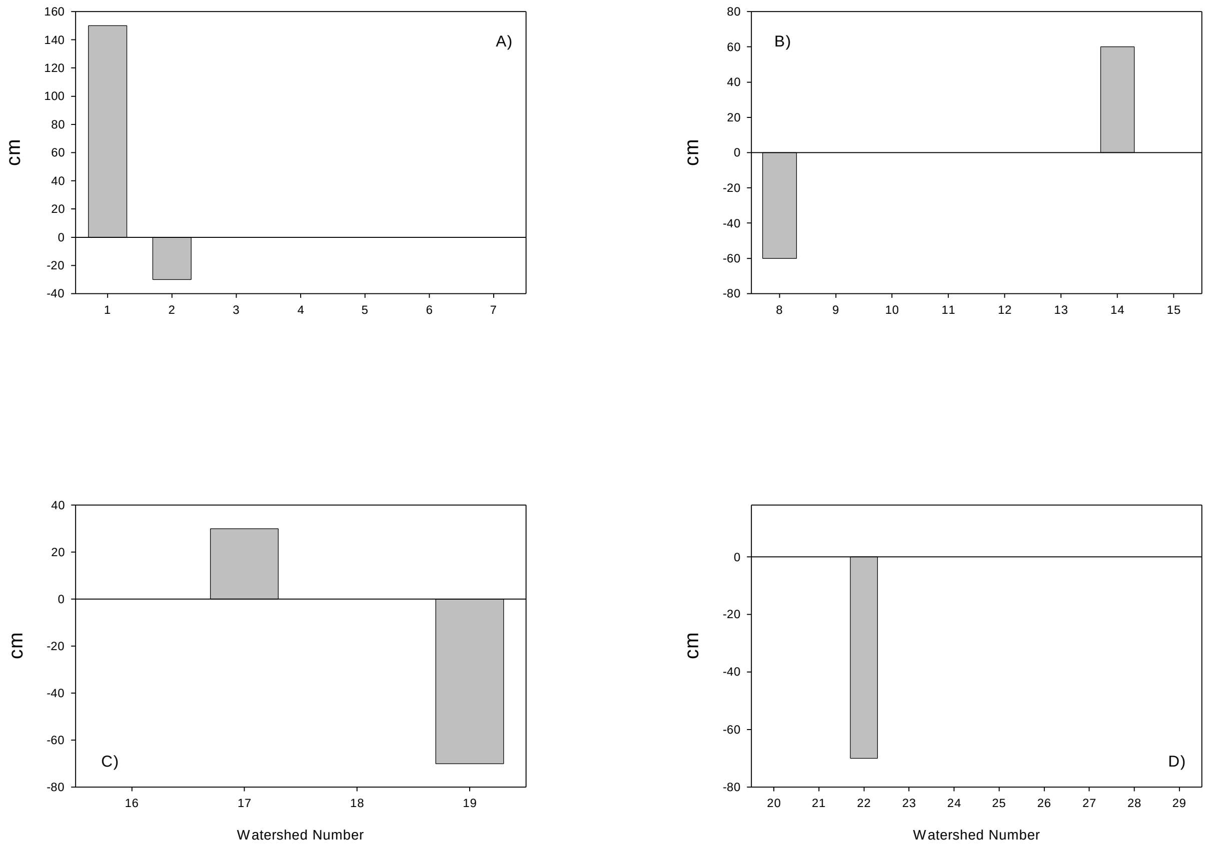

Figure 3.9 Thirty-year Water Budget Residuals. A) Northeast. B) mid-Atlantic. C) Southeast. D) Gulf Coast. Please refer to Table 3.1 for watershed reference numbers

After the groundwater import values were removed from the 30-year residuals, seven of the watersheds showed additional net groundwater influx. Casco Bay, ME averaged 150 cm of groundwater imported into the watershed over the 30 years. This was the largest net groundwater result obtained for the 30-year water budget.

One potential implication for large groundwater imports into these coastal watersheds is significant subsurface transport of contaminants and other harmful substances. Thus, it is necessary to understand these flow volumes and rates to improve management of watersheds and accompanying wetlands. Nevertheless, it is also understood that water budgets are not a perfect tool for assessing groundwater fluxes.

Soil heterogeneity can influence the results obtained from a water budget, where soils with a high hydraulic conductivity may limit evapotranspiration by percolation (Kim et al, 1997). However, this same soil type with high vegetation coverage could act to enhance evapotranspiration. Likewise, a marsh with low hydraulic conductivity (high fine-grained mineral soil content) could have a potentially low evapotranspiration. Future water budget calculations may be much more accurate with better estimates of soil hydraulic conductivity and evapotranspiration.

Regional flow patterns, as well as subsurface geology of these watersheds, affect groundwater discharge. Considering the spatial scale chosen for this study, many different geologic types were encountered. The geology of the aquifer housing each groundwater well determined the value of the specific yield used to calculate the change in storage over the period of record. The range of values used for specific yield was 0.14 to 0.39, the change in storage was evaluated using an average number, and errors may have been introduced. If the sediments were stratified, and only the most common geologic type reported for the area, the wrong value could be used for specific yield. However, exchanging one extreme value of specific yield for the other does not account for all of the variation between calculated and measured fluxes. Overall, change in storage values could account for some error in estimates by using multiple wells.

In the watersheds, inter-annual variation exists in the discharge of groundwater. The empirical data for each watershed was taken at different times of the year, with varying precipitation patterns, producing some variation in the results obtained. However, these seasonal differences are not enough to explain all observed variation. In the case of Tampa Bay, the dry season discharge was 3.15-3.45 x 10^5 m^3/day and in the wet season, fluxes increased to 3.41-3.70 x 10^5 m^3/day (Brooks et al, 1993). The 30-year averages applied here should smooth over any inter-annual variation. However, it is entirely possible that discharge from aquifers may occur during parts of the year, which is not discernible in a method such as this.

Another reason for the large inter-annual differences in groundwater flux may be due in part to the water budget technique itself, as discussed earlier. Large uncertainties persist for all the components included in the budget, and these errors increase when the budget is calculated on a regional basis. However, this method still provides a useful tool for modeling flows of groundwater in such systems. Comparisons have already been made using different techniques to measure fluxes, and the greatest difference shown was a five-fold difference between the water budget calculation and direct measurement of freshwater fluxes in the zone of the watershed influenced by tidal stage (Giblin & Gaines, 1990). The water budget consistently gave the lowest estimate of the groundwater flux. However, even if the calculated water budgets were increased five fold, this would not account for all of the variation in calculated and measured fluxes.

Land use patterns might also affect the groundwater flux quality as evidenced by studies that show large differences between groundwater-derived nitrogen inputs in urban areas and in forested wetlands (Gallagher et al., 1996). Researchers in the Chesapeake Bay watershed also determined that observed fluxes reflected adjacent land use activity (Simmons et al., 1990). However, when the nutrient studies were compared by region, no obvious regional relationship between nutrient concentration and groundwater flux was reported. Possible reasons include heterogeneity of nutrient concentrations in groundwater, both on temporal and spatial scales (Valiela et al., 1978). In addition, it is possible that flux quality, not volume, may be affected by land use.

No obvious trend was observed when we compared watershed wetland area to groundwater flux by region (Table 3.5). Wetlands occupy small areas within these coastal watersheds (range: 2 to 18%), with the northeast having the smallest percentage and the western Gulf the largest. The southeast region had average wetland coverage of 12.5%. No relationship appeared between these areas and the groundwater flux calculated by region.

The ratio of wetlands plus forests to total watershed area showed surprisingly high values (Table 3.5). Wetlands and forests covered more than fifty percent of the northeast, mid-Atlantic, and southeast watersheds. This land use trend was the same for the Gulf coast watersheds - the east end at 42% and the west end 43%. This suggests that, as far as primary land use is concerned, regional differences are not enough to contrast against groundwater fluxes calculated.

Water budget results did not vary with primary land use patterns in individual watersheds, so I compared regional land use patterns with the groundwater flux values and found that the Gulf coast had the highest groundwater flux and the most diverse land use patterns. However, in the northeast, the two extremes in land use exist – urbanization and forests. Yet, the groundwater fluxes in the northeast were generally low. We cannot rule out land use as a possible rationalization for all or part of the differences. However, additional research is required, where an in-depth study of different land use systems might yield some information about this variation.

Population densities show little trend by region (Table 3.1). The mid-Atlantic and the northeast were more densely populated (2.06 to 2.88 people per hectare) than the southeast (0.56 people per ha) and the Gulf coast (0.64 people per ha). When these numbers were compared with the flux values no trends were observed.

This study also compared results of the water budget with observed groundwater fluxes in the field. Empirical studies could not be found for all the regions. By far, most of the studies found in the literature were carried out in the northeast region, but no studies with measured fluxes were found for the western Gulf Coast (Table 3.4). A few of the studies are discussed for use in comparing results of this study with observed groundwater fluxes (Table 3.6). Observed fluxes in every case were much higher than the fluxes calculated using the water budget, in some cases by three orders of magnitude. In Chesapeake Bay, the largest watershed studied, observed fluxes varied widely even among empirical studies.

The calculated annual water budget for Pamlico Sound suggested that a small export of 8 cm from the watershed. However the measured flux indicated an export almost three orders of magnitude higher than that indicated by the budget (Brooks et al, 1993). In Chesapeake Bay, Tampa Bay and Great South Bay, that annual water budget suggested an import requirement of 53, 29 and 47 cm respectively. In the literature, measured fluxes showed an export at all three sites (Table 3-5). In the case of Great South Bay, the measured flux was on the order of 1488.48 m^3/sec.

Table 3.4 Land Use Ratios by Region showing percentage of total watershed area that is water surface area, wetland area and wetland plus forest area.

Region	Primary Land Use	Ratio of Wetland Area to Watershed Area (%)	Wetland & Forest to Watershed Area (%)	Water Surface Area to Watershed Area (%)	No. of Watersheds per Region	No. of Studies found in the Literature
Northeast	Urban, Forests	1.85	58.17	24.84	7	15
Mid-Atlantic	Forests, Agriculture	8.73	54.95	10.56	8	15
Southeast	Forests	12.54	59.04	9.93	4	7
Gulf Coast East	Forests, urban, agriculture, wetlands	7.75	41.59	9.96	5	4
Gulf Coast West	Forests. Urban, rangeland	17.97	43.04	11.76	5	0

The techniques used in determining the fluxes in these studies were benthic flux chambers, seepage meters and mini-piezometers (Lee, 1977; Lee & Cherry, 1978). Depending on the techniques utilized, measured groundwater discharge to a watershed may contain not only freshened groundwater inputs, but also recirculated seawater (Bokuniewicz & Pavlik, 1990). This recycled seawater may be one suggestion for the differences we noted between calculated and measured groundwater fluxes. Already, some evidence exists for this phenomenon. Studies involving the use of benthic flux chambers report greater than 65% saltwater in the chambers, suggesting an overestimation of fresh water flow by up to 50% (Giblin & Gaines, 1990). Assuming that 50% of the measured flux is groundwater, the empirical results begin to approach the calculated water budget flux. Additional evidence from Bokuniewicz and Pavlik (1990) show that re-circulating seawater may account for 30-40% of the seepage in Long Island, New York. Simmons et al. (1992) also suggested that submarine groundwater discharge into Chesapeake Bay was mixed with recycled coastal seawater. Recent measurements in the South Atlantic Bight suggest that recycled seawater may account for as much as 90% of the measured groundwater flux in that region (S. Joye, pers. comm). It is necessary to develop new methodologies for accurately measuring this potentially large component of measured groundwater flux in order to fully understand how new sources of groundwater and what concentrations of accompanying solutes enter these coastal watersheds.

3.6 Summary and Conclusions

1) Annual water budgets may be combined with longer term water budgets to improve estimates of net groundwater flow through watersheds.

Table 3.5 Comparison of Water Budget results to observed fluxes per unit area of watershed.

Watershed Name	R_A GW Flux (m^3/sec)	Measured Flux (m^3/sec)	Reference
Tampa Bay	-29	4	Brooks et al., 1993
Chesapeake Bay	-53	83	Simmons et al., 1990
Pamlico Sound	8	2510	Simmons, 1992
Great South Bay	-47	1488	Bokuniewicz, 1980

2) According to water budgets results for U.S. East and Gulf coast watersheds, net groundwater flow is dominated by import except in the northeast, whereas individual watershed measurements of groundwater flow suggest significant export.

3) This study suggests, based on other recent salinity studies, that differences between calculated and measured groundwater flows might be due to some percentage of recycled seawater along with new, freshened inputs.

4) Other studies have indicated differences in groundwater nutrient concentration depending on prevailing land use, a relationship that was not supported on a regional scale by this water budget study.

Chapter 4

Evaluation of ^{226}Ra and ^{222}Rn Cycling in a Deltaic Estuary of the Mississippi River

4.1 Introduction

4.1.1 Radium-226 and radon-222 behavior in estuaries – For the past two decades, naturally occurring radioisotopes have been utilized to understand surface water-groundwater exchange in marine, estuarine, and salt marsh systems. The behavior of ^{222}Rn ($t_{1/2}$ = 3.8 d) and ^{226}Ra ($t_{1/2}$ = 1620 yr) in estuaries makes them ideal for this type of survey, and their usage has increased, as researchers understand the links between observed nuclide behavior and ecosystem hydrology. These two tracers provide powerful tools for the identification of water masses of different origins.

Radium-226 has been used previously to understand oceanic mixing (Chan et al., 1976; Broecker et al., 1976). In the ocean, ^{226}Ra concentration is found to increase with depth due to a sediment source. In rivers, radium concentrations vary with salinity (Rona and Urry, 1952; Moore, 1967). Radium in coastal freshwater/marine mixing waters does not exhibit strictly conservative behavior. Blanchard and Oakes (1965) first reported that in coastal waters, radium exhibited higher concentrations than in rivers or in open oceans. Theoretically, all conservative elements should vary linearly with salinity. But in the mixing zone between river and ocean, radium concentrations fall above the theoretical mixing line, suggesting an additional influx of radium to waters of the intertidal zone. These higher activities occur within the estuarine mixing zone due to the optimized desorption of radium from river sediments at intermediate salinities of 4-20. This non-conservative behavior in brackish conditions was observed by Li et al. (1977) in Hudson Bay, and similar results were seen in Winyah Bay, Chesapeake Bay, Charlotte Harbor, and Long Island Sound (Elsinger and Moore, 1984; Moore, 1981; Miller et al., 1990; Cochran, 1979). In salt marshes, hydrologic and nutrient studies also utilized tracers, since the origin of observed water column concentration is not always obvious (Krest et al., 2000). Bollinger and Moore (1984,1993) used radium to establish residence times of interstitial waters in a South Carolina salt marsh. They found the marsh itself can be an additional source or sink for radium, and in their case, the marsh supplied a significant input to overlying waters.

Radon-222 concentrations have also been evaluated within shallow water estuaries and in the coastal zone. Activities were found to be higher in the Charlotte Harbor estuary in Florida, as well as in the tidally influenced reaches of the Peace and Myakka Rivers (Miller et al., 1990). Elsewhere in Florida, Fanning et al., (1987) examined variations in concentrations of both ^{222}Rn and ^{226}Ra in shelf waters, and concluded that zones of high concentration might coincide with injection of groundwater, or might be due to the presence of uranium-bearing phosphate sediments. Cable et al. (1996b) further noted that the results of Fanning et al. matched locations of submarine springs along the coasts of Florida, and showed a qualitative relationship between radon concentrations and submarine groundwater discharge (SGD). Thus, like ^{226}Ra, ^{222}Rn is enriched in bottom waters relative to the upper reaches of the water column. However, groundwater is not the only source for bottom water enrichment of ^{222}Rn. Another source is diffusion from sediments, and this occurs both in ocean waters and shallow systems. In the case of ^{222}Rn, an additional sink occurs at the water surface. Gas exchange at the air-sea interface is a sink for dissolved gases such as ^{222}Rn, especially at high wind speed (Broecker and Peng, 1974; Wanninkhof, 1992). It has been suggested that depth-dependent gas transfer might be the primary control on observed water column profiles of ^{222}Rn because of high rates of vertical mixing, temperature, and piston velocity (Broecker et al., 1968).

4.1.2 Geochemistry of Tracers – The two naturally occurring isotopes used as tracers in this study, ^{226}Ra and ^{222}Rn, are part of the ^{238}U decay series. Elemental uranium exists in large concentrations in sediments and sedimentary rock formations, and as decay occurs, radioactive daughters including thorium, radium, and radon, are produced in series ending with stable lead (Friedlander and Kennedy, 1949).

Radon-222 is a dense, gaseous, radioactive element found in group 18 of the periodic table (Dickson, 1971). It is mostly inert, with similar chemical behavior to the other noble gases. Since it is mainly produced in the earth's crust where uranium concentration is highest, the amount of gas that escapes to the atmosphere is dependent either on the permeability of surrounding areas, or on a nearby conduit, such as water. The gas is highly soluble, and groundwater readily transports high concentrations of dissolved ^{222}Rn, allowing atmospheric evasion when the groundwater surfaces (Broecker, 1965).

Radon-222 is useful for tracing groundwater discharge because it is chemically conservative and easily measured. When this radioisotope is measured in streams and standing surface waters within a watershed, groundwater inflows can be detected and quantified, mainly because of an increase in the concentration of ^{222}Rn in the surface waters (Ellins et al., 1990). This increase is due to the presence of groundwater ^{222}Rn concentrations that are several orders of magnitude higher than in surface waters (Cable et al., 1996a). As the groundwater mixes with surface waters, decay of ^{222}Rn occurs, the gas is lost to the atmosphere, and concentration levels quickly dissipate.

Radium-226 is a member of the alkaline earth metals (Group 2) of the periodic table and is chemically reactive. Its properties are similar to those of other +2 elements, particularly barium). As a result, alkaline earths are often studied concurrently in the world's oceans and rivers (Broecker and Peng, 1982; Moore, 1997). Its cation exchange capacity influences observed concentrations under brackish conditions. Radium-226 is not strictly conservative, but is easily measured, and its unique behavior makes it ideal for use as a tracer of water masses. Other isotopes of radium used in hydrology studies include ^{224}Ra ($t_{1/2}$ = 3.6 d), ^{223}Ra ($t_{1/2}$ = 11.4 d), and ^{228}Ra ($t_{1/2}$ = 5.7 yr). For example, these four radium isotopes were used to evaluate groundwater input to the North Inlet salt marsh (Rama and Moore, 1996). The longer lived radium isotopes have proved more useful than the shorter lived isotopes due to the reactivity of radium and the availability of short lived inert ^{222}Rn.

The behavior of these two radioisotopes is a result, not only of their chemistry, but also their position in the uranium decay series. Radium-226, a daughter product of ^{230}Th, seeps into interstitial waters between rocks and becomes concentrated in the pore water. As ^{226}Ra decays to ^{222}Rn, an inert, noble gas, this new daughter tends to be concentrated in groundwater relative to surface water because of entrapment within a confined aquifer. Radium-226 is particle reactive in fresh water (low ionic strength) but quickly desorbs from bottom sediments and suspended solids in brackish salinities (4 to20). Radium then dissolves in interstitial fluid and overlying waters, thus increasing concentrations of both parent and daughter in surface waters (Libes, 1992).

The following equation describes the decay of ^{226}Ra to ^{222}Rn:

$$A_t = A_e (1- e^{-\lambda t})$$

where A_t represents ^{222}Rn activity at time t, A_e is ^{222}Rn activity at secular equilibrium with ^{226}Ra, and λ is the radioactive decay constant for ^{222}Rn. The assumptions on which this tracer method is based are that: (1) the above equation holds true; (2) elemental uranium and ^{226}Ra are evenly distributed at high concentrations throughout the aquifer; (3) loss of ^{222}Rn remain constant over time; and (4) no mixing occurs within the aquifer (Bertin and Bourg, 1994).

The objectives of this study are: (1) to contribute to an overall understanding of the basin hydrology through tracer systematics; and (2) to evaluate spatial and temporal patterns in geochemical tracers.

4.2 Physical Site Description

4.2.1 Geology and Hydrology - Barataria Basin is approximately one half of the largest estuary in the world and is situated south and west of New Orleans, Louisiana at 29^{o}N 50^{o}W (Figure 4-1). It spans about 6,333 square kilometers, and is bounded by the Mississippi River on the east and north sides, Bayou Lafourche on the west side, and barrier islands at the Gulf of Mexico to the south. The Gulf Intracoastal Waterway (GIWW) bisects the basin from northwest to southeast.

South Louisiana was created by thousands of years of delta-switching processes of the Mississippi River. Prior to the early 1900s, the marsh received large sediment loads and high volumes of freshwater by the Mississippi River, and smaller but significant amounts from Bayou Lafourche, producing a vibrant estuary (Roberts, 1997). However, this rich freshwater and sediment source was cut off when the river levees and dams were constructed in the early 1900s. As a result, extensive land loss has occurred. The sediment and freshwater load previously received from the river produced an extensive wetland system, almost 1,900 square kilometers in 1990. This wetland system, once one of the most productive in the state, and indeed, in the nation, with commercial seafood catch from Barataria Basin formerly representing 10% of total U.S. commercial landings, is now dramatically reduced (Keithly, pers. comm.). Currently, little is known about the role of groundwater in marsh and fisheries productivity.

The Mississippi River floodplain is dominated by thick sedimentary deposits of interbedded sand and clay from previous routes taken by the river, which complicate the aquifer systems in the Barataria Basin (Roberts, 1997). Subsurface fluids might enter the wetland system through breaches in the confining layer of an underlying aquifer or river leakage through the Mississippi River levee. The coastal wetlands of Barataria Basin are tidally influenced as far north as the GIWW, depending on the season. As a result, a classic transition exists from salt marshes at the coast to freshwater marshes upstream. This intrusion of salt water has increased the salinity of coastal aquifers in the lower Basin.

