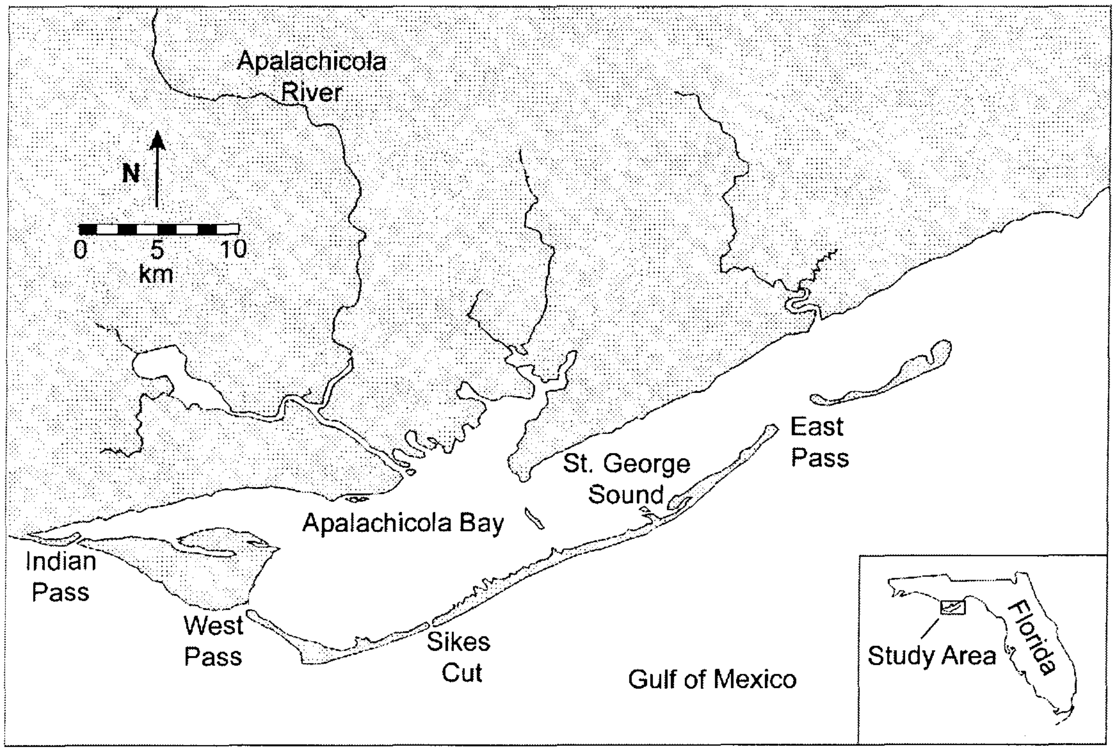
GROUNDWATER AND NUTRIENT DYNAMICS ON A STRIP BARRIER ISLAND SERVED BY ON-SITE SEWAGE TREATMENT AND DISPOSAL SYSTEMS IN THE NORTHEASTERN GULF OF MEXICO



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EXECUTIVE SUMMARY

Groundwater flow on St. George Island, a barrier island in the northeast Gulf of Mexico, was monitored downfield from wastewater systems using artificial tracer techniques. Sulfur hexafluoride and fluorescein dye were used to determine groundwater flow velocity, hydraulic conductivity, and dispersivity at different sites on the island. Monthly monitoring of hydraulic head illustrated the aquifers' dependence on rainfall. However, during periods of little rain, tidal stage also influenced the direction and magnitude of groundwater flow. Estimated hydraulic conductivities ranged from 2.7 to 180 m day^{-1}, with an overall estimate of 36 m day^{-1}. The tracers showed very little dilution and estimated longitudinal dispersivities were approximately 0.1 to 0.5 meters.

The total groundwater flux into an adjacent bay was also evaluated using two independent techniques. Using Darcy's law and an assumed cross sectional area allowed us to use the experimental horizontal transport rates to estimate the volumetric flow. In addition, we used a simple water balance calculation, which accounted for all the sources and sinks of water to the aquifer. The two independent approaches agreed very well, providing an estimated groundwater flux to Apalachicola Bay between 1-8 X 10^6 m^3 yr^{-1}.

In addition, groundwater nutrient concentrations on St. George Island were monitored downfield from three wastewater systems. Background groundwater nutrient concentrations were obtained from an adjacent uninhabited island. Silicate, which was significantly higher in the drinking water relative to the surficial aquifer on St. George Island, was used as a natural conservative tracer, providing insight to the extent of the wastewater plume. Total nitrogen, ammonia, nitrate, nitrite, soluble reactive phosphate

and total phosphate were also measured in all groundwater and surface water samples collected.

Nutrient concentrations were all attenuated rapidly relative to silicate, indicating very little transport of these nutrients to surface waters. Results indicate that the most efficient onsite disposal system would be an aerobic system that was raised approximately 1-m above the natural land elevation. This raised bed provides additional time and material for the wastewater to filter through before reaching the relatively high water table of the island.

Estimates of the total nitrogen and total phosphate flux into Apalachicola Bay from groundwater originating on St. George Island ranged between 0.1-4.3 mmoles m^{-2} yr^{-1} and 0.03-0.8 mmoles m^{-2} yr^{-1}, respectively. This is a conservative estimate since the sites monitored efficiently removed the nutrients before discharge into surface waters. Wastewater systems installed closer to surface waters than the systems monitored in this study, as the law allows, may provide substantially more nutrients to surface waters.

CHAPTER 1

TRACING GROUNDWATER FLOW ON A BARRIER ISLAND IN THE NORTHEAST GULF OF MEXICO

Introduction

Submarine groundwater discharge (SGD) in the coastal zone has been documented as a significant source of the nutrient supply to surface waters (Valiela et al., 1978; Valiela and Teal, 1979; Capone and Bautista, 1985; Lapointe and O'Connell, 1989; Capone and Slater, 1990; Valiela et al., 1990; Corbett et al., 1999). The importance of groundwater discharge to the coastal environment is dependent on several variables, including the amount and type of nutrient enrichment in the groundwater, water column circulation, tidal flushing, porosity and permeability of the underlying strata, hydraulic head, and thus the corresponding groundwater flow. In areas with highly permeable soils, i.e. shallow coastal aquifers, groundwater may be a major pathway for the transport of anthropogenic contaminants, including nutrients, metals, and bacteria (Johannes, 1980; Weiskel and Howes, 1992). Coastal groundwaters contaminated with nutrients from on-site sewage treatment and disposal systems (OSTDS) have become more common and may promote eutrophication of nearshore ecosystems (Johannes, 1980; Nixon, 1986; Nixon, 1992).

Apalachicola Bay is a highly productive bar built estuary in the northeast Gulf of Mexico. St. George Island, like most barrier islands, forms the outer perimeter of the estuary and is critical to the bay's productivity because its orientation determines the salinity distribution as well as other water quality features of the bay. The estuary is an economic resource in north Florida, providing more than 90% of Florida's oyster landings and the third highest catch of shrimp in the state (Livingston, 1984). Although the Apalachicola Bay is the major source of freshwater and nutrients to the estuary, increased population density on St. George Island has raised concerns of potential changes in local water quality and impacts on the oyster industry. Barrier islands throughout the U.S. are experiencing similar increases in permanent and seasonal population. Many estuaries are sensitive to slight physical, chemical, and biological perturbations. Although barrier islands play a critical role in this balance, very little is known about the groundwater dynamics and potential impacts of contaminated groundwaters from these islands. A better understanding of groundwater flow along these islands can help establish responsible comprehensive plans for local development and resource management.

We have employed conservative groundwater tracers (fluorescein and sulfur hexafluoride) to evaluate the subsurface flow directions, velocity, and aquifer characteristics on St. George Island (**Fig. 1.1**). Multi-level samplers (MLS) and 5 cm PVC monitoring wells were placed down-gradient from selected wastewater systems at three locations on the island. The wells were monitored over the course of the study for tracer concentrations along the flow path toward surface waters, as well as mapping of the piezometric surface at each location. In addition, the amount of groundwater entering the bay was estimated from two independent techniques, utilizing the tracer data and a simple box model accounting for all sources and sinks of the groundwater.

St. George Island

St. George Island is a microtidal barrier island in the panhandle of Florida. The island is approximately 48 km long and averages less than 0.5 km in width. Dr. Julian G. Bruce State Park occupies the east end of the island. The climate in the region is mild, with a mean annual temperature of approximately 20 °C (Livingston, 1984). The mean annual rainfall over the area, recorded over the last 42 years by the NOAA weather station in Apalachicola, is approximately 140 cm. The tidal range in the bay and Gulf is less than 0.5 m.

The surficial aquifer on the island is composed of medium to fine sand grains overlying a silty clay impermeable barrier between 7.6 and 9.2 m below the surface that forms a base to the aquifer (Livingston, 1984). Water in the shallow freshwater lens is primarily derived from rainfall, and eventually discharges into Apalachicola Bay or the Gulf of Mexico. The impermeable clay layer separates rain-derived freshwater from the surrounding salt water. St.

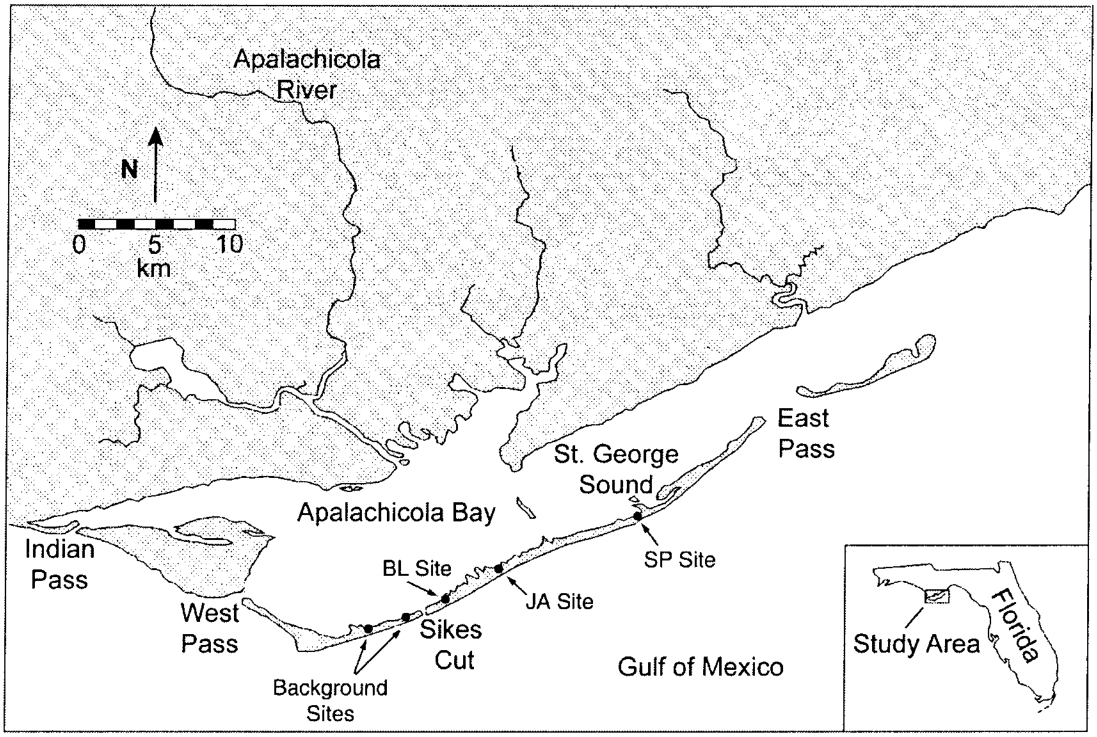


Figure 1.1: St. George Island is a barrier island located in the northeastern Gulf of Mexico. Samples were collected from three different experimental sites located on St. George Island and two control sites located on Little St. George Island, across from Sikes Cut.

