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United States
Environmental Protection
Agency

Relative Risk Assessment of Management Options for Treated Wastewater in South Florida

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WASTEWATER CHALLENGES IN SOUTH FLORIDA

Every day, more than 1.5 billion gallons of wastewater leave municipal treatment facilities in Florida bound for reuse or disposal. Municipalities in South Florida rely less on discharges to surface waters and more on reuse, ocean discharge and deep-well injection. For example, in Miami-Dade County, for every three gallons of wastewater generated, one gallon is treated and sent to deep underground saltwater formations. The other two gallons are piped out to the ocean, three and a half miles offshore. In dry-weather conditions in Pinellas County, for every three gallons of wastewater generated, all three gallons are reclaimed to golf courses, parks, and lawns after high-level treatment and disinfection. However, the Pinellas area receives on average forty-eight inches of rain annually, and deep-well disposal is heavily relied on as the backup during wet weather.

Each municipality in South Florida is faced with its own particular challenges to ensure, safe reuse and disposal of wastewater, safe drinking water and a healthy environment for its 5.8 million residents. Local municipalities are struggling to make sound wastewater management decisions, taking into account the often overwhelming complexities and the range of technical issues associated with different reuse and disposal options.

The State is strongly committed to protecting its surface waters, such as lakes, rivers, streams, wetlands, estuaries, and the ocean. It is equally committed to protecting the highly permeable aquifer systems that provide 94% of the area's drinking water. A major challenge to protecting water resources is Florida's growing population and the accompanying need for safe drinking water, safe reclaimed water reuse, and safe wastewater disposal.

The Environmental Protection Agency (EPA) has established minimum requirements for Class I municipal wells and other underground injection activities through a series of Underground Injection Control (UIC) regulations at Code of Federal Regulations (CFR) Title 40 Parts 144-147, developed under the authority of the Safe Drinking Water Act. These regulations ensure that Class I municipal wells will not endanger USDWs by prohibiting the movement of any contaminant into Underground Sources of Drinking Water (USDW).

On July 7, 2000, EPA proposed revisions to the UIC regulations that would allow continued wastewater injection by existing Class I municipal wells that have caused or may cause movement of contaminants into USDWs in specific areas of Florida (65 FR 42234). Continued injection would be allowed only if owners or operators meet certain requirements that provide adequate protection for USDWs. In the alternative, if new requirements are not promulgated, owners and/or operators of wells targeted by the proposal would be required to close their wells and adopt different wastewater disposal practices, which could consist of surface water disposal, ocean outfall, and/or reuse. Use of these alternative disposal practices would likely require the construction of systems for advanced wastewater treatment, nutrient removal, and high-level disinfection.

CONGRESSIONAL MANDATE FOR RELATIVE RISK ASSESSMENT

EPA, as directed by congressional language in its fiscal year 2000 appropriation, prepared the relative risk assessment presented in this report:

> *Within available funds, the conferees direct EPA to conduct a relative risk assessment of deep well injection, ocean disposal, surface discharge, and aquifer recharge of treated effluent in South Florida, in close cooperation with the Florida Department of Environmental Protection [DEP] and South Florida municipal water utilities.*

Congress directed EPA to conduct this assessment because wastewater injected into deep wells had moved from where it was supposed to be confined to areas where it is prohibited. Congress directed EPA to conduct the relative risk assessment to shed light on the risks posed by fluid movement from deep injection and relate those risks to risks posed by treated effluent from other wastewater management options.

MUNICIPAL WASTEWATER TREATMENT OPTIONS IN SOUTH FLORIDA

To capture all counties with deep-well injection, the South Florida area considered in the relative risk assessment extends south from a line drawn from the northern end of Brevard County on the east coast to the northern end of Pinellas County on the west coast (Exhibit ES-1).

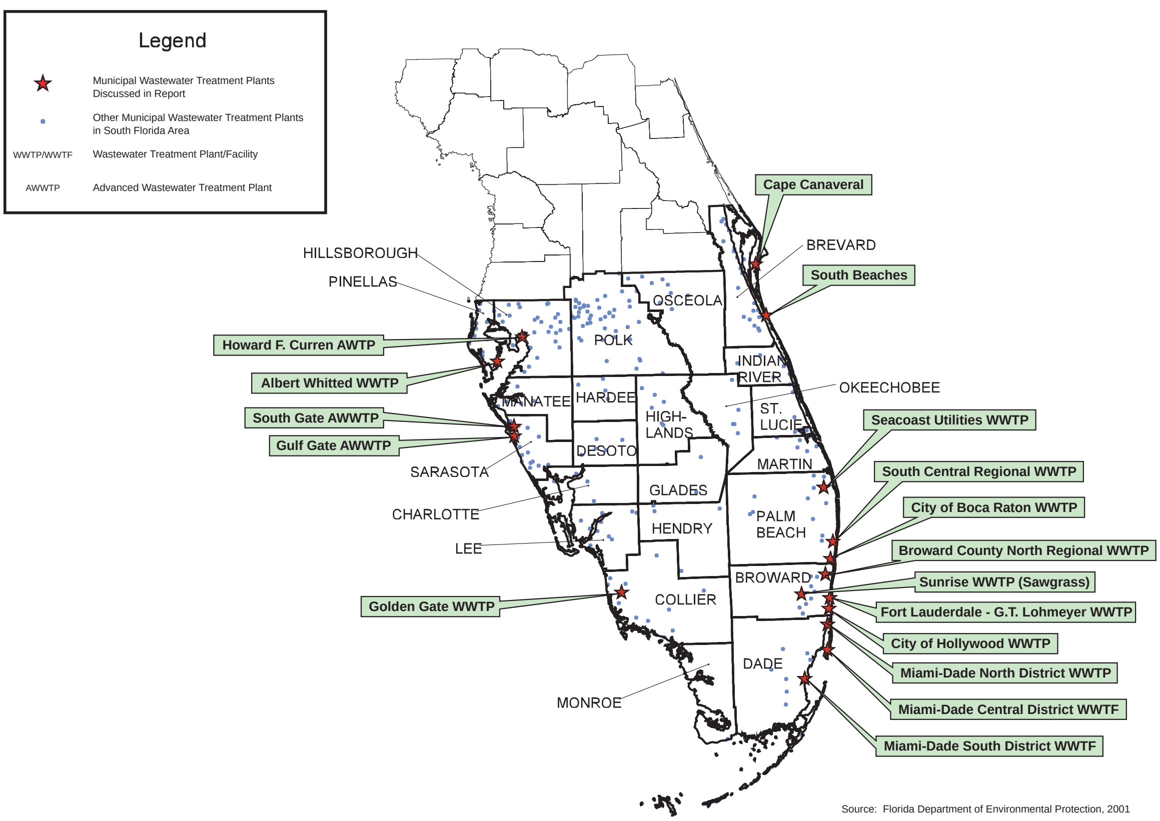


Exhibit ES-1. Municipal Wastewater Treatment Plants in South Florida

Wastewater Treatment Options

Florida primarily uses four options for the management of treated municipal wastewater (Exhibit ES-2):

- **Deep-well injection:** Wastewater is injected by gravity flow or under pressure into deep geological strata below USDWs. Under EPA and State UIC program regulations Class I wells inject fluids beneath the lowermost formation containing a USDW.
- **Aquifer recharge:** Reclaimed water is discharged to land application systems, such as infiltration basins and unlined ponds.
- **Discharge to ocean outfalls:** Treated wastewater is discharged to the ocean via outfall pipes that may extend from almost 1 mile to more than 3.5 miles from shore.
- **Discharge to surface-water bodies:** Wastewater is discharged into canals, creeks, and estuaries.

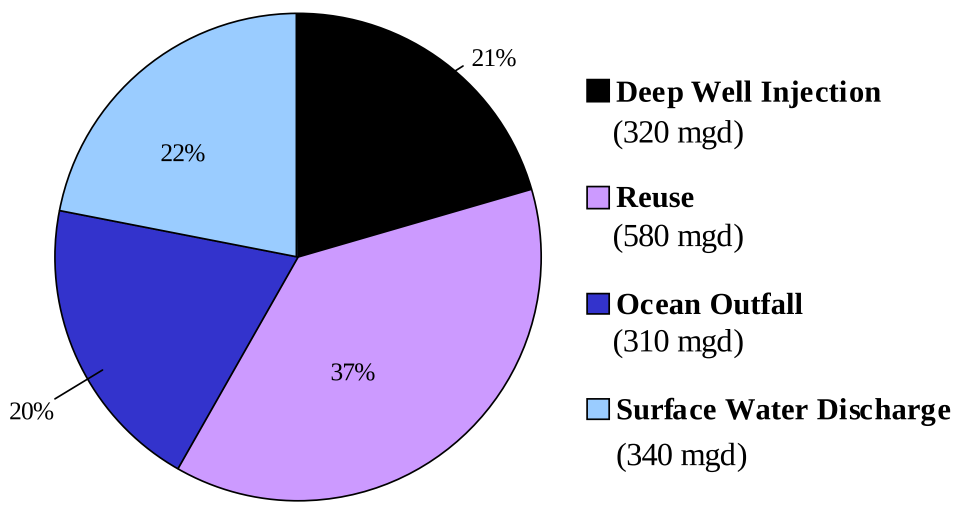


Exhibit ES-2. Use and Disposal of Effluent and Reused Water in Florida[1]

Although the term ***option,*** used to describe the wastewater treatment methods, suggests any of these are available for use by municipalities in South Florida, in fact most municipalities are limited by a variety of critical local conditions, governing regulations and costs in evaluating possible treatment methods. (Exhibit ES-3).

[1] This chart uses data for the entire state of Florida. No specific data was available for the study area only. The distribution of waste treatment options within the study area is likely to be different than that presented in this chart (i.e. all ocean disposal and deep underground injection is in the Study area and there is much less use of surface water disposal in South Florida).

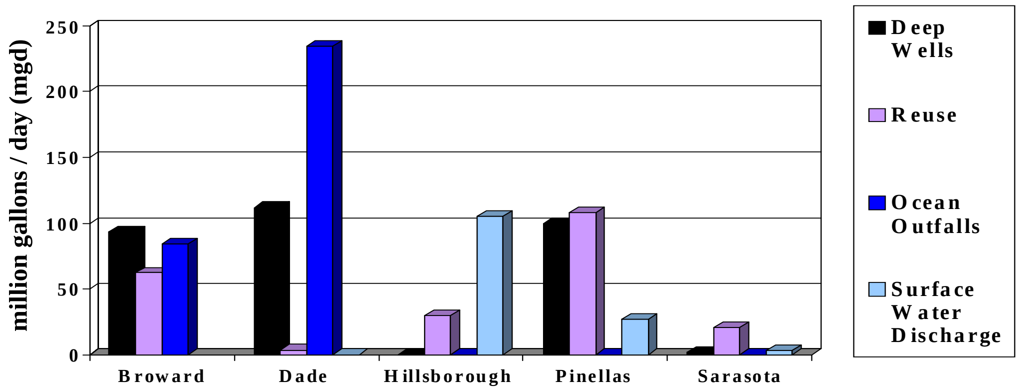


Exhibit ES-3. Wastewater Management for Selected Counties in South Florida

Levels of Wastewater Treatment and Disinfection

Wastewater treatment facilities in South Florida combine various levels of wastewater treatment and disinfection to arrive at effluent concentrations that are appropriate for the local conditions and that comply with State and EPA requirements.

- **Primary Treatment** is a basic treatment process that removes material that will float or settle.
- **Secondary Treatment** is a process in which bacteria consume the biodegradable organic matter and remove suspended solids using chemical and biological processes. The success of treatment may be quantified by its ability to remove Biochemical Oxygen Demand (BOD) and Total Suspended Solids (TSS).
- **Reclaimed Water** in Florida means water has received at least secondary treatment and is reused. Some uses require high-level disinfection that includes filtration.
- **Advanced Water Treatment** (AWT) refers to treatment beyond secondary but in Florida it has specific regulatory meaning for a combination of treatments that includes secondary treatment, high-level disinfection, nutrient removal, and removal of toxic compounds (usually by filtration). AWT is used if there are requirements to remove specific components, such as nitrogen and phosphorus, which are not removed by secondary treatment alone.
- **Disinfection** is the selective destruction of pathogens. The State regulations define basic, intermediate and high-level disinfection with levels of filtration and bacterial deactivation.

Each of the four wastewater management options (deep-well injection, ocean outfall, aquifer recharge, and surface water discharge) provide different levels of treatment and disinfection, depending upon regulatory and site-specific needs. The levels for Biochemical Oxygen Demand, (BOD), Total Suspended Solids (TSS), Total Nitrogen (TN), and Total Phosphorus, (TP) shown in Exhibit ES-4 are required for some required discharges and do not apply universally to all (see Chapters 62-600 and 62-610 F.A.R.).

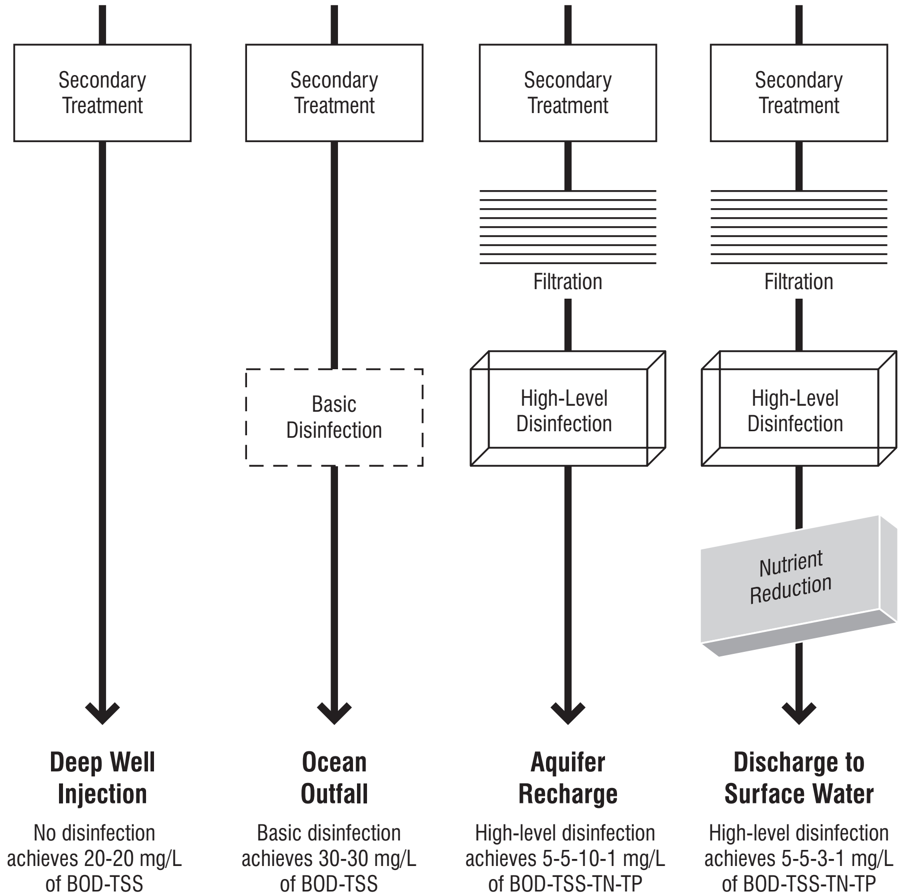


Exhibit ES-4. Levels of Treatment and Disinfection for the Four Disposal Options

RISK ASSESSMENT

Risk assessment is a multistep process. It evaluates the likelihood that adverse human health or ecological effects will occur as a result of exposure to stressors. A ***stressor*** is any physical, chemical, or biological entity that can induce an adverse response. The organism, population, or ecosystem exposed to a stressor is referred to as a ***receptor.*** ***Exposure*** refers to the contact or co-occurrence of a stressor and receptor. If there is no contact or co-occurrence between the stressor and the receptor, then there is no risk.

Risk characterization is the culminating step of the risk assessment process. It conveys the risk assessor's judgment about the existence of human health or ecological risks and their nature (US EPA, 2000). Information from the risk assessment steps is integrated and synthesized into an overall conclusion about risk that is informative and useful for decision-makers and for interested and affected parties.

Approach Used in This Relative Risk Assessment

The risk assessment conducted by EPA involved investigating four very different wastewater disposal options: deep-well injection, aquifer recharge, discharge to ocean outfalls, and discharge to surface-water bodies. Each option has its own specific stressors (hazards), exposure pathways, receptors, and effects.

Data from many sources were used to support the analyses and evaluations. Risk characterization for each wastewater treatment option included identifying and describing the associated risks, the potential magnitude of the risks, and potential effects on human and ecological health. The relative risk assessment then described and compared risks for all four wastewater management options.

This relative risk assessment first used a generalized approach to describe potential risks and identify possible stressors, sources, exposure pathways, and effects on receptors. This step incorporates human health and ecological risk components and provides a conceptual model of potential risk. A conceptual model was developed for each of the four disposal options. Exhibit ES-5 is an example of a conceptual model of potential risks developed for the relative risk assessment. Potential system stressors, exposure pathways, receptors, and the potential effects on receptors are identified in the model.

To assess the risks and to allow comparisons, EPA conducted individual risk assessments for each wastewater disposal option, and the risks associated with each were characterized. The risks and risk factors identified in each disposal option were then evaluated and described. The overall comparisons and conclusions are presented as relative risk assessment matrices. EPA found that the parameters that are relevant to one particular disposal option are not necessarily relevant to the remaining three. Therefore, a strictly quantitative comparison between the four options was not feasible.

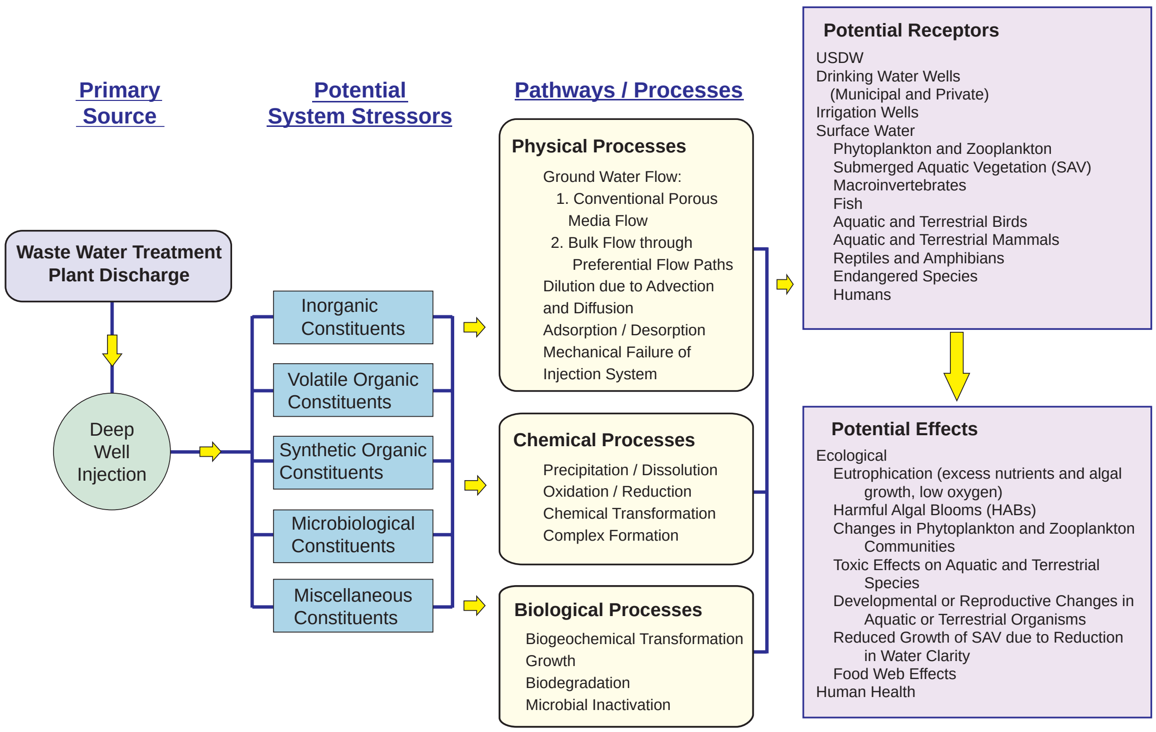
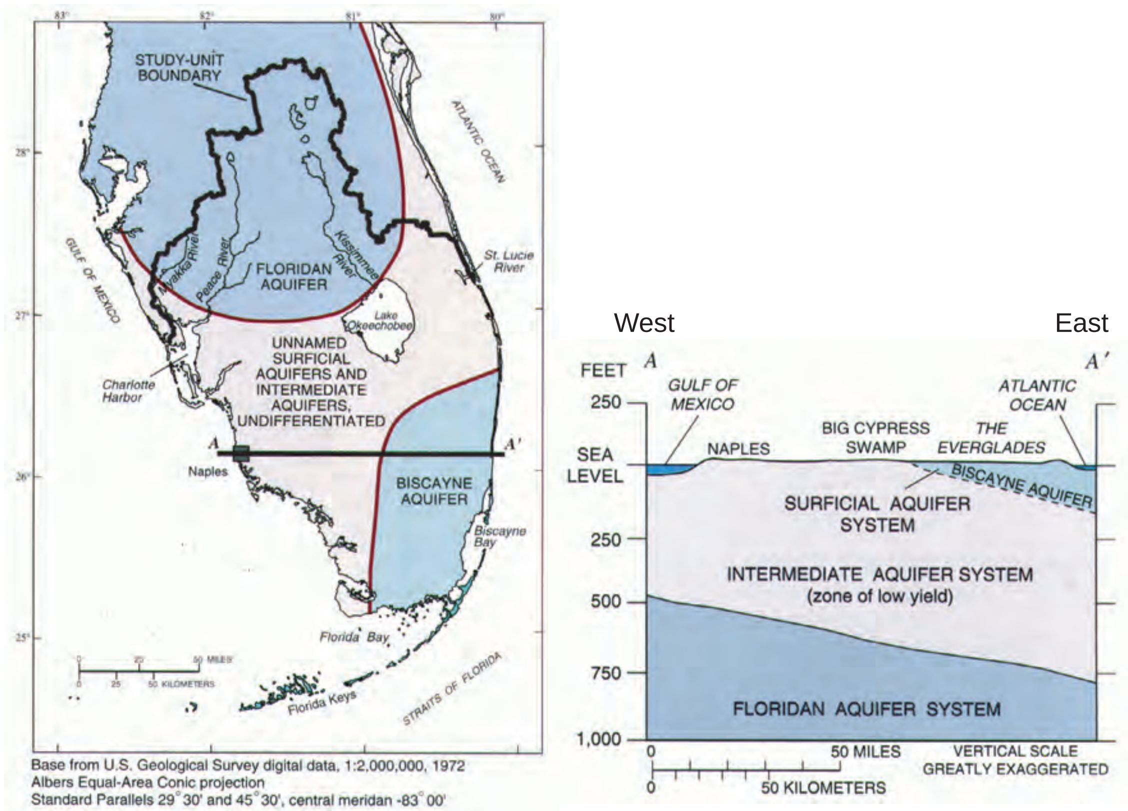


Exhibit ES-5. Conceptual Model of Potential Risks for the Deep-Well Injection Option

DEEP-WELL INJECTION

In South Florida, the most common means of disposal for treated municipal wastewater is by deep-well injection. Deep wells typically inject at depths ranging from 650 to greater than 3,500 feet below land surface, depths that are considerably deeper than the aquifers used for drinking-water supply wells. However, it is acknowledged that in some parts of South Florida, injected water has moved upward into overlying layers and, in some cases, into the base of the area designated as the underground source of drinking water (USDW).