Regional climate is humid sub-tropical with hot, moist summers and cool, mild winters. Average summer temperatures exceed 80^0 F while winter temperatures vary from 50^0 F to below freezing at times. Since the construction of levees along the Mississippi River, precipitation is the dominant source of freshwater input to the Basin, with an average of 125-200 cm per year (Figure 4-2). High humidity in the summer facilitates the development of thunderstorms.

4.2.2 <u>Sampling Stations</u> - Eight stations were sampled seasonally for geochemical tracers and physico-chemical parameters along a transect bisecting the watershed from southeast to northwest (Figure 4-1). The ocean endmember samples were collected from shelf waters off Grande Isle, and the upstream freshwater endmember was located in Bayou Chevreuil. Intermediate salinity stations were in Barataria Bay, Little Lake, Barataria Waterway, the GIWW, Lake Salvador, Lac des Allemands, and the Mississippi River. Additional data collected for the 2-yr study period were: (1) wind speeds from the National Data Buoy Center site at Grande Isle; (2) Mississippi River stage at Algiers Lock from the U.S Army Corps of Engineers (2001); and (3) precipitation data from the weather station from the Southern Regional Climate Center, Baton Rouge, Louisiana.

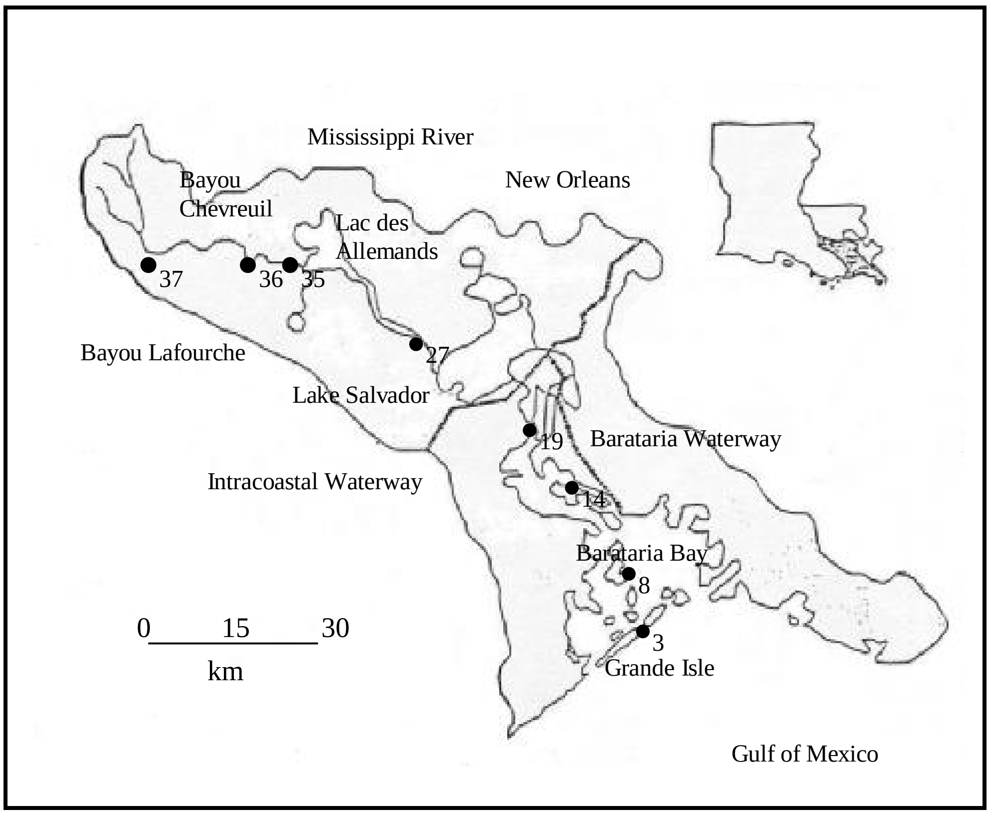

Figure 4.1 Site Map showing sampling stations in Barataria Basin.

4.3 Methods

4.3.1 Water Collection and Analysis - Seasonal sampling was undertaken between May 1999 and August 2001, with bottom water (approximately 0.3 m above sediments) samples collected along a southeast – northwest transect. Samples were collected for ^{222}Rn and ^{226}Ra analysis in 4-L evacuated glass sampling bottles using a peristaltic pump to draw water directly into the bottles. Sample gas loss was carefully controlled during collection - bottles were made airtight and sealed prior to sampling. At each station water temperature, pH, conductivity, salinity, and depth were measured using hand-held meters. Locations of all stations were also recorded using a Global Positioning System (GPS).

Water samples were analyzed first for ^{222}Rn, which was extracted by sparging the sample with helium, forcing gases within the sample into the headspace. Water vapor and carbon dioxide were stripped from the mixture of gases using drierite and ascarite respectively, allowing ^{222}Rn to be collected using cryogenic trapping (Broecker, 1965). The ^{222}Rn was then transferred to an alpha scintillation cell and left for a 3-h period to allow ingrowth and secular equilibrium of radon daughters, ^{218}Po and ^{214}Po with the parent ^{222}Rn (Cable et al., 1996b). Samples were then counted on alpha scintillation counters to yield a total ^{222}Rn activity in the water sample. This analysis was repeated at greater than 5-day intervals, at least twice for ^{226}Ra. After correction for the sample volume, excess ^{222}Rn activities were calculated by subtracting ^{226}Ra activities (a measure of supported ^{222}Rn) from total ^{222}Rn activities. Results were corrected back to the time of sampling to capture all decay during collection and analysis. Water samples of the final sampling trip were also filtered using 20 to 25 micrometer filters and run for ^{226}Ra to give a first approximation of the significance of ^{226}Ra desorbed from sediments within the water sample itself.

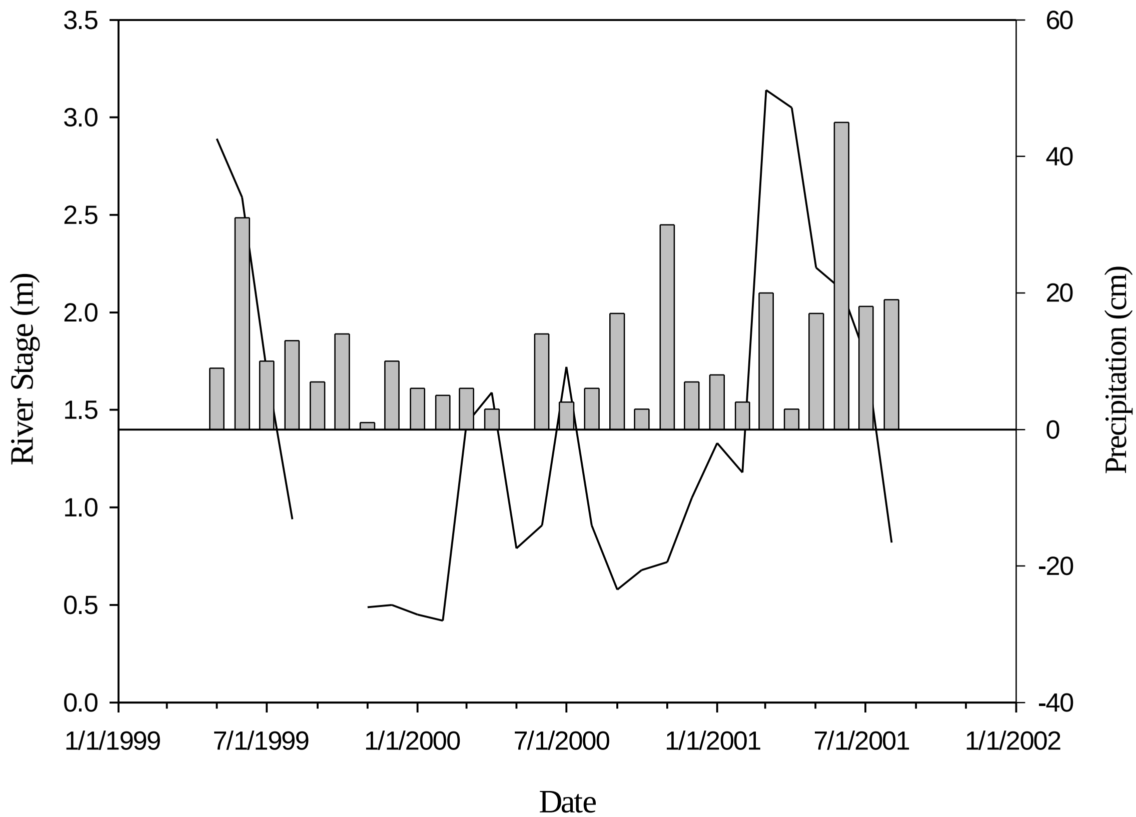

Figure 4.2 Variation in local precipitation (bars) and Mississippi River stage (line) throughout the study period.

For each sampling station, excess water column inventories were calculated according to the equation:

$$(C_{Rn} - C_{Ra})\, z = \text{Excess } C_{Rn} \qquad (1)$$

where C_{Rn} is the measured activity of ^{222}Rn in the water sample (dpm/m^3), C_{Ra} represents the amount of C_{Rn} that is supported by ^{226}Ra in the sample (dpm/m^3), and z is water depth. All values of ^{222}Rn in the study have been reported as excess activities, and were corrected for production and decay of ^{222}Rn during sampling and analysis.

4.3.2 Sediment Collection and Analysis - Bottom sediment grab samples were also taken at each station using a hand auger, and two different analyses of sediments were performed. The moisture content and potential pore space content of sediment samples were measured and used to calculate the porosity. Sediment porosity was measured by weighing wet sediment, then drying for 48 hours and re-weighing. The fraction of water within the sediment (f_{H2O}) and the fraction of dry sediment (f_{dry}) was calculated as:

$$\frac{\text{wet weight (g)} - \text{dry weight (g)}}{\text{wet weight (g)}} = f_{H2O} \qquad (2)$$

The second sediment analysis measured the maximum potential diffusive input of radon from suspended and/or bottom sediments at each site. A pre-weighed 50-g aliquot of wet sediment was placed in 500 ml Erlenmeyer flasks, and mixed with 250 ml water of known ^{226}Ra activity. The samples were sealed and sparged with helium in a similar manner to the bottom water samples described earlier. Samples were then allowed to equilibrate for 30 days before each analysis. During this period, samples were regularly shaken to simulate turbid conditions and enhance radon emanation. Results were used with porosity and wet bulk density to calculate the diffusive flux from the sediments at each sampling site.

Sediment diffusive fluxes (dpm/m^2/d) were determined by equation 2 (Martens et al., 1980):

$$J_{diffusion} = (\lambda D_s)^{0.5} (C_{eq} - C_0) \qquad (3)$$

where D_s is the bulk sediment diffusion coefficient for ^{222}Rn, λ is the decay constant for ^{222}Rn (equaled to 0.181 d^{-1}), C_{eq} is the measured sediment equilibration activity corrected for porosity (dpm/m^3), and Co is the measured water column activity (dpm/m^3).

4.4 Results

Salinity consistently ranged from 25 to 30 for the entire study period at the two ocean endmember stations to less than one at the three furthest upstream stations (Figure 4.3). Seasonal variation in salinity was observed at intermediate stations, where precipitation, tides, GIWW flow, potential subsurface freshwater inputs, or saltwater intrusion might affect the salinity gradient. For example, 50 km upstream at station 19, salinities dropped to less than five in the spring, corresponding to an increase in river discharge (Figure 4-2). Interestingly, at the same site in summer 2001, salinity dropped almost to zero during low river stage, probably due in part to the high levels of precipitation recorded for the watershed during that month. Average soluble ^{226}Ra activity within the Basin for the 2-year study period was calculated as 0.89±0.08 dpm/l, with a range of 0.25 to 2.66 dpm/l.

All activities in excess of 1.0 dpm/l occurred at stations 8 to19. An assessment of radium and salinity showed an increase in radium above the freshwater-saltwater mixing line at those stations with salinities ranging from 4 to 20 (Figure 4-4). This non-conservative behavior for radium is similar to mixing curves obtained in Winyah Bay and Chesapeake Bay (Elsinger and Moore, 1980; Moore, 1981) and suggests an additional source for radium to the water column. The range of activities for Barataria Basin is also comparable to those reported for other estuaries

(Table 4.1). The mixing curves were truncated at the upstream end, suggesting that activities remained higher than the ocean endmember activities at salinities less than 4.

Samples from August 2001 were filtered and radium activities measured to give a first approximation of radium inputs due to suspended sediments. The filter mesh size did not allow separation of the fine clays from the water sample. Nonetheless, stations 8,14, and 19 showed reduced radium activities after filtration, and consistently high ^{226}Ra levels in unfiltered samples for the study period. Average differences in activity between filtered and unfiltered samples at the same three intermediate stations were 0.05±0.01 dpm/l. Li et al., (1977) showed a similar difference between filtered and unfiltered samples in Hudson Bay (0.036 dpm/l).

The highest observed excess ^{222}Rn concentrations occurred at the three upstream stations within the Basin, at distances of 110 km and greater from the open ocean (Figure 4.5). This trend was consistent for the entire study period and varied inversely with salinity (Figure 4.3). These high concentrations (28 to 53 dpm/l) were 47% higher than other measured activities at the same stations, and were measured in the winter months (December to March). In addition, high values coincided with the lowest recorded Mississippi River stage during the study period, which occurred in January and December 2000 (Figure 4-3). However, the March 2001 sampling period reported intermediate levels of excess ^{222}Rn, as well as the highest values of river stage for the study period. The lowest observed values of excess ^{222}Rn throughout the transect were in the fall months. In the downstream half of the Basin, concentrations were low, and ranged from 0.01 to 4.00 dpm/l. The average excess ^{222}Rn value for the Basin was 6.39±0.39 dpm/l (typical marine values = 0.1 to 0.5 dpm/l; Cable et al., 1996a).

Radium-226 activities versus river stage suggested a slight decrease in activity when the river stage was high. The spread of values was wider (0.25 to 2.66 dpm/l) when the river stage

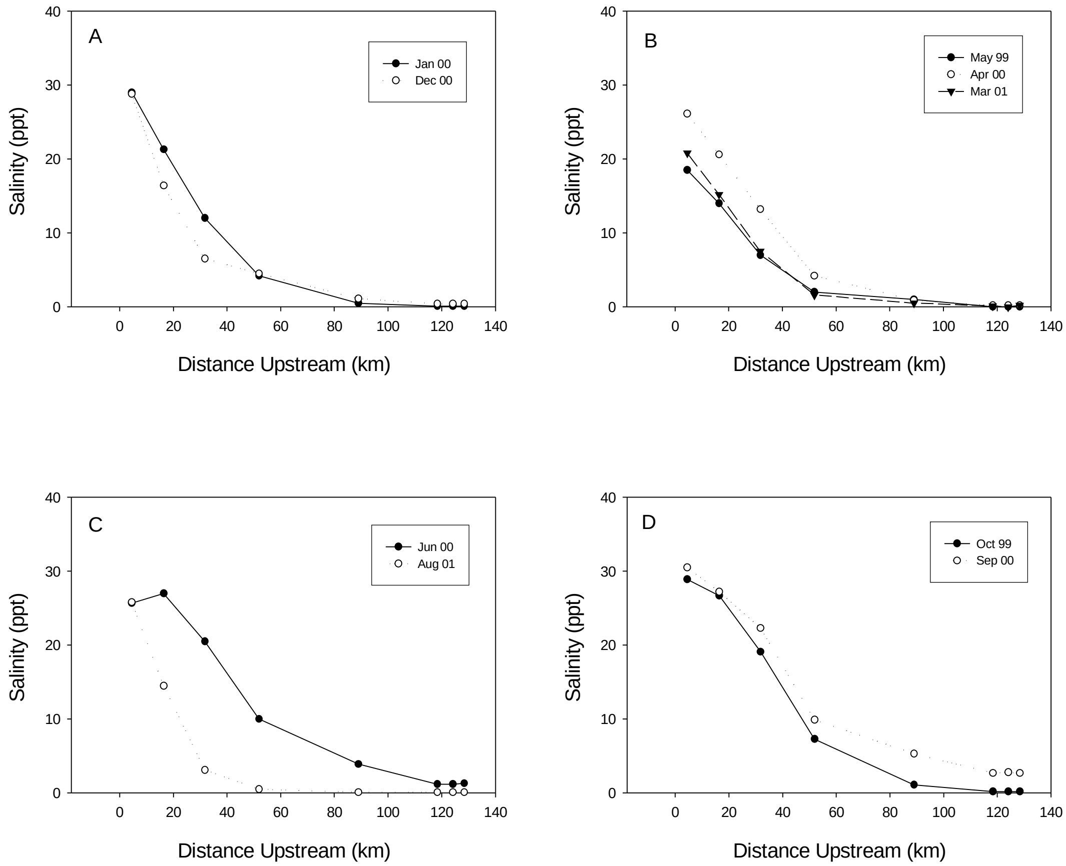

Figure 4.3 Seasonal salinity measured along Barataria Waterway, Louisiana, between May 1999 and August 2001 for A) Winter; B) Spring; C) Summer; and D) Fall.

was low in the summer and winter. Around 1.8 m, the range of activity narrowed by 57% (0.32-1.7 dpm/l), and by another 37% at 2.4 m. Radon-222 activity also seemed to show the same pattern as ^{226}Ra, with a narrower range of activity above 3.0 m river stage. Radium-226 activity within the Mississippi River ranged from 0.42 to 1.1 dpm/l for the study period, while ^{222}Rn activities were 2.1 to 6.3 dpm/l. Both River ^{222}Rn and ^{226}Ra activities were in the low compared to the transect stations. Additional data are found in Appendix B.

4.5 Discussion

The two tracers used in this study have very different chemical signatures in marine, terrestrial, and brackish intertidal environments. Previous studies in other areas indicate that along the salinity gradient from marine to freshwater, ^{226}Ra begins low, then increases and peaks in brackish water, ultimately decreasing in freshwater. For discussion purposes the Basin is subdivided into three zones: downstream or marine zone (stations 3 and 8), intermediate or brackish zone (stations 14, 19, and 27) and upstream or freshwater zone (stations 35, 36, and 37).

Potential sources of ^{226}Ra within the Barataria Bay estuary are: (1) horizontal transport from the Gulf of Mexico; (2) subsurface to surface hydraulic connections and gradients; and (3) desorption from suspended and bottom sediments. Station 3, the Gulf of Mexico endmember station, showed the lowest ^{226}Ra and ^{222}Rn on almost every trip, suggesting that if an ocean source exists, its inputs do not significantly affect the tracer mass balance within the estuary.

A subsurface hydraulic connection may exist between the Mississippi River and Barataria Basin because the Basin is underlain by former distributaries of the river which have been subsequently covered by fine clay sediments. River effects within the estuary were clear during some sampling trips and at certain stations. In the spring seasons during sampling, when the Mississippi River discharge peaked, low salinities within the estuary were pushed further

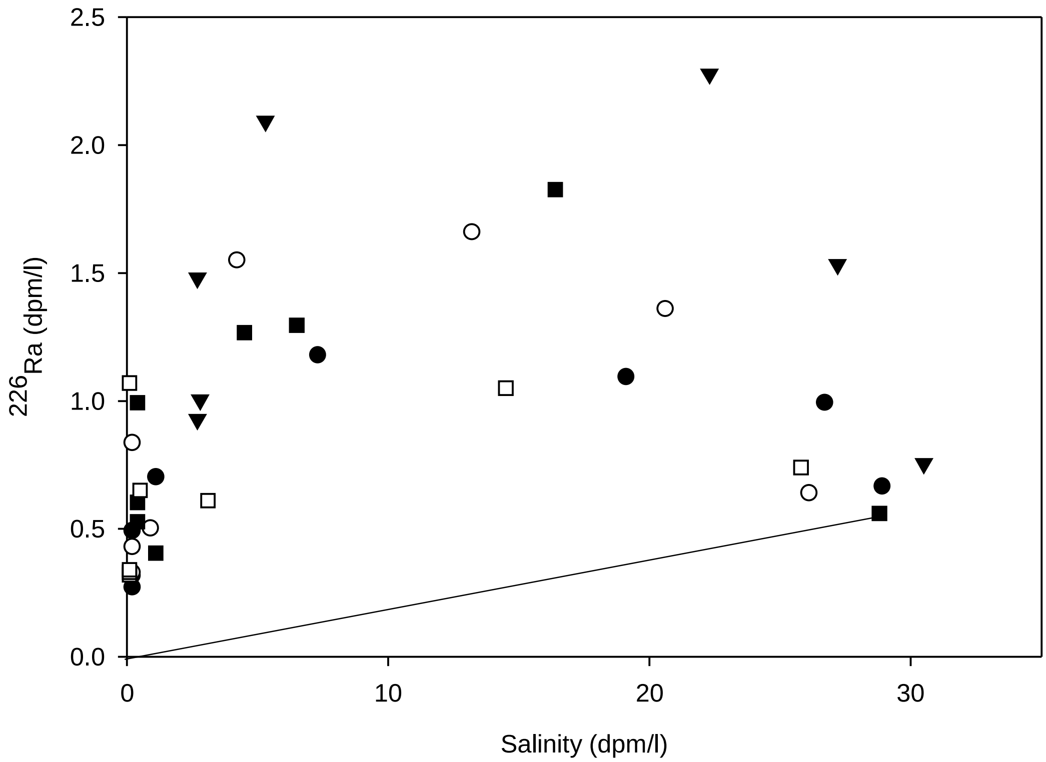

Figure 4.4 Relationship of Ra-226 activities to conservative mixing line (dashed line) along the Barataria Basin Transect for Oct-1999 (closed circle); Apr-2000 (open circle); Sept-2000 (closed triangle); Dec-2000 (closed square); and Aug-2001 (open square).