George Island has a characteristically high water table, which increases the probability of groundwater contamination and transport to the surrounding marine waters.

Study Site

Experimental sites were selected according to location on the island, proximity to Apalachicola Bay, type of OSTDS, and the amount of time the residence is occupied. Care was given to locate sites adjacent to the bay with a similar beachfront, i.e., no canals, natural topographic gradients, etc. Three sites were chosen including a site located within the Dr. Julian G. Bruce State Park (SP Site), on the far eastern end of the island, a private residence near Bob Sikes Cut (BL Site) on the extreme western end of the island, and another private residence approximately 3 km west of the causeway (JA Site), between the other two locations. Each site had an average of 3 residents and was occupied year round.

SP Site. The experimental site within the state park was developed between August and October 1997. Wells were installed primarily down-gradient from the septic system, and consist of twelve 5-cm monitoring wells and thirteen multi-level samplers covering just under 8000 m^2 (**Fig. 1.2**). The wastewater system is set back approximately 100 m from Apalachicola Bay.

BL Site. B.L. Cosey's property is also approximately 8000 m^2 with 60 m of bayfront access (**Fig. 1.3**). The property is generally level and approximately 0.6 m above mean sea level. The septic system and drainfield is raised above ground level approximately 1 m to prevent inundation by surface waters during large storms and rain events. Wells were installed on either side of the drainfield and towards the bay waters during March, 1998. Samples were collected from this site until March, 1999, when the wells were removed due to construction on the property.

JA Site. Jay Abbott's property is located adjacent to Apalachicola Bay with approximately 45 m of bayfront access. The site is just under 6000 m^2 and has an aerobic wastewater system, installed in 1996. Due to the location of the house with respect to the OSTDS, since it is inexpensive, easily detectable, non-toxic, and stable over time (Smart and

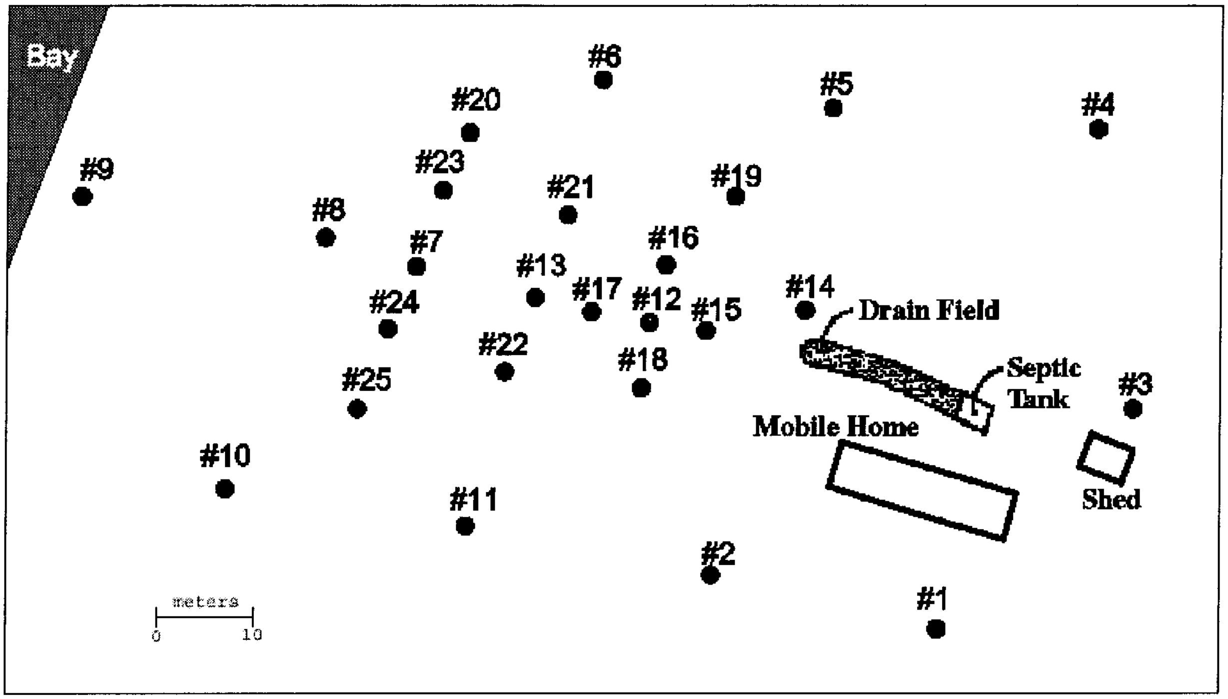


Figure 1.2: The SP study site is located the furthest east of the three sites on the island and is within the Dr. Julian G. Bruce State Park. The SP site contains twelve 5-cm monitoring wells (#1-12) and thirteen multi-level samples (#13-25).

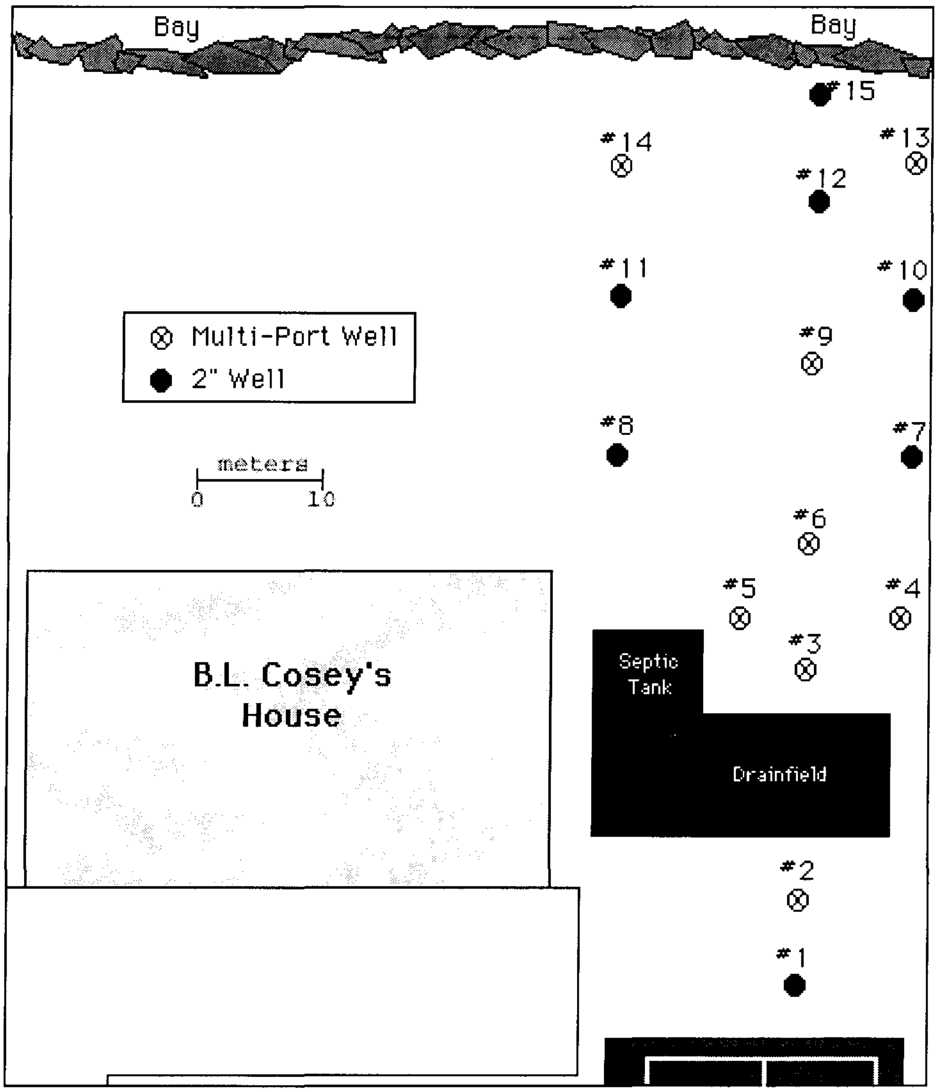


Figure 1.3: The BL study site is located furthest west on the island and is on B.L. Cosey's property. The BL site contains seven 5-cm monitoring wells (solid circle) and eight multi-level samples (x-circle).

wells were not positioned in a grid-type pattern as at the other sites. Wells were placed as close to the OSTDS and around the residence as possible. Fourteen wells including seven monitoring wells and seven multi-level samplers were installed at the site during March, 1998 (**Fig. 1.4**). Samples were collected from this site until December, 1998, when the site had to be abandoned.

Methods

Well Installation

Multi-level samplers (MLS) and 5 cm PVC monitoring wells were installed at all sites to an approximate depth of 3 meters. Wells were installed using a hand auger with a 7.5 cm hollow barrel. To prevent the hole from back-filling during construction, a 10 cm PVC casing (outer-casing) was inserted into the hole and moved downward as the hole was dug deeper. Once the desired depth was reached (~3 m), the well was inserted into the hole, contained by the outer-casing. The outer-casing was then removed from the hole, allowing the aquifer materials to collapse around the sampler, isolating sampling points of the MSL at each level in the borehole. Additional soil material, originally removed from the hole, was backfilled to complete the well as necessary. Wells were typically cut flush to the ground and covered with a removable 15 cm plastic cover. Broward Davis & Assoc., Inc surveyed the elevation of all wells relative to mean sea level.

Multi-level Samplers. Wells that use a slotted interval for sampling tend to provide integrated samples that are a mixture of different zones within the screened interval (Pickens et al., 1978). Nesting wells or piezometers with short screens can be used to obtain samples from different depths, however this approach requires many boreholes and additional expense. The construction of a multi-level sampler allows for sampling of groundwater at closely spaced intervals in a vertical direction from a single borehole. The MLS device used in this study is a slight modification of wells used in previous groundwater studies (Pickens et al., 1978; Boggs et al., 1988; LeBlanc et al., 1991).

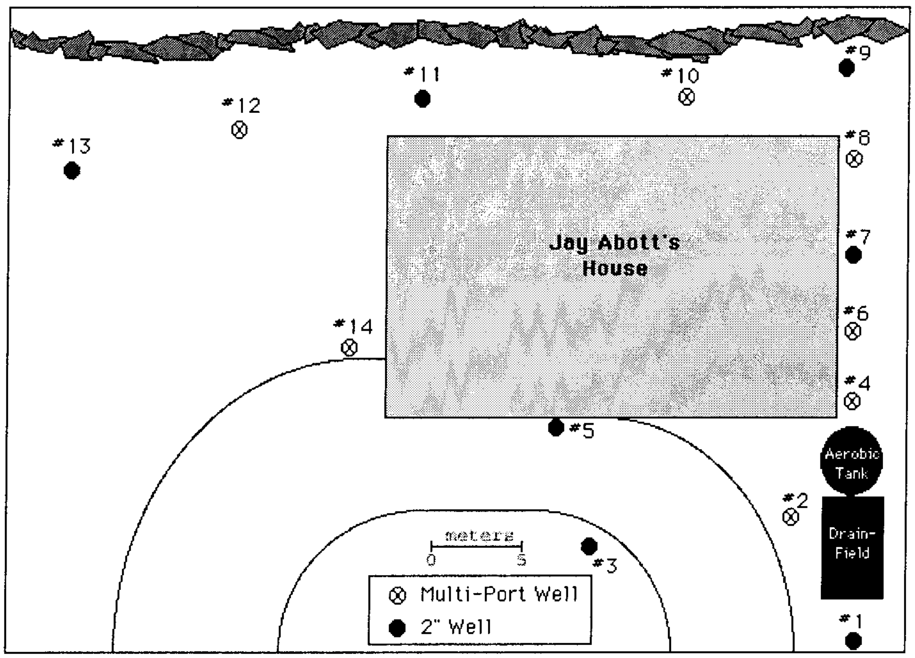


Figure 1.4: The JA site contains seven 5-cm monitoring wells (solid circle) and seven multi-level samples (x-circle).