The Upper Floridan Aquifer and the Biscayne Aquifer are the main water sources in the South Florida region (Exhibit ES-6). The Floridan Aquifer is extensive and underlies parts of Alabama, southeastern Georgia, southern South Carolina, and all of Florida. It is divided into the Upper Floridan and Lower Floridan aquifers, which are separated by a middle confining unit.



Source: McPherson et al (2000)

Exhibit ES-6. Hydrologic Profile of South Florida Aquifer System

In the southeastern part of South Florida, the Floridan Aquifer is overlain by a relatively shallow surficial aquifer, the Biscayne Aquifer. In general, the surficial aquifer is composed of relatively thin layers of sands with some interbedded shell and limestone (Exhibit ES-6). The surficial aquifer in Pinellas County is only about 56 feet thick; in Brevard County, it is only 110 feet thick (Exhibit ES-7). The underlying intermediate confining unit, which separates the surficial and Upper Floridan aquifers, is also relatively thin (about 219 feet thick in Pinellas County and 210 feet thick in Brevard County). These hydrogeologic characteristics mean that the surficial aquifer yields only small amounts of water. Thus, it is not a major source for public water supply, although it is used extensively for private water supplies. However, in southeastern Florida, the Biscayne Aquifer is the principal source of drinking water. In this area, both the aquifer and the underlying intermediate confining unit are thicker (more than 230 and 610 feet thick, respectively), which results in an increased water-bearing capacity.

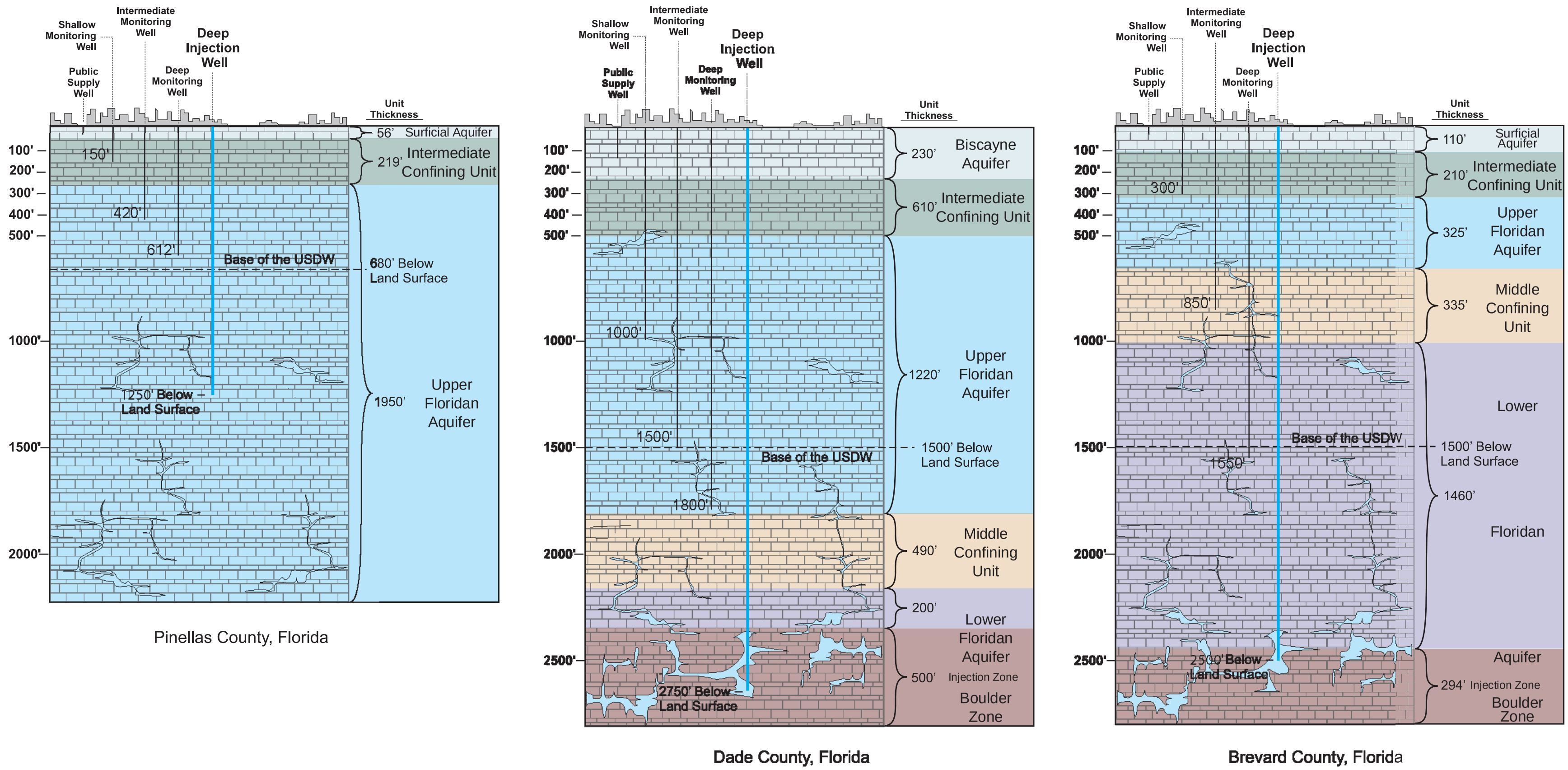


Exhibit ES-7. Representative Hydrogeologic Cross Sections

The presence of the separating confining units (intermediate and middle), combined with the considerable depth to the deep-well injection zones, was considered to provide a sufficient level of protection to the water-bearing strata that supply public water. However, the relative safety of this disposal option is now in question because injected water is known to have migrated up to and, in some cases, into the USDWs.

Deep-well injection fluid is given a secondary level of treatment and the State does not require disinfection, although some facilities may dispose of excess (unused) reclaimed wastewater using Class I deep-well injection. Treatment beyond a secondary level is used to varying degrees in the three other disposal options included in the risk assessment (aquifer recharge, discharge to ocean outfalls, and discharge to surface-water bodies) (Exhibit ES-4).

Many parts of the United States use Class I injection wells for disposal of hazardous and nonhazardous fluids. In Florida, deep-well (Class I) injection is an important management option for treated municipal wastewater and accounts for approximately 20% (0.44 billion gallons per day) of the State's wastewater management capacity (FDEP, 1997). Most of this use occurs in South Florida, particularly southeastern Florida and in coastal areas. The wells inject large volumes of wastes into deep rock formations, which are required to be separated from sources of drinking water by layers of impermeable clay and rock.

The use of Class I wells in South Florida has been considered a safe and effective means of disposing of treated wastewater. However, ground-water monitoring data has indicated that, at some facilities, wastewater is not being adequately confined, resulting in unintended movement of the injected fluid into USDWs. At some locations, injected wastewater has migrated from the injection zone into overlying layers and is compromising USDWs. Of 93 facilities with deep injection wells in South Florida, 18 have been identified as having unintended movement of fluid out of the injection zone: 3 have confirmed fluid movement into the USDW, 6 are reported to have probable movement into the USDW, and 9 have movement into non-USDWs, (layers overlying the injected zone but below the USDW).

Regulatory Oversight of Deep-Well Injection

Federal and State regulations govern the siting, construction, operation, and management of Class I injection wells. A key UIC regulatory requirement prohibits the movement of any contaminant from a Class I injection well into a USDW. UIC regulations also specify well siting requirements, including specifications for constructing wells, for defining hydrologic conditions relative to the site, for ensuring the mechanical integrity of injection wells, and for proper operation and maintenance of wells. Class I injection wells must be cased and cemented to prevent the movement of fluids into or between USDWs. Injection pressures may not cause fractures in the confining zone or cause the movement of injection or formation fluids into a USDW. (40CFR146.12 and 13). In addition, the State requires that all Class I municipal waste disposal wells provide, at a minimum, secondary treatment.

In spite of these many regulations and controls, unintended migration of injected wastewater in South Florida has occurred. Therefore, the ability to maintain sufficient confinement between the injection zone and the USDW is in question.

Option-Specific Risk Analysis for Deep-Well Injection

The risk analysis of deep-well injection focused on Brevard, Pinellas, and Dade counties, because these counties are geographically representative (i.e. they are located in the three corners of the assessment area) and fluid movement, to some degree, has occurred in each location. A large volume of treated wastewater is injected into Class I injection wells. Subsequent migration of this wastewater and any dissolved or entrained wastewater constituents that remain after treatment can lead to exposure for receptors such as USDWs and water-supply wells.

Secondary treatment of wastewater with no disinfection does not remove all potential stressors to human health. Nitrate levels can exceed the Federal and State maximum contaminant level (MCL) for drinking water; pathogenic bacteria and viruses are not inactivated and may exceed standards for drinking water; and *Giardia* and *Cryptosporidium* levels may exceed Florida's health-based (reuse) recommended criteria.

Stressors to ecological health that may remain after treatment are generally limited to nitrates and phosphates. These are considered nutrients for ecological systems. When present in excess concentrations, they can destabilize the natural systems and cause eutrophication of aquatic systems. Given this characterization of the level of contaminants remaining in secondary treated effluent, a next step in the risk assessment was to examine the fate and transport of these contaminants in the sub-surface.

How Injected Wastewater Can Reach Drinking-Water Supplies

In general, injected wastewater can move upwards by porous media flow and by bulk flow. These represent two extremes: porous media flow is a slow fluid movement through connected pores in the rock matrix, and bulk flow is a more rapid flow through preferential paths, such as fissures, fractures, caverns, or channels (Exhibit ES-8). Bulk flow can also occur from improperly constructed and poorly maintained injection-well systems that lead to an incomplete seal between the well and its casing.

In most cases in South Florida, both porous flow and bulk flow mechanisms will contribute to upward migration. However, it is not possible to differentiate the contribution of each for a given location. Bulk flow is likely a major contributing process in South Florida, where there are karst geologic features. The most well known geologic feature in the area that can support bulk flow is the Boulder Zone. Located in the middle section of the Lower Floridan Aquifer (Exhibits ES-7 and ES-8), this highly developed and complex fracture zone has extensive cavernous pores, fractures, and widened joints that allow channelized groundwater flow, sometimes at extremely rapid rates.

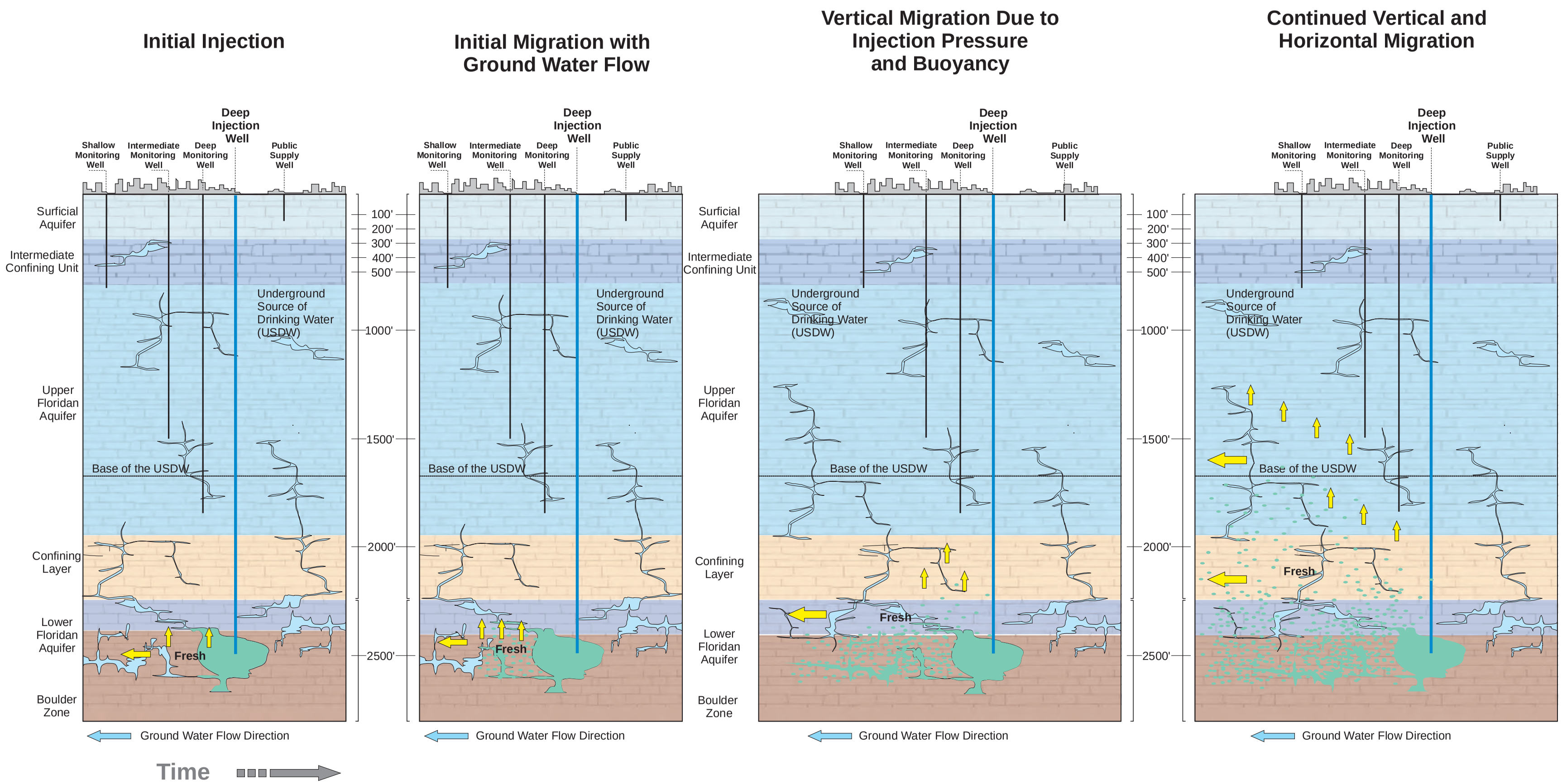


Exhibit ES-8. Migration of Wastewater by Bulk Flow from a Deep-Well Injection Zone. *Bulk flow and porous media flow contribute to migration and are influenced by several factors, including temperature, density, and injection pressure.*

Fluid movement underground is influenced by several factors. Temperature and density differences between native and injected waters affect buoyancy. The fluid density of injected wastewater is roughly equivalent to fresh water. However, wastewater is injected at depths where the native groundwater is saline or hypersaline. Buoyancy tends to force the comparatively lighter, less dense wastewater upward.

Injection pressure also influences fluid movement, but the degree of influence is affected by the geology. In parts of South Florida, where injection zones demonstrate a great capacity to accept injected fluid (for example, the Boulder Zone), the influence of injection pressure may be less significant. Regional differences in the effect of injection pressure were accounted for in the risk analysis by including Dade, Brevard, and Pinellas counties.

The exposure pathway for the stressors found in injected wastewater is upward migration of the injected wastewater into the base of USDWs. In some locations, this upward migration can occur relatively rapidly and with little dilution of stressors. In the area of the Boulder Zone, injected wastewater that has migrated upwards might pose some ecological health risk for the marine environment, were the fluid to migrate more than 2500 feet upward . There is little information currently available to assess such a risk.

Human Health and Ecological Risk Characterization of Deep-Well Injection

Deep-well injection for disposal of treated municipal wastewater has resulted in fluid movement into USDWs. In both Pinellas County and Dade County fluid has moved into the USDW.

The overall human health risk is lower for those USDWs that are deep, and exposure to stressors for currently used drinking-water sources is less likely. The current risk of human exposure is considered lower for Dade and Brevard counties, because the length of time required for contaminants to reach current drinking water supplies is long. However, the time of travel in the Pinellas County area is shorter because of the shallower aquifer depth and lack of confinement. The risk would be therefore higher for Pinellas County and exposure of current water supplies to stressors more likely but for the fact that Pinellas County effluent is subjected to high level disinfection. Failures within the injection system itself clearly increase risk. Improperly constructed or poorly maintained injection-well systems can result in decreased times of travel to receptors and in an associated increase in risks and exposures. However, there is no information to conclude that mechanical failures of Class I municipal waste disposal wells in South Florida have resulted in significant fluid movement into USDWs.

Ecological risk can result from nutrient enrichment of surface waters and the associated ecosystems. However, in South Florida, the risk is considered low because that movement is unlikely. There may be an increased risk in situations where fluid migrates rapidly to surface-water bodies, as in a conduit or a bulk-flow scenario. Nutrient enrichment and other potential impacts to near-shore marine and estuarine environments could occur under such a scenario.

Aquifer Recharge

Any practice that potentially results in the replenishment of a groundwater aquifer can be considered aquifer recharge. Treated municipal wastewater discharged onto the land may percolate through soils and underlying geologic media until it reaches and recharges the surficial aquifer. In Florida, several practices may be considered as aquifer recharge: irrigation, discharge to infiltration basins or absorption fields, and discharge to wetland treatment systems. The State defines reclaimed water as water that has received, at least, secondary treatment and disinfection and is reused after flowing out of a domestic wastewater treatment facility. Reuse is the deliberate application of reclaimed water for a beneficial purpose according to Florida requirements. The final use of the wastewater determines the specific treatment requirements.

Reuse of water for irrigation is significant in Florida. Of a total of 359 reuse irrigation systems, approximately one-half (179) are golf-course irrigation systems, while the other half is divided among irrigation for other public-access areas (98) and residential irrigation (82). Agricultural irrigation systems using reclaimed water number 117.

Reclaimed water is discharged at a rate that prevents surface runoff or ponding and that is within a designated hydraulic loading rate. Loading rates are based on the ability of the plant and soil system to remove pollutants from the reclaimed water, the infiltration capacity, and the hydraulic conductivity of the underlying geology. Slow-rate land application systems must have back-up disposal methods, such as discharge to a storage area or to deep-well injection, for wet-weather conditions and when water-quality treatment standards are not met.

Rapid-rate land application systems discharge reclaimed water to rapid infiltration basins or absorption fields. Infiltration basins operate in series and may include subsurface drains that receive and distribute the water. Absorption fields are subsurface absorption systems covered by soil and vegetation and may include leaching trenches, pipes, or other conduits that receive and disperse water. Rapid-rate systems are potentially high-volume systems. Because of the increased percolation, the loading rates are higher than for slow-rate land application, and rapid-rate systems do not require wet-weather alternatives. For these reasons, EPA focused on rapid-rate infiltration basins (RIBs) for the risk assessment.

Regulatory Oversight of Aquifer Recharge

Aquifer recharge as a wastewater management option is not specifically regulated, but the State regulates the reuse of reclaimed water and land application. State regulations specify system design and operating requirements. Backup treatment and holding capacity is required, in case of system interruption. Slow-rate land application must have back-up wet-weather disposal options. Wastewaters must meet water-quality criteria and must be tested for pathogenic protozoans. Setback distances from surface waters and from potable water sources are required, and Florida’s wastewater-to-wetlands rule controls the quantity and quality of treated wastewater discharged to wetlands.