Table 4.1 Diffusive fluxes of ^{222}Rn from sediment at stations along the Barataria Waterway measured on sediment grab samples collected in January 2000.

Station ID	Porosity	^{222}Rn Sediment Porewater - Ceq (dpm/l)	Rn-222 Bottom water (dpm/l)	Diffusive Flux ($J_{diffusion}$) (dpm/m2/day)
Barataria Transect				
BT#3	0.67	216 ± 5	1.21 ± 0.19	356 ± 9
BT#8	0.75	213 ± 5	3.97 ± 0.16	808 ± 20
BT#14	0.64	668 ± 10	0.53 ± 0.12	1105 ± 16
BT#19	0.88	311 ± 6	0.01 ± 0.12	1195 ± 24
BT#27	0.85	528 ± 8	1.98 ± 0.10	2023 ± 31
BT#35	0.88	532 ± 8	19.15 ± 1.19	1978 ± 31
BT#36	0.92	363 ± 9	16.85 ± 0.42	1334 ± 34
BT#37	0.88	420 ± 10	28.43 ± 2.16	1519 ± 39

The radon diffusion coefficient (D_m) is 5.76 x 10^{-6} cm^2/sec;
porosity (ϕ) is 0.81; and wet bulk sediment density (ρ_{wet}) is 1.31 g/cm^3.
Fluxes range from 356 to 2023 dpm/m^2/day with a mean ($\pm 1\sigma$) of 1290±25 dpm/m^2/day and a coefficient of variation of 53%. Average sediment wet weight was 50 g.

seaward and results showed a reduction in radium activity at high river stage (Figure 4-2). It could be suggested that these observed salinity effects are the result of the river plume wrapping around the coast and affecting the estuarine salinities. This scenario is unlikely because the downstream station salinity continued to be high and radium values reflected conservative mixing consistent with Gulf of Mexico activities. Miller et al. (1990) also reported lower ^{226}Ra activity at high discharge in Charlotte Harbor. Using only the results of this study, river stage does not appear to directly affect tracer concentrations along the Barataria Waterway.

Radium desorption from sediments contributes to elevated water column inventories at intermediate salinities within the estuary. As saltwater moves upstream and mixes with fresh water coming downstream, ^{226}Ra adsorbed on suspended sediment particles is released by ion exchange in brackish water. Researchers have shown that 0.01 dpm/l of ^{226}Ra were released from suspended solids in the Peace River when mixed with Gulf waters (Miller et al., 1990). If Barataria radium desorption is similar, then a significant excess water column inventory of ^{226}Ra still exists at the mid-salinity areas.

Other sediment sources for ^{226}Ra are bottom sediments releasing ^{226}Ra and ^{222}Rn to the overlying water column by ^{230}Th decay to ^{226}Ra, and desorption and dissolution of ^{226}Ra into bottom waters. Results show that sediment diffusion in Barataria Basin accounts for 0.8 to 31.8% of unsupported ^{222}Rn in the upper Basin and 1.9 to 89.2% in the lower Basin. In the intermediate Basin, 100% of the water column inventory may be accounted for by diffusion at stations 14 and 19 during the winter. The highest diffusion rate was at station 27, but this high diffusion could not adequately account for all the unsupported radon observed at that site. Thus, high variation in diffusive fluxes (12 to 100%) and excess ^{222}Rn characterized the intermediate Basin, suggesting that a number of different processes may be influencing the observed activities

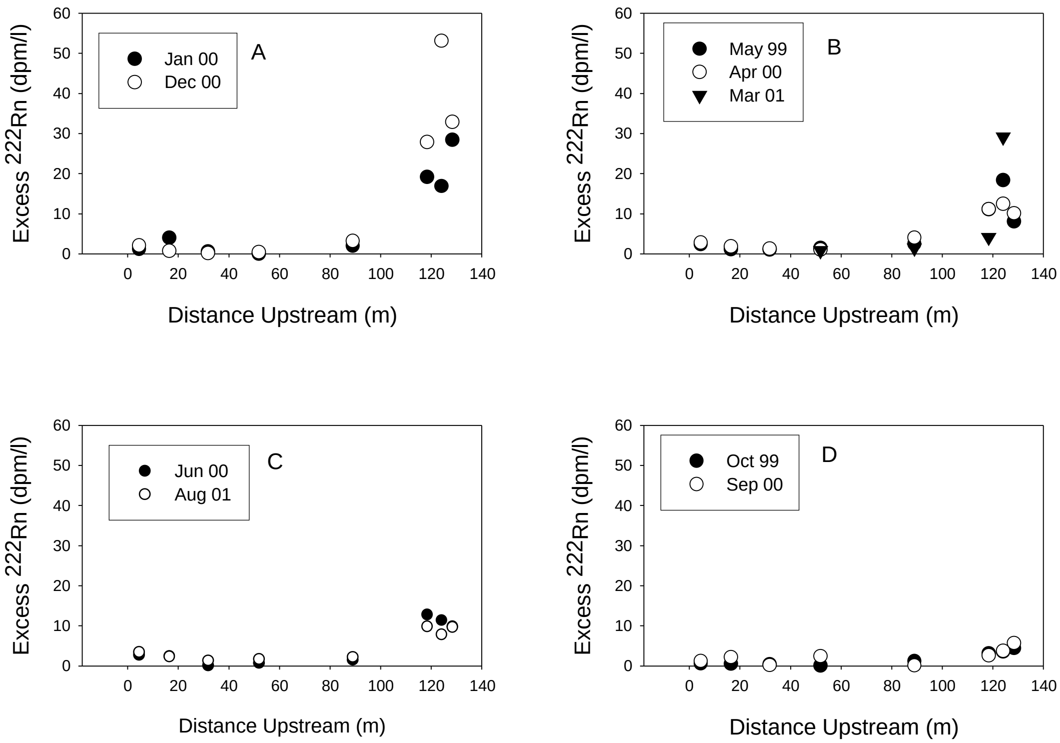

Figure 4.5 Seasonal activities of excess radon from a lateral profile along Barataria Waterway, Louisiana for A) Winter; B) Spring; C) Summer; and D) Fall.

at these locations. All of the above sediment processes may be contributing to the radium salinity curves observed, since the fine-grained clays in the Basin have high ion exchange capacities.

The ^{222}Rn diffusive flux for the eight stations ranged from 356 dpm/m^2/day at the ocean endmember station to 2023 dpm/m^2/day at intermediate station #27 (Table 4-2). When compared with the radon water column inventory, measured fluxes were not high enough to support the observed inventory except in winter at intermediate stations 14 and 19. These supported inventories occurred during months of low river stage and correspondingly low water levels at those stations. Figure 4-6 shows this relationship for a representative sampling trip in October 1999. The inventory at the three upstream stations was only supported minimally by diffusion during the two fall trips (October 1999 and September 2000). Calculated porosities for the transect ranged from 0.64 downstream to 0.92 upstream, reflecting the range quoted by Berner (1980) for initial porosity of fine grained clays at the time of sedimentation.

Throughout the study period, excess ^{222}Rn values varied inversely with wind speed (Figure 4-7), a phenomenon modeled and measured empirically by many groups (Wanninkof et al., 1990; Torgersen et al., 1982; Broecker and Peng, 1982). For individual sampling trips, wind speed did not vary by more than 1 m/s, but the range in radon activities was 0 to 28 dpm/l. The lower part of the Basin was observed to have more boat traffic than upstream stations, which might indicate greater vertical mixing. Mixing in the water column is also enhanced in the lower basin as a result of more open water, thus greater fetch, as well as proximity to the Gulf of Mexico. However, this was not corroborated by values of total suspended solids, which were high for almost all of the trips at the upstream stations (Lee, pers. comm.). During most of the study, Louisiana experienced below normal precipitation. When monthly precipitation was plotted against radon and radium levels, no obvious relationship was observed. Water levels along the transect fluctuated over the study period, with stations 3, 8, and 14 (downstream stations) showing the largest variations but no relationship appeared to exist between those water depth fluctuations and Basin precipitation or river stage.

One additional source which may be the real driving force behind ^{226}Ra and ^{222}Rn inventories along the entire transect is the advective movement of groundwater with high ^{226}Ra and ^{230}Th activities. So far, all the processes described here may account for most of the tracer activities in the intermediate Basin under specific conditions. In the downstream and upstream Basin however, this is not the case. For these regions of the Basin, this is the only process which can occur on a scale large enough to account for the unsupported ^{222}Rn. Groundwater wells measured within the basin at Jean Lafitte National Park showed an excess ^{222}Rn activity range of 0.06 to 100.73 dpm/l. Clearly, subsurface waters in the Basin are elevated with respect to radium and radon, indicating they may be useful for understanding subsurface hydrology in this system. This radon- and radium-rich water may flow from underlying aquifers into surface waters as diffuse seepage over a wide area, or at discrete points where a break in a confining unit may occur. Additionally, recirculating surface waters pumped into and out of bottom sediments can contribute to water column inventories. Since groundwater inputs and recirculated seawater inputs cannot be distinguished by radon and radium, total subsurface fluid inputs are considered together here. High sediment porosities such as were found in this study area may indicate diffuse seepage is the more likely process moving groundwater into the upper and lower Basin.

Atmospheric evasion was the primary sink for ^{222}Rn throughout the sampling period, especially at the downstream stations near the Gulf of Mexico where the fetch is greater. The highest excess ^{222}Rn activities were found in the winter during periods of low wind speed and

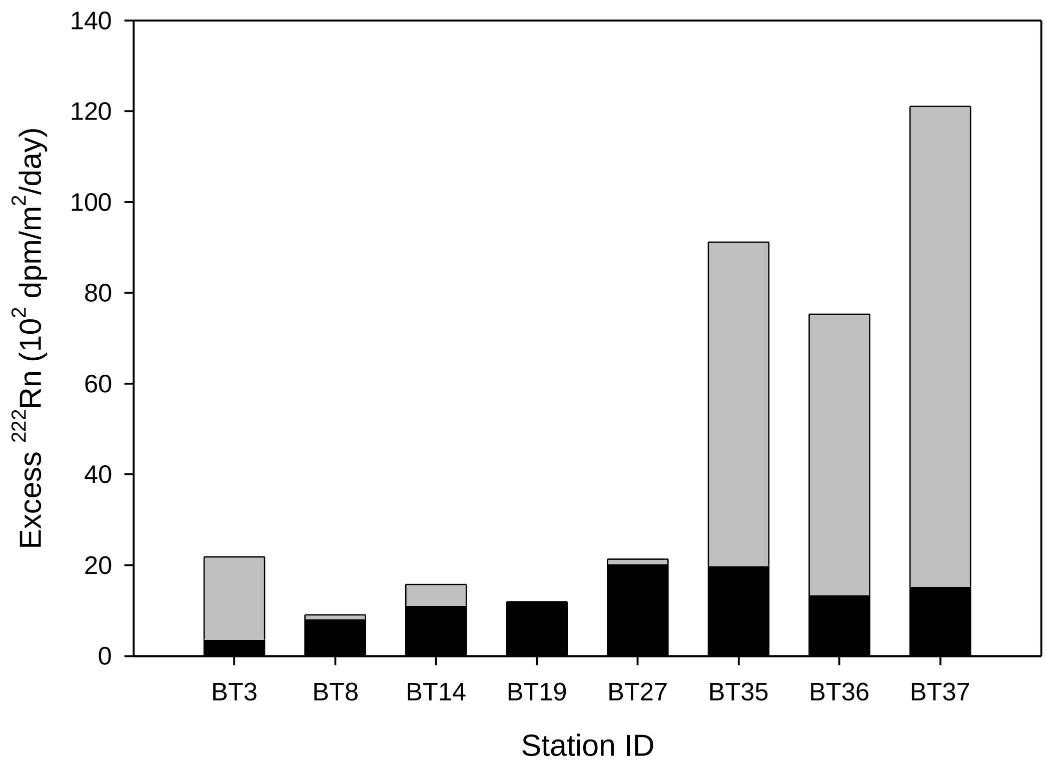

Figure 4.6 Typical relationship between flux based on ^{222}Rn water column inventory (total bar height) and diffusive flux from underlying sediments (black bar) for October 1999. (Gray portion of bar represents water column ^{222}Rn not supported by diffusion)

Table 4.2 Comparison of ^{226}Ra and ^{222}Rn activities in Barataria Basin with other estuaries.

Site Name	Activity (dpm/l)	Reference
Radium-226		
Suwannee River Estuary, FL	0.22	Burnett et al., 1990
Ganges-Brahmaputra River, India	0.12-1.14	Moore, 1997
North Inlet Marsh, SC	0.15	Rama and Moore, 1996
Chesapeake Bay, VA	0.03-0.20	Moore, 1980
Bly Creek, North Inlet, SC	0.14-0.39	Bollinger and Moore, 1993
Charlotte Harbor, FL	0.5-5.5	Miller et al., 1990
Barataria Basin, LA	0.25-2.66	This study
Radon 222		
Lot River, France	3300	Bertin and Bourg, 1994
Charlotte Harbor, FL	29-35	Miller et al., 1990
Florida Bay, FL	5.2-8.6	Burnett et al., 2000
Apalachicola Bay, FL	4	Fanning et al., 1982
Peace River, FL	91-120	Fanning et al., 1982
Little Manatee River, FL	91	Fanning et al., 1982
Barataria Basin, LA	6.3	This study

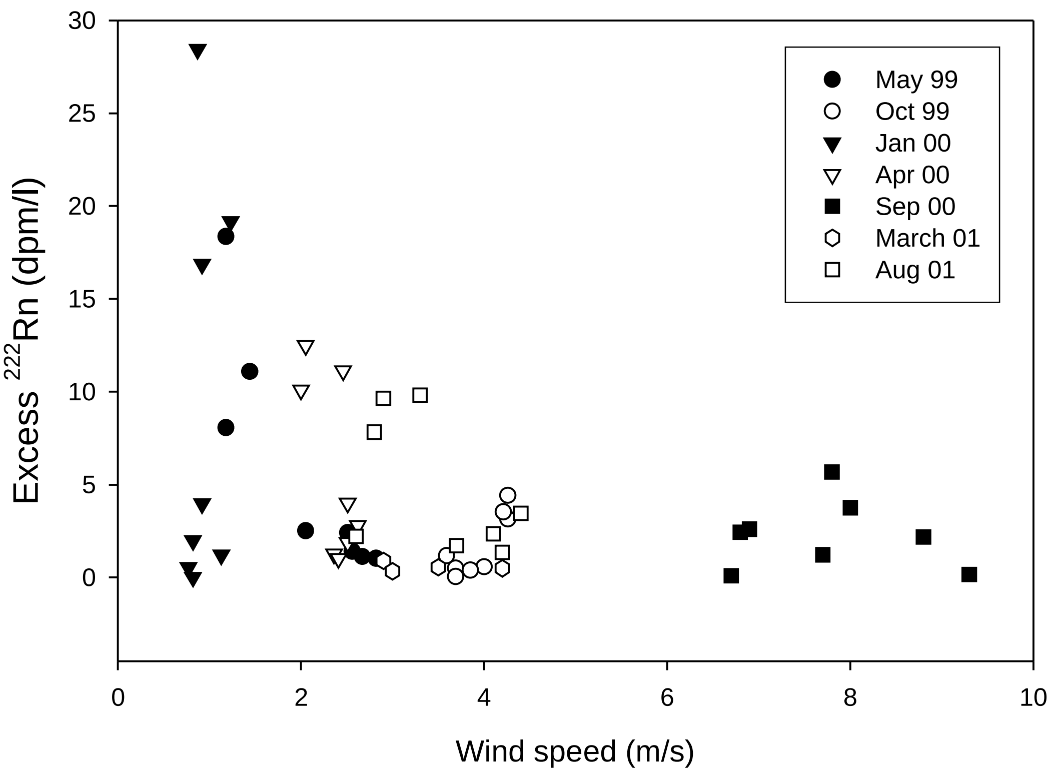

Figure 4.7 Relationship between water column excess ^{222}Rn and wind speed within Barataria Basin during the entire study period.

greatest solubility of radon. During the rest of the year, observed ^{222}Rn water column inventories represented conservative estimates due to atmospheric evasion. Air-sea exchange decreases the inventory and gives the appearance of lower potential input from other sources. It is apparent that wind mixing affects ^{222}Rn (see Figure 4-7) and the potential contribution of radon by groundwater is likely greater than one can observe.

4.6 Summary and Conclusions

Geochemical cycling of ^{226}Ra and ^{222}Rn in Barataria Basin demonstrated an excess presence of the tracers when all sources and sinks were accounted for. In the salinity mixing zone, ^{226}Ra activities are elevated above the mixing line as a result of sediment processes adding to the source terms. Those processes were decay, desorption from particles, and diffusion from bottom sediments. In this intermediate zone, sediment processes could account for all recorded activity in the winter when water levels are low. Highest recorded excess ^{222}Rn activities were in the winter and spring seasons. Low ^{222}Rn values in the summer/fall were likely the result of greater atmospheric evasion, whereas in the spring, water levels in the Basin increased, perhaps bringing new radium-rich waters into the Basin. Based on measurement of tracer sources and sinks in the Basin, an excess geochemical budget exists. The only remaining source that cannot be accounted for in this study is subsurface fluid inputs, but further investigation of sources is warranted.

Chapter 5

Assessment of Implications for Groundwater/Surface Water Exchange within a Deltaic Wetland Using ^{226}Ra and ^{222}Rn

5.1 Introduction

5.1.1 Background – Submarine groundwater discharge (SGD) may be a significant source of dissolved constituents to coastal watersheds. Yet for many years this component of water and geochemical budgets has been largely ignored, mainly because few experimental techniques were available to quantify it. Today, as new groundwater discharge models are developed, tested empirically, and replicated in different ecosystems, patterns of subsurface hydrology are emerging, which allow researchers to identify sources, sinks, and rates of groundwater movement associated with a specific set of geophysical conditions.

Coastal ecosystems in South Louisiana represent a unique and complex set of challenges to the study of its subsurface hydrology, and so far, no recorded attempts have been made to understand or estimate the significance of submarine groundwater discharge in this region. This lack of research may be due to the complication of aquifer heterogeneities and low permeability of deltaic sediments. Research has so far been limited to surface water studies of nutrients, pollutants, and other dissolved constituents within the estuary. As a result, the spatial and temporal variation in solute concentrations, along with their surface sources and sinks are well understood. However, the entire solute transport cycle may not yet be described in this complicated delta system. SGD has proven to be a very significant vehicle for nutrient inputs in similar estuarine systems (Tobias et al., 2001; Lapointe et al., 1990). In several cases, SGD may be as significant a solute source as the rivers associated with an estuary (Moore, 1996).

Groundwater itself may include fluids and constituents from several sources such as meteoric water, wastewater, and recirculated seawater (Burnett et al., 2000). Wastewater may be the least important of these sources within Barataria Basin, since little residential or commercial development has occurred within this wetland. Recirculated seawater, however, is often a potentially large part of quantified groundwater fluxes (Giblin and Gaines, 1990; Simmons, 1992). Hydraulic gradients can increase from the Gulf of Mexico landward, such that saltwater intrusion to the aquifer occurs, increasing aquifer salinities. The resultant brackish subsurface fluid discharges into surface waters via seepage and diffusion.

In many areas, the volume of groundwater is generally considered to be less important than the dissolved solutes it contributes to the overlying water column. Nutrients and other dissolved contaminants percolate down from upland areas and diffuse through subsurface fluids. If solutes are reactive, the chemistry may be altered before groundwater discharges to surface waters (Moore, 1999). Depending on the solute concentration levels in groundwater, contributions to coastal eutrophication may be significant (Libes, 1992).

5.1.2 Indicators of Groundwater - In this study estimates of groundwater inputs to surface waters were obtained using naturally occurring radioactive tracers, ^{222}Rn and ^{226}Ra. For the mass balance of radon, total radon is measured first, a measure of all the ^{222}Rn present in the water sample. Supported ^{222}Rn is that part of the total produced in situ as a result of radioactive decay of ^{226}Ra, the parent of ^{222}Rn. The balance of total radon is considered as excess ^{222}Rn produced elsewhere, decays independent of its parent, and transported into the sampling site by processes such as seepage. The amount of excess ^{222}Rn activity present is quantitatively related to SGD (Cable et al., 1996a). Activity, and hence SGD might show temporal and spatial variations according to salinity, sediment type, and aquifer properties.

5.1.3 Previous Use of Techniques – Martens et al. (1980) developed a mathematical approach to the calculation of diffusive fluxes of ^{222}Rn. This method is independent of sediment depth and allows the maximum possible diffusive flux based on a mixed, turbid environment. The researchers used the secular equilibrium activity of ^{222}Rn and ^{226}Ra, sediment porosity and bottom water concentrations at a given site to calculate diffusion. This method has been adopted in varying environments including river-dominated estuaries such as Barataria Basin, and is utilized in this study as well (Gruebel and Martens, 1984; Berelson et al., 1990; Corbett et al., 1999).

Benthic flux experiments give estimates of the total (advective plus diffusive) flux of SGD (Martens et al., 1980). Radium-226 is particle reactive and is associated with bed sediments. As this parent decays to radon-222, the new product diffuses into the surrounding pore water, increasing concentrations well above that supported by dissolved radium in the overlying water column. The resulting concentration gradient facilitates vertical diffusion, resulting in $^{226}Ra/^{222}Rn$ disequilibrium within the upper sediments. The extent of this deficit is controlled by molecular diffusion (Demas et al., unpublished). Benthic fluxes have been successfully applied in marine embayments such as Florida Bay and Apalachicola Bay (Martens et al., 1980; Cable et al., 1996b; Corbett, et al., 1999) and other river systems like the Potomac River estuary (Callender and Hammond, 1982).