MLS devices were constructed using 1.9 cm OD PVC pipe and 0.6 cm OD polypropylene tubing (**Fig. 1.5**). The PVC pipe acts as a housing for the smaller polypropylene sample tubing. A 3 meter section of PVC pipe is first cut and one end (bottom) is capped. For this study, seven polypropylene tubes were attached to the outside of the PVC pipe by plastic cable ties or other fastening devices. The ends of the sampling tubes attached to the pipe were wrapped twice with 202 μm Nytex mesh and spaced 30 cm apart. Enough tubing was left above the PVC pipe for easy access and sampling (~0.5 m). Upon installation of the well, the PVC pipe was filled with material removed from the borehole and then capped. Sample depths were identified at the top of each piece of tubing. Samples were collected directly from individual sample depths using a peristaltic pump.

Monitor Wells. Wells constructed of 5 cm diameter PVC were installed at the experimental site for monitoring nutrient and tracer concentrations, as well as water table height. The wells were installed to approximately 3 m below ground-level. The last 0.4 m of the well consisted of slotted (254 μm) PVC screen. Water table height was monitored monthly using a Solinst model 101 water level meter. Samples were collected using a peristaltic pump after purging three well volumes.

Hydrologic Tracers

One of the most unequivocal ways to ascertain the rates, pathways through a hydrological system, hydraulic properties of an aquifer, and to link specific sites of contamination to discharge points is *via* artificial tracers. Ideally, tracers should have predictable properties, both intrinsically and in their interaction with the system into which they are introduced, i.e., tracers should be chemically stable, conservative, inexpensive, readily available, and easily detected. The large volume of most hydrological systems means that tracers need to be detectable at low concentrations and have a low natural background concentration.

Fluorescein. Sodium fluorescein ($C_{20}H_{10}O_5Na_2$), referred to as fluorescein dye or simply fluorescein, is an inexpensive highly water soluble fluorescent dye (Quinlan, 1989). Fluorescein

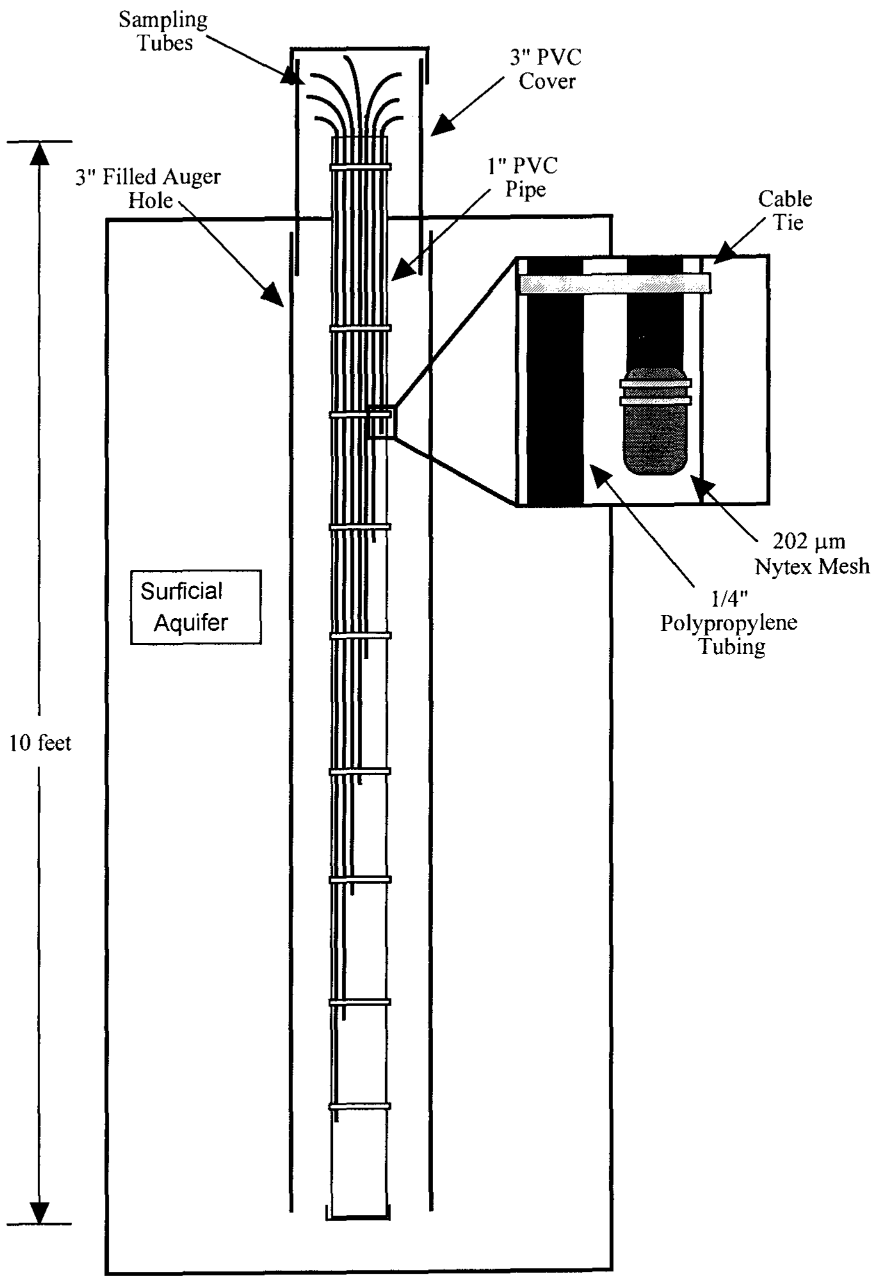


Figure 1.5: This diagram shows the design of multi-level sampler used throughout this study. The center casing is capped on both ends to prevent migration of water through the opening. Sample tubes are held in place with plastic cable ties. The Nytex mesh was wrapped around the sample tubes at least twice then fastened with two small plastic cable ties.

is bright yellow-green to the eye and has a maximum excitation of 491 nm and maximum emission of 513 nm (Gaspar 1987). Many groundwater tracing studies have employed this dye Laidlaw, 1977; Smart, 1984; Duley, 1987; Quinlan, 1989; Reich, 1993). However, the dye will break down if exposed to direct sunlight.

Samples were collected in 125 mL amber polycarbonate containers with a peristaltic pump. Wells were initially purged of three well volumes before sample collection. Samples were returned to the laboratory and analyzed using a Turner Designs 10-AU-005 Fluorometer, which provides exact concentrations after calibration. The fluorometer was set up using the 10-045 daylight white lamp, 10-104R-C Combination Round Excitation Filter, 455-500 nm, 10-109R-C Combination Round Emission Filter, 510-700 nm, and the 10-063 Square Reference Filter as specified by manufacturer. The fluorometer was initially calibrated using fluorescein standards made in the laboratory.

Sulfur Hexafluoride. SF_6 was employed as a conservative tracer for subsequent experiments throughout the study period. Sulfur hexafluoride (SF_6) is an inert water-soluble gas, is biologically and chemically inert, has a low background concentration (10^{-15} mol/L), and can be detected at extremely low concentrations (10^{-16} mol; Wannikhof et al., 1985). SF_6 has been used for gas exchange studies in rivers (Clark et al., 1994) and lakes (Wannikhof et al., 1987; MacIntyre et al., 1995; Upstill-Goddard et al., 1990) as well as applications in atmospheric and oceanic sciences (Brown et al., 1986; Upstill-Goddard et al., 1991; Watson et al., 1991; Ledwell et al., 1993). The strong potential of SF_6 as a geothermal and groundwater tracer has recently been reported (Upstill-Goddard and Wilkins, 1995; Wilson and Mackay, 1993). In those studies, SF_6 compared favorably with fluorescein dye applied in a 7.5×10^{-7} mass ratio of SF_6 to sodium fluorescein.

Sulfur hexafluoride samples were collected in 30-mL serum vials with a peristaltic pump and extracted into a small headspace just before analyses. After purging the well, a sample was pumped into a serum vial and allowed to overflow for three bottle volumes. The vial was then sealed with a rubber septum and a crimp cap. To prevent loss of SF_6 through the septa, the samples were stored on their sides until the samples could be extracted and analyzed. Samples were extracted in the lab by adding a small headspace (typically 4 mL) of argon or ultra-high purity nitrogen to the sample. Simultaneously, a volume of water from the sample had to be removed and discarded to allow for the headspace. The serum vials were slightly over-pressurized with 1 mL of nitrogen to allow several withdrawals for analysis (100 μL or less) by the gas chromatograph (GC).

SF_6 samples were analyzed with a Shimadzu model 8A gas chromatograph equipped with an electron capture detector. Typically, the volume injected was 100 μL or less. The gas chromatograph contained a stainless steel column (180 cm x 0.1 cm I.D.) packed with a molecular sieve 5A (80/100 mesh). Initially, a P5 mixture (95% argon, 5% methane) was used as a carrier gas with a flow rate of 25 mL min-1. After having problems with carrier gas contamination, we switched to ultra-high purity nitrogen as a carrier at the same flow rate. Column and detector temperatures were set at 90°C and 220°C, respectively.

Headspace concentrations in ppmv (parts per million by volume, μL/L) of SF_6 were determined by reference to a 1.04 ppm standard (Scott Specialty Gases). Headspace concentrations were converted to dissolved concentrations in μM using the ideal gas-equation. Replicates were collected for 10% of the samples. In addition, duplicate injections were run on the gas chromatograph every fifth injection. Precision between replicate samples and duplicate injections were usually better than 10%.