Option-Specific Risk Analysis for Aquifer Recharge

Rapid-rate systems have the potential of discharging large volumes of treated wastewater directly to the surficial aquifer. The public water supply in South Florida is generally drawn from wells about 250 feet deep and located in the surficial aquifer. In Pinellas County, the surficial aquifer is shallow, with a depth of about 56 feet. In Brevard County, the surficial aquifer extends to a depth of 110 feet. In Dade County, the surficial Biscayne Aquifer extends to a depth of 230 feet. Depending upon local groundwater conditions, rapid transport of reclaimed water to these shallow aquifers and current drinking water sources may occur. Similarly, surface-water bodies that are under direct influence of groundwater can be exposed to stressors in the discharged wastewater.

Reclaimed water that is bound for rapid-rate land application must have undergone secondary treatment and basic disinfection, and rapid-rate systems must meet, at the base of the discharge zone, groundwater criteria. Projects with permit applications after January 1, 1996 must provide high level disinfection. As a result, the concentrations of stressors are considerably reduced. Potentially remaining stressors in reclaimed water include metals and other inorganic elements (for example, nitrate, ammonium, phosphate), volatile and synthetic organic compounds, and microorganisms resistant to high-level disinfection. Cyst-forming pathogenic protozoans, such as *Cryptosporidium* and *Giardia,* are resistant to chlorination and basic disinfection and require specialized filtration for removal. Concentrations of these pathogenic protozoans typically meet Florida's health-based (reuse) recommendations in rapid-rate land application waters, but some exceptions have been reported. The disinfection byproducts, trihalomethanes, can pose a human health risk, but the concentrations in reclaimed water rarely exceed the health-based standards.

Just as with deep-well injection waters, stressors to ecological health that may remain in reclaimed water after treatment are nitrates and phosphates. Because they are nutrients, they can destabilize the natural systems and, when present in excess concentrations, can cause eutrophication of aquatic systems. Thus, the next step of the risk assessment, the analysis of the fate and transport mechanisms and a determination of the time of travel, was very important.

The time of travel for discharged effluent to move in groundwater to a receptor is site-specific and dependent on required setback distances, location and distance to receptor water-supply wells, direction of groundwater flow, the actual distance to potential receptor wells, and the aquifer's groundwater flow characteristics.

Natural attenuation processes were also analyzed to determine their affect on final constituent concentrations. Sorption, biological degradation, and chemical transformation of constituents can reduce their overall concentration during transport in groundwater. Rapid-rate infiltration and the associated shorter times of travel tend to limit natural attenuation.

Human Health and Ecological Risk Characterization of Aquifer Recharge

Because of the level of treatment, reclaimed water contains relatively few stressors, which generally are at reduced concentrations. Many constituents remaining in the treated wastewater are at levels that meet the respective drinking-water standards (MCLs). The average concentrations of the cyst-forming *Giardia* protozoan meet risk-based criteria. However, monitoring data from reuse facilities indicate the presence of *Giardia* in 58% of the samples, with detections frequently exceeding the stated recommendation of 1.4 cysts per 100 milliliters.

Although time of travel may be relatively short for some locations and indicate a higher potential risk, a high effluent transport rate does not result in a greater overall risk. Dade County, where the Biscayne Aquifer has a high hydraulic conductivity, has the shortest estimated travel times for treated effluent in groundwater to reach drinking-water supply wells: 0.11 year for a 200-foot setback, 0.28 year for a 500-foot setback, and 1.47 years for a 2,640-foot setback. In spite of these relatively short times of travel, there is little overall risk, because the final concentrations of stressors are below the respective drinking water standards (MCLs).

DISCHARGE TO OCEAN OUTFALLS

Six publicly owned wastewater treatment facilities located in coastal southeastern Florida currently use ocean outfalls to dispose of treated municipal wastewater. The total volume discharged is about 310 mgd. Before discharge, the wastewater undergoes secondary treatment, followed by basic disinfection. The treated wastewater is discharged through outfall pipes into the ocean at depths ranging from 27.3 to 32.5 meters and at distances between 0.94 and 3.56 miles from shore.

The outfalls discharge into the Florida Current, which flows northward to join the Gulf Stream. Circulation created by the Florida Current and associated eddy and rotary flows is important and the western boundary of the current is a major nutrient source for ocean productivity. Effluent discharged from the outfall forms a characteristic plume that tends to rise in seawater because it is less saline. However, the effluent is rapidly diluted and mixed with ocean water (Exhibit ES-9). The speed and direction of the currents are the primary factors that govern plume dispersal.

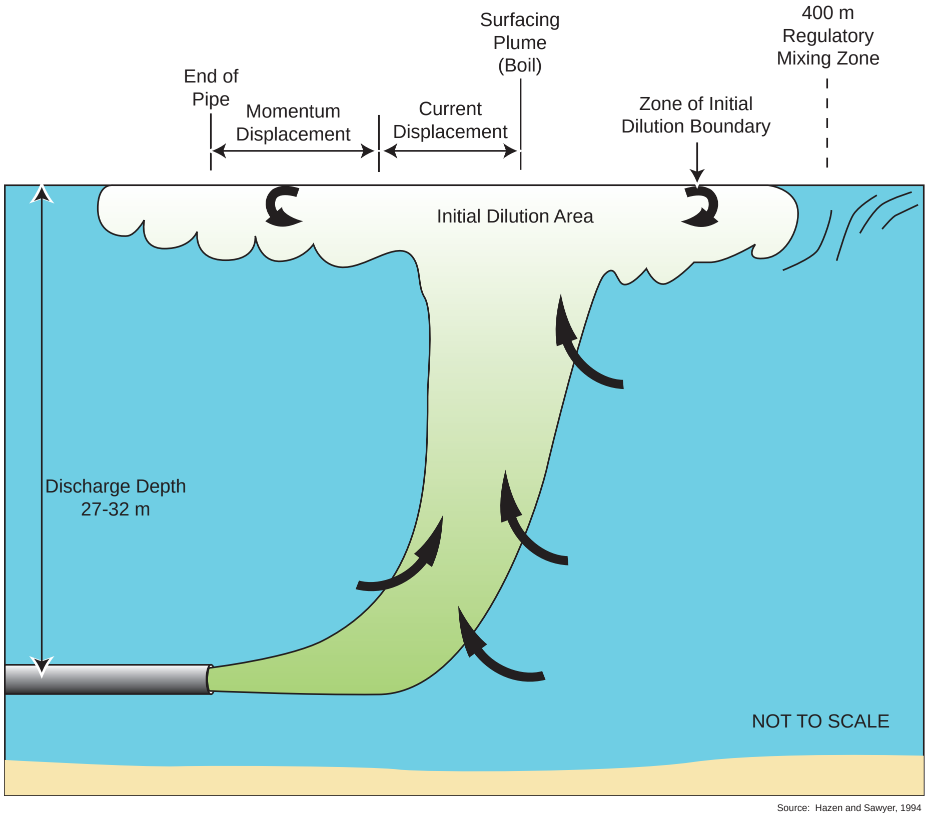


Exhibit ES-9. Effluent Plume Characteristics for Ocean Outfalls

The risk assessment for this option mainly focused on the potential effects on the marine environment. Discharge to the ocean has no effect on sources of drinking water. The receptors considered in this option are those that may have a direct exposure to seawater containing effluent constituents.

Regulatory Oversight of Discharge to Ocean Outfalls

The Clean Water Act and Florida law require that municipal wastewater receive at least secondary treatment before discharge to the ocean. When chlorine is used as a disinfectant, it must be used at the minimum concentration necessary to achieve water-quality standards. Higher concentrations of chlorine may lead to the production of trihalomethanes, which are a human health risk.

State-designated Class III Waters are used for recreation and for the propagation and maintenance of a healthy, well-balanced population of fish and wildlife. Effluent discharged into the ocean must meet the Class III standards for total suspended solids and for a 5-day biological oxygen demand.

There are additional requirements for the effluent when it meets the receiving waters. There are State and Federal water-quality criteria for effluent water at the end of the

outfall pipe, within the mixing zone, and at the edge of the mixing zone. At the edge of the mixing zone, Federal, State, and local regulations require that the water meet surface-water quality standards.

Option-Specific Risk Analysis for Discharge to Ocean Outfalls

The focus of the risk analysis was the potential effects that discharges to the ocean may have on ecological receptors. In Florida, ocean waters are not currently used as a source for drinking water. Therefore, the ocean discharge option is not a human health risk through the drinking-water supply. Human exposure to seawater that contains effluent constituents may occur for recreational users (fishermen, boaters, and swimmers), industrial fishermen, and outfall operators and workers. Exposure may be through dermal contact, incidental ingestion of ocean water, ingestion of contaminated fish or shellfish (near or removed from the point of discharge), or exposure to toxins produced by harmful algal blooms. Ecological receptors include fish and other organisms that occur around the ocean outfall discharge point as well as those that are removed from the outfall but may be affected by the discharge.

Effluent constituents discharged to the ocean are those that typically remain after secondary treatment and basic disinfection: nutrients, inorganic and volatile organic compounds, synthetic organic constituents, metals, and microbial and miscellaneous constituents. The use of disinfection in addition to the secondary treatment reduces the concentrations of the bacterial and viral stressors; however, the disinfection byproduct, trihalomethanes, may occur. Trihalomethanes, a type of organic compound, can pose a human health risk. Although information is lacking, they may also be a health risk to marine life, such as marine mammals. Cyst-forming pathogenic protozoans, such as *Cryptosporidium* and *Giardia,* are resistant to chlorination and require specialized filtration for removal, and therefore, may be present as a stressor.

Potential receptors in the marine environment are numerous and range from submerged aquatic vegetation, plankton (phytoplankton, zooplankton), and larger aquatic organisms, including invertebrates, fish, reptiles, birds, and marine mammals.

Inorganic constituents, such as nitrogen and phosphorus, and metals, such as iron, are nutrients. However, if they are overabundant, they become stressors. In marine and coastal environments, eutrophication can occur when excess nutrients are present. This can produce harmful algal blooms (red tides), change the natural phytoplankton communities, destroy coral reefs, degrade sea grass and algal beds, and destabilize the overall marine community structure.

Dilution and transport, which are controlled for the most part by ocean currents, are important factors included in the risk analysis. Rapid dilution of effluent can reduce or eliminate potential adverse effects on receptors. In addition, chemical and biological processes that have the potential to affect the level of stressors were included in the risk analysis.

Human Health and Ecological Risk Characterization of Discharge to Ocean Outfalls

The risks associated with discharging effluent using ocean outfalls are low for both human and ecological receptors. There is no drinking-water receptors associated with ocean disposal and therefore, exposure through this pathway is unlikely.

Effluent plumes are rapidly dispersed and diluted by the Florida Current, and flows towards coastal areas are infrequent because of the current's prevailing direction and speed. The concentrations of potential stressors in the effluent plume are low, because of the secondary treatment and disinfection, permit effluent concentration limits, and the subsequent dilution of the effluent after discharge. The distances of the outfalls from shore also decrease risk, with those more distant having the lowest risk. Outfalls that have multiport diffuser systems seem to further reduce risk by dispersing the effluent over a wider area further reducing concentrations of potential stressors.

The treatment level used in ocean disposal does not remove certain pathogenic protozoans that could potentially affect human and ecological health. Pathogenic protozoans may pose a risk to marine mammals that come in contact with the effluent constituents. However, there is a lack of ecological health information on the effects of pathogenic protozoans, as well as other stressors, including metals, endocrine disruptors, and surfactants. Although the concentrations of these compounds may meet required water-quality standards, their effect on biological receptors at low concentrations is not understood. For example, endocrine disruptors operate at extremely low concentrations.

Although chlorinated effluent meets water-quality standards generally within 400 meters of the outfall, the long-term ecological effects of discharging effluent into the ocean are not understood. Currently, there are no long-term monitoring data available for these discharges to describe the ecological impacts or to determine what interaction there is, if any, between outfall constituent effects and terrestrial or coastal sources (such as pesticide runoff or river and groundwater inputs).

DISCHARGE TO SURFACE WATERS

Surface water disposal involves discharging treated wastewater directly into canals, creeks, and estuaries that may be brackish, coastal/saline, or fresh water. The wastewater must receive at least secondary treatment and basic disinfection before discharge. Advanced wastewater treatment is required in some locations.

The use of this option in South Florida varies greatly. Treatment facilities in Hillsborough County rely on this option for about 75% of their total design capacity, whereas facilities in Collier County discharge to surface waters about 1% of their design capacity.

Surface waters that receive discharges vary in physical, chemical, and biological characteristics. As a result, the uses and applications of this disposal option are very site-specific. The estuarine and lagoon systems that receive discharges are typically large expanses of mostly shallow water. Tampa Bay is the largest open estuary in Florida,

encompassing over 400 square miles, with an average depth of 12 feet (Pribble et al., 1999). Sarasota Bay is about 56 miles long and about 300 feet to 4.5 miles wide. It has an average depth between 8 and 10 feet (Roat and Alderson, 1990). The Indian River Lagoon is comprised of several water bodies and stretches for about 156 miles, from south of Daytona Beach to near Palm Beach (Adams et al., 1996). Effluent entering these three major surface water systems must undergo advanced wastewater treatment.

These shallow surface-water bodies include many different and extensive features, such as wetlands, lakes, streams, and canals. In South Florida, many of these surface-water bodies have direct hydrologic connections to the underlying surficial aquifers.

Regulatory Oversight of Discharge to Surface Waters

Florida regulations require that wastewater receive at least secondary treatment and basic disinfection before discharge. Discharge to Class I waters (potable water supply) requires principal treatment, (defined within State requirements as secondary treatment, basic disinfection, filtration and high level disinfection) and discharges to the Tampa Bay, Sarasota Bay, and Indian River Lagoon systems require advanced wastewater treatment. Additional permitting requirements may include that effluent meet certain effluent limits, such as technology-based effluent limits or water-quality-based effluent limits.

State-mandated discharge standards apply for overall pollutants, nitrogen, total suspended solids, and fecal coliforms. Currently, there are no Federal or State limits for protozoan pathogens in wastewater but Florida applies its reclaimed water standard (no more than 5.8 cysts or oocysts per 100 liters for *Cryptosporidium* and no more than 1.4 cysts per 100 liters for *Giardia*) to wastewater discharged to surface waters.

Water-quality standards also apply to discharges to surface waters. The standards are dependent on the end-use class of the receiving surface water. The following classes are relevant to the risk assessment: Class I surface waters may be used as a potable water supply; Class II waters may be used for shellfish propagation or harvesting; Class III water may be used for recreation or can support the propagation and maintenance of a healthy, well-balanced population of fish and wildlife.

Option-Specific Risk Analysis for Discharge to Surface Waters

Because of the variability between and within the receiving surface waters and the regulatory standards governing them, the human health and ecological risks associated with this option are site-specific. To overcome this challenge, surface-water quality was the major parameter used in the risk analysis. The water quality of discharges was compared to the relevant surface-water quality standards. The risk analysis also examined the types of adverse effects that might be anticipated when standards are exceeded.

The potential stressors associated with this option can vary substantially, depending upon the level of treatment applied to the wastewater, but may include nutrients (nitrogen and phosphorus), metals, organic compounds, pathogenic microorganisms, and hormonally active agents. Metals remaining in discharged effluent may be taken up and

bioaccumulate in the food chain to potentially toxic levels. Excess nutrients, particularly nitrogen and phosphorus, are stressors and can have a significant effect on aquatic ecosystems. Excess nutrients can change biological productivity and community structure and cause harmful algal blooms.

Before discharge to surface water, wastewater must undergo secondary treatment and basic disinfection. Stressors in wastewater subjected to secondary treatment and disinfection are similar to those remaining in water bound for ocean disposal, that is, inorganic and volatile organic compounds; synthetic organic constituents; microbial and miscellaneous constituents; and trihalomethanes, a disinfection by-product. However, wastewater discharged to Tampa Bay or to Indian River Lagoon must be treated using advanced wastewater treatment. This typically includes secondary treatment, basic disinfection, nutrient removal (nitrification, denitrification, and phosphorus removal), removal of metals and organic compounds, and filtration to remove cyst-forming protozoans.

In many cases, it is not possible to identify the source of stressors in surface waters. In South Florida, surface-water quality shows significant degradation that may be from urban and agricultural activities (McPherson et al., 2000; McPherson and Halley, 1996). Canal water in urban and agricultural areas commonly contains high concentrations of nutrients, coliform bacteria, metals, and organic compounds when compared to water taken from remote areas. The relative contribution of stressors from these sources compared to the contribution from effluent discharge is poorly understood.

Contamination of Florida's coastal environments with enteric viruses, bacteria, or protozoans is a widespread and chronic problem. Potential causes include the prevalence and high density of septic systems, the predominantly porous and sandy soils, the karst topography, and the hydrologic connections between groundwater and coastal embayments and estuaries (Lipp et al., 2001; Paul et al., 1995). The disinfection of treated effluent before discharge eliminates most pathogens. However, pathogenic protozoans are resistant to disinfection and can persist in effluent.

Under optimal natural conditions, estuaries and lagoons are some of the most productive and diverse habitats. Potential receptors are many and range from microscopic phytoplankton and submerged aquatic vegetation to reptiles, birds, marine mammals, and humans. Threatened and endangered species, such as the West Indian manatee and green and loggerhead sea turtles, can be found in these estuary and lagoon areas. Of the almost 800 fish species known to occur in east-central Florida, more than half use the estuaries and lagoons during part of their life cycle (Gilmore et al, 1981; Gilmore 1995). These shallow waters are important breeding and spawning areas for many fish.

USDWs or water-supply wells may be affected where surface waters that receive effluent have a direct hydrological connection to the groundwater resource. In South Florida, there is a strong interconnection of groundwater and surface water, but the processes and hydrologic fluxes are not well understood. Canals, which frequently receive discharge, are often hydrologically connected to groundwater. Whether the canal is being recharged

or is discharging to groundwater depends on the specific hydrologic conditions, but canals that discharge to groundwater provide a pathway for potential contamination of the underground drinking water supply.

In addition to USDWs, human health exposure can include dermal contact with an affected water body, incidental ingestion of affected water, ingestion of contaminated fish or shellfish (near or removed from the point of discharge), or exposure to toxins from harmful algal blooms. Ecological resources can include fish and other organisms present in the surface water body at the point of discharge as well as those that are removed from but may be affected by the discharge. Also, nutrient loading can adversely impact waters, especially sensitive or impaired waters, and this in turn can destabilize the aquatic system.

Human Health Risk and Ecological Risk Characterization of Discharge to Surface Waters

Effluent discharged to surface waters poses limited risks to human health. The volumes discharged in South Florida are not great, there is a generally higher level of effluent treatment, and the discharges are typically intermittent. Although not required at all treatment plants, AWT is used to remove additional nutrients, organic compounds, and total suspended solids. Facilities using this treatment level frequently are within the standard requirements and may be below detection levels for some effluent constituents (for example, pathogenic microorganisms, inorganic compounds, organic compounds, volatile organic compounds). Pathogenic protozoan levels are generally low and usually within recommended standards. However, some facilities did not meet the recommended levels, even when using filtration. In these cases, there is a potential human health risk, albeit a low risk.

Similarly, the overall risk to ecological receptors is low. This is because most facilities use AWT. For example, based on information collected before and after Tampa Bay implemented AWT, the relative risk of AWT-treated wastewater is lower than the risks posed by wastewater treated to a lesser degree.

Although the risk analysis identified limited human health and ecological risks associated with the discharge of treated effluent to surface-water bodies, the receiving surface waters in many cases are already significantly impacted by contamination from urban and agricultural sources. Additional inputs of nutrients, even from effluent containing low nutrient concentrations, are likely to pose some ecological risk. The cumulative effect of the various inputs into these surface waters is not currently understood. Considerable scientific study and public involvement would be needed to identify and address the problems associated with the relative contributions of different sources of stressors to these estuarine and lagoon waters.

OVERALL RISK ASSESSMENT

The degree of treatment of wastewater before its disposal is an important factor that controls the concentrations of stressors present at the receptor. Risk can be significantly reduced by attenuation factors, such as travel time, distance, filtration by geologic media, dispersion by groundwater or ocean currents, biological degradation, and adsorption.

Pathogenic microorganisms pose a significant human health risk for deep-well injection and discharge to ocean outfalls and, to a lesser extent, aquifer recharge and discharge to surface waters. Filtration can significantly reduce the level for pathogenic protozoans in treated water. However, natural water bodies may contain pathogenic protozoans at levels that exceed the recommended levels.

In addition, nutrient levels can still exceed ambient water-quality levels. Excess nutrients can lead to a variety of ecological problems and can affect entire ecosystems.

Most risk analyses have data and knowledge gaps, and it is important to acknowledge and understand their extent and type. This risk assessment identified data and knowledge gaps for all the options (Exhibit ES-10).