Relationships between gas transfer velocities (piston velocity) and wind speed have been modeled, and some field-testing has confirmed general predictions of the models, but with high uncertainty (Wanninkof, 1992). Increases of wind speed and water temperature produce a corresponding increase in gas transfer velocities (Broecker and Peng, 1974). These models show an exponential relationship between wind speed and water column concentration of ^{222}Rn (4-5). Rates of gas transfer also have implications for the residence time of volatile pollutants within shallow water systems (Wanninkof et al., 1985). Piston velocities measured from ^{222}Rn profiles over the ocean ranged from 1.4 to 6.9 m/d (Smethie et al., 1985). The magnitude and direction of gas transfer is controlled by the concentration gradient existing between the water and overlying air for a specific gas (Macintyre et al., 1995). For ^{222}Rn gas, atmospheric concentration is low, measured at 0.22 to 0.89 dpm/l (Gesell, 1983). This low concentration in air creates a large concentration gradient across the surface of water bodies with high ^{222}Rn water column inventories. However, ambient wind speeds must be known in order to predict the resulting flux. In shallow water environments such as JELA where the entire water column is mixed, we would expect high losses of ^{222}Rn to the atmosphere.

Even though ^{222}Rn and ^{226}Ra are ideal for measuring groundwater due to differences in activity in surface water relative to groundwater, other sources of these tracers exist and must be accounted for in any mass balance calculation (Burnett et al., 2000). A mass balance model is used to assess the possible sources and sinks (Figure 5-1). Loss terms (atmospheric evasion) are deducted from source terms (advection, diffusion, in situ production) to arrive at a groundwater flux for the study area. Results of a ^{222}Rn mass balance from Florida Bay estimate advective rates of 0.3 to 2.0 cm/d and a total flux of 2 to 13 times greater than diffusion alone could supply to the water column (Burnett et al., 2000). Cable et al. (1996a) calculated a ^{222}Rn advective rate of 13 cm/d in the northeastern Gulf of Mexico (GOM), contributing greater concentrations to the water column than by diffusion. A rate of 2 to10 cm/d was estimated by mass balance for the inner continental shelf of the northeastern GOM as well, confirming that within the same system, differences exist in vertical flux rates (Cable et al., 1996b). In Par Pond, a freshwater lake in South Carolina, flow rates were measured at 0.15 to 0.64 cm/d (Burnett et al., 1996). Nutrient and ^{222}Rn mass balances have been performed in other river basins such as the Martha Brae River system in Jamaica and the Potomac River system in the U.S. mid-Atlantic coastal region.

Results consistently show significant inputs of SGD and solutes when compared with river inputs (Ellins et al., 1990; Callender and Hammond, 1982).

5.1.4 Aims and Objectives – The purpose of this research was to carry out an evaluation of SGD into a small area of Barataria Estuary using a geochemical mass balance approach, and to assess the implications for nutrient loading to the Basin via groundwater. Specific aims of this study are: (1) to determine whether observed water column inventories of natural tracers could be explained only by diffusive fluxes across the sediment-water interface; (2) to assess the loss of ^{222}Rn gas across the air-sea interface; and (3) to determine whether a subsurface hydraulic connection to the Mississippi River exists in this region of the estuary.

5.2 Site Description

5.2.1 Jean Lafitte National Park and Preserve - A geochemical mass balance research was performed at the Barataria Preserve of Jean Lafitte National Park (JELA). This wetland is located in a bend of the Mississippi River and, should a hydraulic connection exist, would be expected to reflect changes in the river by a chemical or physical signature. The river is east and north of the Preserve, with Lake Salvador to the southwest and Lake Cataouatche to the west (NPS, 2001). A section of the Gulf Intracoastal Waterway (GIWW) bounds the Preserve to the west (Figure 5-2).

The Preserve encompasses approximately 81 km^2 of hardwood forest, cypress swamp and fresh water marsh, and over 32 km of waterways (NOAA, 2001). This site was chosen because: (1) a river diversion is planned at Davis Pond, a site directly upstream; (2) a former Mississippi River distributary bed is adjacent; and (3) research was facilitated by personnel working within the Preserve.

5.2.2 Sampling sites – Sites for groundwater wells were placed along two transects, one running northeast to southwest and the other east to west to take advantage of the full range of sediment types and geologic regions within the study area. For most of the sampling trips, standing water occurred at all well sites except well #1 in the natural levee area. Creek samples were taken from Pipeline Canal and Kenta Canal adjacent to the well sites. Other bottom water sampling sites included Lake Salvador and the GIWW at Jones Point. The sites chosen for benthic fluxes were Kenta Canal dock approximately 100 m upstream from the confluence with the GIWW, and on the eastern end of Twin Canals.

5.3 Methods

5.3.1 Water Column Sampling – The following regions were sampled for ^{222}Rn and ^{226}Ra activity: (1) bottom water from surface water bodies, such as creeks, lakes, larger waterways, and the Mississippi River; (2) total benthic flux (advection and diffusion) was measured at two sites; (3) diffusive fluxes were measured independently at six sites near the wells; and (4) groundwater wells were sampled. Following analyses, a balance of possible inputs and outputs was constructed so that an advective flux could be calculated for JELA (Figure 5.1). Sampling locations and additional data are found in Appendix B.

Surface samples were collected for tracer analysis at seven stations along Pipeline South, Kenta Canal, GIWW, and Lake Salvador. Sediment grab samples were collected at all surface water sites except Lake Salvador and GIWW. Collection and analytical methods for water and sediment samples are described in detail in Chapter 4. Three sites within the Mississippi River were sampled for water column radon – Davis Pond, Belle Chase Ferry Landing, and Luling Bridge. Only Luling Bridge was sampled for the entire study period. All surface water samples were collected 0.3 m above the sediments. Water column inventories and tracer diffusive fluxes were calculated for the geochemical budget (see Chapter 4, Equations 2 and 3).

5.3.2 Benthic Flux – Clear plexiglass chambers (volume = 59 L) were used to measure total flux from sediments. Each chamber covered an area of 0.21 m^2 and

was open at the bottom. Careful insertion into the sediment was required to ensure that no air was trapped within the volume of the chamber. Samples were collected at initial (t=0 hr), intermediate (t=8hrs), and final (t=24 hrs) times to gauge the change in concentration over time. Initial samples were taken from inside the box after stirring for a minute. When final samples were taken, chambers were removed. Salinities, pH, and temperature were also measured at each sample site. Benthic flux was calculated according to Martens et al., (1980) where the change in concentration of ^{222}Rn within the chamber over deployment time results from water column production and decay, as well as potential diffusive or advective influx from the sediments.

5.3.3 Well Installation and Sampling - Background levels of ^{222}Rn and ^{226}Ra in groundwater were determined for the Basin by establishing two transects of wells (Figure 5.2). Highly organized fine clays within this environment required the use of 2.5 cm diameter PVC wells with 30.5 cm screens at different depths. Wells were sampled in conjunction with the water column, and all tracer samples were analyzed using techniques described in Chapter 4.

5.3.4 Atmospheric Evasion – In shallow water environments such as JELA, loss of ^{222}Rn from the water surface is likely to be the driving force behind observed water column inventories (Torgerson et al., 1982). Air-sea fluxes were calculated for the observed wind speeds using the following equation:

$$J_{atm} = k\,(C_w - \alpha C_{atm}) \qquad (2)$$

where J_{atm} is the flux of ^{222}Rn gas to the atmosphere; k is the piston velocity (Wanninkof et al., 1990) calculated as (0.73 m/d); C_w and C_{atm} are radon concentrations in the water column and air respectively; and α is the temperature-dependent solubility coefficient of radon (Broecker and Peng, 1982). The atmospheric concentration of radon used in JELA calculations was an average value of 560 dpm/m^3 (Gesell, 1983).

5.3.5 Mass Balance Calculations – Fluxes based on water column inventories of ^{222}Rn (J_{inv}) were apportioned according to different sources and sinks: (1) molecular diffusive inputs across the sediment-water interface from sediment equilibration experiments ($J_{diffusion}$); (2) loss at the air-sea interface (J_{atm}); and (3) advection of subsurface fluid inputs ($J_{advection}$) calculated in dpm/m^2/d as:

$$J_{advection} = J_{inv} - J_{diffusion} + J_{atmosphere} \qquad (3)$$

If no seepage occurs, the result of this calculation would be zero, balancing all inflows and outflows. If recharge occurs, a negative advection rate would result. This calculated result for $J_{advection}$ was compared with measured total benthic flux (J_{ADF}) obtained using flux chambers.

5.3.6 Groundwater Discharge Calculations – The relative contribution of subsurface fluids to surface water bodies was calculated using the following equation from Ellins et al. (1990):

$$\frac{V_{gw}}{V_s} = \frac{R_s - R_b}{R_{gw} - R_b} \qquad (4)$$

where V_{gw} is the volume of groundwater input to the surface stream; V_s is the volume of fluid in the stream; $R_s - R_b$ is the radon concentration in the stream minus background radon (taken as the in situ radium decay for this study); and $R_{gw} - R_b$ is the radon concentration in groundwater minus background levels. Surface water volume was estimated by multiplying cross-sectional area by stream length for each of three streams within JELA. Equation 4 was solved for V_{gw} to estimate the percentage of surface water originating as subsurface inputs according to the equation:

$$V_{gw} = \frac{(R_s - R_b)}{(R_{gw} - R_b)} Vs \qquad (5)$$

The average seepage velocity was also calculated at Twin Canals for comparative purposes using the flux based on annual average inventory (J_{inv}), subtracting the measured diffusive flux ($J_{diffusion}$) and dividing by the groundwater well ^{222}Rn concentration (R_{gw}) according to the following equation:

$$\text{Seepage Velocity (m/d)} = \frac{J_{inv}\ (dpm/m^2/d) - J_{diffusion}\ (dpm/m^2/d)}{R_{gw}\ (dpm/m^3)}$$

5.4 Results

Spatial and temporal variations were observed in excess ^{222}Rn concentrations in groundwater (Table 5-1). The east - west well transect (W1-W2-W3) showed decreasing excess radon concentrations with increasing distance from the Mississippi River regardless of well depth. No significant difference was observed between results from shallow (0.9 m) and deep (2.7 m) wells at the same sites on the north – south transect (W6-W5-W4). Overall well averages suggested that in shallow wells, tracer concentrations were at least 76% lower than in the mid-depth (1.8 m) and deep wells.

The highest values of surface water radon were observed in the final summer of the study (Figure 5.3). Along the north - south transect, excess radon activities seemed to increase throughout the study period, but Twin Canals was the only site with excess radon values above 10 dpm/l. This increase was not consistent with levels of local precipitation or river stage values. Average surface water activity and inventory per sampling trip confirmed the general trend of increasing activities throughout the study period (Table 5-2). Endmember stations at the GIWW

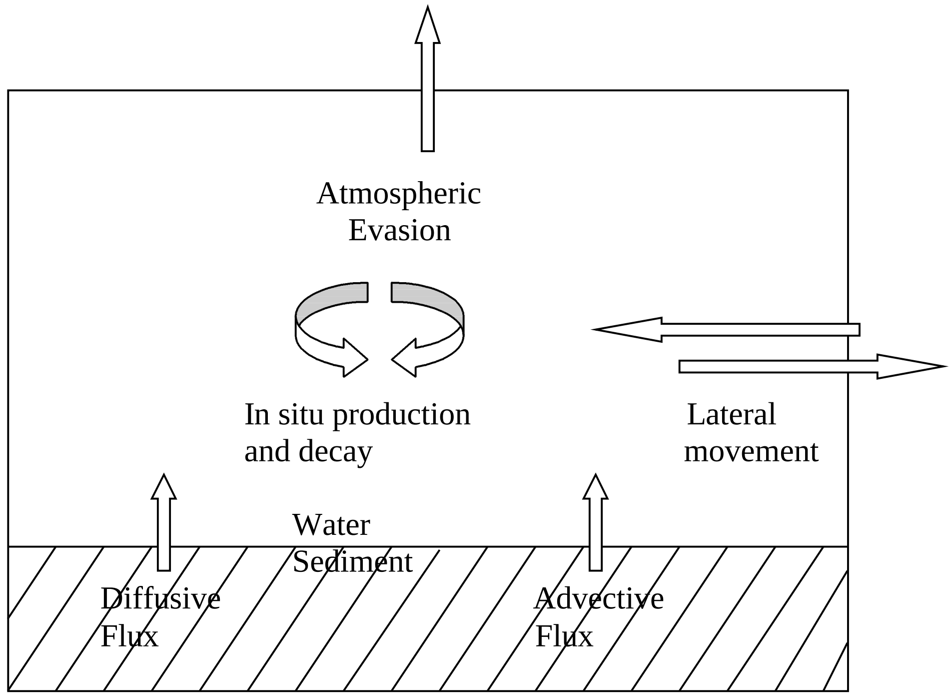

Figure 5.1 Mass balance of ^{222}Rn and ^{226}Ra showing typical sources and sinks.

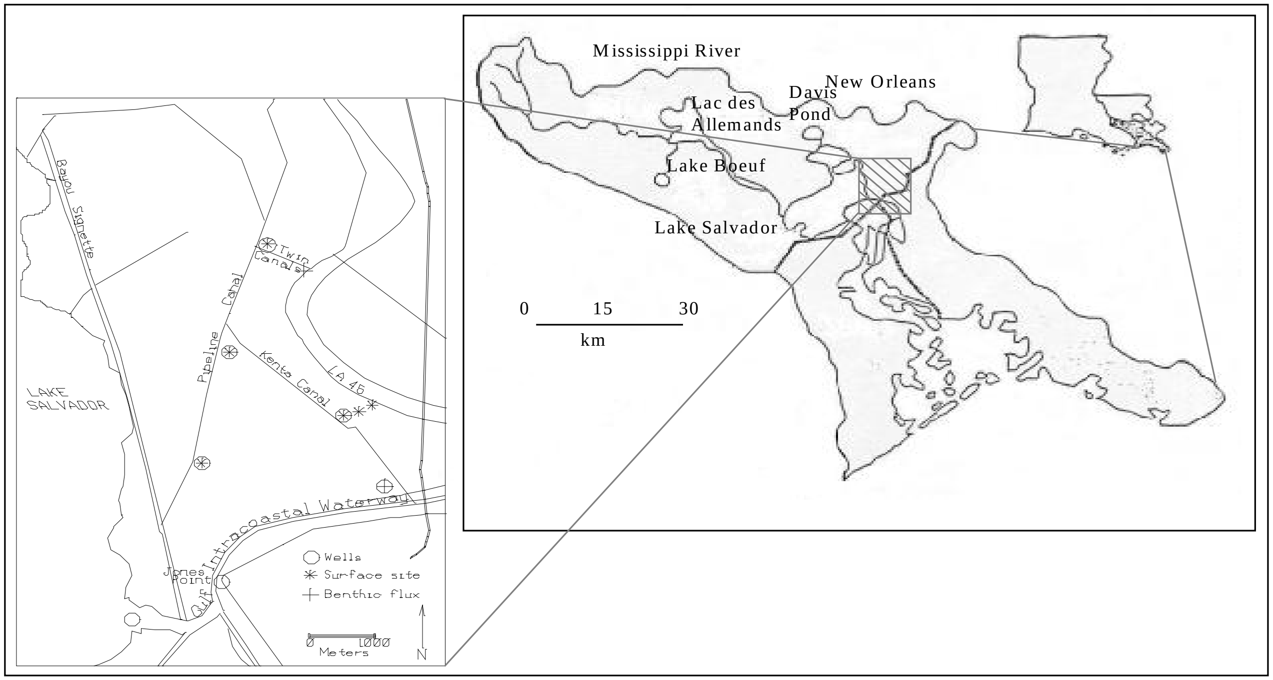

Figure 5.2 Site map of Jean Lafitte National Park showing sampling sites.

Jones point site and Lake Salvador remained low throughout the study period averaging 2 and 1 dpm/l respectively. Mississippi River values at Luling Bridge showed a single high value (6 dpm/l) during the first spring sampling trip and remained below 3 dpm/l for all subsequent trips. Typical values of ^{222}Rn in the ocean have been reported as 0.1-0.5 dpm/l (Cable et al., 1996a), while for this study, typical background radon activities within shallow water estuaries appear to be about 1 to 2 dpm/l.

All JELA groundwater samples exhibited elevated ^{222}Rn concentrations relative to surface waters by up to two orders of magnitude (Table 5.2). Reports of typical alluvial aquifer radon are scarce, but Bertin and Bourg (1994) reported concentrations of 3,300 dpm/l in an alluvial aquifer with significant river water/groundwater mixing. Since the range of values is significantly lower in this study area, little evidence exists that the MR alluvial aquifer is comparable in its subsurface hydraulic connection and mixing processes. However, four orders of magnitude differences were reported in other coastal areas where deep drinking water wells were tested (Corbett et al., 1999; Burnett et al., 1996). In this study, tracers were only measured in shallow water wells installed at the site.

Radium-226 activities within the study area also reflect higher values than previously reported. In the Gulf of Mexico, an annual average value of 0.1 dpm/l was reported, comparable to 0.15 dpm/l in a South Carolina salt marsh (Rama and Moore, 1996). In this study an average ^{226}Ra value of 1.10±0.1 dpm/l was observed during the study period. This high value is not surprising in a deltaic estuary with high clay content, where radium cation exchange capacity increases availability.

5.5 Discussion

Groundwater flow tends to be dispersed as the fluid passes through sediments, complicating the task of identifying specific sites of seepage. To date, however, it has been shown that what may seem to be a patchwork of small, individually inconsequential sites of groundwater upwelling or diffusion, may in fact contribute a significant volume of fluids and solutes to the surface ecosystem. So far, in this relatively small study area, at fewer than ten sampling sites, with comparatively small fluxes, a picture of integrated groundwater flux is emerging that might prove to be critical information for a water budget of the surface ecosystem. In addition, results of decreasing groundwater tracer concentrations along the east-west transect indicate a closer investigation of subsurface river effects within the estuary.

Average (45-year) wind speed for New Orleans was obtained from NOAA (1995) and was used to estimate the loss of ^{222}Rn to the atmosphere (Table 5.3). This average agrees well with the wind speeds observed during the period of study. Piston velocity, the rate at which ^{222}Rn atoms are lost at the air-sea interface, was calculated based on this wind speed and water temperature. Using the concentrations of ^{222}Rn in water and air, flux to the atmosphere at Twin Canals and Kenta Canal was found to be twice as high as the measured water column inventory (Figure 5.4). At Pipeline Canal, atmospheric evasion was three times higher than the measured inventory at that site, but it was approximately 40% lower than the other two sites.

The observed high air sea flux estimates were unsurprising within a shallow water system such as JELA. Calculated piston velocities were within the range for other Gulf of Mexico coastal areas (0.56 m/d). As radon was added to the water column at the sediment-water interface at the rate of 0.016-0.096 m/d, it was lost even more quickly at the air-sea interface (0.73 m/d), constantly creating a deficit in the water column inventory.

Table 5.1 Spatial comparison of groundwater ^{222}Rn as measured in wells.

Well Sites	Depth – 0.9m Excess Rn dpm/l	Depth – 1.8m Excess Rn dpm/l	Depth – 2.7m Excess Rn dpm/l	Average Excess Rn dpm/l
W1 - Terrestrial				
Mean	-	-	338 ± 7	339 ± 7
Range	-	-	0 to 1450	
W2 - Backswamp				
Mean	-	208 ± 2	-	208 ± 2
Range	-	1 to 864	-	
W3 – Marsh				
Mean	-	29 ± 5	-	29 ± 5
Range	-	2 to 53	-	
W4 – Marsh				
Mean	34 ± 1	-	37 ± 3	36 ± 2
Range	14 to 45	-	27 to 59	
W5 – Marsh				
Mean	26 ± 8	-	13 ± 1	20 ± 5
Range	4 to 48	-	4 to 19	
W6 - Marsh				
Mean	14 ± 1	-	32 ± 6	23 ± 4
Range	5 to 20	-	19 to 42	
Total Average =	25 ± 3	119 ± 3	105 ± 4	

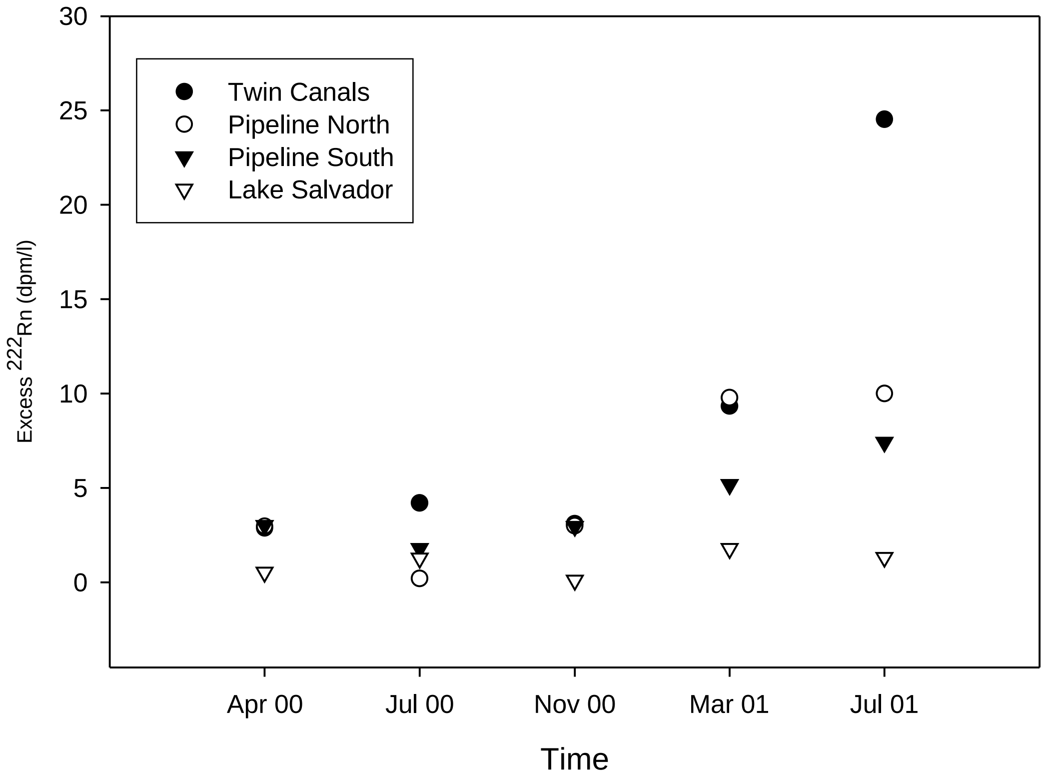

Figure 5.3 North-South surface water transect showing changes in excess ^{222}Rn over time.