Results and Discussion

St. George Island Hydrology

The water level relative to mean sea level (MSL) was monitored monthly throughout the study period in the 5 cm PVC monitoring wells at all the experimental sites. These measurements provide a snap shot of the piezometric surface (**Fig. 1.6**) and demonstrate how the

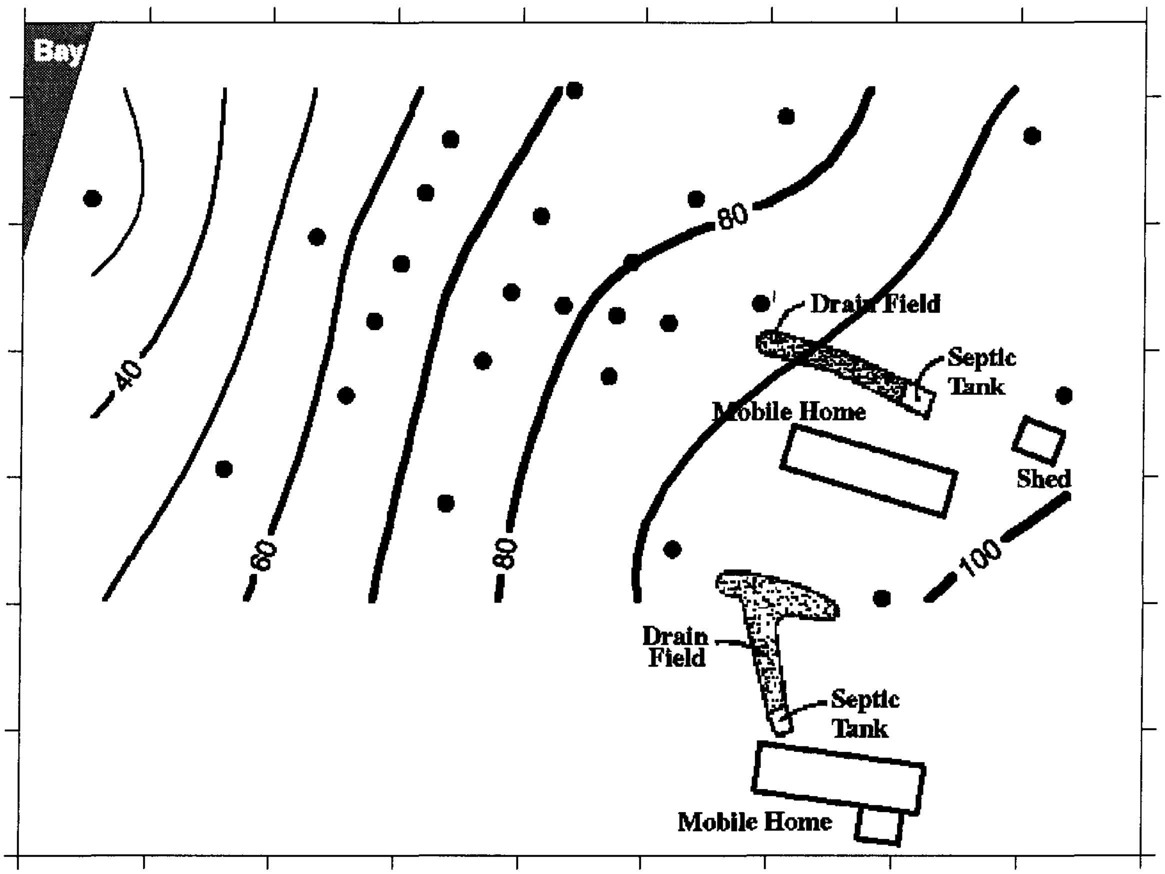


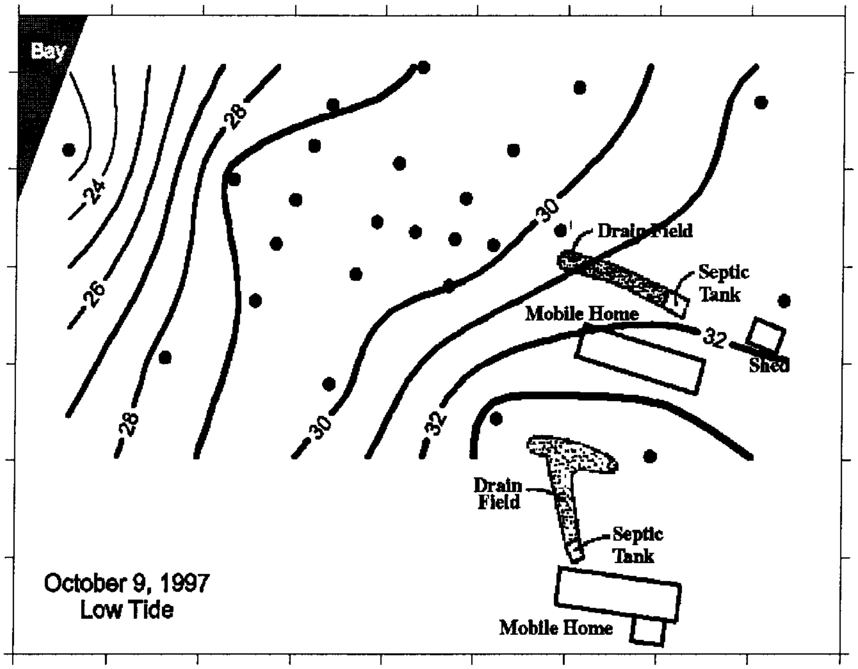
Figure 1.6: The piezometric surface (cm) measured at the SP site during January 1998. The groundwater flow direction may be inferred as perpendicular to the lines of equal hydraulic head. Head measurements are reported as centimeters above mean sea level.

hydraulic gradient changes over time. The hydraulic gradient was calculated using the two inland-most wells to reduce the tidal influence at each site. During periods of little rainfall, groundwater flow near the shoreline (within 15-20 m) is primarily influenced by tidal stage. In fact, groundwater flow can change directions during a 12-hour period due to surface water levels (**Fig. 1.7**). Piezometric surface maps from BL and JA sites during the early summer (June-July), 1998 suggested that the predominant groundwater flow direction near the wastewater systems was toward the interior of the island rather than the bay. It is evident then that the hydraulic gradient, and thus groundwater flow, is greatly dependent on rainfall. In fact, the similarities between rainfall and hydraulic gradient can easily be seen at all three sites (**Fig. 1.8**), illustrating the importance of large rain events on the island hydrology and contaminant transport.

During October 1997, the 5 cm PVC monitoring wells at the SP site were monitored over a twenty-four hour period to evaluate water level changes over a tidal cycle (**Fig. 1.9**). The water level fluctuations in a well in response to changes in sea level may be used to determine certain aquifer characteristics (Carr and Van Der Kamp, 1969). Changes in the water level of a well in response to tidal loading occur as a result of mechanical loading of the aquifer at the oceanic extension, propagation and attenuation of the pressure wave created by the loading, and flow of groundwater from the aquifer to the borehole (Enright 1990). For example, as sea level increases, the aquifer bears a greater load creating a pressure gradient in the immediate vicinity of the loading. This pressure gradient (wave) is propagated inland and is attenuated since the matrix is bearing a portion of the load (**Fig. 1.10**). Ferris (1963) showed the relationship between the time delay that occurs between the tidal high or low and the maximum or minimum water level observed in an observation well at some distance from the tidal source and an aquifer's transmissivity as:

$$T = \frac{x^2 S t_0}{4\pi t^2} \qquad (1)$$

A.



B.

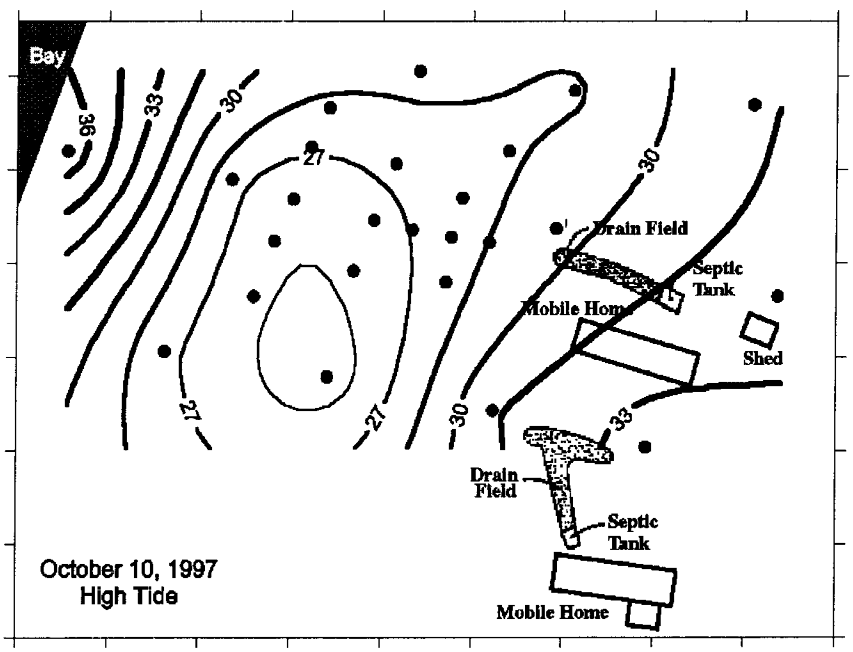


Figure 1.7: The piezometric surface map (cm) during low tide (**A**) and high tide (**B**) shows a shift in groundwater flow near surface waters at the SP site. During periods of little rainfall, the tide controls groundwater flow within at least 50 meters from the shoreline.

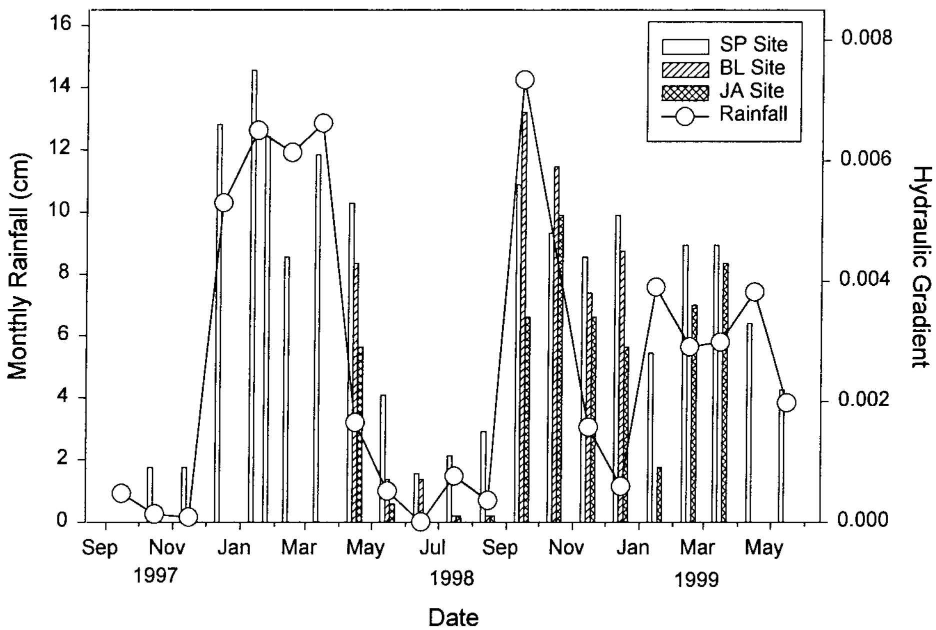


Figure 1.8: Hydraulic gradient measured at the three experimental sites relative to rainfall. The Northwest Florida Water Management District (NWFWMD) measures rainfall on the causeway leading to the island. Hydraulic gradients were measured in wells furthest from surface waters to avoid tidal influence.