Deep Well Injection	Aquifer Recharge (using RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Site-specific mechanisms of transport (for example, porous media flow vs. conduit flow); locations and connectivity of natural conduits such as solution channels. The fate and transport of pathogenic microorganisms; rates of die-off and natural attenuation. The extent of, if any, reduction in inorganic stressor concentration resulting from local geochemical conditions (for example, rate of biologically mediated transformation of ammonia). Groundwater monitoring data to describe transport to (or within) the Biscayne and surficial aquifers.	Site-specific hydrologic data (for example, horizontal hydraulic conductivities); site-specific estimates of horizontal time-of-travel. Groundwater monitoring data to describe transport within the Biscayne and surficial aquifers. Geospatial data to describe proximity to water-supply wells (especially private wells). Fate and transport of pathogenic micro-organisms still present after disinfection; rates and die-off.	The potential for adverse ecological effects near outfalls. The potential for bioaccumulation (such as metals, persistent organic compounds) through food chains. Water-quality and ecological monitoring downcurrent of outfalls (beyond mixing zones). The potential for changes in ocean currents, sea level, or climate that might affect changes in circulation and transportation patterns or exposure.	The potential for adverse ecological effects near points of discharge. The potential for bioaccumulation (such as metals, persistent organic compounds) through food chains. Water-quality and ecological monitoring data for specific potentially impacted water bodies. The nature and extent of recharge to surficial USDWs.

Exhibit ES-10. Data and Knowledge Gaps

Findings on Risk to Human Health

Overall, the risks to human health are generally low for the four disposal options (Exhibit ES-11). The risks are somewhat higher in all options when there is less treatment or when exposure pathways are short. High-level disinfection, combined with filtration for pathogenic protozoans (using an effective process), significantly reduces risk for all the disposal options. There is an increased risk to human health when the disposal location coincides with recreational uses, such as the ocean (outfall location), canals, streams, bays, and lagoons, and when discharges cause harmful algal blooms. Deep-well injection and aquifer recharge disposal options have the potential to directly impact drinking-water supplies, thereby creating a potential risk to human health.

Deep-Well Injection	Aquifer Recharge (using RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Low where proper siting, construction, and operation result in physical isolation of stressors, with no fluid movement. Low where there have been impacts to deep USDWs; however, exposure of current water supplies is unlikely. Increased risk where short times of travel prevail and where exposure of current water supplies is more likely. In all cases, the risk would be further reduced when injected wastewater is treated to reclaimed water standards.	Low because of high-level disinfection, filtration, and treatment to reclaimed-water standards. Increased risk where filtration is not adequate to meet health-based recommendations for Giardia or Cryptosporidium. Increased risk where chlorination results in high levels of disinfection byproducts (that is, failure to dechlorinate).	Low because of rapid dilution and an absence of drinking-water receptors. The low occurrence (less than 4%) of current flow towards the coast means that human exposure along coastal beaches is reduced. Increased risk where recreational use is near the discharge. Increased risk where discharges contribute to stimulation of harmful algal blooms.	Low because of high-level disinfection and additional treatment (e.g. AWT standards). Increased risk where filtration is not provided or is inadequate to meet health-based recommendations for Giardia or Cryptosporidium. Increased risk where surface-water discharges are near recreational use of water bodies. Increased risk where discharges contribute to stimulation of harmful algal blooms.

Exhibit ES-11. Estimate of Risk to Human Health Associated With Each Wastewater Disposal Option

Findings on Risk to Ecological Health

The risk to the ecological health of surface waters is very low for the deep-well injection and aquifer recharge options (Exhibit ES-12). Similarly, the risk to surface waters receiving treated discharge directly is low because of the advanced level of treatment the wastewater receives. However, irrespective of the contribution of contaminants by treated

municipal wastewater, many surface waters in South Florida are considered to be in an impaired status. When a discharge is in close proximity to an impaired water body, there is a higher ecological health risk.

Deep-Well Injection	Aquifer Recharge (using RIBs)	Discharge to the Ocean	Discharge to Surface Waters
The risks from chemical constituents are low but not zero because of possible hydrologic connectivity. Risks related to pathogenic microorganisms are low to moderate for Dade and Brevard counties because of lack of disinfection and filtration. Microbial risk is low in Pinellas County because of use of disinfection and filtration.	Low because of possibility of hydrologic connectivity between wetlands and surficial aquifer. Cumulative and long-term effects are not known.	Low because of the concentrations of nutrients in the discharged effluent. No ecological monitoring is currently conducted. Cumulative and long-term effects are not known.	Low because of the concentrations of nutrients in the discharged effluent.

Exhibit ES-12. Estimate of Risk to Ecological Health Associated With Each Wastewater Disposal Option

Risks are also considered low for ocean outfalls in the areas outside the mixing zones and for marine ecosystems that may be impacted by deep-well injection.

Discharges from ocean outfalls and discharges to surface waters will have increased risk if the discharges cause harmful algal blooms or result in bioconcentration in food webs. Construction of new ocean outfalls may increase risk to coral reefs.

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1.0 INTRODUCTION

This report provides a relative risk assessment of four management options for treated municipal wastewater in South Florida. The four wastewater management options evaluated by the study are the following:

- Disposal via deep-well injection
- Aquifer recharge
- Ocean outfall disposal
- Disposal via surface-water discharge.

The study described in this report compiles new and existing sources of information and provides an evaluation of potential human health and ecological risks associated with the four wastewater management options studied.

1.1 Congressional Mandate

This study was conducted in response to a Congressional mandate included in the fiscal year 2000 appropriation language:

> *Within available funds, the conferees direct EPA to conduct a relative risk assessment of deep-well injection, ocean disposal, surface discharge, and aquifer recharge of treated effluent in South Florida, in close cooperation with the Florida Department of Environmental Protection and South Florida municipal water utilities.*

1.2 Purpose

There is an immediate need for information that will assist EPA, Florida regulatory agencies, and concerned stakeholders to determine an appropriate course for proposed revisions to rules concerning Class I underground injection wells in South Florida. These wells inject treated wastewater below the lower most underground source of drinking water and the surficial aquifers that provide much of Florida's drinking water. Groundwater monitoring information indicates that the injected wastewater has migrated from the injection zone into overlying layers of the subsurface. Stakeholders have expressed concern that such migration may compromise drinking-water sources.

This risk assessment will provide information that regulators, utilities, and communities in South Florida can use to make informed judgments and decisions regarding wastewater management.

Wastewater management involves complex and interrelated issues, many of which are beyond the scope of this risk assessment. Examples of such complex issues include modified wastewater management approaches, changes in the required level of treatment, encouraging flexibility in use of management options and backup methods, economic comparisons relating risks to management costs, and consideration of water conservation

and water quantity. However, a risk assessment that takes all of these issues into account would far exceed the scope and available resources for this study. Accordingly, this risk assessment has been designed to address the Congressional mandate directly. It does not attempt to assess the full range of risk-related considerations that figure into wastewater management decision-making.

Because the purpose of the study is to characterize potential risks to human health and the environment, this study does not incorporate an analysis of cost-effectiveness. As a result, operational lifespan, implementation and maintenance costs, and other economic issues will not be assessed. However, the potential for system failure for each of the four wastewater management options will be addressed, with particular emphasis on the potential for failure of deep injection wells.

The geographic area covered in this study includes areas south of a line drawn from the northern end of Brevard County west to the northern end of Pinellas County (figure 1–1). In an effort to focus data collection within areas exhibiting the most urgent wastewater management needs, the heavily populated counties of Dade, Palm Beach, Broward, Pinellas, Brevard, Sarasota, and Hillsborough were selected.

EPA acknowledges that this study area may or may not be entirely consistent with what has been traditionally considered as South Florida. However, EPA collected data and conducted this risk assessment within a study area that provides for the fullest and most informative evaluation of the human health and ecological risks associated with the four studied management options.

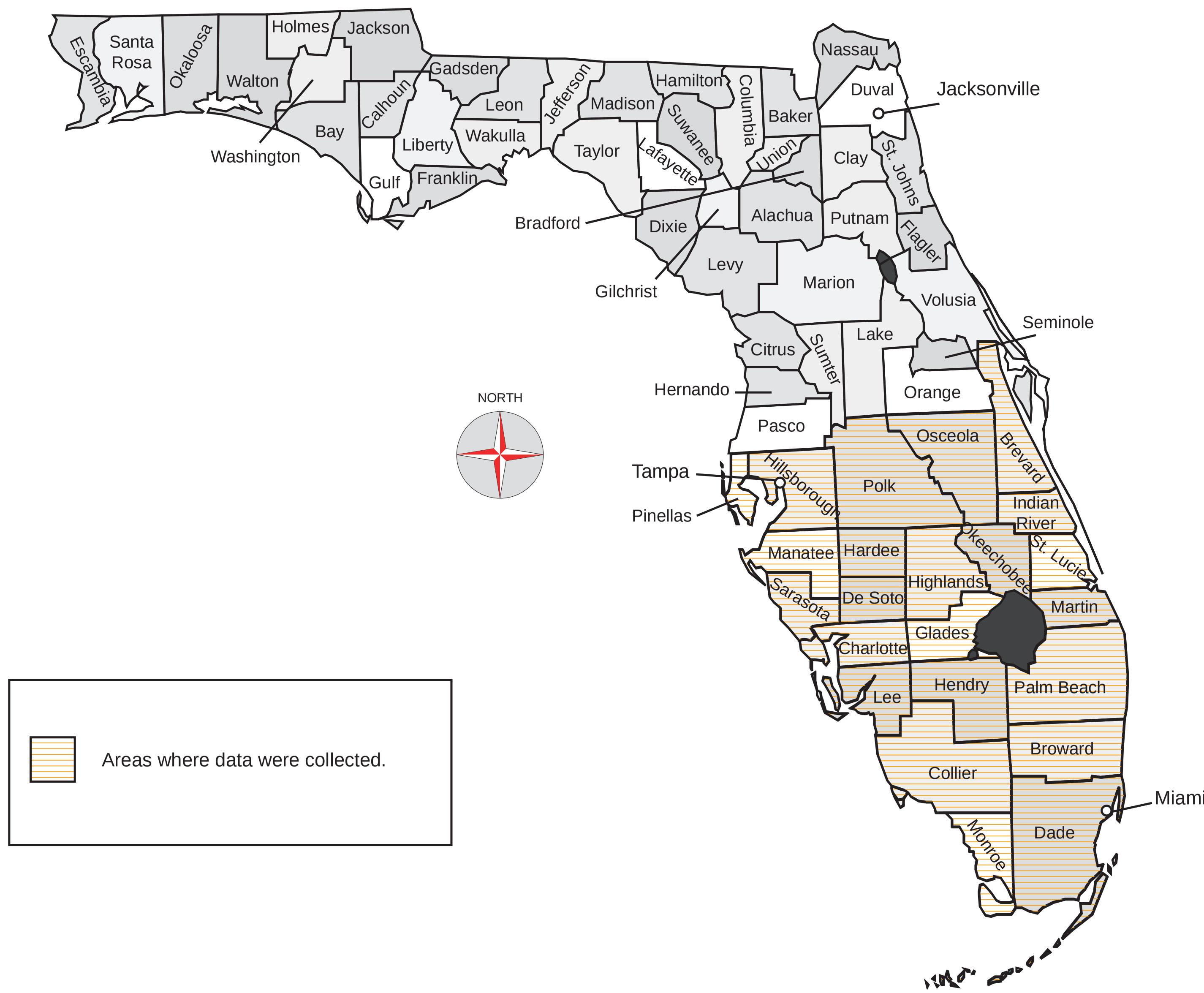


Figure 1-1. The South Florida Study Area

2.0 BACKGROUND

This report analyzes risk in areas that are the most densely populated or that exhibit hydrogeologic conditions that will affect the risks associated with different wastewater management options. Wastewater management needs in South Florida are most critical in southeast Florida and in the more densely populated cities along both the Atlantic and Gulf coasts of Florida. The interior of South Florida and the Everglades have the lowest density of wastewater treatment plants. The distribution of public municipal wastewater treatment plants in South Florida is shown in Figure 2-1 (FDEP, 2002). Municipal wastewater treatment plants reviewed for this study are listed in Table 2-1, according to the county in which they are located.

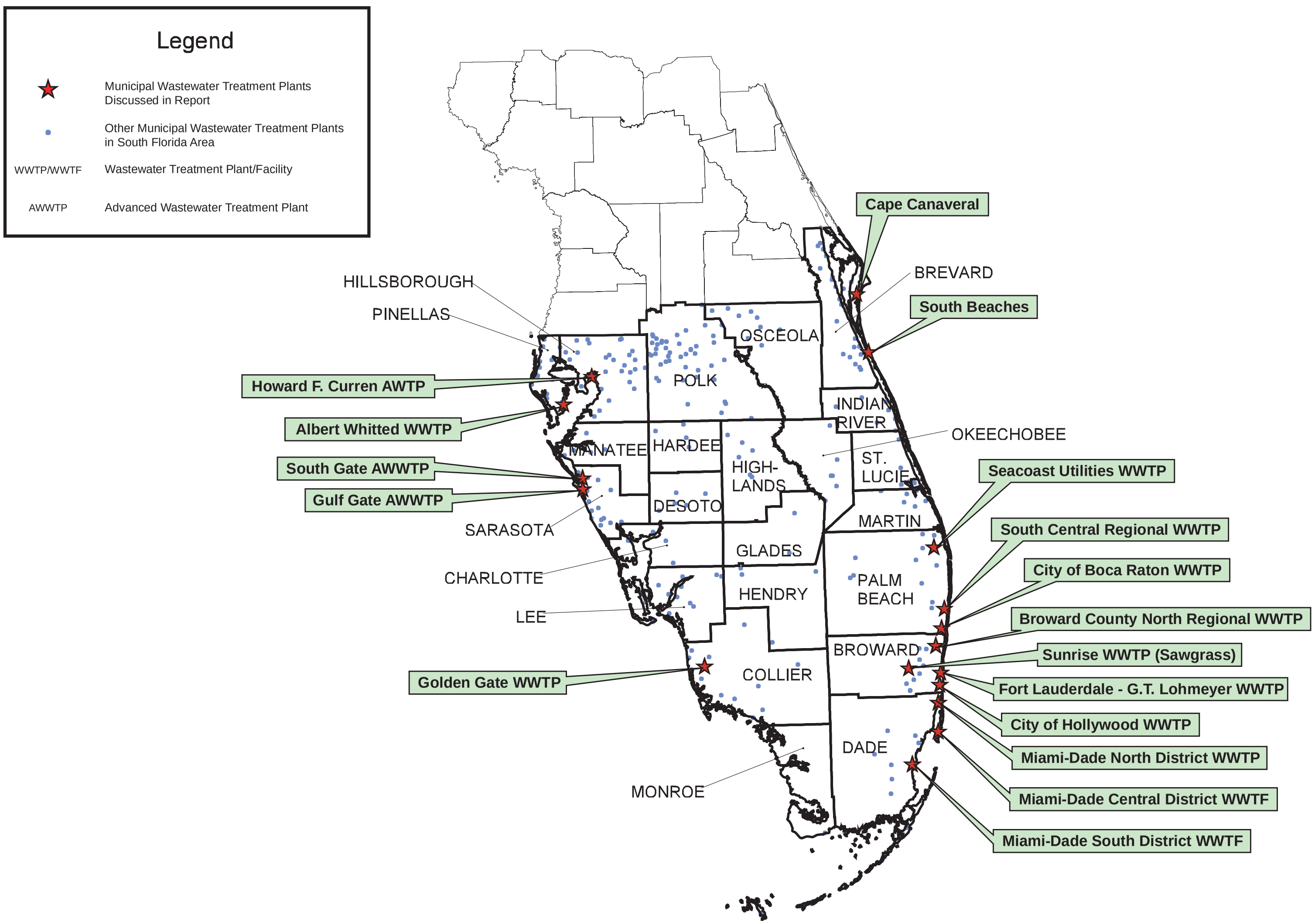
The tables in Appendix 1 provide data on the water quality of treated wastewater. Other data used in this study are also presented in Appendix 1, including data on the following topics:

- Chemical constituents (Appendix Table 1-1)
- The Southeast Florida Outfall Experiment or SEFLOE (Appendix Table 1-2)
- Microorganisms in wastewater (Appendix Table 1–3)
- Groundwater monitoring of fecal coliforms (Appendix Tables 1-4 and 1-5)
- Injection well locations, capacities, and treatment (Appendix Table 1-6).

2.1 Wastewater Management Options Used in South Florida

Wastewater treatment facilities often incorporate multiple management options to ensure continuous operation. The capacity of South Florida counties to manage treated wastewater using different management options is illustrated in Figure 2-2. Discharge volume capacities, not actual flow volumes, are represented in this figure. Information for this figure was obtained from the Florida Department of Environmental Protection (DEP) wastewater facilities database (FDEP, 2002) and the Florida DEP (personal communication, Kathryn Muldoon, February, 2002). Note that the DEP database does not always distinguish between Class I deep injection wells and Class V shallow injection wells.

Broward, Palm Beach, and Dade counties discharge the majority of their treated wastewater through ocean outfalls and deep injection wells. In Hillsborough, Sarasota, Pinellas, and Collier counties, aquifer recharge can be done using reclaimed water, surface water discharge, or deep injection well disposal, depending on irrigation needs and weather conditions. Facilities in Brevard County discharge reclaimed water to the Indian River Lagoon only when there is no demand for irrigation water. Dade, Broward, and Palm Beach counties primarily use Class I deep injection wells and ocean outfalls to dispose of wastewater treated to secondary standards, but they also reuse a small amount of reclaimed water.



Source: Florida Department of Environmental Protection, 2001

Figure 2-1. Municipal Wastewater Treatment Plants in South Florida

Table 2-1. Wastewater Treatment Plants Discussed in This Report

Wastewater Treatment Plant	County	Type of Disposal	Treatment	Design or Current Capacity in mgd)[a, b]
Cape Canaveral	Brevard	Surface water, reuse	Secondary and High-level disinfection	1.80
South Beaches	Brevard	Surface water, reuse, deep-well injection[c]	Secondary and High-level disinfection	12.4
City of Fort Lauderdale[d]	Broward	Deep-well injection	Secondary	43
City of Sunrise (Sawgrass)[d]	Broward	Deep-well injection	Secondary	13
City of Hollywood[d, e]	Broward	Some Reuse, Ocean outfall	Secondary and High-level disinfection	42
Broward County North Regional[d, e]	Broward	Some Reuse, Ocean outfall, deep-well injection	Secondary and High-level disinfection	80
Golden Gate (Naples)[d]	Collier	Reuse	Secondary and High-level disinfection	0.95
Miami-Dade South District[d]	Dade	Deep-well injection	Secondary	112.5
Miami-Dade Central District[e]	Dade	Ocean outfall, deep-well injection	Secondary	121
Miami-Dade North District[d, e]	Dade	Ocean outfall, deep-well injection Some reuse	Secondary and High-level disinfection	112.5
Howard F. Curren (Tampa)	Hillsborough	Surface water, reclaimed	Advanced	96
Seacoast[d]	Palm Beach	Reuse and Deep-well injection	Secondary and High-level disinfection	12
Boca Raton[d, e]	Palm Beach	Some reuse, Ocean outfall	Secondary and High-level disinfection	20
South Central Regional/Delray Beach[d, e]	Palm Beach	Ocean outfall	Secondary and High-level disinfection	24
Albert Whitted	Pinellas	Deep-well injection, Some reuse[c]	Secondary and High-level disinfection	12.4
Gulf Gate[d]	Sarasota	Surface water	Advanced	1.80
South Gate[d]	Sarasota	Surface water, reuse	Advanced	1.36

[a] mgd = million gallons per day
[b] FDEP, 2001
[c] US EPA, 1997
[d] Englehardt et al., 2001
[e] Hazen and Sawyer, 1994

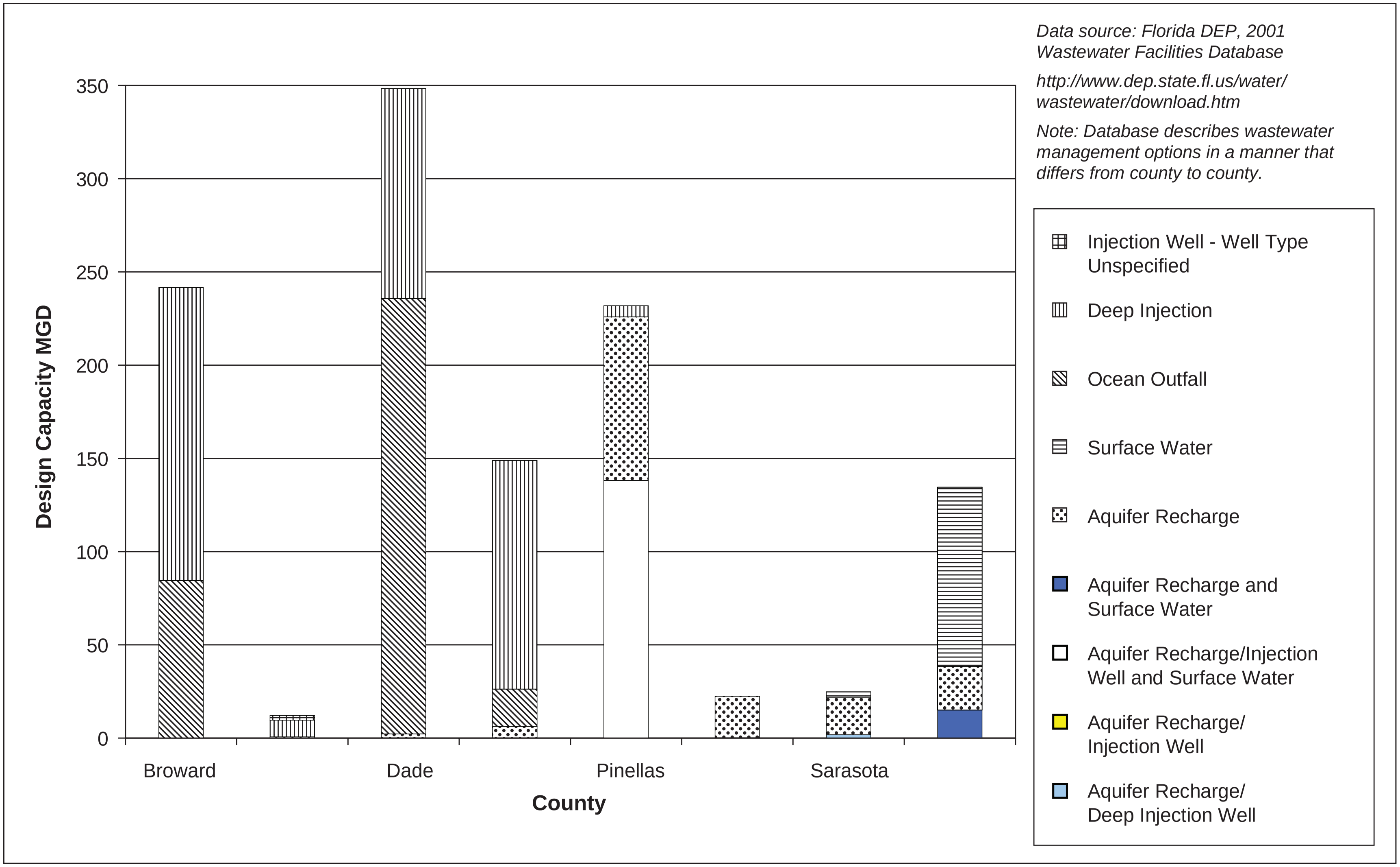


Figure 2-2. Wastewater Management Options and Design Capacities for Counties in South Florida

Approximately 1.2 million people are served by the Dade Central and Dade North District wastewater treatment plants, which discharge a total of approximately 230 million gallons per day (mgd) to the open ocean (Marella, 1999). Both outfalls have multi-port diffusers. In Broward County, approximately 80 mgd are treated and discharged to the Atlantic Ocean (Marella, 1999). (Note: This is 1995 data and may not reflect the impact of Class I injection wells that became operative in 1996; at this time, discharge to the ocean may have been diverted to the Class I wells.)