Sediment equilibration results showed high diffusive fluxes at Twin Canals and Kenta Canal (Table 5.4). Measured diffusion at the Pipeline north and south sites were one order of magnitude less than at Kenta and Twin Canals, and for the balance of the calculations, the Pipeline north and south sites were averaged. No trend was observed when measured diffusive fluxes were compared with water column inventory at each site per trip. However, in mass balance estimates, averages were used and it was clear that at all sites, water column inventories were higher than could be accounted for by diffusion as measured via sediment equilibration (Table 5.5). At Twin Canals in the winter, disparities between inventories and diffusion were greater than in the summer. From these estimates of diffusion, advection appears to be important within this system as a source of fluids and dissolved constituents, especially at Twin Canals. The previous estimates of sources and sinks for ^{222}Rn were input to the mass balance equation (3) so that an estimate of total benthic (advective plus diffusive) flux could be calculated. Inventories were averaged according to Louisiana wet (June to September) and dry (October to May) seasons to assess any seasonal differences in benthic flux. Table 5.5 summarizes the results of the radon budget. Losses to the atmosphere were added back to the inventory to estimate pre-atmospheric evasion levels. The interdependence of the two source terms, advection and diffusion, excluded the measured diffusive flux from being used to calculate advection (Burnett et al., 1996). However, when comparisons were made at the three canal sites, the required advective flux was two to ten times greater than the measured diffusive flux. To put these values in perspective, in Florida Bay, total benthic flux was found to be 100 times greater than diffusive flux alone (Burnett et al., 2000).

A comparison of measured total benthic flux (J_{ADF}) to calculated benthic flux ($J_{benthic}$) showed that measured benthic flux at Kenta Canal was only 14% of the geochemical mass

Table 5.2 Comparison of ^{222}Rn in surface and well water samples by trip.

Sampling Trip	Surface Excess ^{222}Rn Activity dpm/l	Well Excess Rn Activity dpm/l	Flux (J_{inv}) based on ^{222}Rn inventory dpm/m^2/day
April 2000			
Mean	3.57 ± 0.17	-	698 ± 30
Range	0.53 to 8.55	-	
July 2000			
Mean	2.39 ± 0.36	18.85 ± 0.84	682 ± 65
Range	0.2 to 4.58	1.60 to 34/50	
November 2000			
Mean	2.55 ± 0.24	32.34 ± 0.81	1898 ± 120
Range	1.1 to 3.4	0.07 to 100.73	
March 2001			
Mean	6.22 ± 0.43	39.23 ± 7.76	2298 ± 77
Range	0.37 to 1.18	18.70 to 56.80	
July 2001			
Mean	8.40 ± 1.49	38.80 ± 2.92	2834 ± 270
Range	1.32 to 24.52	1.45 to 178.84	

Table 5.3 Air-Sea calculation for sites of the two benthic flux experiments in Jean Lafitte National Park. (Co and Catm represents water column and atmospheric concentrations respectively).

Parameters	Twin Canals	Kenta Canal	Pipeline Canal
Water Temp (^oC)	22.10	22.05	23.20
Wind Speed * (m/s)	3.67	3.67	3.67
Piston velocity** (m/d)	0.73	0.73	0.73
Co (dpm/l)	8.28 ± 0.51	8.81 ± 1.21	4.63 ± 0.59
Average Catm (dpm/l)	0.56	0.56	0.56
Air Sea Flux (J_{atm}) (dpm/m2/d)	6013 ± 370	6391 ± 879	3410 ± 435

* Average 45-year wind speed from New Orleans was used and equation for average K for long term average wind speeds.
** Normalized to CO^2 at 20^{o}C in freshwater (k, 600) (Macintyre et al, 1995)

balance estimate. At Twin Canals, measured results were similar when compared with calculated summer benthic flux, and 28% higher compared with the winter flux.

Table 5.6 shows stream characteristics, groundwater estimates, and seepage velocity for the three main sites. Twin Canals showed higher SGD and hence, a higher percentage of groundwater in its streamflow than did Pipeline Canal or Kenta Canal. Pipeline, the longest of the three canals, exhibited the lowest inventory, SGD, estimated benthic flux and measured diffusive flux, which may be an initial indication of limited groundwater/ surface water interaction in this region.

Water levels showed large variation in all three canals during the sampling period. Even though no seasonal trend was obvious, each individual canal reflected inventories that could sometimes be accounted for by diffusion alone, and other times, advection seemed to be more important. This switch may suggest additional processes occurring during some periods of the year when one set of hydrologic conditions exists versus another. For example, at Kenta Canal, periodic elevations in salinity suggested a backflow of higher salinity water from the nearby GIWW, a much deeper channel that may have its own subsurface tracer sources. However, initial results from this study do not show elevated tracer levels at the site measured in the GIWW. In addition, at Pipeline Canal, where a low current velocity of 0.22 m/s exists in the direction of the confluence with Bayou Segnette, backflow may have affected results at the south site.

5.6 Summary and Conclusions

This study represents a first approximation of the interactions within Barataria Basin. Clearly the system is as complex hydrologically as it is geologically. From this first approximation, advective flux appears to be important in the Jean Lafitte wetland system,

Table 5.4 Diffusive fluxes of ^{222}Rn from sediment at stations in Jean Lafitte National Park

Station ID	Wet Sediment Weight (g)	Porosity	Rn-222 Ceq (dpm/L)	Rn-222 Bottom water (dpm/L)	Diffusive Flux (J_{diff}) (dpm/m2/day)
Kenta (W3)	50.10	0.85	273± 7	6±0.2	1030 ± 26
Twin Canal	50.14	0.57	701± 13	3±0.2	2687 ± 48
Pipeline Nth	50.75	0.92	123± 5	3±0.2	463 ± 18
Pipeline Sth	50.40	0.93	105± 5	3±0.2	393 ± 18

The radon diffusion measured on sediment grab samples collected in April 2000. coefficient (D_m) is 5.76 x 10^{-6} cm^2/sec; average porosity (ϕ) is 0.82; average wet bulk sediment density (ρ_{wet}) is 1.34 g/cm^3. Fluxes range from 393 to 1030 dpm/m^2/day with a mean ($\pm 1\sigma$) of 1143±27 dpm/m^2/day and a coefficient of variation of 53%.

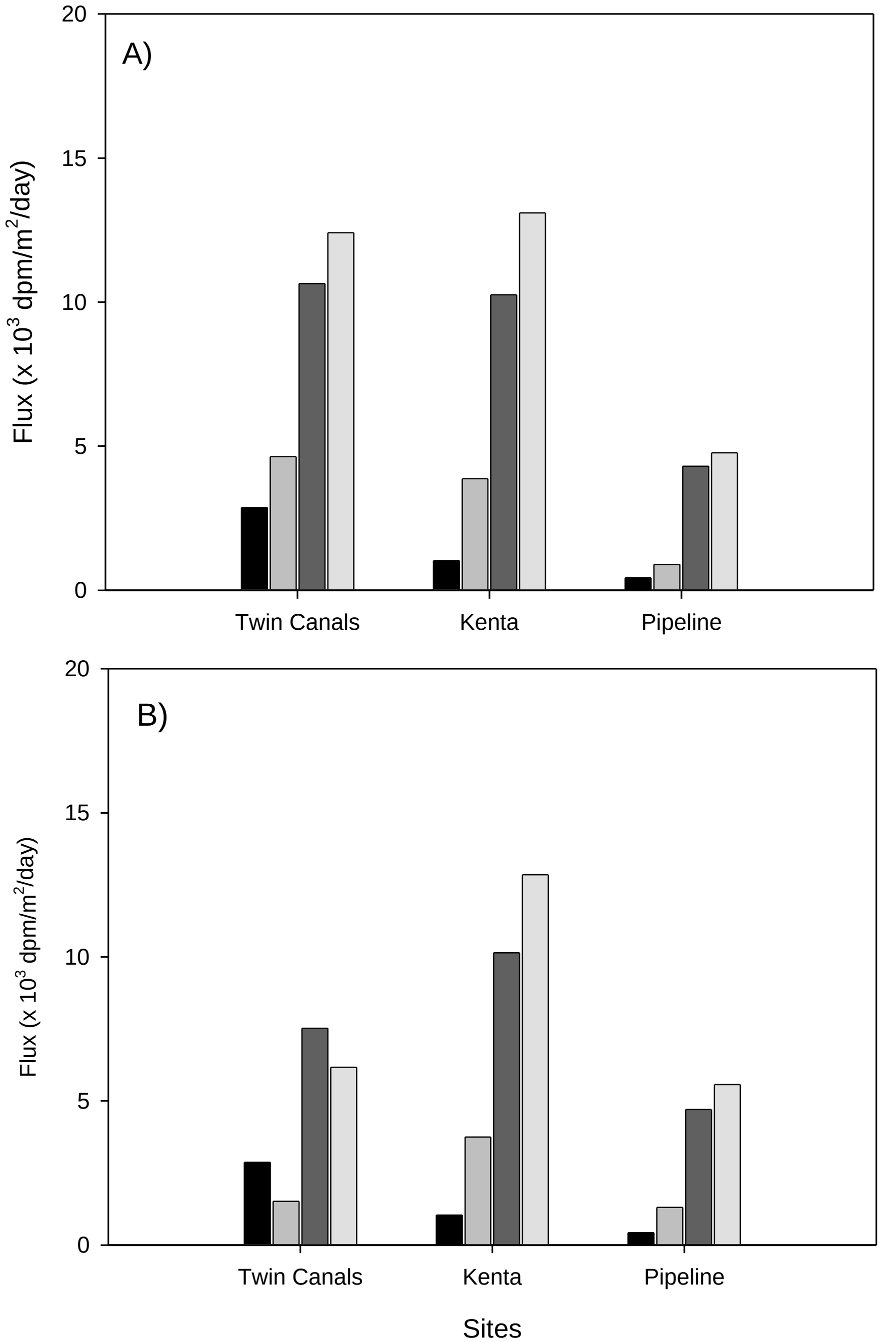

Figure 5.4 Relationship between flux based on inventory (light gray center bar), atmospheric evasion (dark gray bar), and diffusive flux (black bar) at three sample sites in winter (Graph A) and summer (Graph B). The gray bar on the right represents net flux.

Table 5.5 Mass balance calculations for canals in JELA and estimates for benthic flux.

Site Parameters	Twin Canals (dpm/m^2/day)	Kenta Canal (dpm/m^2/day)	Pipeline Canal (dpm/m^2/day)
Flux (J_{inven})based on Inventory*			
winter/dry	1508 ± 81	3743 ± 153	1295 ± 40
summer/wet	4633 ± 425	3867 ± 58	891 ± 206
Sinks			
Atm Evasion (J_{atm})	6013 ± 370	6391 ± 879	3410 ± 435
Sources			
Benthic Flux ($J_{benthic}$)**			
winter/dry	7521 ± 451	10134 ± 1032	4705 ± 475
summer/wet	10646 ± 795	10258 ± 937	4301 ± 641
Sediment samples			
Diffusive Flux ($J_{diffusion}$)	2687 ± 48	1030 ± 26	428 ± 18
Chamber Experiments			
Benthic Flux (J_{ADF})	10390 ± 5246	1400 ± 1747	

* In situ decay has already been accounted for by using excess radon values in the inventory calculation.
** Total benthic flux is the sum of flux based on inventory and air-sea flux (For comparison, measured diffusive and benthic fluxes are added at the bottom of the table.

Table 5.6 Estimates of groundwater outputs and seepage velocity for JELA streams.

Site	Stream Length (m)	Stream Volume Vs (x 1000 m^3)	Surface Excess Rn Rs (dpm/l)	GW Excess ^{222}Rn Rgw (dpm/l)	Percent GW in stream	Average Seepage Velocity (cm/d)	GW Influx Estimate (m^3/sec)
Twin Canals	966	31.67	8.3 ± 0.5	23 ± 4	40	1.6	0.08×10^{-2}
Pipeline Canal	5796	235.32	4.6 ± 0.6	28 ± 4	16	2.3	0.98×10^{-2}
Kenta Canal	4830	105.87	8.8 ± 1.2	29 ± 5	30	9.6	2.3×10^{-2}

especially at the Twin Canals site where high inventories and total benthic fluxes were observed. However, it is also important to note that seasonally and spatially, large variations in tracer concentrations were observed, both in surface water and groundwater. Contrasts between results at the three canal sites, six well sites, and seven surface water sites suggest that big differences exist even at small distances within this system. One additional rationale for the variation observed during this study may have been its relatively short duration. A system where little is known about the processes driving the surface/groundwater interactions may require intensive study for a longer time. Changes in river stage, precipitation, as well as an in depth look at stratigraphy, would be necessary as baseline information for assessing groundwater inputs within such a complex system.

Chapter 6

Summary and Conclusions

The potential usefulness of accurate information about hydrologic budgets, nutrient inputs, and fluid and sediment geochemistry become more obvious as more groundwater studies are performed within coastal wetlands. Hydrologic budgets are adjusted to reflect what is usually a significant loss or gain of fluids to the system. Nutrient loading, often underestimated in the absence of substantiated groundwater-derived nutrient estimates, is usually revised upward. SGD may affect the geochemistry within sediments and in the water column, and thereby influence the ecological character of estuaries. However, groundwater estimates from these new studies, while supplying much needed insights into groundwater/surface water exchange, have also raised management uncertainty regarding the application of such information within large non-uniform systems.

Aquifer heterogeneity has been generally accounted for in groundwater studies by measuring and correcting for permeability. However, surface ecosystem heterogeneity may still limit system-wide application of groundwater flux estimates in wetlands planning. Intensive study of many small sites within a single wetland allows managers to overcome this obstacle, exemplified by the case of Sippewissett Marshes in Massachusetts (Chapter 2). Even in the relatively small area that comprises these Marshes, researchers have found dramatic spatial and temporal differences in hydrologic parameters, and hence, controls on SGD. Sustained long-term investigations enabled them to provide the coastal wetlands managers with a comprehensive and useful body of SGD information. The level of groundwater research and collaboration between scientists and managers working in Sippewissett Marshes is not found in many other areas. Groundwater studies are still limited spatially, with more than 80% of studies performed in industrial nations. In addition, the results of Chapter 2 suggest non-uniformity in ecosystems does not facilitate extrapolation of groundwater estimates to regions where no subsurface fluid studies have been performed. Precipitation and tides were found to influence the magnitude and direction of groundwater flow, but predictions could not be made regarding SGD by examining the dominant sediment type.

A review of groundwater-derived nutrient studies indicates a limitation in scope, where groundwater is implicated as a possible conduit for dissolved and suspended constituents. Supporting site investigations into the quantity and direction of flow are seldom available. However, the nutrient and SGD studies reviewed confirm that:

1) Values of SGD may range from zero to several orders of magnitude both temporally and spatially;
2) Precipitation and tidal regimes influence SGD flux; and
3) A more intensive study of the mixture of sediment types at each site is required to establish that substrate is a controlling factor in determining SGD levels.

The water budget is a modeling tool that might assist managers in assessing the significance of groundwater in large areas where empirical groundwater data are not yet available. This technique was used to estimate annual and long-term net groundwater flux within U.S. coastal watersheds, as well as on a regional scale. An evaluation of the 29 local watershed annual budgets showed that approximately 90% were dominated by groundwater import. By contrast, 73% of the long-term water budgets showed zero net flux, suggesting that over decadal periods groundwater import and export processes balance. Of the seven watersheds showing a long-term flux, five demonstrated a further net import after annual groundwater import was accounted for. The long-term regional averages for the northeast suggest a small, likely insignificant groundwater export, whereas the eastern Gulf region showed an equally small groundwater import. All other regional averages for long-term flux were zero.

Results of this water budget study were in contrast to many empirical studies within the same watersheds. It was clear that, while high cumulative errors for groundwater values were observed, a clear disparity existed between calculated (showing net import) and measured (showing net export) values. Previous studies indicate that some percentage of this variation was due to seawater recycling. When SGD was measured in the field all subsurface fluid inputs were presumed to be groundwater. Thus, seawater circulation within the sediments artificially increases the observed magnitude of groundwater flux. This conclusion may further impact the usefulness of such observed results by coastal managers who need an estimate of freshened inputs only.

An evaluation of the radioactive tracer technique in Barataria Basin (Chapter 4) supports previous evidence showing that tracers are useful tools for measuring SGD in shallow water systems. Radium-226 exhibits a distinct signature across a salinity gradient, and desorbs from sediments in brackish water, thereby increasing concentrations above a conservative mixing line. Other sediment processes which add to the source terms for ^{226}Ra are decay of ^{230}Th and diffusion from bottom sediments. Temporal differences occurred in excess ^{226}Ra. In the winter sediment processes could account for all excess ^{226}Ra, whereas in the summer, a larger concentration of ^{226}Ra was present than the maximum produced from all sediment processes.

Along the same gradient, ^{222}Rn increases in the upper reaches of the basin where salinity is low. Major losses of ^{222}Rn occurred due to atmospheric evasion, particularly at the downstream stations with greater areas of open water. An evaluation of all ^{222}Rn sources and sinks yielded an excess of ^{222}Rn activity in the upper Basin, suggesting a groundwater source. This type of information is critical for coastal wetland planners in Louisiana, where efforts to reduce wetland loss have focused on hydrology. As plans for Mississippi River diversions are being developed, planners require information on subsurface fluid flows, a previously unknown parameter for this estuary.

Further spatial resolution of SGD flux within Barataria tested at Jean Lafitte Park using tracers showed that even in such a small area advection is significant only at certain sites. At Kenta Canal, where surface water is very shallow, atmospheric evasion was highest, accounting for the greatest loss of ^{222}Rn. The average seepage velocity was almost 90% higher at Kenta Canal (9.6 cm/d) than at Twin Canals (1.6 cm/d), producing the highest estimate of groundwater influx (2.3×10^{-2} m^3/s). Pipeline Canal demonstrated the lowest atmospheric evasion and excess ^{222}Rn in surface waters. Also, a relatively low seepage velocity (2.3 m^3/s), coupled with an estimated groundwater influx that is an order of magnitude lower than Kenta Canal, might account for the low water column inventory observed at Pipeline Canal. A comparison of the Barataria Basin groundwater results, which were based on a water budget calculation (0.26 cm/d) and measured fluxes in Jean Lafitte based on geochemistry (1.6-9.6 cm/d), support results of other studies noting that the water budget technique tends to give the lowest SGD estimates.

Suggestions for future research in Barataria Basin are:

- A detailed evaluation of results of sediment coring projects in Barataria to better understand stratigraphy and contour permeability;
- A large scale salinity study identifying areas of anomalously high salinity, and assessing the impact of large conduits like the GIWW to surface salinities within the wetland;
- A ^{222}Rn well sampling project where private and commercial wells are sampled for groundwater ^{222}Rn background levels, improving the efficacy of measured fluxes; and
- Measurements of SGD volume and direction in areas of Barataria Basin where SGD was implicated but not measured.
- Coastal eutrophication, another issue for wetland planners, has been detailed through nutrient research. To augment this current knowledge a study of SGD-derived nutrients is needed.

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Appendix A

Hydrologic Information for Change in Storage Calculation

Table A-1: Groundwater well data for Northeast Watersheds as found in the USGS website (http://water.usgs.gov/usa/nwis/gw) for use in calculating annual change in storage. Land elevation is reported for each well as feet above mean sea level for NGVD29.