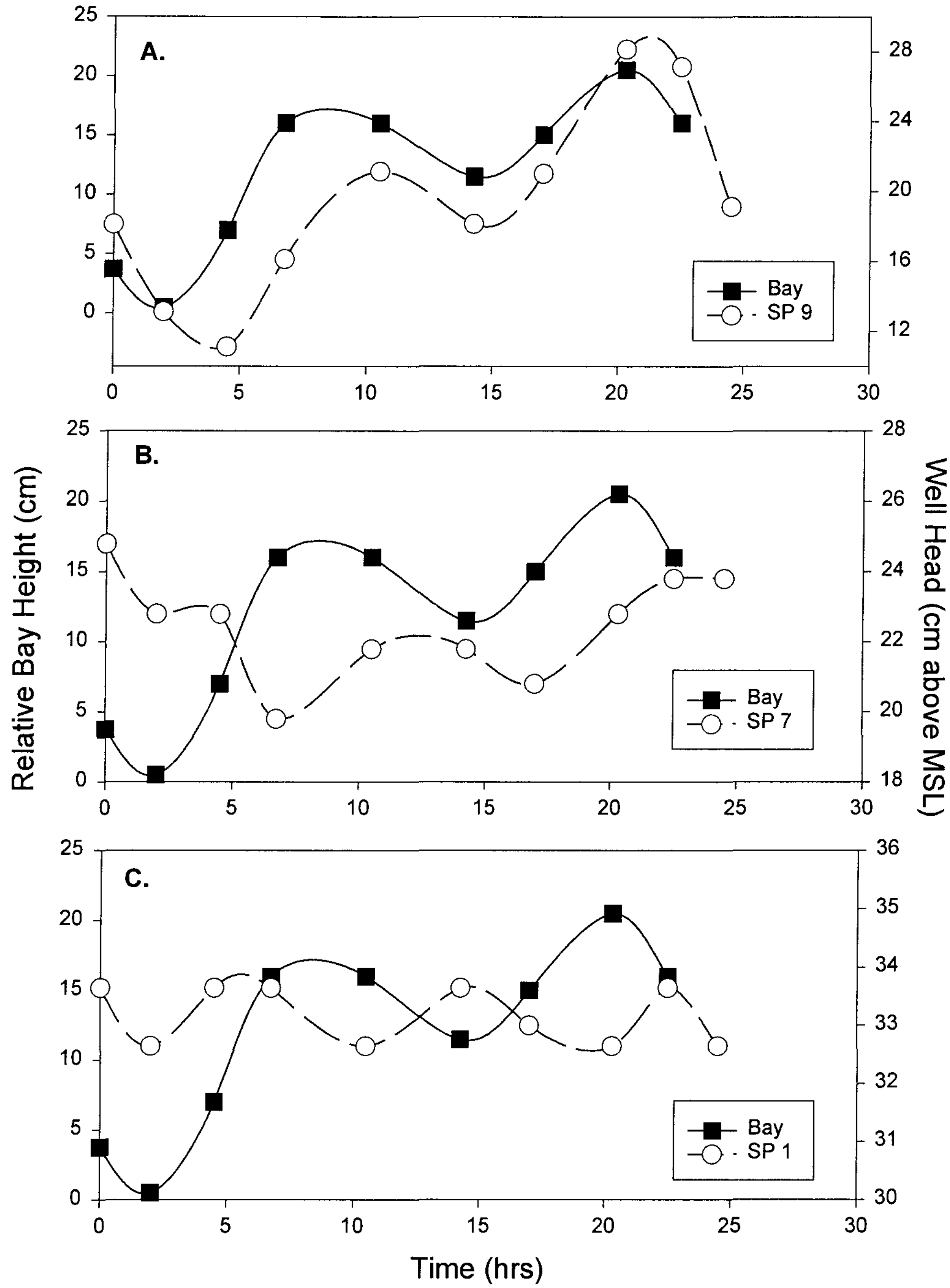


Figure 1.9: The hydraulic head of three monitoring wells (open circles) relative to tidal changes (closed square) shows the attenuation of the tidal signal as it propagates inland. Well SP-9 (**A**) is located 9 meters from surface waters. Well SP-7 (**B**) is located 39 meters from surface waters. Well SP-1 (**C**) is located 97 meters from surface waters. Note the difference in scales of the well head as the tidal signal propagates inland.

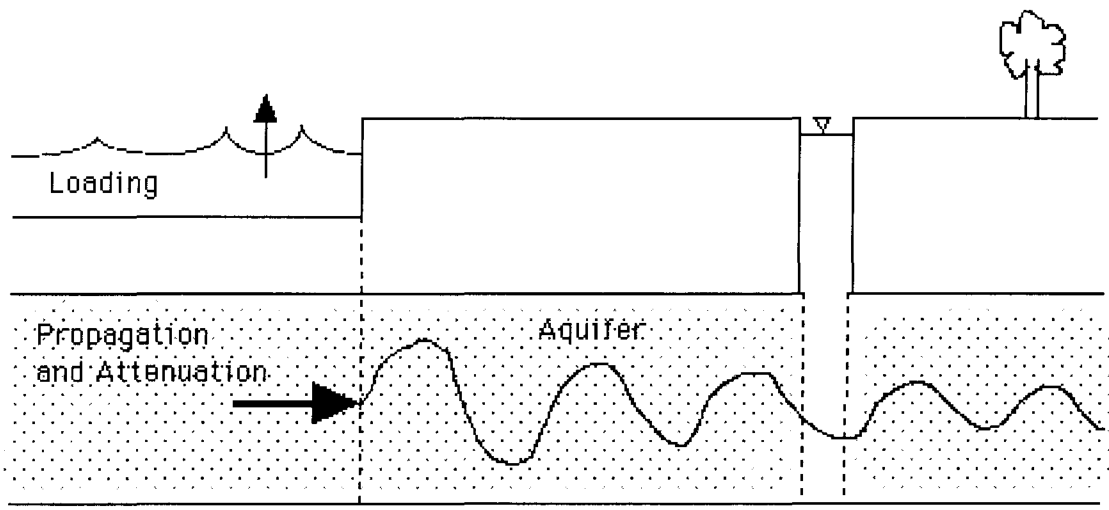


Figure 1.10: Propagation and attenuation of pressure wave created by an increase in sea level. The response in a well will depend on the linear distance of the well from the area that was loaded. Diagram modified from Enright (1990).

where T is the transmissivity (m^2 day^{-1}), x is the distance from the tidal source (m), S is the specific yield of the aquifer (dimensionless), t_o is the period of the tidal signal (day), and t is the time delay to successive maxima or minima between the surface water body and the wells (day). The transmissivity (T) is the rate of groundwater flow through a vertical strip of aquifer one unit wide, extending the full saturated thickness of the aquifer, under a unit hydraulic head and is related to hydraulic conductivity by:

$$T = Kb \qquad (2)$$

where K is the hydraulic conductivity (m day^{-1}) and b is the saturated aquifer thickness (m). The specific yield is the ratio of the volume of water that drains by gravity to the total volume of aquifer. Domenico and Schwartz (1990) suggest that the specific yield for fine to coarse grain sands ranges from 0.23 to 0.28. I have assumed a conservative value of 0.20 for specific storage on all calculations. The thickness of the surficial aquifer used in these calculations is 8 meters, based on data from boreholes (Livingston, 1984) and geophysical data (Ruppel, pers. comm.).

The lag time of the well response and the tidal attenuation increases with increasing distance from shore. Three wells (#1, #7, #9), which provided a transect from the shoreline to the residence (**Fig. 1.2**), were used to calculate a hydraulic conductivity based on the methodology given above. Results suggest that the hydraulic conductivity varies from 20 – 180 m day^{-1} (**Table 1.1**). The largest (180 m day^{-1}) and the smallest (20 m day^{-1}) conductivity were observed in the wells furthest inland and shoreward, respectively. Schultz and Ruppel (1999) observed a similar trend of aquifer heterogeneity on a barrier island off the coast of Georgia. Results from their tidal experiments and grain size analyses suggested that the hydraulic conductivity ranged from 0.07 m day^{-1} nearest the surface water body to 12 m day^{-1}, approximately 35 meters inland. Millham and Howes (1995) reported similar results from their study and earlier studies of coastal aquifers. They attributed the lower hydraulic conductivity

Table 1.1. Tidal experiment at the SP site conducted in October 1997.

Well #	Distance from Tidal Signal (m)	Lag Time (day)	Hydraulic Conductivity (m day^{-1})
Well #1	97	0.20 ± 0.06	180 ± 100
Well #7	39	0.13 ± 0.04	70 ± 40
Well #9	9	0.06 ± 0.02	20 ± 10

along the shoreline to increased organic material in the sediment. The organic material is several times less dense than the inorganic and occurs in smaller particle sizes, which could occupy pore spaces in the aquifer matrix. Unfortunately, sediments from St. George Island were not analyzed for organic content, although a similar trend is likely.

Tracer studies have also provided information about the hydrologic characteristics of all three experimental sites as well as providing groundwater flow paths and velocities. Two tracer experiments were conducted at the SP site, while only one experiment was conducted at each of the other two sites (**Table 1.2**). In each case, the tracer was dissolved in 190 liters of tap water from the site. The tracer was then injected directly into a 5 cm PVC monitoring well (fluorescein) or directly into the drainfield (SF_6), bypassing the wastewater tank. The tracer was then followed by approximately 100 liters of tap water, acting as a "chaser." Wells were monitored at least monthly downfield from the injection sites over extended time periods (8-18 months).

The first experiment was conducted using fluorescein dye at the SP site. The dye was initially dissolved in approximately 190 L of tap water and injected on October 6, 1997 into SP-12, a 5 cm PVC monitoring well near the middle of the site. Wells were then monitored over an 18 month period. The tracer was only detected in two of the wells on the site, indicating very little vertical or horizontal dispersion of the plume. The tracer was first recorded in SP-17 after 30 days (**Fig. 1.11**) at a depth of approximately 45 cm below mean sea level (MSL), about the same depth the tracer was injected into the aquifer. The tracer reached its peak concentration (1660 $\mu g\ L^{-1}$) after 41 days (**Table 1.3**), diluted by 3 orders of magnitude. Based on these observations, the horizontal transport rate calculated for the tracer to this well is 0.11 m day^{-1}. However, when this tracer was initially injected into the subsurface there was very little hydraulic head (gradient <0.001) driving the water (**Fig. 1.8**). Soon after the experiment commenced, the island received enough rain to increase the hydraulic gradient by more than 5 times (0.0052) the initial gradient. If the horizontal transport rate is calculated from the time of

Table 1.2. Summary of tracer experiments conducted on St. George Island.

Experimental Site	Tracer Employed[1]	Date Injected	Injection Concentration[2]
SP Site	Fluorescein	06-Oct-97	1.6×10^6 μg L^{-1}
	SF_6	23-Jun-98	1250 ± 270 nM
JA Site	SF_6	23-Jun-98	1360 ± 400 nM
BL Site	SF_6	23-Jun-98	1560 ± 190 nM

[1]Tracers were injected directly into the wastewater drainfield, bypassing the wastewater tank, except for the fluorescein experiment. Fluorescein was injected into well SP-12.
[2]The volume of injection was 189 liters during each experiment.