Wastewater treatment facilities located in Brevard, Collier, and Pinellas counties are permitted to discharge to surface waters. However, these facilities often use other management options, such as spray irrigation, in conjunction with discharges to surface water. When there is no need for spray irrigation, treated wastewater may be discharged into a surface-water body or injection well. For example, the South Beaches wastewater treatment facility in Brevard County discharges into the Indian River Lagoon when there is no demand for irrigation water.

In Sarasota, two wastewater treatment plants, Gulfgate and Southgate, discharge into freshwater canals (Phillippe Creek and Methany Creek). These eventually drain to Roberts Bay (Camp, Dresser, McKee, 1992; Roat and Alderson, 1990). The Sarasota facilities have no alternative for discharging wastewater and thus treat to advanced wastewater standards at all times.

In Pinellas County, the City of Clearwater and the City of Bellaire have permits to discharge to surface waters. Bellaire discharges to Clearwater Bay, and the City of Clearwater Northeast Wastewater Treatment Plant discharges to Tampa Bay. These facilities also have the option of reusing treated or reclaimed wastewater.

Each of the four studied methods of managing treated wastewater is described briefly below and in more detail in Chapters 4 through 7.

2.1.1 Class I Deep-Well Injection

Class I underground injection wells are used to dispose of secondary treated municipal wastewater to deep geologic strata. Injection zones are selected so that they are situated beneath the lowermost geologic formation that contains an underground source of drinking water (FDEP, 1999). An underground source of drinking water (USDW) is defined as an aquifer, or portion of an aquifer, with a sufficient quantity of ground water to supply a public water system and containing a total dissolved solids concentration of less than 10,000 milligrams per liter (mg/L) (FDEP, 1999; 40 CFR 144.3).

Class I wells are located throughout the South Florida study area, including Dade, Brevard, and Pinellas counties. Wastewater is injected at depths ranging from 650 to 3,500 feet below the land surface (US EPA, 1998). Management of treated municipal wastewater by Class I deep-injection wells constitutes approximately 20 percent (0.44 billion gallons per day) of the total wastewater disposal capacity in Florida, based on design capacity (FDEP, 1997).

Movement of injected fluids into USDWs by Class I is prohibited by Federal and State requirements. A major purpose of the Federal and State regulations is to protect the quality of USDWs by regulating the construction and operation of injection wells to ensure that the injected fluid remains in the injection zone. 40 CFR 146 establishes criteria and standards that apply to the construction, operation, and monitoring of Class I wells. Many specific regulations governing the construction and operation of injection wells serve to prevent fluid movement into USDWs.

Chapter 4 discusses deep-well injection in greater detail and examines potential human and ecological risks associated with this wastewater management option.

2.1.2 Aquifer Recharge

Aquifer recharge involves the infiltration of water into the ground and includes such practices as infiltration basins, percolation ponds, wetland treatment systems, and irrigation of turf, landscaped areas, and crops. Ultimately, these result in recharging groundwater aquifers and may benefit wetlands habitat as well. For these reasons, aquifer recharge using reclaimed wastewater is widely considered to be a beneficial reuse of treated wastewater.

Under the State of Florida's regulatory framework (the Florida Administrative Code [FAC]), Chapter 62-600 contains definitions of secondary treatment, disinfection levels, and requirements for effluent disposal systems; and Chapter 62-610 contains detailed requirements for a wide range of reuse options; and that Chapter 62-611 regulates discharges to wetlands.

Chapter 5 discusses aquifer recharge in greater detail and examines potential human and ecological risks associated with this wastewater management option. Wastewater treatment and disinfection is discussed in detail in Section 2.3.

2.1.3 Ocean Outfalls

There are six existing publicly owned treatment facilities that use ocean outfalls for management of treated wastewater in South Florida (Hazen and Sawyer, 1994). A seventh ocean outfall with limited discharge capacity is located in the Florida Keys, according to Hoch et al. (1995). The six major ocean outfalls in southeast Florida discharge effluent from the Dade Central District, Dade North District, City of Hollywood, Broward County, Delray Beach, and Boca Raton treatment facilities. The outfalls discharge secondary-treated chlorinated wastewater effluent at ocean depths ranging from 27.3 meters to 32.5 meters. Discharge points are located between 1,515 and 5,730 meters offshore.

The southeast Florida outfalls discharge along the western boundary of the Florida Current, a tributary of the Gulf Stream. The Florida Current is a fast-flowing current, with maximum current speeds occurring in the Florida Strait between southeast Florida and the Bahamas, in the vicinity of the southeast Florida outfalls. Maximum current

speeds measured at the outfall sites during the Southeast Florida Outfall Experiment (SEFLOE) were upwards of 60 to 70 centimeters per second. The speed and strength of the Florida Current causes effluent plumes to be rapidly dispersed (Huang et al., 1998; Proni et al., 1994; Proni et al., 1996; Proni and Williams, 1997).

Chapter 6 discusses ocean outfall disposal in greater detail and examines potential human and ecological risks associated with this wastewater management option.

2.1.4 Surface-Water Discharges

Surface-water disposal consists of discharge of treated municipal wastewater into estuaries, lagoons, canals, rivers, or streams. Surface-water discharge of treated municipal wastewater is limited and discouraged in South Florida because of potential ecological and health concerns. There are no known permitted discharges into fresh water lakes or ponds in South Florida (personal communication, K. Muldoon, Florida DEP). Discharge into canals is the predominant form of surface-water discharge (Marella, 1999; Kapadia and Swain, 1996; Englehardt et al., 2001; personal communication, K. Muldoon, Florida DEP). Discharges into estuaries may also be permitted. Tampa Bay, Roberts Bay, and the Indian River Lagoon each receive surface-water discharges through discharges into canals or estuaries that empty into these coastal embayments (City of Tampa Bay Study Group, 2001).

Wastewater intended for discharge to certain coastal embayments generally must be treated to advanced wastewater treatment standards. Advanced wastewater treatment refers to secondary treatment, plus further removal of nitrogen and phosphorus to attain the 5mg/L $CBOD_5$, 5 mg/L TSS, 3 mg/L total nitrogen (as N) and 1 mg/L total phosphorus (as P) or treatment to water-quality-based effluent standards. Discharge to Tampa Bay and Indian River Lagoon areas must be treated to these standards. While it represents a reasonable assumption for the level of treatment required for surface water discharges, it is not a formal statewide requirement.

Most surface-water discharges are also subject to water-quality-based effluent limits (WQBELs) established using the processes outlined in Chapter 62-650, F.A.C. WQBELs generally include nutrient limits for nitrogen and phosphorus established to protect water quality in the receiving waters. This may include very stringent nutrient limits. While filtration may be needed to achieve the TSS limit, it is not specifically designed to remove pathogenic protozoa, nor is it required to do so. In addition, any new or expanded surface water discharge is subject to Florida’s Antidegradation Policy.

Chapter 7 discusses surface water discharges in greater detail and examines potential human and ecological risks associated with this wastewater management option.

2.2 Drinking Water in South Florida

Concerns about potential effects on drinking-water quality lie at the heart of stakeholder anxieties regarding management of treated wastewater. In order to evaluate potential

human health risks associated with these management options, it is important to understand the sources of drinking water used by South Florida communities.

The USGS National Water Quality Assessment Program (NAWQA) has estimated that ground water accounts for approximately 94 percent (872 million gallons per day, or mgd) of the water used by 5.8 million people in South Florida as of 1990, generally from wells less than 250 feet deep in the surficial aquifer. The remaining 6 percent of drinking water is supplied by surface water sources (McPherson et al., 2000). (Note that the NAWQA report encompasses an area of South Florida that is approximately similar to the area of this risk study, with the exclusion of a portion of Sarasota County and the inclusion of several other counties not addressed in this study.)

Most Community Water Systems within the geographic area covered by this study are supplied by ground water. As of October 18, 2001, a total of 133 Community Water Systems in five counties (Brevard, Broward, Dade, Palm Beach, and Pinellas Counties) provide ground water from their own wells or purchase ground water from nearby utilities. Current figures indicate that only 12 Community Water Systems provide surface water to their customers (US EPA, 2001).

Water suppliers that use ground water generally use either the Floridan Aquifer or the Biscayne Aquifer as a water source. The Biscayne Aquifer underlies 4,000 square miles in Broward, Dade, and Palm Beach Counties. The Miami-Dade Water and Sewer Department withdraws approximately 330 mgd from the Biscayne Aquifer for distribution to the City of Miami and surrounding communities. The City of Fort Lauderdale draws water from the Biscayne Aquifer as well. The City of St. Petersburg, in Pinellas County, purchases ground water (from the Floridan Aquifer).

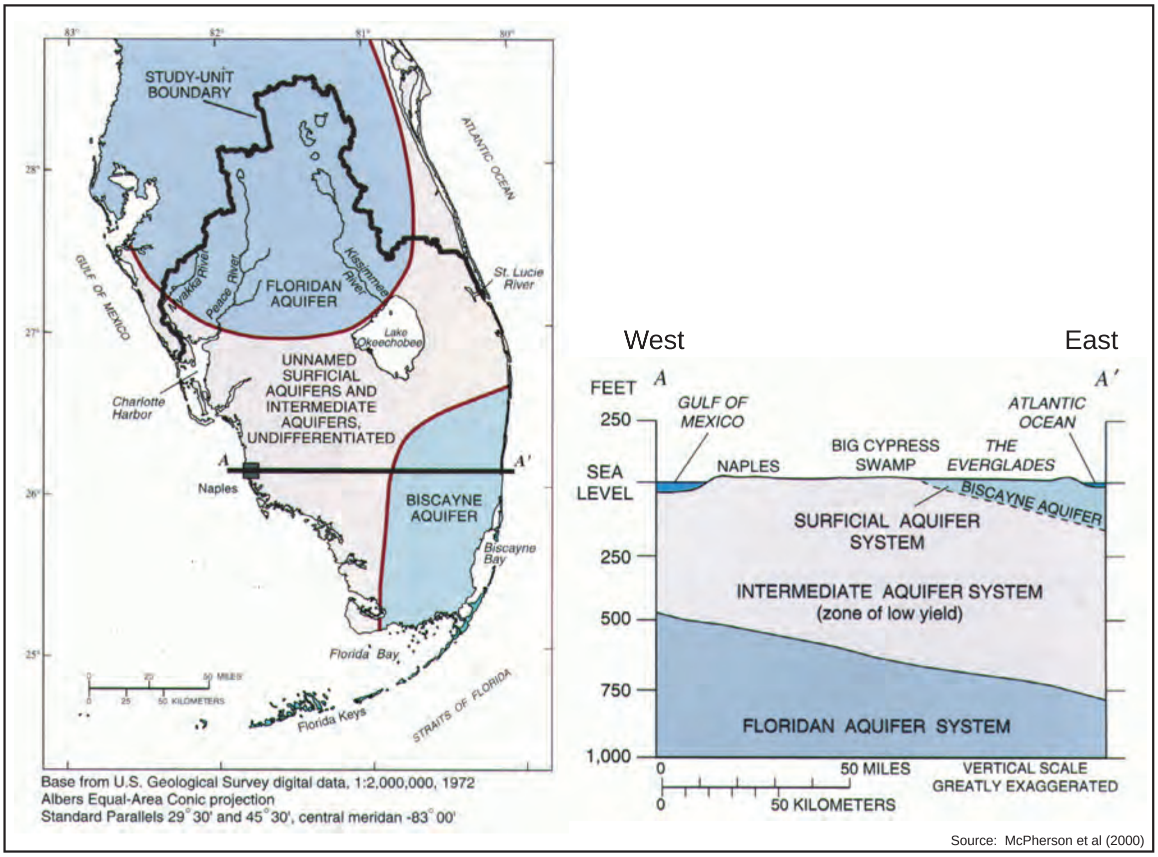


Figure 2-3. Hydrologic Profile of South Florida Aquifer System

2.2.1 Floridan Aquifer System

The Floridan Aquifer System underlies approximately 100,000 square miles in southern Alabama, southeastern Georgia, southern South Carolina, and all of Florida. Several large cities in the southeastern United States use the Floridan Aquifer as a drinking water source, including St. Petersburg in Florida. In addition, the aquifer is a source of water for many smaller communities and rural areas. During 1985, approximately three billion gallons per day of fresh water were withdrawn from the Floridan Aquifer (USGS, 2000).

In most places, the Floridan Aquifer can be divided into two aquifers (the Upper and Lower Floridan aquifers) with a confining layer of material in between. The hydraulic properties and geology of the Upper Floridan aquifer are better known than the properties of the Lower Floridan because the Lower Floridan occurs at greater depths than the Upper Floridan, and therefore fewer borehole data are available. Most of the fresh water that is withdrawn from the Floridan Aquifer is pumped from the Upper Floridan.

Since 1988, approximately 320 million gallons per day of wastes are injected into disposal wells that empty into the Lower Floridan; about 97 percent of this volume is municipal wastewater.

2.2.2 Biscayne Aquifer System

The Biscayne Aquifer system, the main source of water for Dade, Broward, and southeastern Palm Beach Counties, underlies approximately 4,000 square miles (USGS, 2000). In 1985, approximately 786 million gallons per day of fresh water were withdrawn from the aquifer for all purposes; withdrawals as of 1990 were somewhat greater. About 70 percent of the water is estimated to be withdrawn for public supply. Major population centers that depend on the Biscayne aquifer for water supply include Boca Raton, Pompano Beach, Fort Lauderdale, Hollywood, Hialeah, Miami, Miami Beach, and Homestead. Water from the Biscayne Aquifer also supplies the Florida Keys with water transported from the mainland by pipeline.

Because the Biscayne Aquifer lies at shallow depths and is highly permeable, it is highly susceptible to contamination. According to the USGS, this aquifer is the sole source of drinking water for 3 million people.

The Biscayne Aquifer lies on top of the Floridan Aquifer, and is separated from that deeper aquifer by approximately 1,000 feet of low-permeability clay deposits. The Biscayne Aquifer ranges in thickness from a few feet in the west to about 240 feet near the Florida coast.

2.2.3 Surficial Aquifer

In areas of South Florida outside the Biscayne Aquifer, the unnamed surficial aquifer is used locally for community and public water supply.

2.2.4 Drinking-Water Quality in South Florida Communities

The City of St. Petersburg purchases ground water pumped by the City of Tampa from the Floridan Aquifer. Routine monitoring reported in the city’s 2000 Water Quality Report indicates that the water system produces drinking water that meets all Federal and State drinking water standards. According to data in the report, the concentrations of all constituents in the water were below Federal and State Maximum Contaminant Levels (MCLs). The maximum concentration of arsenic (MCL 50 ug/l) was 3.3 ug/l and the maximum concentration of nitrate (MCL for nitrate is 10 mg/l) was 0.05 mg/l during the latest round of water quality testing.

Dade County withdraws approximately 330 million gallons per day of fresh water for distribution to Miami and surrounding communities. The Miami-Dade 2000 Water Quality Report indicates that concentrations of all constituents detected in the water were below Federal and State MCLs. The concentration of nitrate as measured at nine water

treatment plants ranged from ND (not detected) to 7 mg/l; the concentration of arsenic at the nine plants ranged from ND to 2 ug/l.

The Biscayne Aquifer is used by millions as a source of drinking water and is suitable for most other purposes. In some areas in Broward county and portions of Dade County, however, the water is colored as a result of decomposing organic material in the aquifer. While this coloration is an aesthetic issue, it does not present a risk to human health.

Canals managed by the South Florida Water Management District have been used in South Florida to control flooding and drainage. These canals are hydraulically connected to the Biscayne Aquifer and present a potential contamination route. Major sources of contamination to the Biscayne Aquifer include salt water intrusion and infiltration of contaminants from the canal system. Other potential sources of contamination include the infiltration of substances spilled on the ground, fertilizer carried in surface runoff, septic tanks, and improperly constructed disposal wells.

2.3 General Description of Wastewater Treatment

2.3.1 Wastewater Treatment Methods Used in Florida

In the State of Florida, there are four primary means of managing treated municipal wastewater:

- Release of treated wastewater effluent to ocean outfalls
- Release of treated wastewater effluent to surface waters
- Aquifer recharge of reclaimed wastewater
- Underground injection of treated wastewater into subsurface geologic formations using Class I injection wells.

A precise knowledge of the regulation, treatment, and disinfection of municipal wastewater is important for evaluating and understanding human health and ecological risks associated with the four different wastewater management alternatives. Treatment and regulatory oversight are two critically important risk management tools that greatly affect the final risk determination.

Regulations governing water-quality treatment and the quality of water in receiving water bodies are important because they require that wastewater be treated to a certain standard that depends on its management method; therefore, treated wastewater is likely to have a composition that falls within a predictable range. Risk assessment is made simpler when the quality of treated wastewater can be expected to be fairly predictable. Furthermore, regulations concerning water quality are based upon rational evidence that human health or ecological entities would be better protected if such standards were met. Risk assessment is made easier when such standards exist. In addition, comparison of risks of different management options may depend to a large extent upon the kind and amount of treatment required. Regulations for treatment of wastewater and standards for receiving

waters are discussed generally in Chapter 3 and in Chapters 4 through 7 for each wastewater management option.

In order to understand how wastewater treatment reduces risks, it is helpful to understand the composition of untreated wastewater and to compare it with that of treated wastewater. Typical untreated (raw) municipal wastewater contains a variety of constituents, the concentration of which depends on the type and size of commercial and industrial flows added and on the amount and quality of ground water infiltrating into the sewage system. For instance, food-handling wastewater (for example, from food stores and restaurants) can have higher concentrations of organic matter than typical domestic wastewater, while industrial flows may exhibit higher levels of metals. Untreated wastewater typically contains notably high concentrations of pollutants, including organic and inorganic compounds, microorganisms and metals (WPCF, 1983; Metcalf and Eddy, 1991; Richardson and Nichols, 1985; Krishnan and Smith, 1987; and Williams, 1982). Table 2-2 lists typical concentrations and ranges of several raw wastewater constituents as well as the percent removal of these constituents that can be achieved using primary and secondary wastewater treatment methods.