Casco Bay, Maine

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS (m)	Std Dev	S DS m/sec
Oxford	444637070552301	1994	11.41	10.63	0.14	0.03		1.05471E-09
Oxford2	440823070291501	1994	9.62	7.02	0.39	0.31		9.79375E-09
Cumberland	435453070013601	1994	30.66	27.78	0.29	0.25		8.0668E-09
York	434822070482501	1960-1982	16.1	14.84	0.39	0.15		4.7462E-09
Franklin	443831070002601	1994	17.63	14.03	0.39	0.43		1.35606E-08
					Watershed DS	0.23	0.15	7.44441E-09
							(n = 3; %cv = 53)	4.78076E-09
						% error	64.22	

Great Bay, New Hampshire

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS (m)	Std Dev	S DS m/sec
Rockingham	430527071140101	1995	39.89	38.55	0.25	0.10		3.23561E-09
Strafford	430721071005001	1995	31.92	30.51	0.25	0.11		3.40463E-09
Strafford2	432534071095601	1995	20.27	18.29	0.25	0.15		4.78097E-09
Belknap	431916071125901	1995	3.33	3.15	0.25	0.01		4.34634E-10
Grafton	434952071390901	1995	14.23	10.82	0.25	0.26		8.2339E-09
					Watershed DS	0.13	0.09	4.01795E-09
							(n = 3; %cv = 53)	2.83658E-09
						% error	70.598	

Table A-1 continued

Merrimack River, Massachusetts

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std dev	S DS m/sec
Hillsborough	424800071295301	1995	29.90	27.41	0.25	0.19		6.01243E-09
Hillsborough2	425024071413001	1995	9.02	6.05	0.25	0.23		7.17146E-09
Merrimack	430235071275501	1995	50.36	47.16	0.18	0.18		5.56331E-09
Merrimack2	432428071390701	1995	15.67	13.27	0.25	0.18		5.79512E-09
Essex	423505070491702	1995	4.65	1.90	0.25	0.21		6.64024E-09
Middlesex	424055071435301	1995	14.47	12.37	0.39	0.25		7.91034E-09
					Watershed DS	0.21	0.03	6.51548E-09
							(n = 6; %cv = 53)	9.0223E-10
						% error	13.847	

Massachusetts Bay, Massachusetts

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Barnstable	420313070061901	1995	56.1	55.2	0.29	0.08		2.5209E-09
Barnstable2	414517070393102	1995	49.39	47.99	0.29	0.12		3.9214E-09
Middlesex	424055071435301	1995	14.47	12.37	0.39	0.25		7.9103E-09
Middlesex2	421852071220501	1995	18.1	15.39	0.25	0.21		6.5437E-09
Middlesex3	422627071154002	1995	4.05	1.76	0.39	0.27		8.626E-09
Suffolk	422133071033801	1995	32.4	30.08	0.19	0.13		4.2575E-09
Plymouth	415217070393102	1995	23.31	22.8	0.19	0.03		9.3591E-10
					Watershed DS	0.16	0.09	4.9594E-09
							(n = 7; %cv = 53)	2.8392E-09
						% error	57.25	

Table A-1 continued

Buzzards Bay, MA

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Barnstable	413522070373601	1995	23.13	22.31	0.39	0.10		3.0888E-09
Barnstable2	413525070291904	1995	9.9	8.85	0.29	0.09		2.941E-09
Bristol	415447071155301	1995	4.96	3.46	0.39	0.18		5.6502E-09
Norfolk	420432071151201	1995	20.92	18.44	0.39	0.29		9.3417E-09
Plymouth	414518070435701	1995	10.61	7.71	0.39	0.34		1.0924E-08
Plymouth2	415228070554601	1995	20.56	13.57	0.25	0.53		1.6878E-08
					Watershed DS	0.26	0.17	8.1373E-09
							(n =6; %cv = 53)	5.372E-09
						% error	66.02	

Narragansett Bay, RI

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Bristol	414705071045301	1995	15.51	14.15	0.39	0.16		5.1229E-09
Providence	415317071220601	1995	5.31	4.14	0.19	0.07		2.1471E-09
Providence2	414420071422301	1995	11.4	3.65	0.19	0.45		1.4222E-08
Kent	413645071332901	1995	6.82	3.8	0.14	0.13		4.0836E-09
Washington	413400071363101	1995	13.43	11.72	0.14	0.07		2.3123E-09
					Watershed DS	0.18	0.16	5.5776E-09
							(n = 5; %cv = 53)	4.99E-09
						% error	89.46	

Table A-1 continued

Long Island Sound, CT

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS (m)		S DS m/sec
Fairfield	411256073153101	1995	8.75	7.51	0.39	0.15		4.67086E-09
Newhaven	413245072584201	1995	21.44	13.15	0.39	0.99		3.1227E-08
New London	412013072030601	1995	16.94	13.7	0.39	0.39		1.22045E-08
Litchfield	420125073193001	1995	11.51	8.59	0.39	0.35		1.09991E-08
Tolland	414548072114501	1995	14.3	10.28	0.39	0.48		1.51426E-08
Windham	414054071552001	1995	31.55	29.44	0.39	0.25		7.948E-09
Hampshire	421355072322001	1995	10.57	6.12	0.39	0.53		1.67624E-08
Windsor	431551072350601	1995	5.93	3.42	0.29	0.22		7.03044E-09
Cheshire	425543072175801	1995	5.09	2.45	0.25	0.20		6.37463E-09
					Watershed DS	0.39	0.26	1.24844E-08
							(n = 9; %cv = 53)	8.12128E-09
						% error	65.05143	

Table A-2: Groundwater well data for Mid-Atlantic Watersheds as found in the USGS website (http://water.usgs.gov/usa/ nwis /gw) for use in calculating change in storage. Land elevation is reported for each well as feet above mean sea level for NGVD29.

Peconic/Gardiners Bays, Long Island, New York

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Suffolk1	410858072171501	1985	2.42	2.20	0.18	0.01		3.82478E-10
Suffolk2	410356072260301	1985	2.76	2.54	0.18	0.01		3.82478E-10
Suffolk3	405756072173501	1985	15.53	13.89	0.26	0.13		4.1184E-09
Suffolk4	405628072164701	1985	10.56	9.73	0.25	0.06		2.00414E-09
Suffolk5	405347072494001	1985	42.33	41.13	0.25	0.09		2.89756E-09
					Watershed DS	0.06	0.05	1.95701E-09
							(n = 7; %cv =53)	1.62148E-09
						% error	82.85	

Hudson River/Raritan Bay, NY/NJ Harbor

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Dutchess	414128073475201	1995	18.79	14.07	0.25	0.36		1.13971E-08
Hamilton	432832074122201	1995	13.16	7.65	0.25	0.42		1.33046E-08
Mercer	401552074501801	1995	20.07	16.45	0.25	0.28		8.74097E-09
Middlesex	402015074275702	1995	95.75	91.31	0.25	0.34		1.0721E-08
Montgomery	430141074423501	1995	7.68	5.32	0.25	0.18		5.69853E-09
Putnam	412450073413101	1995	13.91	7.61	0.25	0.48		1.52122E-08
Renselaer	423834073391001	1995	13.76	9.87	0.25	0.30		9.39292E-09
Saratoga	430013073370401	1995	11.44	9.44	0.25	0.15		4.82926E-09
Ulster	414425074213601	1995	23.93	22.56	0.25	0.10		3.30805E-09
Westchester	411421073481201	1995	15.66	10.87	0.25	0.36		1.15661E-08
					Watershed DS	0.30	0.12	9.41707E-09
							(n = 7; %cv =53)	3.82374E-09
						% error	40.60	

Table A-2 continued

Barnegatt Bay, New Jersey

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Ocean	394742074142001	1995	9.09	7.16	0.26	0.15		4.84665E-09
Ocean2	395323074225501	1995	10.71	9.83	0.25	0.07		2.12488E-09
Middlesex	402536074201801	1995	37.31	18.92	0.26	1.46		4.61813E-08
Middlesex2	402623074212701	1995	23.71	13.65	0.26	0.80		2.52628E-08
Middlesex3	403242074161701	1995	1.01	1.13	0.26	0.01		3.01346E-10
					Watershed DS	0.50	0.62	1.57434E-08
							(n = 7; %cv =53)	1.97468E-08
						% error	125.43	

New Jersey Inland Bays, Great South Bay, New Jersey

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Atlantic	391827074371001	1995	78.35	61.41	0.26	1.34		4.254E-08
Atlantic2	392754074270101	1995	72.10	61.83	0.26	0.81		2.57902E-08
Cape May	390058074524270	1995	35.66	28.15	0.26	0.60		1.88592E-08
Cape May2	390425074544601	1995	11.00	9.00	0.26	0.16		5.02244E-09
Gloucester	395232075094201	1995	57.78	51.90	0.26	0.47		1.4766E-08
					Watershed DS	0.68	0.44	2.13956E-08
							(n = 7; %cv =53)	1.40077E-08
						% error	65.47	

Table A-2 continued

Delaware Bay, Delaware

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Burlington	395122074301702	1997	20.92	18.53	0.26	0.19		6.00181E-09
Camden	395229074571201	1997	250.75	216.94	0.25	2.58		8.16387E-08
Cumberland	392732075092401	1997	6.04	4.25	0.26	0.14		4.49508E-09
Montgomery	401733075171401	1997	11.46	9.96	0.25	0.11		3.62195E-09
Salem	393348075275701	1997	34.58	33.25	0.25	0.10		3.21146E-09
					Watershed DS	0.62	1.09	1.97938E-08
							(n = 7; %cv =53)	3.45889E-08
						% error	174.746	

Delaware Inland Bays, Delaware

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Sussex	383138075260201	1997	11.25	6.42	0.25	0.37		1.16627E-08
Sussex2	384639075353101	1997	13.74	11.18	0.25	0.20		6.18146E-09
Sussex3	384955075192801	1997	13.22	8.80	0.25	0.34		1.06727E-08
Worcester	382022075072401	1997	5.60	2.35	0.25	0.25		7.84755E-09
Worcester2	382215075041801	1997	44.48	4.85	0.25	3.02		9.56919E-08
					Watershed DS	0.83	1.22	2.64112E-08
							(n = 7; %cv =53)	3.87907E-08
						% error	146.872	

Table A-2 continued

Maryland Inland Bays, Maryland

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Accomack	374324075443201	1969-2000	11.58	8.07	0.25	0.27		8.47536E-09
Northampton	371735075572601	1956-1995	25.30	22.68	0.25	0.20		6.32634E-09
Somerset	380616075380701	1949-2000	5.84	1.07	0.25	0.36		1.15178E-08
Somerset2	381156075412501	1952-2000	61.27	53.45	0.25	0.60		1.88824E-08
Somerset3	373059075484502	1979-2000	29.47	19.12	0.25	0.79		2.49914E-08
					Watershed DS	0.44	0.24	1.40387E-08
							(n = 7; %cv =53)	7.74936E-09
						% error	55.2001	

Chesapeake Bay

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Anne Arundel	385905076293601	1996	16.47	12.69	0.25	0.29		9.12731E-09
Caroline	385310075503601	1996	16.36	15.14	0.25	0.09		2.94585E-09
Caroline2	390333075504501	1996	2.99	2.24	0.25	0.06		1.81097E-09
Chemung	420829076484801	1996	25.16	19.78	0.25	0.41		1.29907E-08
Dauphin	402118076462201	1996	4.65	3.82	0.25	0.06		2.00414E-09
Frederick	393156077135701	1996	36.31	29.51	0.25	0.52		1.64195E-08
Garret	393749079190301	1996	14.78	5.79	0.25	0.69		2.17075E-08
Lancaster	400506076235201	1996	29.63	28.98	0.25	0.05		1.56951E-09
Loudoun	391542077423801	1996	58.86	57.3	0.25	0.12		3.76683E-09
Montgomery	390434076573002	1996	12.95	9.82	0.25	0.24		7.5578E-09
Prince Georges	384131076533301	1996	215.8	210.1	0.25	0.43		1.37634E-08
Talbot	384643076043801	1996	25.59	0.47	0.25	1.91		6.06556E-08
Washington	392904077371501	1996	41.05	17.98	0.25	1.76		5.57056E-08
					Watershed DS	0.51	0.62	1.61557E-08
							(n = 7; %cv =53)	1.97086E-08
						% error	121.991	

Table A-3: Groundwater well data for Southeast Watersheds as found in the USGS website (http://water.usgs.gov/usa/nwis/gw) for use in calculating annual change in storage. Land elevation is reported for each well as feet above mean sea level for NGVD29.

Pamlico Sound, North Carolina

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Beaufort	352252077050707	1982	17.37	15.02	0.25	0.18		5.67439E-09
Craven	352309077102901	1982	25.03	24.37	0.25	0.05		1.59366E-09
Lenoir	351600077381001	1982	71.9	68.44	0.25	0.26		8.35463E-09
Martin	355734077180001	1982	15.43	12.53	0.25	0.22		7.00243E-09
Orange	355522079043001	1982	45.7	40.07	0.25	0.43		1.35944E-08
Wayne	352002077581001	1982	10.75	6.49	0.25	0.32		1.02863E-08
					Watershed DS	0.24	0.13	7.75097E-09
							(n = 7; %cv =53)	4.09201E-09
						% error	52.79	

Sapelo Island/Doboy Sound, Georgia

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Appling	314028082220901	1984	131.28	130.07	0.25	0.09		2.92171E-09
Liberty	313901081234101	1984	9.52	7.32	0.25	0.17		5.31219E-09
Long	313845081361701	1984	49.77	47.76	0.25	0.15		4.85341E-09
McIntosh	313054081245501	1984	8.79	7.92	0.25	0.07		2.10073E-09
Toombs	321110082131501	1984	131.3	131.2	0.25	0.01		2.41463E-10
					Watershed DS	0.10	0.07	3.0859E-09
							(n = 7; %cv =53)	2.07176E-09
						% error	67.14	

Table A-3 continued

Altamaha Sound, Georgia

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Brantley	311217081580501	1995	24.68	23.2	0.25	0.11		3.57366E-09
Camden	304512081343601	1995	2.33	-0.24	0.25	0.20		6.2056E-09
Glynn	310658081250101	1995	-18.43	-19.98	0.25	0.12		3.74268E-09
Pierce	312355082084201	1995	61.7	54.65	0.25	0.54		1.70232E-08
Telfair	314858082573901	1995	85.05	84.27	0.25	0.06		1.88341E-09
Turner	313913083375201	1995	160.2	157.79	0.25	0.18		5.81926E-09
					Watershed DS	0.20	0.17	6.37463E-09
							(n = 7; %cv = 53)	5.45292E-09
						% error	85.54	

Indian River Lagoon, Florida

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Osceola	274856080594401	1999	45.4	44.04	0.25	0.10		3.2839E-09
Orange	282348080564701	1999	36.61	35.41	0.25	0.09		2.89756E-09
Ornage2	283333081233501	1999	48.84	47.33	0.25	0.12		3.64609E-09
Orange3	283333081233502	1999	49.55	47.8	0.25	0.13		4.22561E-09
Orange4	283249081053202	1999	48.91	48.38	0.25	0.04		1.27976E-09
					Watershed DS	0.10	0.04	3.06658E-09
							(n = 7; %cv =53)	1.11217E-09
						% error	36.27	

Table A-4: Groundwater well data for Gulf Watersheds as found in the USGS website (http://water.usgs.gov/usa/nwis/gw) for use in calculating annual change in storage. Land elevation is reported for each well as feet above mean sea level for NGVD29.

Charlotte Harbor, Florida

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Charlotte	270133082034602	1995	22.36	18.02	0.18	0.24		7.54524E-09
Desoto	270410081565201	1995	49.68	45.98	0.18	0.20		6.43258E-09
Hardee	273103081363701	1995	42.4	20.15	0.18	1.22		3.86824E-08
Hardee2	272041081562301	1995	45.49	4.6	0.18	2.24		7.10887E-08
					Watershed DS	0.98	0.97	3.09372E-08
							(n = 10; %cv = 53)	3.06583E-08
						%error	99.10	

Tampa/Sarasota Bay, Florida

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Hillsborough	275802082044701	1991	73.25	56.94	0.18	0.89		2.83555E-08
Hillsborough2	280112082270101	1991	10.01	8.9	0.18	0.06		1.92977E-09
Manatee	271832082064802	1991	69.79	65.81	0.26	0.32		9.99465E-09
Pasco	281037082071801	1991	87.79	82.12	0.18	0.31		9.85749E-09
Polk	274155081573201	1991	67.1	46.67	0.18	1.12		3.55183E-08
					Watershed DS	0.54	0.45	1.71311E-08
							(n = 10; %cv = 53)	1.41332E-08
						%error	82.50	

Table A-4 continued

Appalachee/Ochlocknee Bays, Florida

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Colquitt	311730083412501	1990	202.56	207.76	0.25	0.40		1.25561E-08
Mitchell	312253084100001	1990	21.74	21.35	0.25	0.03		9.41707E-10
Mitchell2	311328084130701	1990	34.16	27.17	0.25	0.53		1.68783E-08
Thomas	304646083443401	1990	139.75	136.16	0.25	0.27		8.66853E-09
Worth	312149083511801	1990	201.09	199.36	0.25	0.13		4.17731E-09
					Watershed DS	0.27	0.20	8.64438E-09
							(n = 7; %cv = 53)	6.37152E-09
						%error	73.71	

Apalachicola Bay, Florida

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Chattahoochee	322036084590301	1989-90	32.86	30.91	0.25	0.15		4.70853E-09
Decatur	310215084325201	1989-90	60.16	54.03	0.25	0.47		1.48017E-08
Early	311704084474101	1989-90	34.66	30.9	0.25	0.29		9.07902E-09
Seminole	305356084534601	1989-90	33.96	16.53	0.25	1.33		4.2087E-08
Webster	320401084320801	1989-90	102.45	99.3	0.26	0.25		7.91034E-09
					Watershed DS	0.50	0.48	1.57173E-08
							(n = 10; %cv = 53)	1.51856E-08
						%error	96.62	

Table A-4 continued

Mobile Bay, Alabama

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Baldwin	311018087473001	1980-82	75.07	71.04	0.25	1.01		3.19257E-08
Blount	340959086305901	1980-82	18.74	18.29	0.25	0.11		3.56491E-09
Choctaw	315151088174101	1980-82	29.22	28.40	0.25	0.21		6.49606E-09
Clarke	314950088052001	1980-82	28.14	26.27	0.26	0.49		1.54067E-08
Elmore	323138086184201	1980-82	57.57	55.92	0.25	0.41		1.30713E-08
Hale	325308087264301	1980-82	75.69	50.82	0.25	6.22		1.97021E-07
Monroe	314852087193501	1980-82	100.28	96.04	0.25	1.06		3.35894E-08
					Watershed DS	1.36	2.17	4.30107E-08
							(n = 7; %cv = 53)	6.88941E-08
						%error	160.18	

Barataria/Terrbonne Basin, Louisiana

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Ascension	302008090541601	1981	1.84	3.04	0.26	0.31		9.88668E-09
Assumption	295918091030101	1981	17.32	16.8	0.25	0.13		4.11945E-09
Iberville	301227091101301	1981	11.42	8.69	0.25	0.68		2.16271E-08
Pointe Coupee	303402091325501	1981	5.4	8.5	0.26	0.81		2.55406E-08
St Landry	303108092041201	1981	96.25	85.38	0.25	2.72		8.61124E-08
St Martin	301304091424002	1981	7.85	8.82	0.25	0.24		7.68436E-09
W Baton Rouge	302652091121401	1981	127.9	110.8	0.26	4.45		1.40885E-07
					Watershed DS	1.33	1.63	4.22651E-08
							(n = 7; %cv = 53)	5.17259E-08
						%error	122.38	

Table A-4 continued

West Mississippi Sound, Mississippi

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Attala	325844089462701	1994	5.06	4.96	0.25	0.01		2.41463E-10
Calhoun	335118089175901	1994	103	103.04	0.25	0.00		9.65853E-11
Choctaw	331849089105101	1994	15.35	15	0.25	0.03		8.45121E-10
Hancock	302227089373001	1994	41.55	40	0.25	0.12		3.74268E-09
Jackson	302244088325801	1994	82.1	61.59	0.25	1.56		4.95241E-08
Jones	314143089083901	1994	181.56	160.55	0.25	1.60		5.07314E-08
Leake	324427089295201	1994	63.25	61.84	0.25	0.11		3.40463E-09
Lowndes	332517088235601	1994	4.55	2.7	0.25	0.14		4.46707E-09
Madison	322413090101701	1994	219.78	216.32	0.25	0.26		8.35463E-09
Noxubee	330645088332801	1994	16.5	15.93	0.26	0.05		1.43139E-09
Oktibbeha	332710088471701	1994	240.91	240.21	0.25	0.05		1.69024E-09
Rankin	321846089475101	1994	136.12	135.31	0.25	0.06		1.95585E-09
Simpson	315256089392401	1994	356.49	355.12	0.26	0.11		3.44037E-09
Wayne	314115088392301	1994	25.12	20.72	0.25	0.34		1.06244E-08
					Watershed DS	0.32	0.68	1.00393E-08
							(n = 7; %cv = 53)	1.72423E-08
						%error	215.63	

Table A-4 continued

Calcasieu/Sabine Rivers, Louisiana and Texas

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Allen	303004092541101	1993	32.22	24.12	0.25	2.03		6.41684E-08
Bossier	321702093293914	1993	0.6	0.3	0.25	0.08		2.37661E-09
Calcasieu	300353093210201	1993	61.98	50.7	0.26	2.93		9.29348E-08
Natchitoches	313139092465001	1993	22.78	20.79	0.25	0.50		1.57648E-08
Sabine	312206093311001	1993	24.44	23.49	0.25	0.24		7.52592E-09
Vernon	311201093080203	1993	177.5	172.8	0.25	1.18		3.72335E-08
Orange	300322093452601	1993	42.98	41.8	0.25	0.30		9.34799E-09
					Watershed DS	1.03	1.08	3.27646E-08
							(n = 7; %cv = 53)	3.42159E-08
						%error	104.43	

Galveston/Matagorda Bay, Texas

County	Well #	Period of Record	High Water (ft)	Low Water (ft)	Sy	DS	Std Dev	S DS m/sec
Bosque	320019097272701	1999	95.28	91.35	0.26	1.02		3.23789E-08
Comal	293636098190901	1999	143.04	142.95	0.18	0.02		5.13347E-10
Coryell	312558097435201	1999	438.35	437.33	0.25	0.26		8.08046E-09
Harris	294253095352701	1999	397.51	386.96	0.25	2.64		8.35773E-08
Harris2	294901095221001	1999	256.77	249.74	0.25	1.76		5.56918E-08
Hays	300510097504001	1999	132.2	107.72	0.18	4.41		1.3963E-07
Kendall	295819098534001	1999	116.99	114.07	0.26	0.76		2.40576E-08
Travis	301356097473301	1999	252	227.51	0.18	4.41		1.39687E-07
					Watershed DS	1.91	1.75	6.04522E-08
							(n = 7; %cv = 53)	5.54887E-08
						%error	91.79	

Table A-4 continued

Coastal Bays and Estuaries, Texas

County	**Well #**	**Period of Record**	**High Water (ft)**	**Low Water (ft)**	**Sy**	DS	**Std Dev**	S DS **m/sec**
Frio	285324099043001	1994	369.7	333.1	0.26	9.52		3.01544E-07
Medina	292117098524701	1994	70.26	46	0.18	4.37		1.38376E-07
Medina2	292618099165901	1994	123.88	94.25	0.18	5.33		1.69005E-07
Medina3	292045099081801	1994	183.48	152.97	0.18	5.49		1.74025E-07
Uvalde	291237099471201	1994	30.56	27.54	0.18	0.54		1.72256E-08
					Watershed DS	5.05	3.20	1.60035E-07
							(n = 7; %cv = 53)	1.01445E-07
						%error	63.39	