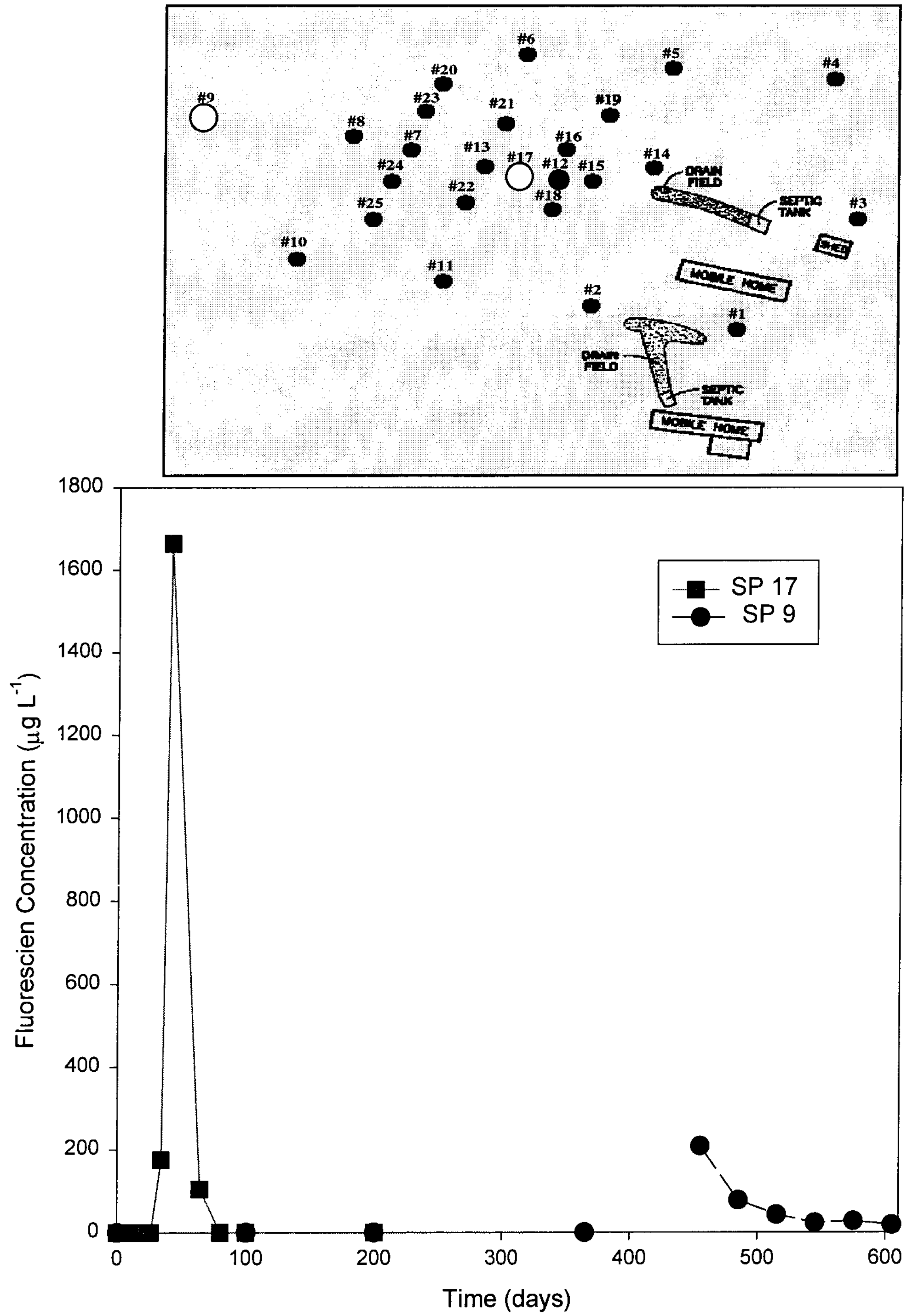


Figure 1.11: Results of tracer experiment in which fluorescein was injected into SP-12 and wells downgradient were sampled over approximately 18 months. The tracer only appeared above detection limits in two wells, SP-17 and SP-9. Since the peak concentration in well SP-9 may have occurred between sampling events, the points were not connected.

Table 1.3. Times of peak tracer concentrations and horizontal transport rates (HTR) for each sampling location at the SP site during both experiments.

Tracer Experiment	Sample Location	Distance from Injection (m)	Time (days)	Max. Tracer Concentration[1]	HTR[2] (m day^{-1})
Fluorescein	SP-17a	4.5	41	1660	0.11
06-Oct-97	SP-17b[3]	4.5	12	1660	0.40
	SP-9[4]	55	410-450	210	0.12-0.13
SF6	SP-12	31	73	640	0.42
23-Jun-98	SP-21	54	210	830	0.26
	SP-21b	23	137	830	0.17
	SP-23a	67	290	210	0.23
	SP-23b	13	80	210	0.16
SF6	JA-2	3	175	240	0.02
23-Jun-98	JA-4a	7	230	453	0.03
	JA-4b	4	55	453	0.07

[1]Concentrations of fluorescein are in μg L^{-1} and SF_6 are in nM.
[2]Horizontal transport rates were calculated for each sampling location, except SP-21b, SP-23b, and JA-4b, by dividing the distance from the site of injection by the time of the peak tracer concentration at that sampling location. SP-21b, SP-23b, and JA-4b were calculated using the distance between the upfield well and the difference in time of maximum concentration between the upfield well.
[3]SP-17b is based on the possibility of rain events influencing the groundwater transport.
[4]The time until the maximum concentration is presented as a range, since the actual peak was not measured.

the rain event (November 2, 1997), a horizontal transport rate of 0.40 m day^{-1} is estimated. It is unlikely that the tracer had moved a significant distance from the point of injection until this rain event occurred.

The tracer was not detected in any wells until it reached SP-9, even though there are other monitor wells in between the injection point and this well. This is probably due to very little vertical migration. Monitoring well SP-9 has a screened interval of 10-50 cm below MSL, approximately the same as the injection point. The wells sampled between SP-9 and SP-17 have a deeper screened interval (>50 cm below MSL), so the tracer could pass above the sampling interval without detection. By the time the fluorescein dye reached detectable concentrations in SP-9, samples were only collected monthly. The exact time of peak concentration is not known, since there was no detectable amount of fluorescein in the well one month (October 1998) followed by an easily detectable amount (210 $\mu g\ L^{-1}$) in the subsequent month (November 1998). Therefore, the horizontal transport rate is reported as a range (0.12-0.13 m day^{-1}), accounting for the possibility that the concentration may have peaked any time between the two sampling periods (**Table 1.3**). The dye was never detected in any of the MLS wells (SP-21, 22, 23, or 24) on either side of the center transect (SP-12 to SP-9).

Sulfur hexafluoride was injected into the drainfield at all three experimental sites in June 1998. The tracer was not detected in any wells at the BL site throughout the study period. This is believed to have occurred due to the design of the drainfield. The elevated drainfield increases the residence time of the wastewater, and thus the trace gas in the unsaturated zone. The tracer may have degassed before ever reaching the water table, decreasing the concentration below the detection level. The drainfields at the other two sites are not raised, reducing the amount of time the gas tracer may emanate from solution.

At the SP site, the tracer was found to be above detection levels in several wells, with the highest concentrations measured in SP-12, SP-21, and SP-23 (**Fig. 1.12**). The peak concentrations measured in wells SP-12 (640 nM) and SP-21 (830 nM) are diluted by less than a factor of two, approximately the same volume as the chaser injected into the drainfield

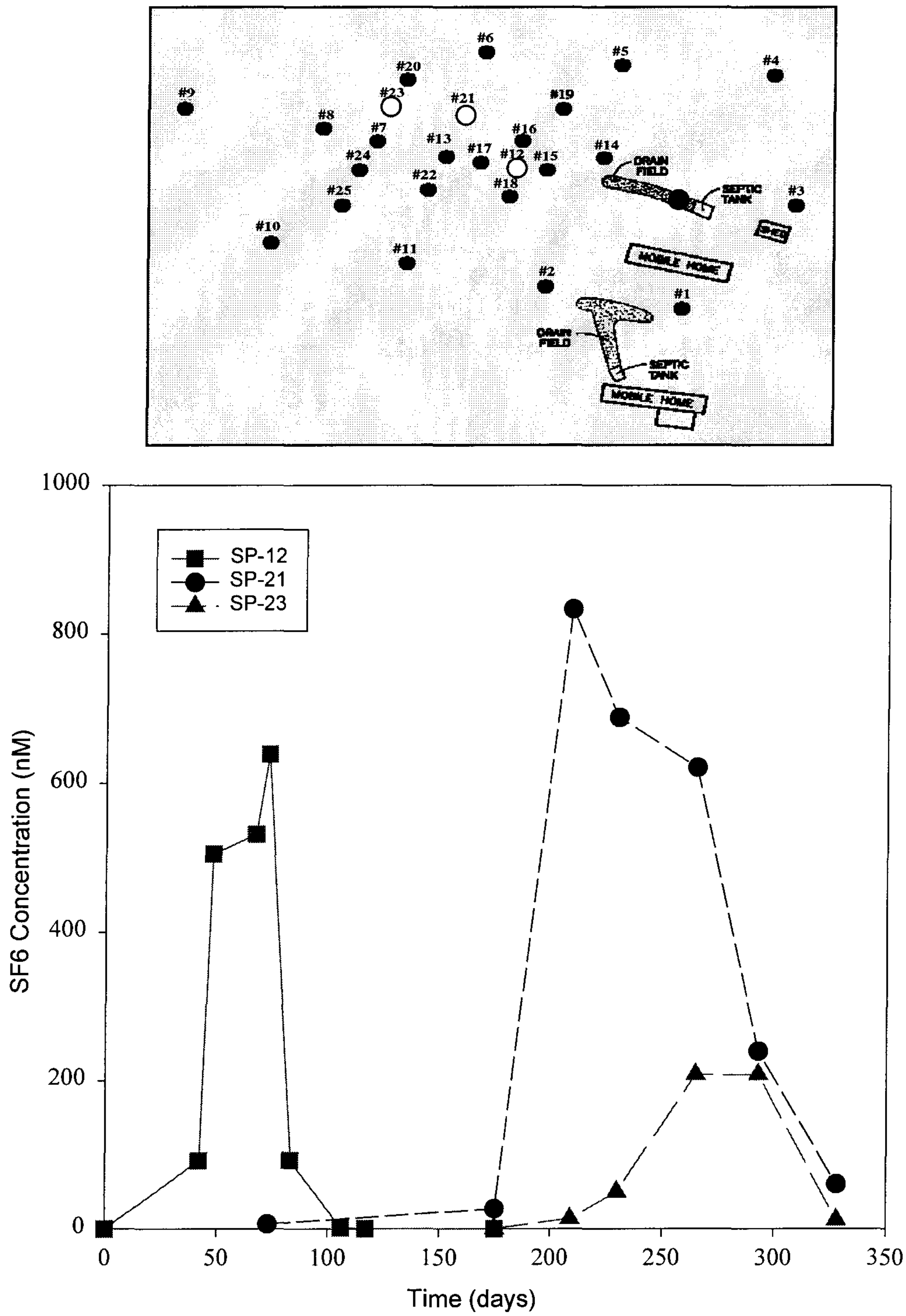


Figure 1.12: Sulfur hexafluoride tracer curves from the SP site. Well locations that had measurable SF_6 concentrations are shown on the site map (open circle). The tracer was injected directly into the drainfield, bypassing the septic system.

immediately following the tracer. The tracer concentration was only 210 nM at SP-23, which could be associated with dilution of the tracer or may indicate that the center of mass of the plume did not move through the area of the well. In either case, there was not a significant amount of dilution of the tracer over a long period of time (>290 days). Horizontal groundwater velocities calculated for each well range from 0.16 to 0.42 m day^{-1} (**Table 1.3**). Two estimates of velocity were calculated for wells SP-21 and SP-23. The higher value corresponds to the average over the entire experiment, including periods of increased flow, i.e., heavy rainfall. The slower rate is indicative of the groundwater movement between two different wells. For instance, an average velocity for SP-21 was calculated to be 0.26 m day^{-1} over the course of the experiment. However, between the period of time that the maximum concentration was measured at SP-12 and then reached SP-21, the groundwater velocity was only 0.17 m day^{-1}. Estimating the velocity between wells provides a more realistic flow field and may provide a more accurate estimate of aquifer characteristics, since the hydraulic gradient may change due to rain events in the time the tracer travels between wells. The hydraulic gradient ranged from 0.0005 to 0.0045 throughout this tracer study, with the lowest gradients measured during June and July 1998.