Table 2-2. Typical Levels of Constituents in Wastewater and Percent Removal Using Treatment (Primary and Secondary)

Constituent	Raw Wastewater (mg/L)		Percent Removal		Secondary Effluent (mg/L)	
	Typical	Range	Primary	Secondary	Typical	Range
BOD_5	220	110–400	0–45	65–95	20	10–45
COD	500	250–1000	0–40	60–85	75	35–75
TSS	220	100–350	0–65	60–90	30	15–60
VSS	165	80–275	—	—	—	—
NH_4-N	25	12–50	0–20	8–15	10	<1–20
NO_3 + NO_2-N	0	0	—	—	6	<1–20
Org-N	15	8–35	0–20	15–50	4	2–6
TKN	40	20–85	0–20	20–60	14	10–20
Total N	40	20–85	5–10	10–20	20	10–30
Inorg P	5	4–15	—	—	4	2–8
Org P	3	2–5	—	—	2	0–4
Total P	8	6–20	0–30	10–20	6	4–8
Arsenic	0.007	0.002–0.02	34	28	0.002	—
Cadmium	0.008	<0.005–0.02	38	33–54	0.01	<0.005–6.4
Chromium	0.2	<0.05–3.6	44	58–74	0.09	<0.05–6.8
Copper	0.1	<0.02–0.4	49	28–76	0.05	<0.02–5.9
Iron	0.9	0.10–1.9	43	47–72	0.36	0.10–4.3
Lead	0.1	<0.02–0.2	52	44–69	0.05	<0.02–6.0
Manganese	0.14	0–0.3	20	13–33	0.05	—
Mercury	0.001	<0.0001–0.0045	11	13–83	0.001	<0.0001–0.125
Nickel	0.2	—	—	33	0.02	<0.02–5.4
Silver	0.022	0.004–0.044	55	79	0.002	—
Zinc	1.0	—	36	47–50	0.15	<0.02–20

Note: Partially adapted from WPCF, 1983; Metcalf and Eddy, 1991; Richardson and Nichols, 1985; Krishnan and Smith, 1987; and Williams, 1982.
[Please insert note explaining meaning of cells occupied by em dash (—): for example, "— = no data was collected for this consitutent."]

Raw wastewater must be treated at a wastewater treatment facility prior to discharge, regardless of the disposal method. Wastewater treatment facilities provide what is known as primary, secondary, and/or tertiary or advanced treatment. The dividing boundaries between these levels of treatment can become blurred, especially in recent years with the development of new processes that can accomplish several treatment objectives at once. As Table 2-2 indicates, percent removal of raw wastewater constituents depends largely on the level of treatment, though it is important to note that even primary treatment alone will produce a much cleaner effluent. Treatment facilities are designed to meet national, state and local treatment standards, and the processes are chosen on the basis of those standards and local wastewater composition. Most importantly, the level of treatment is dictated by the disposal or reuse option chosen.

Wastewater treatment and disinfection methods and levels are summarized below. A summary of treatment methods used in South Florida is presented in Table 2-4. Disinfection methods are summarized in Table 2-5. Treatment and disinfection for different wastewater management options are discussed fully in Chapters 4 through 7.

2.3.2 Definitions of Wastewater Treatment Methods and Levels of Disinfection

Primary wastewater treatment generally consists of physical separation of solids from the wastewater and includes screening and grinding operations, as well as sedimentation.

Secondary wastewater treatment provides for the removal of suspended solids and biodegradable organic matter using chemical and biological processes before discharge to receiving waters. Secondary treatment, which often includes basic disinfection (described below), is required for ocean discharge but disinfection is not required for underground injection via Class I injection wells. Pursuant to the Clean Water Act, EPA first issued its definition of secondary treatment in 1973. Current Federal standards for secondary treatment are included in 40 CFR Part 133 and presented in Table 2-3. The State's requirements for secondary treatment are contained in Chapter 62-600, F.A.C.

Table 2-3. National Standards for Secondary Treatment

Parameter	Minimum % Removal	Maximum 7-Day Avg.	Maximum 30-Day Avg.
BOD_5, mg/L	85	45	30
TSS, mg/L	85	45	30
pH, units	Within range of 6.0 to 9.0 at all times		

Most secondary treatment of domestic wastewater is accomplished using activated sludge processes. These processes utilize microorganisms already present in the wastewater. The wastewater is aerated and mixed vigorously, which increases contact between the microorganisms and both organics and oxygen. The microorganisms oxidize the dissolved and suspended organics into carbon dioxide and water. Inorganic and organic nitrogen, sulfur, and phosphorus are oxidized to nitrates, sulfates, and phosphates. Some suspended organic and mineral solids are not broken down; these are settled out in

clarifiers or a clarification step. The liquid flows out of the top of the clarifier, and after undergoing whatever final treatment is required, it is on its way out of the wastewater treatment facility.

Principal treatment and disinfection (more advanced secondary) requires secondary treatment and high-level disinfection. The reclaimed water must meet a standard of 5.0 mg/L of total suspended solids before application of the disinfectant and total nitrogen is limited to 10 mg/L. Filtration is also required for total suspended solids control, increasing the ability of the disinfection process to remove protozoan pathogens.

Reclaimed water treatment requires secondary treatment, filtration, and high-level disinfection. The quality of water discharged via reclaimed water treatment systems is intended to be high so that it may be reused. Reclaimed water treatment is required if wastewater is being reclaimed for reuse. A standard of 5.0 mg/l TSS (a single sample maximum applied after the filter and before the application of the disinfectant) is required for reuse projects permitted under Part III of Chapter 62-610, F.A.C. Part III imposes a number of additional operational and reliability requirements.

Advanced (or tertiary) wastewater treatment is a term of art that simply means wastewater treatment beyond secondary treatment such as processes that are used if there are requirements to remove specific components, such as nitrogen and phosphorus, which are not removed by the secondary treatment.

Basic disinfection must result in not more than 200 fecal coliforms per 100 ml of reclaimed water of effluent sample. Where chlorine is used, facilities must provide for rapid and uniform mixing and a total chlorine residual of at least 0.5 milligram per liter shall be maintained after at least 15 minutes contact time at the peak hourly flow. Higher residuals or longer contact times may be needed. (See Rule 62-600.440(4) F.A.C.)

High-level disinfection includes additional removal of total suspended solids (TSS) beyond secondary treatment, to achieve a TSS concentration of 5.0 mg/L or less before the application of disinfectant, in order to maximize disinfection effectiveness. It results in reclaimed water in which fecal coliform values (per 100 ml of sample) are below detectable limits (at least 75% of all observations: with no single sample above 25/100 mL. Where chlorine is used, facilities must provide for rapid and uniform mixing and a total chlorine residual of at least 1.0 milligram per liter must be maintained at all times. Larger residuals or longer contact times may be required and as well as minimum contact times if chlorine is used as the disinfectant. This requirement does not preclude an additional application of disinfectant prior to filtration for the purpose of improving filter performance. (See Rule 62-600.440(5) F.A.C.)

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3.0 METHODOLOGY FOR RELATIVE RISK ASSESSMENT

Risk assessment is a process that evaluates the likelihood that adverse ecological or human health effects will occur as a result of exposure to stressors (US EPA, 1998a). It is a process for organizing and analyzing data, information, assumptions, and uncertainties. Risk assessment involves identification of hazards or stressors, analysis of the linkage between exposure to stressors and effects on receptors, and risk characterization (US EPA, 1998a). Risk assessments are used to help risk managers determine priorities for actions that are designed to manage or reduce risk. Risk management is a decision-making process which involves such considerations as risk assessment, technological feasibility, statutory requirements, public concerns, and other factors.

In this study, the terms *risk analysis*, *risk characterization* and *relative risk assessment* refer to the processes of analyzing risks, describing risks, and the final comparison of relative risk assessment, respectively.

For this study, risk analysis and relative risk assessment of four different wastewater management options involved three steps:

1. Creation of a generic risk analysis framework (GRAF) for each wastewater management option
2. Conducting a risk analysis of each management option using the GRAF and characterizing the risk associated with each option
3. Comparing the risks associated with all four wastewater management options, based on the results of risk analysis of each management option, to arrive at a relative risk assessment.

3.1 Generic Risk Analysis Framework and Problem Formulation

In order to provide a consistent and comprehensive procedure for analyzing risk, a generic risk analysis framework (GRAF) was developed. The GRAF is a procedure for describing all potential risks and identifying all possible hazards, sources, exposure pathways, and effects on receptors, based on a generalized approach to the issue. This framework, also known as problem formulation, outlines potential issues to be analyzed for risk, using site-specific information. In this study, the GRAF was used to develop a conceptual model of potential risk for each management option. The GRAF incorporates human health and ecological risk components.

The use of a GRAF to analyze risks of individual wastewater management options is based upon the *Guide for Developing Conceptual Models for Ecological Risk Assessments* (Suter, 1996), a risk assessment framework outlined in EPA's *Residual Risk Report to Congress* (US EPA, 1999a), and EPA's ecological risk assessment framework, presented in *Guidelines for Ecological Risk Assessment* (US EPA, 1998a).

The first step in developing a GRAF is formulating the problem and developing a conceptual model of potential risk. In formulating the problem, the purpose for

conducting the risk assessment is articulated, data are collected and assessed, and potential stressors, receptors, and exposure pathways are selected for further analysis. This information is then organized within a conceptual model, which is a “written description and visual representation of predicted relationships between ecological [or other] entities and the stressors to which they may be exposed” (US EPA, 1998a). For each wastewater management option, a conceptual model helps to define the information necessary to complete the risk analysis. The analyses necessary to characterize risk are then conducted as part of the next step, the option-specific risk analysis and characterization (see below).

Potential stressors include constituents of concern, such as compounds and elements, present in treated wastewater and their degradation byproducts or other derivatives. Potential secondary stressors include other effects of stressors that may pose additional risks themselves. Secondary stressors and the risks they pose can be particularly difficult to anticipate and describe.

Receptors are the human and ecological entities that are exposed to stressors and that may suffer potential adverse effects. Exposure to a stressor must be demonstrated before the linkage between a stressor and an adverse effect can be evaluated. Exposure pathways are the ways in which stressors and receptors are brought into contact with each other. Assessment endpoints provide yardsticks for measuring the effects of stressors. Important assessment endpoints selected for this study included drinking-water quality standards, surface- and marine-water quality standards, and other human health and environmental indicators. Where no assessment endpoints existed, potential adverse effects were evaluated using a weight-of-evidence approach.

3.2 Option-Specific Risk Analysis and Risk Characterization

The second step in risk assessment is conducting an analysis and evaluation of the conceptual model of risk for each wastewater management option. In this step, specific information concerning stressors, receptors, and exposure pathways is used to analyze relationships and anticipate potential adverse effects (or risks). In this study, such information included site-specific data on hydrogeology, water quality, wastewater treatment plant effluent, and wastewater management options used in South Florida. In order to evaluate exposure pathways, information concerning properties of stressors (for example, concentration, solubility, half-life, tendency to bioaccumulate) and the environment they pass through (groundwater, surface water, ocean, subsurface geology, and soils) were compiled and analyzed. Information about large-scale physicochemical processes that determine exposure pathways is also essential for determining whether receptors will actually be exposed to stressors. Such information was used to evaluate and refine the conceptual model for each wastewater management option.

Evaluation of the conceptual model involves an exposure analysis and risk characterization. Exposure analysis is critical to risk analysis and risk assessment; without exposure to a stressor, there is no risk (US EPA, 1998a). In this study, as the conceptual models were evaluated and refined, pathways that did not result in exposure

of a receptor to stressors or exposure pathways that were insignificant or improbable were eliminated. Areas of uncertainty and data gaps were identified. Whenever appropriate, conservative assumptions were made that may result in overstating, rather than understating, exposure and risk. A conservative approach will be more protective of human and ecological health.

Risk characterization involves describing the potential adverse ecological and human health effects (risks) that may result from exposure to stressors (US EPA, 2000). Risks may be estimated, compared, or qualitatively described. In this study, risk characterization was performed at assessment endpoints for each conceptual model of a wastewater management option. Upon completion of the risk characterization, issues that pose actual risks were identified, while issues that pose little or no risk were eliminated or assigned lower priority in the final conceptual model of risk. Other risk factors were also taken into account, such as receptor sensitivity, response to change, and potential for recovery if the stress is removed or decreased (Brickey, 1995; GMIED, 1997).

3.3 Relative Risk Assessment

Risks and risk factors may be compared using a variety of methods; comparisons may be quantitative, semiquantitative, or qualitative. Frequently, such an assessment requires that professional judgment be applied to evaluate the relative magnitude of effects (US EPA, 1998a; Suter, 1999a, 1999b).

In this study, relative risk assessment relied upon results of the option-specific risk analysis and risk characterization to compare the risks and risk factors of the four wastewater management options. This relative risk assessment used a qualitative approach to prioritizing risk factors and describing the relative risks and risk factors. There are many risk factors that could have been used in the relative risk assessment. Risk factors were chosen on the basis of how they contributed to making useful comparisons between the potential risks to human and ecological health. Chapter 8 compares risk findings for each wastewater management option and discusses their priorities.

The following sections provide detailed descriptions of the risk methodology used.

3.4 Detailed Description of Problem Formulation

This study of relative risk develops conceptual models of risk that are based on the physical, chemical, and biological processes that govern the fate and transport of discharged wastewater constituents. Developing an understanding of such large-scale fate and transport processes is critical for providing the risk manager with the necessary information to make informed decisions on managing and decreasing risks. Without an understanding of the physical, chemical, biological, and human factors that influence risk, a risk manager may expend time and resources on managing risk symptoms without addressing and eliminating the causes of risk.

3.4.1 Selection of Potential Exposure Pathways

Once treated wastewater is released into the environment, the processes that determine fate and transport of wastewater constituents (stressors) define the large-scale nature of exposure pathways that must be evaluated. These large-scale processes act upon stressors in ways that are strongly governed by the specific chemical and physical properties of the stressors themselves. How these large-scale processes operate in the environments that receive discharges of treated wastewater also varies. Descriptions of the receiving environments (for example, groundwater, surface water, subsurface geology, and soils) are given in Chapters 4 through 7, which examine each of the wastewater management options in detail.

Advection, dispersion, and dilution are large-scale physical processes that play an important role in determining the transport of wastewater constituents. Advection involves mixing and transport by bulk movement of water and is often the single most important mechanism responsible for migration of wastewater constituents. Dispersion refers to slow spreading of constituents in response to gradients in concentration (molecular diffusion) and other phenomena. Dilution is a reduction in concentration of a stressor or other wastewater constituent, which may result from advection or dispersion.

In groundwater, the large-scale movement of wastewater constituents in the subsurface is strongly influenced by the characteristics of the geological media through which discharged effluent and groundwater flows. Porous media flow occurs where primary porosity exists, and it can result in widely varying rates of groundwater flow, depending on the size of pores, amount of pore space, and interconnection of pore spaces. Secondary porosity refers to larger fractures or solution channels in sediments or rock, where groundwater and effluent can move along solution channels, fractures, and other preferential flow paths. In such preferential flow, advective transport rates may be greater than porous media flow rates. In this case, dispersion frequently results from mixing at intersections of fractures and as a result of variations in fracture openings.

The eventual fate of wastewater constituents in the environment determines the final concentrations of stressors to which receptors may be exposed. *Attenuation* describes a variety of processes involving interactions between wastewater constituents and the environment that cause concentrations of constituents to decrease as time passes. Examples of processes that may result in attenuation include filtration, precipitation, settling, biological uptake, chemical transformation, dissolution and adsorption of constituents. Porous media may allow filtration of small bacteria and viruses, which can result in attenuation of these microorganisms, although very small microorganisms may be transported over long distances in porous media. Such attenuation may not occur if open fractures and solution channels are present, which may allow more rapid transport of both chemical compounds and microorganisms (US EPA, 1989).

Other important physical and chemical properties that influence the behavior of wastewater constituents include the stressor’s solubility in water; tendency to adsorb to soil, sediments or geologic media; and half-life. Wastewater constituents with higher

solubilities may remain longer in effluent and groundwater and may also be present at higher concentrations in the initial effluent. The tendency to adsorb or bind to soil, sediments, or geologic media is determined by complex interactions between wastewater constituents and the physical and chemical environment. Adsorption of a constituent can result in retardation, or slowing, of the transport of stressors. For organic components, organic carbon partition coefficients (k_{oc}) provide measures of this tendency. Chapters 4 and 5 use such characteristics, in conjunction with distribution coefficients (k_d) and other measures, to determine rates of retardation for wastewater constituents.

The residence time of a compound or element in the environment is equivalent to the lifetime of the compound or element before attenuation or other processes cause it to dilute or disappear. The half-life ($t_{1/2}$) of a compound or element is the time required for it to decrease to half of its initial concentration. Half-life values take into account biodegradation and hydrolysis. Biodegradation is a geological process whereby microorganisms bring about chemical changes that can reduce the concentration of a specific wastewater constituent. Hydrolysis is a chemical reaction that adds water to the chemical structure of a compound, disrupting existing bonds or adding new bonds. Hydrolysis can increase solubility of a compound in water and enhance biodegradation, but it may also make a constituent more biologically available (Suthersan, 2001).

3.4.2 Definition of Potential Receptors

For this risk assessment, several potential receptors were selected. Drinking-water receptors are groundwater or surface-water resources that are potential receptors of underground or surface-water contaminants derived from treated wastewater. Potential drinking water receptors include underground sources of drinking water (USDWs), shallow public-water supply wells, private drinking-water wells, and some surface-water bodies used for drinking water sources (the latter are very uncommon in South Florida). Potential ecological receptors in surface water and ocean environments include organisms and ecosystems. Potential human receptors are people who may be exposed to treated wastewater constituents through recreational or occupational activities that bring them into contact with the disposed water.

3.4.3 Selection of Assessment Endpoints

The assessment endpoints chosen for this study are related to the type of receptor chosen. The first category of assessment endpoints pertains to USDWs and public and private drinking-water supply wells. For these drinking-water receptors, drinking-water standards were used as assessment endpoints. Federal drinking-water standards, also known as maximum contaminant levels, or MCLs, were designed to protect human health by establishing minimum standards for drinking water. MCLs are assumed to be protective of human health, although they may not be relevant to ecological standards. The Florida Department of Environmental Health (DEP) also regulates water quality of Class I surface waters intended for drinking water sources. In addition to treatment and disinfection requirements for the different wastewater management options, DEP

regulations ensure protection of groundwater quality by establishing minimum criteria for groundwater according to Florida Administrative Code (FAC) 62-520.400.

Construction, operation, and monitoring of wastewater treatment facilities to certain standards are also considered in this category of drinking water-related assessment endpoints. All management methods are also subject to regulations concerning operation, maintenance, and monitoring.

The second category of assessment endpoints is used for ecological risk assessment in fresh surface-water bodies and the ocean. Surface-water quality standards for fresh water and marine water are intended to protect human recreation and ecological values. DEP regulations protect surface-water quality through an extensive set of regulations contained in FAC 62-302. These include state surface-water quality standards for fresh water and nearly marine or marine waters (Class III standards).

The third category of assessment endpoints addresses unregulated substances that may be present in drinking-water supplies, treated municipal wastewater effluent, and other water bodies. For unregulated substances, a weight-of-evidence approach was used, based on examination of the scientific literature concerning the effects of these substances. Examples of unregulated substances include emerging contaminants, such as hormonally active substances (endocrine-disrupting compounds), surfactants, and a wide range of other organic and inorganic compounds. Emerging contaminants are of concern because there is some evidence, based on a limited number of studies, that they may cause adverse effects in humans or other organisms. However, extensive and definitive testing under controlled conditions has generally not been conducted. Where possible, a range of concentrations that may have adverse effects is defined, and the concentration in USDWs or other water bodies is compared with this range-of-effects levels.

Assessment endpoints and the regulatory standards for surface water, groundwater, drinking water, and other operational standards are described more fully in Chapters 4 through 7 for each wastewater management option.

3.4.4 Selection of Potential Stressors

General characteristics of the potential human health or ecological stressors selected for this study are described in this section. Understanding the behavior and characteristics of stressors and their response to wastewater treatment is critically important in the analysis of risk. The stressors considered for this risk assessment were selected based on their occurrence in treated wastewater, scientific information concerning their toxic properties or other potential adverse effects, whether they are representative of a larger group of similar compounds, and their long-term fate in the environment.

In order to conduct a focused risk assessment, suitable representatives of each major category of stressors were chosen. Criteria for selection of representative stressors that might affect human health included severity of effects, level and efficacy of wastewater treatment, representative behavior, and whether the representative stressor provides a

conservative (that is, protective) approach to evaluating risk. Contaminants of concern to public health also included substances for which human health effects are not yet fully understood, but for which there may be adverse human health effects, based on laboratory tests, observed effects, or other evidence.

General categories of human health stressors selected for this study include the following:

- Pathogenic microorganisms (for example, viruses, bacteria, protozoans)
- Inorganic compounds and elements (for example, metals and inorganic nutrients)
- Synthetic organic compounds (for example, pesticides and surfactants)
- Volatile organic compounds (VOCs)
- Hormonally active agents (for example, endocrine modulators and disruptors).

Representative ecological stressors that may cause adverse effects on organisms or ecosystems were selected based on a review of the scientific literature. Ecological stressors were chosen if they are known or suspected stressors to aquatic ecosystems, cause toxic effects in aquatic species, and are commonly found in wastewater effluent. Because many similar physical, chemical, and biological processes occur in both fresh-water and marine systems, the contaminants of concern are similar in both environments. Categories of ecological stressors selected for this study include the following:

- Inorganic compounds and elements (for example, inorganic nutrients and metals)
- Synthetic organic compounds (for example, pesticides, surfactants)
- Volatile organic compounds (VOCs)
- Hormonally active agents (for example, endocrine modulators and disruptors)
- Pathogenic microorganisms.

The general categories of human health and ecological stressors and the representative stressors selected to represent different stressor categories are listed in Table 3-1.