Appendix B

Geochemical and Hydrographic Data for Barataria Basin, Louisiana

Table B-1: Barataria Basin Sample Locations

Description	Latitude (N)	Longitude (W)	Station ID	Distance Upstream (km)
Red Buoy #4 Barataria Bay	29^o17.060'	89^o57.990'	**BT#3**	4.5
Red Pole #26 Barataria Bay	29^o22.963'	89^o59.141'	**BT#8**	16.4
Bayou St Dennis	29^o30.312'	90^o03.245'	**BT#14**	31.8
Bayou Perot	29^o35.664'	90^o09.401'	**BT#19**	51.9
Bayou de Allemands	29^o46.239'	90^o23.816'	**BT#27**	89.0
Bayou Chevreuil	29^o53.739'	90^o37.729'	**BT#35**	118.4
Bayou Chevreuil	29^o53.926'	90^o41.278'	**BT#36**	124.1
B. Chev. Boat Landing	29^o54.694'	90^o43.766'	**BT#37**	128.4

Table B-2: Barataria Basin Transect Hydrography

Station ID	Water Depth (m)	Salinity (ppt)	Conductivity (mS)	Temperature (C)	Wind Speed (m/s)
25 May 1999 Barataria Transect					
BT#3	3.5	18.5	N/A	26.9	2.51
BT#8	1.5	14.0	N/A	28.0	2.67
BT#14	1.8	7.0	N/A	27.0	2.82
BT#19	2.0	2.0	N/A	28.0	2.56
BT#27	1.8	1.0	N/A	32.2	2.05
BT#35	2.1	0.0	N/A	31.1	1.44
BT#36	2.7	0.0	N/A	31.4	1.18
BT#37	2.0	0.0	N/A	30.7	1.18
06 October 1999 Barataria Transect					
2BT#3	4.1	28.9	48.67	24.6	4.00
2BT#8	2.1	26.7	40.67	23.9	3.69
2BT#14	>4.3	19.1	30.30	24.3	3.85
2BT#19	3.0	7.3	12.22	23.2	3.69
2BT#27	2.1	1.1	2.19	24.5	3.59
2BT#35	3.2	0.2	0.34	24.2	4.26
2BT#36	2.4	0.2	0.36	25.1	4.21
2BT#37	3.0	0.2	0.33	23.8	4.26
11January 2000 Trip to Barataria Transect					
3BT#3	3.7	29.0	38.22	17.3	1.13
3BT#8	2.0	21.3	28.89	17.3	0.92
3BT#14	1.8	12.0	17.34	17.4	0.77
3BT#19	1.8	4.2	6.51	17.6	0.82
3BT#27	1.8	0.5	0.87	17.6	0.82
3BT#35	3.0	0.1	0.25	17.8	1.23
3BT#36	1.8	0.1	0.24	17.3	0.92
3BT#37	2.7	0.1	0.23	17.5	0.87
18 April 2000 Trip to Barataria Transect					
4BT#3	6.7	26.1	40.26	23.4	2.62
4BT#8	1.8	20.6	32.25	24.1	2.51
4BT#14	7.3	13.2	21.87	24.8	2.36
4BT#19	1.5	4.2	7.50	24.1	2.41
4BT#27	1.5	0.9	1.79	25.2	2.51
4BT#35	3.0	0.2	0.35	23.5	2.46
4BT#36	1.8	0.2	0.33	23.5	2.05
4BT#37	2.4	0.2	0.40	26.7	2.00

Table B-2 continued

Station ID	Water Depth (m)	Salinity (ppt)	Conductivity (mS)	Temperature (C)	Wind Speed (m/s)
27 June 2000 Trip to BT Transect					
5BT#3	1.2	25.7	44.20	29.9	3.1
5BT#8	N/A	27.0	45.93	29.5	3.2
5BT#14	7.0	20.5	35.86	29.6	2.5
5BT#19	0.3	10.0	18.54	29.6	1.5
5BT#27	1.8	3.9	8.02	31.4	4.5
5BT#35	2.7	1.2	2.63	30.2	5.2
5BT#36	3.0	1.2	2.63	29.3	4.6
5BT#37	2.4	1.3	2.68	30.0	4.7
6 September 2000 Trip to BT Transect					
6BT#3	4.0	30.5	51.30	29.7	7.7
6BT#8	N/A	27.2	46.29	29.7	8.8
6BT#14	4.3	22.3	39.16	30.1	9.3
6BT#19	1.8	9.9	18.42	29.5	6.8
6BT#27	1.5	5.3	10.76	31.2	6.7
6BT#35	2.4	2.7	5.58	30.6	6.9
6BT#36	1.8	2.8	6.02	32.8	8
6BT#37	2.4	2.7	5.74	31.7	7.8
5 December 2000 Trip to BT Transect (Rough trip - meters stopped working after BT8)					
7BT#3	N/A	28.8	N/A	11.0	4.9
7BT#8	N/A	16.4	N/A	9.0	5.5
7BT#14	N/A	6.5	N/A	N/A	6.4
7BT#19	N/A	4.5	N/A	N/A	7.5
7BT#27	N/A	1.1	N/A	N/A	7.6
7BT#35	N/A	0.4	N/A	N/A	5.7
7BT#36	N/A	0.4	N/A	N/A	5.3
7BT#37	N/A	0.4	N/A	N/A	5.7
14 March 2001 Trip to BT Transect					
8BT#3	4.3	20.8	29.86	19.8	6.4
8BT#8	2.4	15.2	22.85	20.6	N/A
8BT#14	4.3	7.5	11.99	20.6	2.9
8BT#19	1.5	1.6	2.79	20.5	3.5
8BT#27	2.1	0.5	1.08	20.5	3
8BT#35	2.7	0.1	0.27	20.4	N/A
8BT#36	3.0	0.0	0.01	19.0	N/A
8BT#37	2.4	0.2	0.30	18.8	4.2

Table B-2 continued

Station ID	Water Depth (m)	Salinity (ppt)	Conductivity (mS)	Temperature (C)	Wind Speed (m/s)
21 August 2001 Trip to Barataria Transect					
9BT3	4.6	25.8	44.18	29.6	2.57
9BT8	1.8	14.5	26.13	29.5	0.67
9BT14	2.7	3.1	6.27	29.9	0.21
9BT19	2.1	0.5	1.03	30.1	1.39
9BT27	2.4	0.1	0.23	32.0	0.82
9BT35	2.7	0.1	0.18	30.2	1.13
9BT36	2.7	0.1	0.20	31.8	0.51
9BT37	2.1	0.1	0.17	29.5	0.77

Table B-3: Barataria Basin Transect Chemistry

Station ID	Sample Depth (m)	Sample Time	PH	Ra-226 Activity dpm/L	EXCESS Rn Activity dpm/L	Excess Rn Inventory dpm/m2	Diffusive flux dpm/m2/day	Unsupported Inventory dpm/m2/day
25 May 1999 Barataria Transect								
BT#3	3.2	8:37	8.20	0.42 ± 0.07	2.41 ± 0.12	7707	356	1040.22
BT#8	1.2	9:34	8.15	1.00 ± 0.05	1.13 ± 0.09	1373	808	-559.15
BT#14	1.5	10:21	7.74	1.26 ± 0.06	1.03 ± 0.11	1566	1105	-821.13
BT#19	1.7	11:08	8.45	0.63 ± 0.06	1.41 ± 0.11	2357	1195	-768.29
BT#27	1.5	12:21	8.23	0.38 ± 0.03	2.51 ± 0.11	3830	2023	-1328.91
BT#35	1.8	14:19	7.38	0.25 ± 0.02	11.09 ± 0.28	20282	1978	1696.32
BT#36	2.4	14:39	7.23	0.34 ± 0.02	18.35 ± 0.68	44745	1334	6772.25
BT#37	1.7	14:56	7.61	0.69 ± 0.20	8.07 ± 0.34	13524	1519	930.98
06 October 1999 Barataria Transect								
2BT#3	3.8	8:54	8.43	0.67 ± 0.04	0.57 ± 0.12	2185	356	39.76
2BT#8	1.8	9:39	8.32	0.99 ± 0.09	0.50 ± 0.14	906	808	-643.87
2BT#14	4.0	10:40	8.06	1.09 ± 0.10	0.40 ± 0.14	1581	1105	-818.41
2BT#19	2.7	11:16	8.24	1.18 ± 0.24	0.05 ± 0.29	124	1195	-1172.92
2BT#27	1.8	12:38	8.31	0.70 ± 0.04	1.16 ± 0.09	2130	2023	-1637.00
2BT#35	2.9	14:24	8.22	0.27 ± 0.03	3.15 ± 0.13	9114	1978	-326.98
2BT#36	2.1	14:40	7.99	0.32 ± 0.02	3.53 ± 0.16	7526	1334	29.08
2BT#37	2.7	14:54	8.12	0.49 ± 0.05	4.41 ± 0.25	12109	1519	674.62
11January 2000 Trip to Barataria Transect								
3BT#3	3.4	9:13	8.55	0.89 ± 0.04	1.21 ± 0.19	4048	356	377.24
3BT#8	1.7	9:49	8.52	0.67 ± 0.07	3.97 ± 0.16	6649	808	396.60
3BT#14	1.5	10:28	8.18	1.37 ± 0.07	0.53 ± 0.12	812	1105	-957.77
3BT#19	1.5	11:13	8.39	1.39 ± 0.08	0.01 ± 0.12	18	1195	-1192.15
3BT#27	1.5	12:27	8.13	0.29 ± 0.02	1.98 ± 0.10	3025	2023	-1474.82
3BT#35	2.7	13:54	7.75	0.73 ± 0.03	19.15 ± 1.19	52541	1978	7540.91
3BT#36	1.5	14:11	7.40	0.95 ± 0.04	16.85 ± 0.42	25686	1334	3319.32
3BT#37	2.4	14:25	7.58	0.94 ± 0.09	28.43 ± 2.16	69330	1519	11041.85

Table B-3 continued

Station ID	Sample Depth (m)	Sample Time	PH	Radium-226 Activity dpm/L			Excess Radon-226 Activity dpm/L			Excess Rn Inventory dpm/m2	Diffusive flux dpm/m2/day	Unsupported Inventory dpm/m2/day
18 April 2000 Trip to Barataria Transect												
4BT#3	6.4	8:43	7.52	0.64	±	0.04	2.79	±	0.20	17853	356	2878.38
4BT#8	1.5	9:16	7.95	1.36	±	0.04	1.87	±	0.05	2853	808	-291.12
4BT#14	7.0	9:55	8.09	1.66	±	0.07	1.26	±	0.11	8843	1105	497.33
4BT#19	1.2	10:36	8.43	1.55	±	0.16	1.03	±	0.17	1253	1195	-968.33
4BT#27	1.2	11:42	8.42	0.50	±	0.03	4.00	±	0.05	4880	2023	-1138.82
4BT#35	2.7	13:16	8.08	0.84	±	0.16	11.13	±	0.58	30533	1978	3553.54
4BT#36	1.5	13:33	7.81	0.43	±	0.02	12.48	±	0.74	19018	1334	2111.10
4BT#37	2.1	13:45	7.91	0.33	±	0.02	10.09	±	0.14	21538	1519	2383.03
27 June 2000 Trip to Barataria Transect												
5BT#3	4.0	8:41	8.20	0.66	±	0.04	2.74	±	0.20	10949	356	1627.57
5BT#8	4.2	9:08	7.85	0.97	±	0.04	2.46	±	0.14	10312	808	1060.26
5BT#14	4.2	9:44	7.46	2.66	±	0.05	0.10	±	0.06	403	1105	-1031.73
5BT#19	1.0	10:44	7.82	2.14	±	0.06	0.69	±	0.10	692	1195	-1070.05
5BT#27	1.8	11:32	7.82	0.69	±	0.05	1.50	±	0.10	2698	2023	-1534.16
5BT#35	2.7	13:16	7.21	0.84	±	0.04	12.76	±	0.49	34452	1978	4263.63
5BT#36	3.0	13:29	7.09	0.67	±	0.03	11.38	±	0.71	34141	1334	4851.02
5BT#37	2.4	13:41	7.03	0.69	±	0.03	9.87	±	0.42	23697	1519	2774.21
6 September 2000 Trip to Barataria Transect												
6BT#3	4.0	8:54	7.80	0.75	±	0.03	1.23	±	0.14	4904	356	532.35
6BT#8	4.2	9:33	7.83	1.53	±	0.13	2.18	±	0.24	9160	808	851.63
6BT#14	4.2	10:12	7.77	2.28	±	0.39	0.17	±	1.09	699	1105	-978.23
6BT#19	1.8	10:57	7.85	N/A		N/A	N/A		N/A			
6BT#27	1.5	12:08	7.81	2.09	±	0.09	0.09	±	0.12	131	2023	-1999.18
6BT#35	2.4	13:33	7.57	1.48	±	0.07	2.60	±	0.16	6243	1978	-847.20
6BT#36	1.8	13:48	7.59	1.00	±	0.05	3.76	±	0.28	6759	1334	-109.86
6BT#37	2.4	13:57	7.36	0.93	±	0.10	5.68	±	0.19	13635	1519	951.21

Table B-3 continued

Station ID	Sample Depth (m)	Sample Time	PH	Ra-226 Activity dpm/L			Excess Rn Activity dpm/L			Excess Rn Inventory dpm/m2	Diffusive flux dpm/m2/day	Unsupported Inventory dpm/m2/day
05 December 2001 Trip to Barataria Transect												
7BT3	4.0	9:05	8.06	0.56	±	0.07	2.04	±	0.27	8175	356	1125.06
7BT8	4.1	9:43	7.11	1.83	±	0.15	0.69	±	0.18	2812	808	-298.46
7BT14	1.3	10:27	N/A	1.30	±	0.22	0.16	±	0.24	210	1105	-1066.78
7BT19	1.4	11:09		1.27	±	0.29	0.42	±	0.32	582	1195	-1089.94
7BT27	2.3	12:16		0.40	±	0.02	3.16	±	0.08	7271	2023	-705.53
7BT35	2.7	13:59		0.99	±	0.09	27.83	±	2.53	75154	1978	11637.89
7BT36	3.0	14:14		0.53	±	0.05	53.09	±	0.66	159274	1334	27522.29
7BT37	2.4	14:25		0.60	±	0.16	32.87	±	0.60	78881	1519	12772.27
14 March 2001 Trip to Barataria Transect												
8BT3	4.3	8:48	8.12									
8BT8	2.4	9:21	8.25									
8BT14	4.3	9:59	8.00	0.89	±	0.02	0.81	±	0.04	3474	1105	-475.34
8BT19	1.5	10:40	8.10	0.55	±	0.05	1.44	±	0.16	2165	1195	-803.07
8BT27	2.1	11:48	7.45	0.34	±	0.02	4.10	±	0.27	8601	2023	-464.65
8BT35	2.7	13:23	7.71									
8BT36	3.0	13:37	7.56									
8BT37	2.4	13:49	7.29	0.49	±	0.04	29.08	±	2.47	69797	1519	11126.44
21 August 2001 Trip to Barataria Transect												
9BT3	4.6	8:41	7.43	0.74	±	0.06	3.46	±	0.26	15910	356	2526.43
9BT8	1.8	9:08	7.75	1.05	±	0.15	2.36	±	0.20	4239	808	-39.87
9BT14	2.7	9:42	7.97	0.61	±	0.05	1.35	±	0.14	3642	1105	-444.97
9BT19	2.1	10:22	8.00	0.65	±	0.04	1.71	±	0.21	3599	1195	-543.33
9BT27	2.4	11:30	6.95	0.32	±	0.03	2.21	±	0.53	5298	2023	-1063.02
9BT35	2.7	13:11	6.72	0.33	±	0.06	9.81	±	2.26	26498	1978	2822.57
9BT36	2.7	13:25	6.79	1.07	±	0.24	7.83	±	0.51	21139	1334	2495.47
9BT37	2.1	13:41	6.87	0.34	±	0.05	9.65	±	0.19	20271	1519	2153.42

Table B-4: Comparison of results for filtered and unfiltered radium Samples – August 21, 2001.

Station ID	Unfiltered Ra-226 Activity dpm/L	Filtered Ra-226 Activity dpm/L	Difference due to Filtration
9BT 3	0.74 ± 0.06	0.72 ± 0.06	0.02 ± 0.00
9BT 8	1.05 ± 0.15	0.96 ± 0.24	0.09 ± -0.09
9BT 14	0.61 ± 0.05	0.58 ± 0.07	0.03 ± -0.02
9BT 19	0.65 ± 0.04	0.61 ± 0.04	0.04 ± 0.00
9BT 27	0.32 ± 0.03	0.23 ± 0.02	0.08 ± 0.01
9BT 35	0.33 ± 0.06	0.34 ± 0.02	0.00 ± 0.04
9BT 36	1.07 ± 0.24	1.11 ± 0.23	-0.03 ± 0.00
9BT 37	0.34 ± 0.05	0.35 ± 0.08	-0.01 ± -0.03

Table B-5: Upper Barataria Basin: Lac des Allemands (LDA) Hydrography
Trip Date: 11/11/1999

Station ID	Station Description	Water Depth (m)	Salinity (ppt)	Conductivity (uS)	Temperature (C)
LDA#1	East Inlet of LDA	1.2	0.2	430.5	20.2
LDA#2	NE End of Lake	1.5	0.3	501.0	20.7
LDA#3	North End of Lake	0.9	0.3	505.0	20.9
LDA#4	Haut Point	2.1	0.3	530.0	20.4
LDA#5	Herbes Point	1.5	0.2	466.0	20.9
LDA#6	Bayou Chevreuil #1	2.7	0.1	272.0	18.8
LDA#7	Bayou Chevreuil #2	3.2	0.1	257.3	18.7
LDA#8	Bayou Chevreuil #3	2.7	0.1	242.0	19.1
LDA#9	Bayou Chevreuil #4	2.7	0.1	245.0	18.7
LDA#10	Bayou Boeuf	4.1	0.2	302.2	20.1

Table B-6: Upper Barataria Basin: Lac des Allemands (LDA) Chemistry
Trip Date: 11/11/1999

Station ID	Sample Depth (m)	Sample Time	PH	Dissolved Oxygen (MG/L)	Ra-226 Activity dpm/L	Excess Rn Activity dpm/L
LDA#1	0.9	11:59	7.15	7.53	0.57 ± 0.19	2.25 ± 0.30
LDA#2	1.2	12:41	7.03	8.84	0.35 ± 0.02	1.28 ± 0.15
LDA#3	0.6	13:00	7.67	10.67	0.30 ± 0.02	1.22 ± 0.12
LDA#4	1.8	13:17	7.98	9.96	0.38 ± 0.02	0.89 ± 0.25
LDA#5	1.2	13:30	8.08	9.84	0.48 ± 0.03	2.09 ± 0.13
LDA#6	2.4	13:46	7.12	4.11	0.30 ± 0.02	3.87 ± 0.11
LDA#7	2.9	13:56	6.85	N/A	0.44 ± 0.02	3.48 ± 0.19
LDA#8	2.4	14:15	6.94	6.06	0.51 ± 0.03	5.99 ± 0.54
LDA#9	2.4	14:32	6.96	5.40	0.52 ± 0.03	8.87 ± 0.65
LDA#10	3.8	15:18	7.06	7.80	0.54 ± 0.03	2.96 ± 0.65

Table B-7: Jean Lafitte Surface Water Sampling Locations

Description	Latitude (N)	Longitude (W)
Pipeline North (W5)	29°47.464'	90°08.129'
ICW-Jones Point	29°44.662'	90°08.263'
Kenta Canal (W3)	29°46.189'	90°06.519'
Pipeline South (W4)	29°45.829'	90°08.665'
Kenta Canal (Brdge)	29°45.425'	90°06.519'
Twin Canals (W6)	29°48.526'	90°07.822'
Lake Salvador	29°44.568'	90°09.077'
Miss River (Luling)	29°54.019'	90°11.465'
Belle Chase	29°51.300'	90°58.855'

Table B-8: Site Characteristics for JELA wells

Station ID	Coordinates		Depth (m)	Sediment	Description
East to West Transect					
Well 1 - woods	$29^o46.939'$	$90^o06.741'$	2.7	Peat, Fine clays	Terrestrial, natural levee zone
Well 2 - deep woods	$29^o46.899'$	$90^o06.788'$	1.8	Peat, clays	Swamp, with typical vegetation (maple, ash)
Kenta Canal	$29^o46.213'$	$90^o06.517'$	1.8	Peat, silt clays	Swamp to marsh transition zone
North to South Transect					
Twin Canals	$29^o48.568'$	$90^o07.717'$	0.9, 2.7	Peat, clay	Creek edge site
Pipeline North	$29^o47.517'$	$90^o08.070'$	0.9, 2.7	Peat, silt, clay	Floating marsh site, creek edge
Pipeline South	$29^o45.782'$	$90^o08.618'$	0.9, 2.7	Peat, mud, organics	Floating marsh site, creek edge