Results obtained from the JA site are similar to those observed during the first experiment at the SP site. When the tracer was injected in June 1998, there was a very small hydraulic gradient (<0.0008). In addition, the hydraulic gradient measured in early July 1998 was negative, moving water toward the center of the island rather than toward the bay and the monitoring wells. During this experiment, the tracer was only detected in two wells (JA-2 and JA-4), and it is not certain if the tracer's center of mass moved through the area of the wells (**Fig. 1.13**). If a horizontal transport rate is calculated from the time of injection to the time when the maximum concentration in the monitoring wells appeared, an average velocity of 0.02 and 0.03 is obtained for wells JA-2 and JA-4, respectively (**Table 1.3**). However, this includes the periods of very little transport (low gradient) and the time the groundwater was moving in the opposite direction and should, therefore, be considered a conservative estimate. Calculating the time between corresponding tracer maximum and distance between the two wells provides an

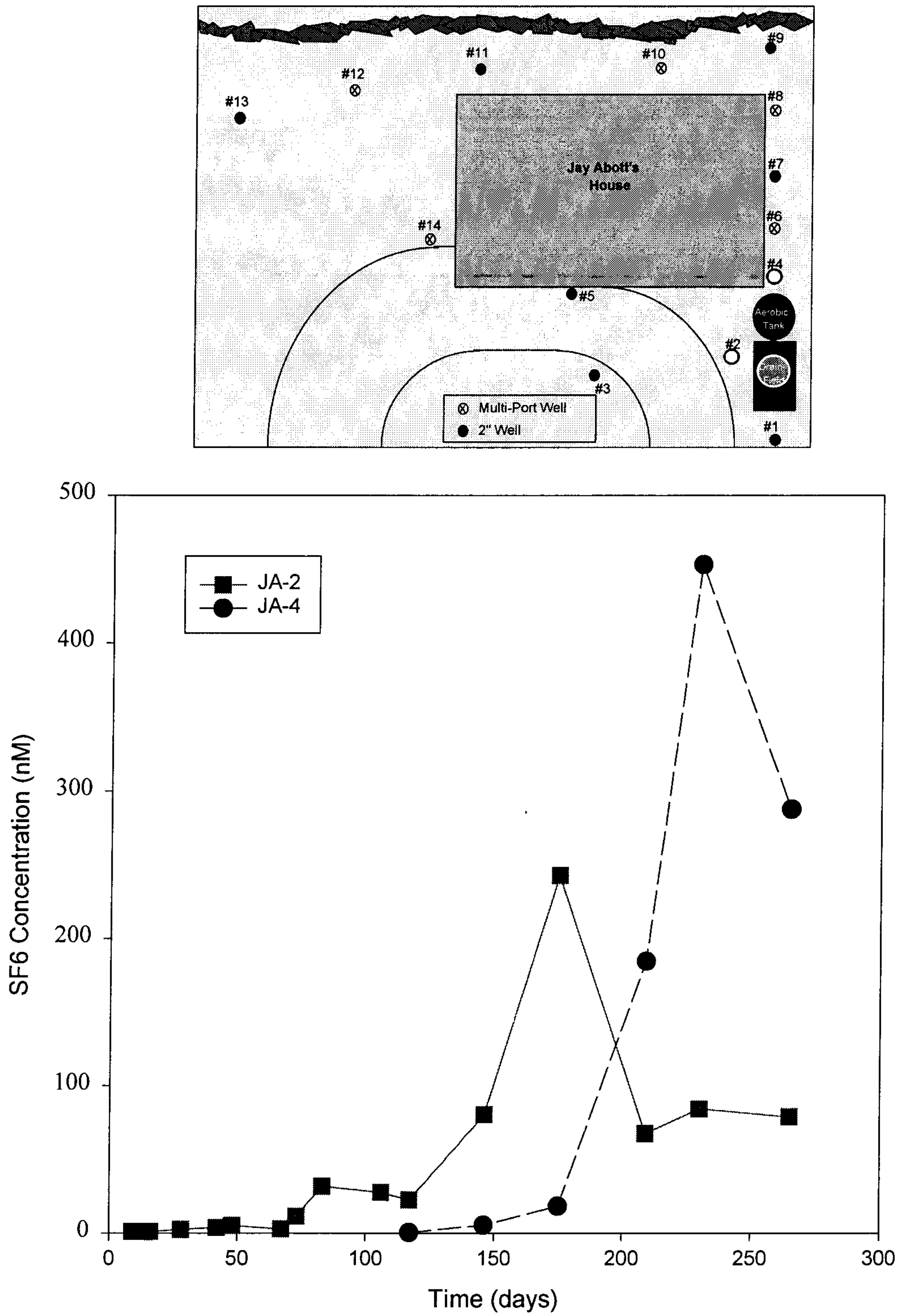


Figure 1.13: Sulfur hexafluoride tracer curves from the JA site. Well locations that had measurable SF_6 concentrations are shown on the site map (open circle). The tracer was injected directly into the drainfield, bypassing the aerobic system.

estimated groundwater velocity of 0.07 m day^{-1} between these wells. Again, there was very little dilution of the injected tracer, on the order of 3-5 times the injected value.

Many aquifer characteristics may be obtained from tracer experiments like those performed on St. George Island. Estimates of these characteristics are needed before further prediction tools may be utilized. For instance, an estimate of the aquifer's hydraulic conductivity may be calculated using the relationship

$$v = \frac{Ki}{n} \tag{3}$$

where v is the horizontal transport rate (m day^{-1}), K is the hydraulic conductivity (m day^{-1}), i is the hydraulic gradient (dimensionless), and n is the sediment porosity (dimensionless). The porosity of St. George Island sediment is approximately 0.30-0.35, measured gravimetrically from samples collected during installation of the monitoring wells. This also agrees with values reported by Domenico and Schwartz (1990) for fine-medium grain sands. The average groundwater velocity is calculated using the tracer results. Unfortunately, the hydraulic gradient measured at the site has been shown to change dramatically throughout the course of the tracer experiment, increasing the error associated with this calculation. To reduce these errors, the hydraulic conductivity can also be calculated using the estimated groundwater velocity obtained between two wells. In this case, the hydraulic gradient is constrained to the time period between the arrival of the maximum tracer concentration in the two wells, greatly reducing the associated error. Estimates of hydraulic conductivity using **Eq. 3** range from 4.5 to 75 m day^{-1} and 2.7 to 7.1 m day^{-1} for the SP site and JA site, respectively **(Table 1.4)**. These values compare well with other studies of coastal sandy aquifers (Garabedian et al., 1991; Robertson et al., 1991; Mas-Pla et al., 1992; Millham and Howes, 1995) and with the estimates obtained from the tidal method at the SP site (**Table 1.1**). The results from the SP site show a slight decrease in hydraulic

Table 1.4. Aquifer characteristics estimated from tracer results.

Sampling Site	Tracer Employed	Hydraulic Gradient	Hydraulic Conductivity (m day^{-1})	Longitudinal Dispersivity (m)
SP Site				
SP-17a	Fluorescein	0.0079 ± 0.0067	4.5 ± 3.5	0.10
SP-17b	Fluorescein	0.0079 ± 0.0067	16 ± 14	0.30
SP-9	Fluorescein	0.0043 ± 0.0038	9.4 ± 8.2	0.20
SP-12	SF_6	0.0018 ± 0.0012	75 ± 50	0.20
SP-21a	SF_6	0.0024 ± 0.0011	23 ± 11	0.10
SP-21b	SF_6	0.0031 ± 0.0005	27 ± 4	
SP-23a	SF_6	0.0024 ± 0.0009	22 ± 8	0.50
SP-23b	SF_6	0.0024 ± 0.0003	31 ± 4	
JA Site				
JA-2	SF_6	0.0024 ± 0.0021	2.7 ± 2.3	0.03
JA-4a	SF_6	0.0026 ± 0.0020	3.8 ± 2.9	0.04
JA-4b	SF_6	0.0032 ± 0.0005	7.1 ± 1.1	

conductivity toward the shore when calculated using **Eq. 3**, providing additional evidence of a non-homogeneous aquifer.

Hydrodynamic dispersion, another important aquifer characteristic, refers to the mechanical and diffusive mixing of solutes within the porous media. Diffusive dispersion is a consequence of a concentration gradient, while mechanical dispersion is mixing that occurs due to local variations in velocity around some mean velocity of flow (Dominico and Schwartz, 1990). Hydrodynamic dispersion may act to dilute the solutes, spreading them both vertically and horizontally, as they travel downgradient. Many studies have shown that the values of dispersivity are scale dependent, increasing with the scale of measurement (Theis, 1963; Fried, 1975; Lee et al, 1980; Pickens and Grisak, 1981; Gelhar et al., 1985). Zou and Parr (1994) described a mathematical approach to estimate longitudinal (same direction of flow) and transverse (perpendicular horizontally and vertically to flow) dispersivity using single well tracer breakthrough curves, like those shown in **Figures 1.11-1.13**. Longitudinal dispersivity (α_x; m) may be calculated using

$$\alpha_x = \frac{v\Delta t^2}{\left[4\left(t\ln(R) - \Delta t\right)\right]} \qquad (4)$$

where

$$R = \frac{C_{\max} t_{\max}}{Ct} \qquad (5)$$

and v is the linear velocity of the flow field (m day^{-1}), t_{max} is the time of maximum concentration (C_{max}) observed (day), t is the time of some observed concentration (C) at the same well as C_{max} (day), and Δt is the difference between t_{max} and t. Using this approach, the longitudinal dispersivity at the SP site ranged from 0.1 to 0.5 meters throughout both experiments (**Table 1.4**). The largest value obtained (0.5 m) was calculated for the well furthest from the injection site, potentially showing the scale dependence of this parameter as described above.

Longitudinal dispersivities calculated for the JA site were an order of magnitude lower than those observed at the SP site (**Table 1.4**). However, the scale of observations at the JA site were much smaller (7 m) compared to that of the SP site (67 m), again showing a scale dependence of this parameter. These values of lateral dispersivity are comparable to those reported in several studies of coastal sandy aquifers under similar field-scales (Gelhar et al., 1985; Knopman and Voss, 1991; Mas-Pla et al., 1992). Unfortunately, calculation of the transverse dispersivity was not possible since the tracer never consistently appeared in any well except for those along the center transect. The calculation of transverse dispersivity requires a breakthrough curve from a well some distance downgradient from the injection site (x) and a distance perpendicular to the center of mass (y). However, it can be inferred that the transverse dispersivity is relatively small, since the tracer only appeared in one well along each "picket fence" (wells were spaced <5 m apart), including the fence 67 meters from the injection site. The small dispersivities estimated for St. George Island indicate that very little dilution of contaminants occurs in the subsurface before being released into surface waters. This finding may be significant since regulators frequently rely on dilution for attenuation of wastewater contamination in groundwater (Robertson et al., 1991).