Table 3-1. Representative Human Health and Ecological Stressors Selected for this Study

Stressor Category	Representative Human Health Stressors	Representative Ecological Stressors
Pathogenic microorganisms	Rotavirus, total coliform, fecal coliform, enterococci, *Cryptosporidium parvum*, *Escherichia coli*, *Giardia lamblia*	*Cryptosporidium parvum*
Inorganic compounds (metals, metalloids)	Arsenic, copper	Arsenic, copper, lead, silver, cyanide
Inorganic nutrients	Nitrate, ammonia	Nitrate, total nitrogen, ammonia, total phosphorus, orthophosphate
Volatile organic compounds (VOCs)	Tetrachloroethene (PCE)	Tetrachloroethene (PCE)
Synthetic organic compounds (SOCs)	Chloroform (trihalomethanes) and chlordane (pesticides)	Methylene blue anionic surfactant (MBAS)
Hormonally active agents (endocrine-disrupting compounds)	Di(2-ethylhexl)phthalate (DEPH)	Estrogen equivalence

The characteristics of the selected stressors are described below.

3.4.4.1 Pathogenic Microorganisms

Microbial pathogens in water pose a high-priority public health concern (Raucher, 1996). In the United States, the number of microbiological diseases originating in contaminated drinking water is estimated to be as high as 40 to 50 million cases per year. While the total number of outbreaks of diseases caused by contaminated drinking water has decreased by 20% since the mid-nineties, the proportion of outbreaks associated with groundwater sources has increased by almost 30% (PSR, 2000). The emergence of new pathogens (for example, *Escherichia coli* O157:H7 and *Cryptosporidium parvum*), antibiotic-resistant strains of microorganisms, and a larger sensitive population have resulted in increased public health concerns (Rose et al., 2001). Microbiological diseases caused by ingestion of contaminated shellfish are included in this category of waterborne infections because contaminated water is often the major carrier (Wittman and Flick, 1995). Enteric microbial pathogens (that is, microbes that live in the intestinal tracts of humans and animals and that cause disease) present in wastewater are listed in Table 3-2.

Table 3-2. Microbial Pathogens Potentially Present in Untreated Domestic Wastewater

Bacteria	**Protozoa**
Campylobacter jejuni	*Cryptosporidium parvum*
Escherichia coli	*Giardia lamblia*
Legionella pneumophila	*Balantidium coli*
Salmonella typhi	*Entamoeba histolytica*
Shigella	**Viruses**
Vibrio cholerae	*Adenovirus* (51 types)
Helminths	*Astrovirus* (5 types)
Ancylostoma duodenale (hookworm)	*Calicivirus* (2 types)
Ascaris lumbricoides (roundworm)	*Coronavirus*
Echinococcus granulosis (tapeworm)	Enteroviruses (72 types)
Enterobius vermicularis (pinworm)	Hepatitis A
Necator americanus (roundworm)	Norwalk agent
Schistosoma	*Parvovirus* (3 types)
Strongyloides stercoralis (threadworm)	*Reovirus* (3 types)
Taenia (tapeworm)	Rotavirus (4 types)
Trichuris trichiura (whipworm)	

Source: York et al., 2002.

Depending upon the level of treatment and disinfection, concentrations of microbial pathogens in treated wastewater discharged to the environment can vary widely. An aggressive treatment combines disinfection with filtration to kill or physically remove microbial pathogens present in drinking water. For example, most bacteria and viruses in wastewater are generally effectively inactivated by disinfection with chlorine and filtration (York et al., 2002). However, disinfection byproducts, such as trihalomethanes, that are formed when chlorine reacts with organic compounds can pose human health concerns as well.

Survival of pathogenic microorganisms in soil and water generally is limited to days, weeks, or months, depending on the microorganism and whether it can form cysts or spores that persist in the environment. Survival is affected by factors such as temperature, availability of water and oxygen, and whether an animal host is needed for survival or growth of the microorganism. There is a small but growing body of information concerning survival of pathogenic microorganisms in the shallow subsurface and other microbial processes in geologic formations such as microbial denitrification in the shallow subsurface in northeast Florida (USGS, 2000). If viruses are not inactivated by treatment and are released, their small size and longevity may allow them to be distributed widely through the environment. Viruses may survive in surface water and groundwater, although most viruses typically cannot reproduce outside the human host (PSR, 2000). Viral contamination of wells, especially private wells with no treatment, poses concerns.

Potential human exposure pathways to pathogenic microorganisms include the following:

- Ingestion of water contaminated by exposure to wastewater
- Ingestion of contaminated food (such as shellfish, fish, produce, or foods processed in contaminated water)
- Dermal contact with contaminated water or soil through swimming, showers, spray irrigation, or occupational exposure
- Inhalation of contaminated water or soil (aerosols, shower spray, spray irrigation, dust, occupational exposure).

Secondary spread may also be possible, which includes person-to-person contact, use of public swimming facilities, and transmission from food handlers and care facilities (Chick et al., 2001).

Microbial growth in groundwater is not well characterized in general because of the difficulty of obtaining microbiologically representative samples without introducing surface contaminants. There are many gaps in knowledge concerning potential human health effects from ingestion of pathogenic microorganisms in water:

- Whether indicator organisms for microbial pathogens, such as coliform, are representative of pathogenic microorganisms
- Whether environmental sources other than wastewater exist for pathogenic microorganisms
- Exposure factors
- Modes of transmission
- Modes of environmental transport of microorganisms
- Survival potential of microorganisms in groundwater.

Three representative pathogenic microorganisms were selected to evaluate human exposure to treated wastewater: rotavirus, *Cryptosporidium parvum,* and pathogenic strains of *Escherichia coli* (*E. coli*). These are described below.

<u>Rotaviruses</u>

Rotaviruses are highly infective viruses that can be transmitted in water, causing a highly contagious disease that induces vomiting and diarrhea. In the United States, rotavirus has been estimated to cause 3 million cases of childhood diarrhea, resulting in 500,000 doctor visits, 100,000 hospitalizations, and up to 100 deaths annually (EHP, 1998a; SAIC, 2000). Because of the easily transmitted and highly contagious nature of the illness, rotaviruses were selected as a representative of pathogenic enteric viruses.

Other enteric viruses that are associated with poor-quality or untreated wastewater have been detected in near-shore waters and canals, including coxsackie viruses, Hepatitis A viruses, and Norwalk-like virus (Rose et al., 2000). These viruses, if ingested, can cause diarrhea, aseptic meningitis, and myocarditis. Their small size (in the nanometer range) and structure enhances viral survival and transport in water; these viruses can survive in

groundwater for more than 2 months (Rose et al., 2000). Plankton and marine sediments may serve as reservoirs for pathogenic microorganisms, which can emerge to become infective when conditions are favorable (Henrickson et al., 2001).

Cryptosporidium parvum

Cryptosporidium parvum, an enteric protozoan, is considered to be a major risk to U.S. water supplies because it is highly infectious, forms cysts and oocysts that are resistant to chlorine disinfection, and is difficult to filter because of its small size. *Cryptosporidium* poses significant challenges to public health and water authorities (Gostin et al., 2000). If it is present in drinking water, it poses a high risk of waterborne disease (particularly for immunocompromised individuals). There have been 12 documented waterborne outbreaks of *Cryptosporidium* in North America since 1985; in two of these (Milwaukee and Las Vegas), mortality rates among exposed immunocompromised individuals ranged from 52% to 68% (Rose, 1997). Similar enteric protozoans include *Giardia lamblia*, *Entamoeba histolytica*, and *Balantidium coli* (York et al., 2002).

Protozoan cysts and oocysts are very persistent in the environment, particularly where water exists. Dormant oocysts may remain viable for months in sewage or groundwater until they find a new host. *Cryptosporidium* infects both humans and animals and can be transmitted through ingestion of contaminated water or food. Secondary infection can also occur. *Cryptosporidium* forms a reproductive oocyst that, once ingested, continues its life cycle in the gastrointestinal tract, causing the disease Cryptosporidiosis. The parasite can also be spread through the fecal-oral route by infected food handlers or in day-care settings. As few as 10 to 25 oocysts can cause infection; however, the disease is usually self-limiting with 2 to 10 days of symptoms in healthy persons. In sensitive populations and individuals, the disease can be life threatening.

Chlorine, the traditional water disinfectant for killing water-borne pathogenic bacteria and viruses, is not as effective against *Cryptosporidium* as other waterborne organisms, for example, *Giardia* (Joyce, 1996). Standard screening methods have proven ineffective as well. Filtration is the accepted method of removing *Cryptosporidium*.

Because of the severity of the disease, its widespread occurrence in nature, and because water and wastewater treatment does not always address *Cryptosporidium*, it was chosen for use as a representative pathogenic protozoan for evaluating human health risks from pathogenic protozoans in discharged treated wastewater.

Fecal Coliforms (*Escherichia coli*)

Fecal coliforms are bacteria that are normally found in human and animal wastes. *Escherichia coli*, or *E. coli*, is a type of fecal coliform bacteria. The presence of *E. coli* in water is a frequently used indicator of recent sewage or animal waste contamination, although it is not a reliable indicator of human sewage. It is important to note that sewage-indicator bacteria such as fecal coliforms have short survival times in the environment and may not be good indicators of the presence of protozoans and viruses in

some environments (Henrickson et al., 2001). For example, one injection well monitoring study performed in Florida found that indicator bacteria and coliphages were not detected, while *Cryptosporidium* oocysts were detected at very low concentrations (Rose et al., 2001).

Most strains of *E. coli* are harmless and live in the intestines of healthy humans and animals without causing illness. However, *E. coli* O157:H7 is one strain of *E. coli* that produces a powerful toxin that can cause severe gastrointestinal illness. Infection by *E. coli* O157:H7 may cause hemolytic uremic syndrome, in which red blood cells are destroyed and kidney failure occurs. About 2% to 7% of infections lead to this complication. In the United States, most cases of hemolytic uremic syndrome are caused by *E. coli* O157:H7 (US EPA, 2001a). Exposure may occur through ingestion, recreational contact, or consumption of contaminated water or food (Schmidt, 1999). Sensitive human receptors include children, the ill, the immunocompromised, and the elderly. Because of the severity of illness that may occur upon exposure to *E. coli* O157:H7, fecal coliforms were selected as a representative human health stressor.

Pathogen fate, transport, and survival in the environment are discussed more in Chapter 4. Data on concentrations of pathogenic and indicator microorganisms in treated wastewater and from groundwater monitoring are provided in Appendix 1 (Appendix Tables 1-3, 1-4, and 1-5).

3.4.4.2 Inorganic Stressors

Wastewater contains a large number and variety of inorganic constituents, including metals, salts, nutrients, and other substances. Many, if not all, of these inorganic constituents are natural in origin (that is, they are ultimately derived from natural materials and are not “manmade” in the sense of being synthesized by humans), but their concentrations in wastewater may be elevated because of human activities. Many inorganic substances, if present at high enough concentrations, can pose some risk to human health. For this reason, many drinking-water standards (maximum contaminant levels, or MCLs) address the maximum amount of a given inorganic substance allowed in drinking water. Removal of these constituents will depend upon the level and type of wastewater treatment that is used.

Metals

Like nutrients, metals are naturally occurring and play a necessary biological role in the environment. However, in excessive amounts, metals can be toxic to wildlife, fish, and aquatic organisms. Metals have complex and dynamic physical and chemical reactions in the environment and can occur in different chemical forms or species. Metal speciation is important in understanding biological uptake by fish and wildlife. Factors that affect chemical speciation of metals include pH, alkalinity, the presence of organic matter and colloidal particles, and the oxidation-reduction potential of the environment (Stumm and Morgan, 1981). Organisms also differ in their capacity to store, remove, and detoxify metal contaminants (Wallace and Braasch, 1997).

Copper is an example of an essential micronutrient metal that is required by plants, animals, and most microorganisms in trace amounts. However, at higher concentrations, copper is toxic to algae, inhibiting growth and photosynthesis; copper sulfate and other copper-containing compounds have been used to control algal blooms in fresh water bodies and reservoirs since the early 1900s. The bioavailability of copper, or its ability to be taken up by organisms, depends in large part on its speciation. Total copper is not a good measure of bioavailablity. Reduced copper, or Cu^{2+}, is more readily taken up by organisms than the oxidized form and is therefore a better indicator of potential stress.

Arsenic is a metalloid element that is often present in groundwater where underlying rocks and soil contain arsenic salts or arsenic-containing minerals. A variety of industrial and agricultural activities also generate or release arsenic-containing compounds, including production and use of wood preservatives (for example, copper chromium arsenate), mining of arsenic-containing ores, and manufacture and use of arsenic-containing pesticides (for example, lead arsenate). Since arsenic is highly soluble, particularly under reducing conditions (which are often found in groundwater), it may also be highly mobile. Movement of surface water and groundwater provide important potential transport pathways for arsenic and other metals.

Chronic arsenic exposure causes a variety of human health effects, including carcinogenic and noncarcinogenic effects (Chowdhury et al., 2000; Morales et al., 2000). The population cancer risks from arsenic in U.S. water supplies may be comparable to those from environmental tobacco smoke and radon in homes (Smith et al., 1992). Noncarcinogenic effects of low levels of arsenic include adverse effects on the gastrointestinal system, central nervous system, cardiovascular system, liver, kidney, and blood (Abernathy et al., 1999; Tseng et al., 2000; Kaltreider et al., 2001; and Hopenhayn-Rich et al., 2000). At higher oral doses (600 milligrams per kilogram per day or more), poisoning and death will result.

Human exposure to inorganic arsenic results primarily from ingestion of contaminated drinking water or ingestion of contaminated food. Examples of food that can contain elevated arsenic levels include fish, shellfish, crustaceans, and some cereals, such as rice, taken from water or soils with high arsenic concentrations. Consumption of fish and shellfish from waters that contain elevated amounts of arsenic may be an important source of arsenic in humans. In food, the highest levels of arsenic in the U.S. Food and Drug Administration’s total diet survey were found in fish, with a mean level of about 15 parts per million (ppm) As_2O_3 in the edible portion of finfish (Jelinek and Corneliussen, 1977).

Approximately 5% of large and small regulated water-supply systems in the United States are estimated to have arsenic concentrations that exceed 20 micrograms per liter (μg/L) (Morales et al., 2000). The MCL for arsenic was formerly 50 parts per billion (ppb). In January 2001, the EPA lowered the MCL to 10 ppb. This lower standard was reviewed in 2001 and early 2002. After considerable public comment and deliberation, the 10 ppb MCL level was determined to be appropriate, and the Final Arsenic Rule went

into effect in February 2002. The World Health Organization also recognizes an arsenic standard for drinking water of 10 ppb.

In the marine environment, arsenic typically occurs in seawater at concentrations ranging from 1 to 8 ppb and in sediments at 2 to 20 ppm. The distribution of arsenic in terrestrial environments is not nearly so homogeneous, as indicated by the higher levels of arsenic in marine organisms than terrestrial organisms; the biological concentration factor may vary by orders of magnitude between aquatic and terrestrial organisms (Fishbein, 1981). Arsenic may bioaccumulate in aquatic organisms. However, there is considerable variability in aquatic food-web bioaccumulation (Penrose et al., 1977; Vallette-Silver et al., 1999; Woolson, 1977). Organisms containing high levels of arsenic tend to be those that ingest sediment particles while feeding; that is, benthic filter-feeders or detritus-feeders exhibit higher concentrations of arsenic than pelagic fish.

As with copper, factors that govern biological effects of arsenic include its bioavailability, the quantity ingested or respired, the degree of assimilation, and the extent of retention in tissues.

Gaps in knowledge concerning arsenic and human health and ecological effects concern detailed transport mechanisms, mobility in the environment, carcinogenesis, whether there are cumulative or synergistic effects in combination with other contaminants, differences in bioaccumulation by different species, and the proper dose-response relationship to use in ecological risk assessment.

Inorganic Nutrients

Wastewater is a source of nutrients such as nitrogen, phosphorus, and other substances that act as nutrients. Secondary treatment removes only a portion of the nitrogen and phosphorus that may be present (see Chapter 2).

Nitrogen is the most important nutrient to consider in an ecological risk assessment for a marine environment because nitrogen limits primary production in marine environments. While many studies focus on total nitrogen (all forms of nitrogen), nitrate is the form that is most readily available for uptake by algae and plants. Excess nitrate in drinking water can potentially affect the health of infants, young children, and pregnant women and can cause methemoglobinemia (Knobeloch et al., 2000; Gupta et al., 2000). Human exposure to excess nitrate can occur through drinking or accidentally ingesting water that has elevated concentrations of nitrate. Little is known about the potential health effects of long-term exposure to excess nitrate in drinking water. Some studies of chronic nitrate ingestion have suggested connections to reproductive and developmental health effects, certain cancers, childhood diabetes, and thyroid disease.

The Safe Drinking Water Act established an MCL for nitrate of 10 milligrams per liter (mg/L), or 10 parts per million (ppm). This federal standard is used to ensure the safety of public water supplies, but does not apply to private wells. An estimated 2 million private

household water supplies in the United States today may fail to meet this federal standard for nitrate (Knobeloch et al., 2000).

Excessive nitrate in the marine environment can stimulate phytoplankton and macroalgal growth. This can create adverse effects such as eutrophication (reduction of available oxygen), loss of eelgrass, dead zones because of low dissolved oxygen concentration from decomposing organic matter, and increases in harmful algal blooms (Nixon, 1998; Joyce, 2000). It is important to note that the 10 ppm drinking water standard for nitrate is generally much higher than the concentration of nitrate typically present in seawater or coastal waters, which ranges from several tenths of a part per million to several parts per million.

Excess nutrients may create secondary stressors, such as harmful or nuisance algal blooms. The algal toxins that may be produced by harmful algal blooms (HABs) can cause adverse effects on humans, aquatic mammals, fish, shellfish, and other organisms. Human ingestion of seafood contaminated by HABs can result in respiratory illness, gastroenteritis, and skin irritation. Paralytic shellfish poisoning is one example of an illness caused by toxin-producing dinoflagellates that form red tides. However, most scientists agree that, although excess nutrients may be a factor in some blooms, other environmental factors such as changes in temperature or circulation may cause many algal blooms (Tibbetts, 2000).

Phosphorus is a nutrient of concern in freshwater ecosystems because it is frequently the limiting nutrient for algal and plant growth, in contrast to nitrogen which tends to be the limiting nutrient in marine waters. Excess phosphate in freshwater can cause excessive algal growth, eutrophication, and low dissolved oxygen, just as excess nitrate in coastal waters can result in similar effects. Excess phosphate already exists in many of South Florida’s fresh water aquatic ecosystems, and a phosphate-based water quality standard is being considered for Lake Okeechobee, which is heavily affected by fertilizer runoff from adjacent agricultural lands.

Different forms of phosphorus exist in the aquatic environment; the most important are orthophosphate, total phosphorus, and particulate phosphorus. Orthophosphate (also known as soluble reactive phosphorus) is the major inorganic form of dissolved phosphorus most readily available for biological assimilation. Total phosphorus, as the name implies, refers to all the phosphorus in a volume of water including dissolved and particulate forms. Orthophosphate was chosen as a representative nutrient stressor in fresh water ecosystems.

3.4.4.3 Organic Compounds

<u>Pesticides</u>

Pesticides in wastewater primarily originate from stormwater runoff from lawns and gardens and other areas where pesticides are used. Human exposure to pesticides can occur through ingestion of contaminated drinking water, food, or dermal contact with

contaminated water (Moody and Chu, 1995). Potential human receptors include adults, children, subsistence fishermen, farmers, and sensitive portions of the population, such as the elderly and ill. A number of pesticides, including chlordane, were evaluated for deep-well injection, while chlordane alone was used as a representative pesticide in other wastewater management options.

Chlordane is a chlorinated insecticide that was widely used in agricultural, industrial, and domestic applications; about one-third of the chlordane used in the United States was applied to control pests in homes, gardens, lawns, and turf (ATSDR, 1995). The EPA in 1983 banned all use of chlordane, except for control of termites. In 1988, because of concerns about carcinogenicity, toxicity, and harmful effects on wildlife, the EPA banned its use except for fire-ant control in power transformers. Chlordane is no longer distributed in the United States.

Despite having been banned years ago, chlordane is extremely persistent in the environment and may remain in soil for 20 years (ATSDR, 1995a). It is associated with many human health effects: chlordane may be carcinogenic, toxic, and impair human immune and neurological systems (IARC, 1997; Hardell et al., 1998; Kilburn and Thornton, 1995). Gaps in knowledge concerning human health risks posed by chlordane include the effects of long-term, low-dose exposure, whether it is carcinogenic, and whether it affects fertility, development, or neural systems.

Chlordane binds strongly to particles, does not dissolve easily in water, and may concentrate in the surface microlayer of surface water or in aquatic sediments. Because it is highly lipophilic, chlordane bioaccumulates in aquatic organisms. For compounds such as chlordane, groundwater transport is minimal (Thomann, 1995). The solids on which the chemical is adsorbed are stationary for the most part in groundwater. In surface water, the solids are transported during advection, and there may be significant interactions with aquatic sediments (Thomann, 1995).

Volatile Organic Compounds

Tetrachloroethene (PCE) is a VOC that may be formed in small quantities during chlorination of water or wastewater. Due to its volatility, tetrachloroethene does not remain long in surface or marine waters and will evaporate to the atmosphere; therefore, it has little potential for accumulating in aquatic organisms (US EPA-OW, 2002). However, in groundwater, tetrachloroethene is very mobile and persistent, which enables it to travel significant distances. Research studies have concluded that PCE-contaminated drinking water can be linked to elevated incidence rates of leukemia, bladder, lung and colorectal cancers in humans and experimental animals.