Table B-9: Jean Lafitte National Park Hydrography

Station ID	Salinity (ppt)	Conductivity (mS)	Temperature (C)	Sample Depth (m)
02-Mar-00				
Miss R.-Davis P.	0.2	0.32	11.0	0.9
Miss R.-Belle Chase	0.2	0.32	12.3	1.8
Kenta Canal	2.0	3.63	22.5	0.3
ICW-Jones Pt	3.0	5.10	23.0	0.6
21-Apr-00				
Pipeline North (W5)	2.4	4.52	26.3	1.1
Pipeline South (W4)	3.0	6.22	31.1	1.4
Kenta Canal (W3)	0.6	1.27	26.7	0.6
Kenta Canal (Brdge)	0.3	0.71	25.3	0.9
Twin Canals (W6)	2.5	4.71	25.0	0.9
ICW-Jones Point	1.4	2.65	27.0	3.7
Lake Salvador	3.8	7.07	26.0	1.5
Miss River (Luling)	0.2	0.31	18.2	0.3
Belle Chase	0.2	0.32	17.1	0.6
31-Jul-00				
Pipeline Nth	3.5	7.04	30.0	0.6
Pipeline Sth	6.0	11.75	30.6	1.9
Kenta (W3)	2.1	4.16	28.1	0.6
Twin Canals	2.1	3.97	27.6	0.5
ICW-Jones Pt	6.6	11.61	30.6	4.3
Lake Salvador	6.7	11.67	30.0	3.4
MR (Luling)	0.2	0.48	29.5	0.3
10-Nov-00				
3ICW-JP	4.1	2.04	15.9	3.0
3Pipeline Sth	3.9	6.99	9.1	2.0
3Lake Salvador	4.7	8.21	13.3	4.3
3Twin Canals Surf	1.9	3.65	10.5	0.9
3Pipeline Nth Surf	2.7	3.98	11.6	1.7
3Kenta Surf (W3)	1.9	1.86	11.0	2.1
3MR Luling	0.3	0.32	15.2	0.5

Table B-9 continued

Station ID	Salinity (ppt)	Conductivity (mS)	Temperature (C)	Sample Depth (m)
30-Mar-01				
4PipeLine Nth Surf	1.1	2.46	19.6	1.3
4Pipeline Sth Surf	2.8	5.04	18.1	1.5
4Twin Canals Surf	0.8	2.25	19.9	0.8
4Lake Salvador	4.1	7.09	20.2	4.1
4ICW Jones Pt.	4.0	1.96	17.9	2.1
4Kenta Dock	3.8	2.52	17.7	0.6
4Kenta (W3)	3.2	2.88	17.6	2.0
4MR Luling	0.1	0.26	17.0	0.6
13-14 July 2001				
4PipeLine Nth Surf	1.0	2.12	27.4	0.8
4Pipeline Sth Surf	1.5	3.07	27.7	1.1
4Twin Canals Surf	1.0	2.12	27.5	0.5
4Lake Salvador	0.7	1.58	28.3	4.3
4ICW Jones Pt.	0.3	6.84	29.5	2.0
4Kenta Dock	0.6	1.24	26.3	0.3
4Kenta (W3)	0.7	1.37	26.4	1.4
4MR Luling	0.2	0.49	29.0	0.3

Table B-10: Jean Lafitte National Park Surface Chemistry

Station ID	Sample Depth (m)	Sample Time	PH	Diss. Oxygen (MG/L)	Ra-226 Activity dpm/L	EXCESS Rn Activity dpm/L	Flux based on Inventory (dpm/m2/day
02 March 2000 Trip							
Miss R.-Davis P.	0.9	7:59	5.90	10.96	2.24 ± 0.13	4.51 ± 0.38	735.2 ± 68.5
Miss R.-Belle Chase	1.8	16:09	8.13	12.65	0.85 ± 0.04	3.05 ± 0.12	995.8 ± 22.0
Kenta Canal	0.3	15:14	7.34	7.24	0.52 ± 0.03	15.74 ± 0.74	855.5 ± 133.2
ICW-Jones Pt	0.6	14:08	8.01	11.10	0.57 ± 0.06	2.77 ± 0.57	300.9 ± 103.7
21-22 April 2000 Trip							
Pipeline North	1.1	15:06	7.21	6.14	0.77 ± 0.03	2.97 ± 0.24	592.1 ± 43.4
Pipeline South	1.4	16:04	7.64	5.02	0.99 ± 0.13	3.02 ± 0.18	765.3 ± 32.0
Kenta Canal (W3)	0.6	17:11	8.26	6.54	0.44 ± 0.02	6.06 ± 0.21	658.9 ± 37.4
Kenta Canal (Dock)	0.9	16:31	8.12	6.00	0.44 ± 0.03	8.18 ± 0.13	1333.2 ± 24.4
Twin Canals	0.9	11:42	6.94	3.59	1.02 ± 0.05	2.87 ± 0.23	467.8 ± 41.6
ICW-Jones Point	3.7	13:43	8.15	8.35	0.42 ± 0.02	1.37 ± 0.07	921.6 ± 12.3
Lake Salvador	1.5	13:58	8.08	9.08	0.96 ± 0.03	0.53 ± 0.10	145.0 ± 18.8
Miss River (Luling)	0.3	13:12	8.34	10.32	0.63 ± 0.03	6.27 ± 0.06	340.6 ± 11.1
Belle Chase	0.6	11:57	8.39	9.80	0.69 ± 0.03	8.55 ± 0.25	929.3 ± 45.5
31 July 2000 Trip							
2Pipeline Nth	0.6	15:43	7.26	N/A	1.87 ± 0.10	0.20 ± 0.15	21.6 ± 27.4
2Pipeline Sth	1.9	13:46	6.96		1.78 ± 0.10	1.78 ± 0.16	614.3 ± 29.6
2Kenta (W3)	0.6	17:08	7.27		1.33 ± 0.33	4.58 ± 0.43	497.6 ± 77.6
2Twin Canals	0.5	14:56	7.06		1.89 ± 0.28	4.20 ± 0.50	380.2 ± 90.3
2ICW-Jones Pt	4.3	16:16	7.46		1.44 ± 0.25	2.28 ± 0.30	1778.8 ± 54.5
2Lake Salvador	3.4	13:16	7.29		2.48 ± 0.50	1.28 ± 0.61	799.9 ± 110.2
2MR (Luling)	0.3	19:33	8.1		0.43 ± 0.09	2.29 ± 0.14	124.6 ± 25.3

Table B-10 continued

Station ID	Sample Depth (m)	Sample Time	PH	Diss. Oxygen (MG/L)	Ra-226 Activity dpm/L	EXCESS Rn Activity dpm/L	Flux based on Inventory (dpm/m2/day
11 November 2000 Trip							
3Pipeline Nth	3.0	13:46	8.08	N/A	1.95 ± 0.14	3.00 ± 0.21	1630.0 ± 38.7
3Pipeline Sth	2.0	14:27	7.42		1.48 ± 0.09	2.96 ± 0.23	1073.8 ± 41.6
3Kenta Surf (W3)	4.3	15:53	7.06		1.78 ± 0.18	4.40 ± 0.60	3427.4 ± 109.4
3Twin Canals Surf	0.9	13:05	8.19		1.57 ± 0.08	3.10 ± 0.15	505.8 ± 26.5
3ICW Jones Pt	1.7	14:51	7.42		1.47 ± 0.08	1.75 ± 0.11	540.2 ± 20.7
3Lake Salvador	2.1	14:41	8.01		1.96 ± 0.11	0.11 ± 0.14	41.2 ± 24.6
3MR Luling	0.5	18:02	7.21		0.43 ± 0.04	2.25 ± 0.15	203.5 ± 26.9
30-31 March Trip							
4PipeLine Nth Surf	1.3	17:55	6.92	N/A	1.00 ± 0.08	9.77 ± 0.18	2302.3 ± 32.6
4Pipeline Sth Surf	1.5	18:18	7.99		1.18 ± 0.16	5.17 ± 0.29	1405.9 ± 52.1
4Kenta Dock	0.8	8:38	7.72		0.37 ± 0.03	5.08 ± 0.19	737.0 ± 34.9
4Kenta (W3)	4.1	9:07	8.07		0.86 ± 0.06	9.62 ± 0.84	7142.8 ± 153.0
4Twin Canals Surf	2.1	17:19	6.53		0.54 ± 0.11	9.33 ± 0.96	3549.0 ± 173.8
4ICW Jones Pt.	0.6	18:31	8.29		0.65 ± 0.04	2.79 ± 0.16	303.2 ± 29.0
4Lake Salvador	2.0	18:26	8.31		0.78 ± 0.22	1.79 ± 0.37	647.6 ± 66.3
4MR Luling	0.6	12:10	7.63		1.10 ± 0.07	2.86 ± 0.15	310.5 ± 28.0
13-14 July 2001							
4PipeLine Nth Surf	0.8	11:56	7.01	N/A	0.91 ± 0.06	10.00 ± 2.85	1448.9 ± 517.2
4Pipeline Sth Surf	1.1	12:48	6.98		0.78 ± 0.04	7.42 ± 1.38	1479.3 ± 250.5
4Kenta Dock	0.5	14:39	7.64		0.61 ± 0.06	3.16 ± 0.20	285.9 ± 36.2
4Kenta (W3)	4.3	14:01	8.09		2.05 ± 0.12	9.29 ± 0.21	7237.4 ± 37.8
4Twin Canals Surf	2	12:10	8.31		0.78 ± 0.13	24.52 ± 4.20	8884.8 ± 760.1
4ICW Jones Pt.	0.3	13:21	7.46		0.42 ± 0.02	3.12 ± 0.76	169.4 ± 138.4
4Lake Salvador	1.4	13:12	6.96		1.87 ± 0.14	1.32 ± 0.82	335.5 ± 148.1
4MR Luling	0.3	18:16	7.3			1.84 ± 0.10	0.0 ± 17.4

Table B-11: Results of Benthic Flux Experiments at Twin Canals
Twin Canals 30 March 2001 Trip

Sample ID	Hour of Sampling	Collection Date/Time	Ra-226 (dpm/L)		Rn-222 (dpm/L)		Flux Time (min)	Rn-222 Benthic Flux (dpm/m2/day)	
Chamber A									
TCBF1A	0	03/30/2001 11:53	0.474	± 0.108	19.451	± 1.843			
TCBF2A	3	03/30/2001 14:11	0.447	± 0.025	25.662	± 0.696	138	20571	± 6247
TCBF3A	7	03/31/2001 6:42	0.657	± 0.035	41.192	± 1.092	1129	9726	± 921
TCBF4A	11	03/31/2001 10:12	0.647	± 0.033	58.230	± 5.367	1339	13811	± 2042
							Mean Flux =	14703	± 3070
Chamber B									
TCBF1B	0	03/30/2001 12:11	0.301	± 0.034	17.135	± 0.256			
TCBF2B	3	03/30/2001 14:18	0.483	± 0.033	24.901	± 2.334	127	26856	± 7908
TCBF3B	7	03/31/2001 6:46	0.497	± 0.074	32.314	± 3.007	1115	6980	± 1298
TCBF4B	11	03/31/2001 10:24	0.562	± 0.032	33.882	± 0.381	1333	6435	± 166
							Mean Flux =	13424	± 3124

Table B-11 continued
Twin Canals 08 August 2001 Trip

Sample ID	Hour of Sampling	Collection Date/Time	Ra-226 (dpm/L)	Rn-222 (dpm/L)	Flux Time (min)	Rn-222 Benthic Flux (dpm/m2/day)
Chamber A						
2TCBF1A	0	08/08/2001 10:35	0.455 ± 0.030	9.590 ± 0.237		
2TCBF2A	1	08/08/2001 11:18	0.618 ± 0.084	9.333 ± 0.581	43	-2099 ± 6436
2TCBF3A	3	08/08/2001 13:29	0.572 ± 0.121	10.895 ± 0.655	174	3725 ± 15795
2TCBF4A	7	08/08/2001 16:34	0.842 ± 0.132	11.869 ± 0.640	359	3262 ± 843
					Mean Flux =	1629 ± 7691
Chamber B						
2TCBF1B	0	08/08/2001 10:35	1.154 ± 0.171	9.516 ± 0.605		
2TCBF2B	1	08/08/2001 11:23	1.381 ± 0.076	9.030 ± 0.609	48	-3851 ± 7740
2TCBF3B	3	08/08/2001 13:25	1.415 ± 0.166	10.554 ± 0.647	170	2964 ± 2142
2TCBF4B	7	08/08/2001 16:40	2.334 ± 0.122	42.760 ± 10.326	365	36286 ± 11407
					Mean Flux =	11800 ± 7096

Table B-12: Results of Benthic Flux Experiments at Kenta Canal

Kenta Canal Benthic Fluxes 18 Sep 2000 Trip

Sample ID	**Hour of Sampling**	**Collection Date/Time**	**Ra-226** (dpm/L)	**Rn-222** (dpm/L)	**Flux Time (min)**	**Rn-222 Benthic Flux (dpm/m2/day)**
Chamber A						
BTBF1A	0	09/18/2000 10:06	1.160 ± 0.022	6.393 ± 0.516		
BTBF1B	5	09/18/2000 14:52	1.358 ± 0.060	6.247 ± 0.244	286	126 ± 909
Chamber B						
BTBF2A	0	09/18/2000 10:16	1.013 ± 0.018	3.906 ± 0.156		
BTBF2B	5	09/18/2000 14:59	1.113 ± 0.086	4.182 ± 0.159	283	639 ± 351
Chamber C						
BTBF3A	0	09/18/2000 10:12	1.143 ± 0.050	5.567 ± 3.398		
BTBF3B	5	09/18/2000 15:05	0.816 ± 0.069	6.503 ± 0.166	293	1518 ± 4623
					Mean Flux =	761 ± 1961

Table B-12 continued

Kenta Canal Benthic Fluxes 30 March 2001 Trip

Sample ID	Hour of Sampling	Collection Date/Time	Ra-226 (dpm/L)	Rn-222 (dpm/L)	Flux Time (min)	Rn-222 Benthic Flux (dpm/m2/day)
Chamber A						
2BTBF1A	0	03/30/2001 12:56	0.504 ± 0.021	8.822 ± 0.178		
2BTBF2A	3	03/30/2001 15:38	0.607 ± 0.042	8.240 ± 0.702	162	-1061 ± 1922
2BTBF3A	7	03/30/2001 19:43	0.402 ± 0.038	10.356 ± 0.849	407	2081 ± 938
2BTBF4A	18	03/31/2001 7:08	0.326 ± 0.035	10.861 ± 0.208	1092	1265 ± 117
					Mean Flux =	762 ± 992
Chamber B						
2BTBF1B	0	03/30/2001 13:04	0.557 ± 0.047	7.245 ± 0.629		
2BTBF2B	3	03/30/2001 15:36	0.259 ± 0.022	8.555 ± 0.137	152	4134 ± 1855
2BTBF3B	7	03/30/2001 19:45	0.483 ± 0.017	11.523 ± 3.775	401	4826 ± 4088
2BTBF4B	18	03/31/2001 7:12	0.321 ± 0.034	8.806 ± 0.172	1088	985 ± 276
					Mean Flux =	3315 ± 2073

Table B-13: Jean Lafitte National Park Benthic Flux Hydrography Information

Station ID	Description	Sample Date	Sample Time	Period	Salinity (ppt)	Conductivity (mS)	Temp (oC)	pH
(Kenta Canal Flux Measurements)								
18 September 2000 Trip								
BTBF1A	Chamber A, t=0 hrs	18-Sep	10:06	t=0	2.6	4.9	25.1	7.55
BTBF1B	Chamber A, t=5 hrs		14:52					
BTBF2A	Chamber A, t=0 hrs		10:16					
BTBF2B	Chamber B, t=5 hrs		14:59					
BTBF3A	Chamber C, t=0 hrs		10:12					
BTBF3B	Chamber C, t=5 hrs		15:05	t=5	2.9	5.9	29.4	7.75
30 March 2001 Trip								
2BTBF1A	Chamber A, t=0	30-Mar	12:56	t=0	0	0.68	20	7.72
2BTBF1B	Chamber B, t=0		13:04					
2BTBF2A	Chamber A, t=2		15:38	t=2	0.4	0.68	19.9	6.72
2BTBF2B	Chamber B, t=2		15:36					
2BTBF3A	Chamber A, t=7		19:43	t=7	0.4	0.73	15.2	6.65
2BTBF3B	Chamber B, t=7		19:45					
2BTBF4A	Chamber A, t=22	31-Mar	10:42	t=22	0.5	0.97	16.8	8.07
2BTBF4B	Chamber B, t=22		10:47					

Table B-13 continued

Station ID	Description	Sample Date	Sample Time	Period	Salinity (ppt)	Conductivity (mS)	Temp (oC)	pH
(Twin Canals Flux Measurments)								
30 March 2001 Trip								
TCBF1A	Chamber A, t=0	30-Mar	11:53	t=0	0.7	0.11	16.8	6.5
TCBF1B	Chamber B, t=0		12:11					
TCBF2A	Chamber A, t=2		14:11	t=2	0.75	0.13	19.9	6.53
TCBF2B	Chamber B, t=2		14:18					
TCBF3A	Chamber A, t=19	31-Mar	6:42	t=19	0.5	0.06	14.9	7.89
TCBF3B	Chamber B, t=19		6:46					
TCBF4A	Chamber A, t=23		10:12	t=23	0.6	1.32	16.7	7.96
TCBF4B	Chamber B, t=23		10:24					
08 August 2001 Trip								
TCBF1A	Chamber A, t=0	08-Aug	10:15	t=0	0.7	0.11	16.8	6.5
TCBF1B	Chamber B, t=0		10:35					
TCBF2A	Chamber A, t=2		11:18	t=2	0.75	0.13	19.9	6.53
TCBF2B	Chamber B, t=2		11:23					
TCBF3A	Chamber A, t=19		13:29	t=19	0.5	0.06	14.9	7.89
TCBF3B	Chamber B, t=19		13:25					
TCBF4A	Chamber A, t=23		16:34	t=23	0.6	1.32	16.7	7.96
TCBF4B	Chamber B, t=23		16:40					

Table B-14: Advective Diffusive fluxes of Rn-222 based on inventories and sediment equilibration experiments

Station ID	Rn-222 Bottom water (dpm/L)	Flux based on Rn-222 inventory (dpm/m2/day)	Diffusive Flux (dpm/m2/day)	Difference (dpm/m2/day)
21-22 April 2000 Trip				
Pipeline North	2.97 ± 0.24	592.11 ± 43.40	462.73 ± 18.25	129.38 ± 25.15
Pipeline South	3.02 ± 0.18	765.26 ± 31.96	392.96 ± 17.64	372.30 ± 14.31
Kenta Canal (W3)	8.18 ± 0.13	658.91 ± 37.43	1029.76 ± 25.97	-370.85 ± 11.46
Twin Canals	2.87 ± 0.23	467.84 ± 41.64	2686.83 ± 48.07	-2218.99 ± -6.44
31 July 2000 Trip				
Pipeline North	0.20 ± 0.15	21.58 ± 27.43	462.73 ± 18.25	-441.15 ± 9.18
Pipeline South	1.78 ± 0.16	614.34 ± 29.59	392.96 ± 17.64	221.38 ± 11.94
Kenta Canal (W3)	4.58 ± 0.43	497.57 ± 77.63	1029.76 ± 25.97	-532.19 ± 51.66
Twin Canals	4.20 ± 0.50	380.21 ± 90.32	2686.83 ± 48.07	-2306.62 ± 42.25
11 November 2000 Trip				
Pipeline North	3.00 ± 0.21	1629.98 ± 38.73	462.73 ± 18.25	1167.25 ± 20.48
Pipeline South	2.96 ± 0.23	1073.81 ± 41.60	392.96 ± 17.64	680.85 ± 23.95
Kenta Canal (W3)	4.40 ± 0.60	3427.37 ± 109.42	1029.76 ± 25.97	2397.61 ± 83.45
Twin Canals	3.10 ± 0.15	505.81 ± 26.46	2686.83 ± 48.07	-2181.03 ± -21.61

Table B-14 continued

Station ID	Rn-222 Bottom water (dpm/L)	Flux based on Rn-222 inventory (dpm/m2/day)	Diffusive Flux (dpm/m2/day)	Difference (dpm/m2/day)
30 March 2001 Trip				
Pipeline North	9.77 ± 0.18	2302.28 ± 32.57	462.73 ± 18.25	1839.56 ± 14.32
Pipeline South	5.17 ± 0.29	1405.92 ± 52.09	392.96 ± 17.64	1012.96 ± 34.45
Kenta Canal (W3)	5.08 ± 0.19	7142.82 ± 153.04	1029.76 ± 25.97	6113.05 ± 127.07
Twin Canals	9.33 ± 0.96	3549.00 ± 173.79	2686.83 ± 48.07	862.16 ± 125.72
13-14 July 2001 Trip				
Pipeline North	10.00 ± 2.85	1448.92 ± 517.25	462.73 ± 18.25	986.19 ± 498.99
Pipeline South	7.42 ± 1.38	1479.30 ± 250.49	392.96 ± 17.64	1086.34 ± 232.84
Kenta Canal (W3)	3.16 ± 0.20	7237.41 ± 37.76	1029.76 ± 25.97	6207.65 ± 11.79
Twin Canals	24.52 ± 4.20	8884.82 ± 760.08	2686.83 ± 48.07	6197.99 ± 712.01

Vita

Lorna Inniss was born in Barbados, West Indies, in 1964. She graduated from the University of the West Indies, Cave Hill Campus, in 1989 with a Bachelor of Science degree in biology with honors. She further completed a postgraduate diploma in management studies in 1991 at the same university.

Miss Inniss joined the team at the Coastal Conservation Project Unit in 1991 as water quality analyst, and was promoted to marine biologist in 1994. In 1996, she was awarded a Fulbright scholarship to read for a Master of Science degree in environmental planning and management at Louisiana State University. In 1998, she was also awarded a fellowship from the Organization of American States, along with a tuition award from the university.

In 1999, Miss Inniss graduated with her master's degree and began a doctoral program in oceanography and coastal sciences with funding from the Louisiana Board of Regents and the Department of Oceanography. She will be graduating in May 2002. Meanwhile, she has returned to Barbados to assume the post of deputy director of the Coastal Zone Management Unit, a Government Agency.