Groundwater Flux into Apalachicola Bay

Estimating the groundwater flux from St. George Island into Apalachicola Bay may allow estimates of nutrient fluxes from wastewater systems, thus offer more information for better management decisions in the future. We have used two independent techniques to evaluate the volume of groundwater exiting into the bay. Tracer experiments have provided an estimate of groundwater velocity and hydraulic conductivity at two different locations along the bay side of the island. Assuming this water eventually enters Apalachicola Bay, an estimate of the volumetric flow rate (Q; $m^3\ yr^{-1}$) may be calculated, if the cross sectional area of the aquifer is known (A; m^2), with Darcy's law:

$$Q = vnA \qquad (6)$$

where v is the groundwater velocity (m yr^{-1}) estimated from the tracer experiments, and n is the effective porosity. The cross sectional area of the aquifer was calculated by assuming the length of the island (48000 m) and the approximate depth to the clay layer near mean high tide (8 m). In order to provide a more conservative estimate, the volumetric rate is presented as a range assuming the entire island's hydrology mimics the SP site (long term average v = 66 m yr^{-1}) or the JA site (long term average v = 26 m yr^{-1}). This gives a range in the groundwater flux into the bay of between 3 – 8 X 10^6 m^3 yr^{-1}. Although small in comparison to the Apalachicola River, which discharges an average 2.2 X 10^{10} m^3 yr^{-1} (Fu and Winchester, 1994), groundwater could provide a significant amount of nutrients and bacteria to smaller local bodies of water, i.e. semi-enclosed embayments.

We also estimated this flux using a simple box model and assuming the aquifer is at steady state on a yearly basis. Using this assumption, the groundwater flux can be calculated by totaling the sources to the surficial aquifer (rainfall and potable water from the mainland) and subtracting the sinks (evapotranspiration), accounting for the area of the island. The average rainfall for the island is 140 cm yr^{-1}. Multiplying by the area of the island provides a flux of 34 X 10^6 m^3 yr^{-1}. The total amount of water pumped over to the island in 1998 was 0.5 X 10^6 m^3 yr^{-1}, less than 2% of the rainfall. Although all of the water pumped over to the island never makes it to the surficial aquifer, the uncertainty is minimal since the volume is so much less than the other parameters. Evapotranspiration is one of the most difficult parameters to obtain using this model. Again, we have presented our estimates as a range to try and include the uncertainties of each parameter. Assuming evapotranspiration is between 30-80% of the total rainfall, the total groundwater that must exit the aquifer to continue steady state conditions is between 8-24 X 10^6 m^3 yr^{-1}. However, some of this groundwater moves toward and ultimately discharges into the Gulf of Mexico and the rest enters Apalachicola Bay. Conservatively, we can assume half of the groundwater moves in each direction. Preliminary data suggests that the highest part of the aquifer lies on the Gulf side of the island. This suggests that more than 50% of the groundwater moves toward the bay. Assuming the estimate of 50%, the groundwater flux into the bay is around 4-12 X 10^6 m^3 yr^{-1}. The agreement between the two independent estimates suggests that the average value probably lies within these extreme values.

Summary

The transport and ultimate discharge of groundwater into coastal zones may be a significant process in the geochemical and nutrient budgets of many marine nearshore waters, especially in areas served by onsite sewage treatment and disposal systems. In order to better understand the potential impacts of wastewater systems on surface water environments, a better understanding of the groundwater flow and general hydrology of these islands is needed. The groundwater movement, downgradient from three different onsite sewage treatment and disposal systems, was monitored on St. George Island, a barrier island in the northeastern Gulf of Mexico. It is hoped that results from this study may help in the development of better management plans to protect this and other delicate ecosystems.

Tracer experiments and results from a tidal experiment suggest that the hydraulic conductivity averages 36 m day^{-1}, ranging from 2.7 to 180 m day^{-1}. The hydraulic conductivity also appears to increase moving inland, away from surface waters. Calculated longitudinal dispersivities suggest that there is very little dilution of the tracer plumes as they move downgradient. This is also evident by the small change in concentration in the monitoring wells relative to the injected concentrations. Horizontal transport rates ranged from 0.11-0.42 m day^{-1} and 0.02-0.07 m day^{-1} at the SP and JA field sites, respectively. Differences in flow velocity are associated with large deviations of hydraulic head in response to tides and rain events. The flow velocities, as well as an independent mass balance, were used to estimate the total flux of groundwater entering Apalachicola Bay. Both calculations provide an estimate of between 3-12 X 10^6 m^3 yr^{-1}.

CHAPTER 2

GROUNDWATER NUTRIENT DYNAMICS DOWNFIELD FROM ONSITE SEWAGE TREATMENT AND DISPOSAL SYSTEMS ON A BARRIER ISLAND IN THE NORTHEAST GULF OF MEXICO

Introduction

Groundwater discharge may provide a significant amount of nutrients and contaminants into the coastal zone (Valiela et al., 1978; Valiela and Teal, 1979; Capone and Bautisa, 1985; Lapointe and O'Connell, 1989; Capone and Slater, 1990; Lapointe et al., 1990; Valiela et al., 1990). In areas with a shallow freshwater system, groundwater may easily be contaminated from onsite sewage treatment and disposal systems (OSTDS), which are typically installed less than 1 meter above the water table and may be flooded during frequent rain events. Effluent originating from OSTDS will follow streamlines within the freshwater lens. Transport is easily modeled theoretically for simplified islands (i.e., those with a sharp saltwater/freshwater interface and homogeneous medium [Van der Veer, 1977; Vacher, 1988]). In principle, these streamlines converge in a narrow submarine outflow face where all discharge is focused. This direct path between wastewater systems and surface waters could potentially contaminate local water bodies, creating problems within an estuary or small embayment.

St. George Island, like most barrier islands, forms the outer perimeter of the estuary and is critical to the bay's productivity because its orientation determines the salinity distribution as well as other water quality features of the bay. Apalachicola Bay is one of the most economically important estuarine systems in Florida due to oyster and shrimp harvesting. This coastal community, located within the Apalachicola National

Estuarine Reserve, is developing at record pace. This has resulted in a high density of individual wastewater treatment systems on the island. Growth in these and surrounding communities is of major concern with regard to the health of the estuary. Many estuaries are sensitive to slight physical, chemical, and biological perturbations. Although barrier islands play a critical role in this balance, very little is known about the groundwater dynamics and potential impacts of contaminated groundwaters from these islands.

In order to prevent the possible deterioration of Apalachicola Bay and other estuarine systems, including economic zones (oyster beds and areas of dense shrimp populations), contaminants of any type must be monitored closely. Groundwater may be an important pathway of harmful bacteria and nutrients to local nearshore areas of the bay. Although the Apalachicola River provides the majority of the nutrients to the bay, those supplied by the groundwaters of St. George Island may be important to small local embayments. Without knowledge of the groundwater contribution, interpretation and management decisions on the treatment of sewage may be faulty and lead to future environmental threats. Thus, monitoring of OSTDS in an area of increasing development and density is necessary to help in future wastewater treatment decisions.

We have monitored the groundwater nutrient concentrations and estimated the nutrient flux to bay waters along St. George Island (**Fig. 1.1**). Multi-level samplers (MLS) and 5 cm PVC monitoring wells were placed downgradient from selected wastewater systems at three locations on the island. Experimental sites include two types of onsite sewage treatment and disposal systems, aerobic and anaerobic treatment. Differences in the effluent discharged and possible groundwater contaminants from both types of systems have been evaluated. The wells were monitored over the course of the study for nutrient concentrations along the flow path toward surface waters. Surface water samples were collected from selected sites in Apalachicola Bay and analyzed for nutrient concentrations and bacterial densities. A sampling location on Little St. George Island (**Fig. 1.1**), a small barrier island adjacent to St. George Island, was established to assess reasonable background levels for select nutrients and bacteria, and allowed comparison to our experimental sites.

Study Site

St. George Island

St. George Island is a microtidal barrier island in the panhandle of Florida. The island is approximately 48 km long and averages less than 0.5 km in width. Dr. Julian G. Bruce State Park occupies the east end of the island. The climate in the region is mild with a mean annual temperature of approximately 20 °C (Livingston, 1984). The mean annual rainfall over the area, recorded over the last 42 years by the NOAA weather station in Apalachicola, is approximately 140 cm. The tidal range in the bay and Gulf is less than 0.5 m. The surficial aquifer is composed of medium to fine sand grains overlying a silty clay impermeable barrier between 7.6 and 9.2 m below the surface that forms a base to the aquifer (Livingston, 1984). Water in the shallow freshwater lens is primarily derived from rainfall and eventually discharges into Apalachicola Bay or the Gulf of Mexico. The impermeable clay layer separates rain-derived freshwater from the surrounding salt water. St. George Island has a characteristically high water table, which increases the probability of groundwater contamination and transport to surrounding marine waters.

Apalachicola Bay has an area of 260 km^2 and a mean depth of approximately 2 m. The Apalachicola River, part of the greater Apalachicola-Chattahoochee-Flint River system, empties directly into the bay. The river is the third largest in the northern Gulf of Mexico and the largest in Florida and provides the largest source of freshwater and nutrients to the bay. Mean dissolved inorganic nitrogen (DIN) input from the river between 1972 and 1990 was 2.4 moles m^{-2} yr^{-1}, ranging from 1.2 to 3.4 moles m^{-2} yr^{-1} (Frick et al., 1996). Mortazavi et al. estimated the DIN (1999a) and total phosphate (1999b) input into the bay, between June 1994 and May 1996, to be 2.7 ± 0.1 moles m^{-2} yr^{-1} and 0.15 ± 0.02 moles m^{-2} yr^{-1}, respectively. Their results indicated that the DIN was 61% of the total nitrogen contribution. Approximately 60% of the phosphate was in the particulate form, 24% was dissolved organic phosphorous, and the remainder was soluble reactive phosphate. Nitrogen was shown to be limiting mainly in the summer, while phosphate was limiting in the winter. The patterns of nutrient limitation were shown to be dependent on river discharge and changes in N:P inputs (Fulmer, 1997), similar to other estuaries (D'Elia et al., 1986; Webb and Eldridge, 1987).

Experimental Sites

Experimental sites were selected according to location on the island, proximity to Apalachicola Bay, type of OSTDS, and the amount of time the residence was occupied. Care was given to locate sites adjacent to the bay with a similar beachfront, i.e., no canals, natural topographic gradients, etc. Three sites were chosen, including a site located within the Dr. Julian G. Bruce State Park (SP Site) on the far eastern end of the island, a private residence near Bob Sikes Cut (BL Site) on the extreme western end of the island, and another private residence approximately 3 km west of the causeway (JA Site) between the other two locations. Each site had an average of 3 residents and was occupied year round.

SP Site. The experimental site within the state park was developed between August and October 1997. Wells were installed primarily downgradient from the septic system and consist of twelve 5-cm monitoring wells and thirteen multi-level samplers covering just under 8000 m^2 (**Fig. 1.2**). The wastewater system is set back approximately 100 m from Apalachicola Bay. Nutrient samples were collected from September 1997 to May 1999.

BL Site. B.L. Cosey's property is also approximately 8000 m^2 with 60 m of bayfront access (**Fig. 1.3**). The property is generally level and approximately 0.6 m above mean sea level. The septic system and drainfield is raised above ground level approximately 1 m to prevent inundation by surface waters during large storms and rain events. Wells were installed on either side of the drainfield and towards the bay waters during March, 1998. Samples were collected from this site until March, 1999, when the wells were removed due to construction on the property.

JA Site. Jay Abbott's property is located adjacent to Apalachicola Bay with approximately 45 m of bayfront access. The site is just under 6000 m^2 and has an aerobic wastewater system, installed in 1996. Due to the location of the house with respect to the OSTDS, wells were not positioned in a grid-type pattern as at the other sites. Wells were placed as close to the OSTDS and around the residence as possible. Fourteen wells, including seven monitoring wells and seven multi-level samplers, were installed at the