Human exposure pathways for VOCs could include drinking water, ingestion of water during recreational or occupational activities, and exposure to vapor in water. Potential human receptors include private well owners, who may be operating wells that are neither monitored nor treated to national drinking water standards.

The Florida Class III Marine water quality standards for tetrachloroethene are ≤8.85 μg/L on an annual average. The estimated half-lives of trichloroethylene (3.2 μg/L) from an experimental marine mesocosm during the spring (8 to16ºC), summer (20 to 22ºC), and winter (3 to 7ºC) were 28, 13, and 15 days respectively (Wakeham, et al., 1983, in Montgomery, 2000). Toxicity tests indicate toxic levels range from 22 mg/L (LC_{50} (24 hours) for *Daphnia magna* (LeBlanc, 1980, in Montgomery, 2000) to 3,760 milligrams per kilogram (mg/kg) acute oral LD_{50} in rats (TECS, 1985, in Montgomery, 2000).

Surfactants

Gaps in knowledge concerning the human health effects of surfactant compounds in drinking water include the effects of chronic low-dose exposures, suitable critical endpoints for risk estimates to represent sensitive populations, and the exact biological mechanisms by which these compounds affect human health.

Surfactants were chosen as a potential ecological stressor to evaluate because of their widespread use, occurrence in wastewater, their effects upon organic matter, and the relative lack of information concerning their ecological effects, in comparison to compounds currently regulated under the Safe Drinking Water Act. Surfactants are found in laundry detergents and in wastewater and are known to persist in wastewater, sewage sludge, and the environment (Dental et al., 1993). Surfactants have also been suggested as a potential precursor to an endocrine-disrupting agent or estrogenic substance. Estrogenic substances, such as alkylphenol-polyethoxylates (APE), and other alkylphenols, such as nonylphenol, in sewage effluent may also originate from biodegradation of surfactants and detergents during wastewater treatment (Purdom et al., 1994 and Jobling and Sumpter, 1993, both in US EPA, 1997). The representative surfactant chosen for this study is methylene blue anionic surfactant (MBAS), which is an anionic surfactant found in commercially available detergents (Dental et al., 1993).

Hormonally Active Agents

Estrogenic hormones and potential endocrine disrupters include pharmaceuticals (for example, estrogens and their degradation products), surfactants, some pesticides, dioxins, and plasticizers. Scientific opinion is mixed concerning whether such compounds disrupt normal endocrine function, reproductive and developmental processes, or immunological processes (Birnbaum, 1994; Colborn, 1995; vom Saal, 1995). Not all scientists agree that exposure to hormonally-active agents represents cause for alarm. Authors of one paper reported that “there is little direct evidence to indicate that exposures to ambient levels of estrogenic xenobiotics are affecting reproductive health” (Daston et al., 1997). In addition, they state that “estrogenicity is an important mechanism of reproductive and developmental toxicity; however, there is little evidence at this point that low level exposures constitute a human or ecologic risk.” The picture regarding hormonally active agents is therefore complex.

Hormonally active agents found in wastewater and in surface water elsewhere include estradiols (an active component of oral contraceptives), as well as alkylphenols

(biodegradation products of nonionic surfactants). Industrial and pharmaceutical compounds with hormonally active effects include butylbenzylphthalate (BBP), di-n-butylphthalate (DBP), tributylphosphate, butylated hydroxyanisole (BHA), dimethylphthalate, and 4-nonylphenol, dioxin (2,3,7,8-TCDD), bisphenol A, PCBs, PBBs, pentachlorophenol, penta- to nonylphenols, phthalates, and styrenes (Daughton and Ternes, 1999; Jobling et al., 1995).

Scientific studies suggest that these chemicals may cause adverse effects in aquatic organisms and that wastewater is one source of such chemicals (Rodgers-Gray et al., 2000; Nichols et al., 1998). Studies in Florida have documented potential endocrine exposure effects on the Florida panther (Facemire et al., 1995) and American alligator (Semenza, 1997). However, the sources of endocrine disruptors were not documented in these studies.

These substances have been identified in concentrations in the nanograms-per-liter (or parts-per-trillion) range in secondary-treated municipal wastewater effluent and receiving waters (Huang and Sedlak, 2000; and Harries et al., 1998). Because these substances are often highly soluble in water, they may be difficult to remove using conventional technology; estrogenicity has been identified primarily in the water-soluble fraction of wastewater (Raloff, 2000). Municipal wastewater treatment may remove these compounds if they are associated with other organic particles or substances that are removed by treatment.

Environmental monitoring indicates that such chemicals can be present in drinking water as well (Potera, 2000). Potential human exposure pathways include ingestion of water containing such substances, dermal contact with water, and inhalation of volatile compounds from water vapor. Potential human receptors include people consuming or drinking water containing such substances and those exposed to such water as a result of recreational or occupational activities, including subsistence fishermen and farmers.

Significant gaps in knowledge exist concerning the human health and ecological effects of these compounds because they have only recently been recognized as potential contaminants of concern. Comprehensive and long-term epidemiological studies are needed to critically evaluate the effects of exposures to these compounds. Other gaps in knowledge include the concentrations of hormonally active substances in treated municipal wastewater effluent, whether they present an ecological concern, effects of exposures to mixtures, and cumulative effects of all sources of such compounds. Better monitoring methods need to be developed in order to conduct such studies.

The EPA requires testing of commercial chemicals to determine their endocrine disruption potential. Screening techniques to test chemicals for endocrine disruption are being developed. Because of the relative newness of the science, no regulatory guidelines have yet been established for concentrations of hormonally active agents in wastewater.

The hormonally active substance selected to evaluate potential human health risk was di(2-ethylhexyl)phthalate, or DEPH. DEPH is a plasticizer, used to make polymers (such

as PVC) flexible. The threshold limit value for constant 8-hour exposure in air (OSHA, ACGIH) is 5 ppm. DEPH poses some human health concerns, but because it is mostly insoluble in water and is biodegradable in small quantities, it is not considered a critical ecological risk stressor. Large quantities can cause liver damage and reproductive problems in lab animals, but the effects are reversible if the stressor is removed.

One advanced wastewater treatment plant in South Florida also provided data on estrogen equivalence in treated wastewater. Estrogen equivalence is a measure of the response of breast cancer cells to exposure to strongly estrogenic substances, such as hormone replacement and birth control pills (Frederic Bloettscher, Consulting Professional Engineer. September 13, 2001. E-mail communication to Jo Ann Muramoto, Horsley & Witten, Inc.).

3.5 Analysis Plan

This relative risk assessment focused on characterizing and evaluating the major fate and transport processes that determine where the vast majority of discharged effluent and effluent constituents will end up. The focus is on the major exposure pathways that could lead to potential exposure of receptors to effluent constituents that act as stressors.

One of the goals of the risk assessment team was to determine whether final dilutions of wastewater stressors could be predicted or modeled for the ends of major exposure pathways (that is, at the USDW, surface water, or ocean receptors). There are many other potential sources of these stressors in the South Florida environment; wherever possible, evidence linking the stressor to the wastewater management option was sought. Analysis of fate and transport pathways is particularly important for singling out the concentration of stressors that can be ascribed to discharged treated wastewater. Without an analysis of fate and transport, it would be difficult to rule out other sources of the same stressor in surface-water receptors or the ocean or even in drinking-water receptors, such as the USDW or surficial aquifer.

In order to evaluate human health risks, concentrations of representative stressors in treated wastewater at the treatment plant and in drinking water or other receptors were compared with the assessment endpoints: drinking-water standards such as the federal drinking-water standards (MCLs) or Florida’s water quality standards for Class I waters intended to protect drinking-water sources. If there was no human exposure pathway involving a particular water resource, then the standards for that pathway were not used (for example, as there is no human exposure pathway involving ingestion of seawater, then the drinking-water standards were not used). To evaluate ecological risks, monitoring data for treated wastewater were likewise compared with water quality standards intended to protect ecological values. Examples include Florida’s regulations pertaining to Class III coastal and marine waters.

For unregulated compounds, a weight-of-evidence approach based on general scientific literature was used to determine whether disposal of treated wastewater containing such compounds could pose a risk to human health or aquatic ecosystems.

3.6 Final Conceptual Model of Probable Risk

When the conceptual model of potential risk was evaluated using site-specific information, stressors, receptors, or exposure pathways that were insignificant or improbable were eliminated. Criteria for elimination of exposure pathways, stressors, or receptors included the following:

- The transport or exposure pathways that would expose a receptor to a stressor never or hardly ever exist or occur
- The time it would take for a stressor to be transported from the discharge point to the receptor is longer than the residence time of the stressor in the environment
- Wastewater treatment or other attenuation processes routinely decrease the concentration of a particular stressor well below required standards or assessment endpoints
- Attenuation processes that would in all probability result in a significant decrease in concentration of a stressor are known to exist in the receiving environment
- A receptor does not exist in the receiving environment
- There is little or no evidence that adverse effects occur from exposure of receptors to stressors, despite the fact that exposure must occur, using site-specific information.

The risk to human health or the environment from stressors in treated effluent was described to be nonexistent to very low, when either of the following occurs—

- A stressor, receptor, or exposure pathway is eliminated
- It is demonstrated that adverse effects do not occur.

The risk was judged to be low or moderate when any of the following occurs—

- There is a small chance of exposure
- Assessment endpoints (standards) are usually but not always met
- Adverse effects are possible.

The risk was judged to be moderate to high when any of the following occurred—

- There is a moderate-to-high chance of exposure
- Assessment endpoints were almost always exceeded for some stressor
- Adverse effects can occur.

The risk was judged to be very high when there is a high chance of exposure and monitoring indicates that adverse effects have already occurred.

The final conceptual model for each option describes in narrative form the risk findings and conclusions for each wastewater management option.

3.7 Relative Risk Assessment

The risk findings for each wastewater management option were compared and evaluated. Ecological and human health risk factors were compared across all four wastewater management options. A final set of criteria for risk prioritization was developed. The product of the relative risk comparison of wastewater management options is a prioritized list of risk factors for each wastewater management option.

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4.0 DEEP-WELL INJECTION

In this chapter, human health and ecological risks associated with the deep-well injection wastewater management option are described and evaluated. Sources of data and information are used to develop a conceptual model of potential risks. A wastewater fate and transport analysis examines the factors that may be most important in determining risk and levels of risk. This evaluation results in a refined final conceptual model that describes the risks that are most probable.

4.1 Definition of the Deep-Well Injection Option

Deep wells are used in South Florida to dispose of secondary-treated municipal wastewater. These wells are permitted as Class I municipal wells, which by definition dispose of wastewater beneath the lowermost formations containing, within a minimum of one-quarter mile of the well bore, an underground source of drinking water (USDW) (FDEP, 1999a). Deep municipal wells in South Florida inject at depths ranging from approximately 1,000 feet to greater than 2,500 feet below surface of the land.

4.2 Deep-Well Capacity and Use in South Florida

Class I injection wells are used in various regions of the United States for disposal of hazardous and nonhazardous fluids. In South Florida, they provide an important means of managing treated municipal wastewater. The Florida Department of Environmental Protection (DEP) estimates that deep-well injection accounts for approximately 20% (0.44 billion gallons per day) of the total wastewater management capacity in the State of Florida (FDEP, June 1997).

Although deep-well injection is practiced throughout much of South Florida, these wells are concentrated in southeastern portions of the State and in the coastal areas (Figure 4-1; Figure 2-2; Appendix Table 1-6). Dade, Pinellas, and Brevard counties serve as three areas of focus for this risk analysis and are at three corners of the triangular study area. These counties present unique geologic environments and differences in injection system operation that may have a substantial bearing on risk.

4.3 Environment into Which Treated Wastewater is Discharged

To evaluate risk, it is critical to understand regional variations in geology and hydrogeology that influence subsurface fate and transport of injected wastewater. Hydrogeologic units vary in thickness and in their characteristics (for example, porosity and conductivity) across various regions of South Florida. A description of the hydrologic system and hydrogeologic units in South Florida is provided below.

Hydraulic conductivity (“K”) is a measure of a formation’s capability to transmit water under pressure. Aquifer units or layers that exhibit low hydraulic conductivity typically slow the rate at which groundwater flows.

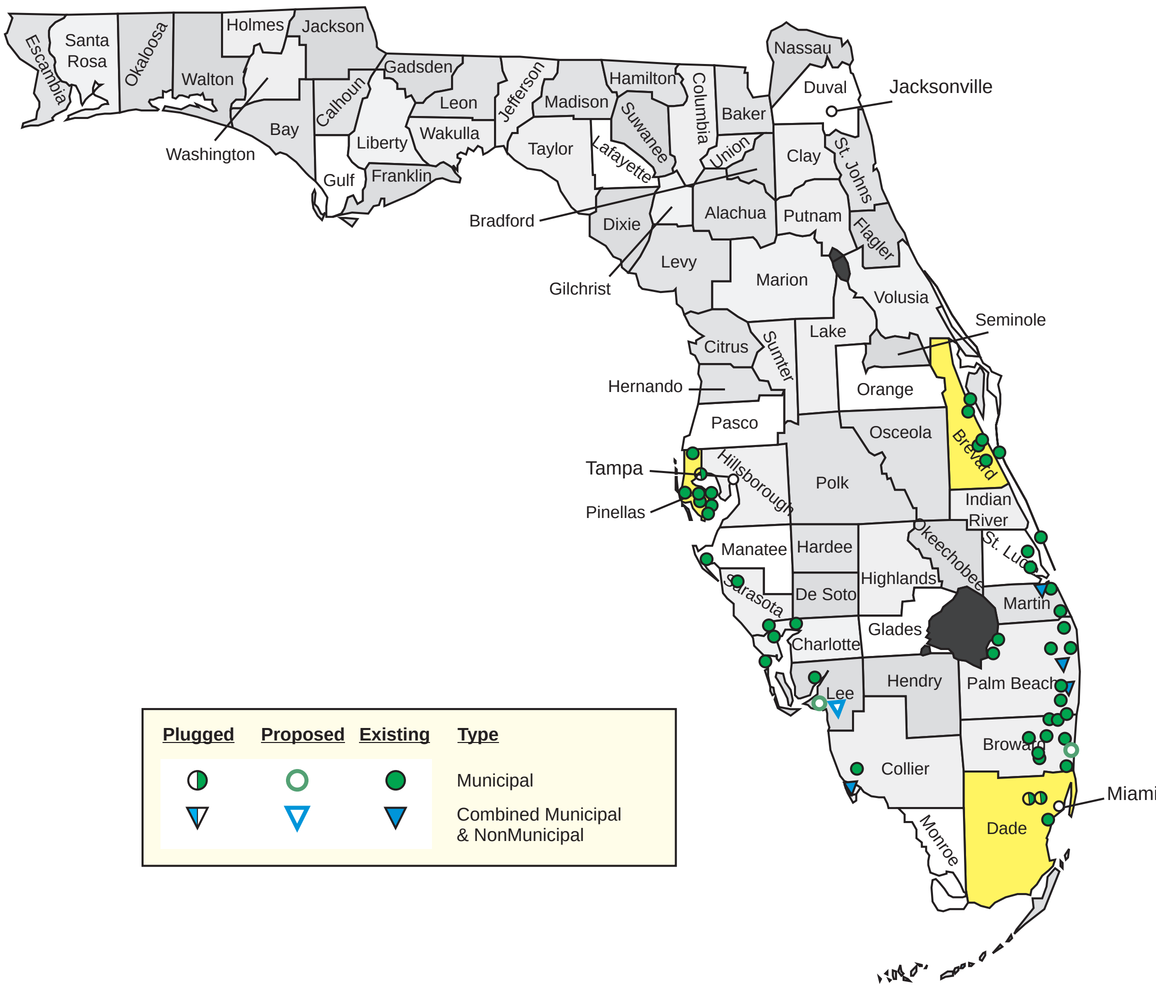

Figure 4-1. Locations of Class I Injection Wells in South Florida

The hydrogeologic system throughout much of South Florida consists of thick sequences of carbonate rocks overlain by clastic deposits (Tibbals, 1990; Broska and Barnette, 1999; Tihansky and Knochenmus, 2001). Three hydrogeologic features are common to Dade, Pinellas and Brevard counties: the presence of a relatively shallow surficial aquifer (called the Biscayne Aquifer in Dade County), the presence of a unit with lower relative hydraulic conductivity (the intermediate confining unit), and the presence of the Floridan Aquifer System. Figure 4-2 presents representative hydrogeologic cross sections that illustrate these and other features in the three counties.

The surficial aquifer (and the Biscayne Aquifer in Dade County) represents the uppermost hydrogeologic unit. These shallow aquifers lie above sequences exhibiting lower relative hydraulic conductivity **(**the intermediate confining unit) which, in turn, overlie the Floridan Aquifer System. The Floridan Aquifer System is divided into three distinct units, referred to as the Upper Floridan Aquifer, the middle confining unit, and the Lower Floridan Aquifer. Each of these aquifers is described in more detail below.

Deep-well injection is conducted within the Lower Floridan Aquifer in Dade and Brevard counties and within the Upper Floridan Aquifer in Pinellas County (Hutchinson, 1991; Hickey, 1982; Florida Department of Regulation, 1989; FDEP, 1999a).

4.3.1 Aquifers in South Florida

The Biscayne and surficial aquifers are the uppermost aquifers in South Florida. The surficial aquifer is composed of relatively thin layers of sands with some interbedded shells and limestone. Thickness of the surficial aquifer ranges from 20 to 800 feet, with the greatest thickness occurring in southeastern Florida (Adams, 1992; Barr, 1996; Lukasiewiez and Adams, 1996; Reese and Cunningham, 2000). The surficial aquifer yields relatively small volumes of water and is thus of limited use for public water supply; however, it is an important source of private water supplies (Miller, 1997).

The Biscayne Aquifer is the only formally named surficial aquifer unit in South Florida. The Biscayne Aquifer is the principal source of drinking water in Dade County. This aquifer extends along the eastern coast from southern Dade County into coastal Palm Beach County. The Biscayne Aquifer varies in thickness from a few feet to 240 feet and is composed of highly permeable limestone or calcareous sandstone (Meyer, 1989; Reese, 1994; Maliva and Walker, 1998; Reese and Memburg, 1999; Reese and Cunningham, 2000).

The intermediate confining unit lies beneath the surficial aquifers in Dade, Pinellas, and Brevard counties. Thick upper and lower clay layers confine depositional layers within this aquifer and limit, but do not eliminate the aquifer’s hydraulic conductivity (Miller, 1997).

The intermediate confining unit consists of sedimentary deposits from the Arcadia Formation of the upper Hawthorn Group and the Tamiami Formation. Figure 4-3 presents a geologic profile of South Florida. Unit thickness varies across a broad range, with the greatest unit thickness generally occurring in southeast Florida. Sedimentary layers are composed mostly of sand, sandy-limestone, and shell beds, with interlayered dolomite and clayey beds.

The intermediate confining unit is characterized by low hydraulic conductivity and acts as a confining unit, preventing or slowing migration between the overlying surficial aquifer and the underlying Floridan Aquifer System (Duerr and Enos, 1991; Barr, 1996; Knochenmus and Bowman, 1998). Similarly, the intermediate confining unit present in Dade County separates the Biscayne Aquifer from the Floridan Aquifer System.

The Floridan Aquifer System is subdivided into three distinct hydrogeologic units: the Upper Floridan Aquifer, the middle confining unit, and the Lower Floridan Aquifer. In general, the rocks of the Upper and Lower Floridan Aquifers consist of fractured and karstified limestones and dolomites of varying but generally high permeability. The hydrologic units of the Upper Floridan Aquifer correlate to the geologic units identified as the Suwannee Limestone, the Ocala Limestone, and the upper portion of the Avon Park Formation. The portions of the Upper Floridan Aquifer that yield lower amounts of water are typically associated with the Avon Park Formation (Hickey, 1982; Hutchinson, 1991; Hutchinson and Trommer, 1992; Reese, 1994).

The Upper and Lower Floridan Aquifers are separated by the middle confining unit, which contains lower-permeability rocks and clays (Meyer, 1989; Tibbals, 1990; Duncan et al., 1994; Reese, 1994; Reese and Memburg, 1999). The middle confining unit is comprised of rocks from the lower portion of the Avon Park Formation and upper part of the underlying Oldsmar Formation. These rocks consist of low-permeability clays, fine-grained limestones, and anhydrous dolomite, ranging in thickness across South Florida from 900 to 1,100 feet (Bush and Johnston, 1988; Duncan et al., 1994; Miller, 1997; Reese and Memburg, 1999).

The Lower Floridan Aquifer consists of three distinct layers within one depositional unit. The upper portion of this aquifer consists of dolostones and limestones of the Upper Oldsmar Formation (Duncan et al., 1994). The middle portion is commonly referred to as the Boulder Zone and consists of heavily karstified limestone and dolomite (Duncan et al., 1994; Maliva and Walker, 1998). Below this middle portion, the Lower Floridan Aquifer has properties that are largely consistent with the upper portion of the aquifer.

Within the Boulder Zone, solution channels, fractures, and widened joints allow channelized groundwater flow, sometimes at extremely rapid rates. Flow through fractures, solution channels, or other large voids are referred to as bulk flow through preferential flow paths, fracture flow, or channel flow.

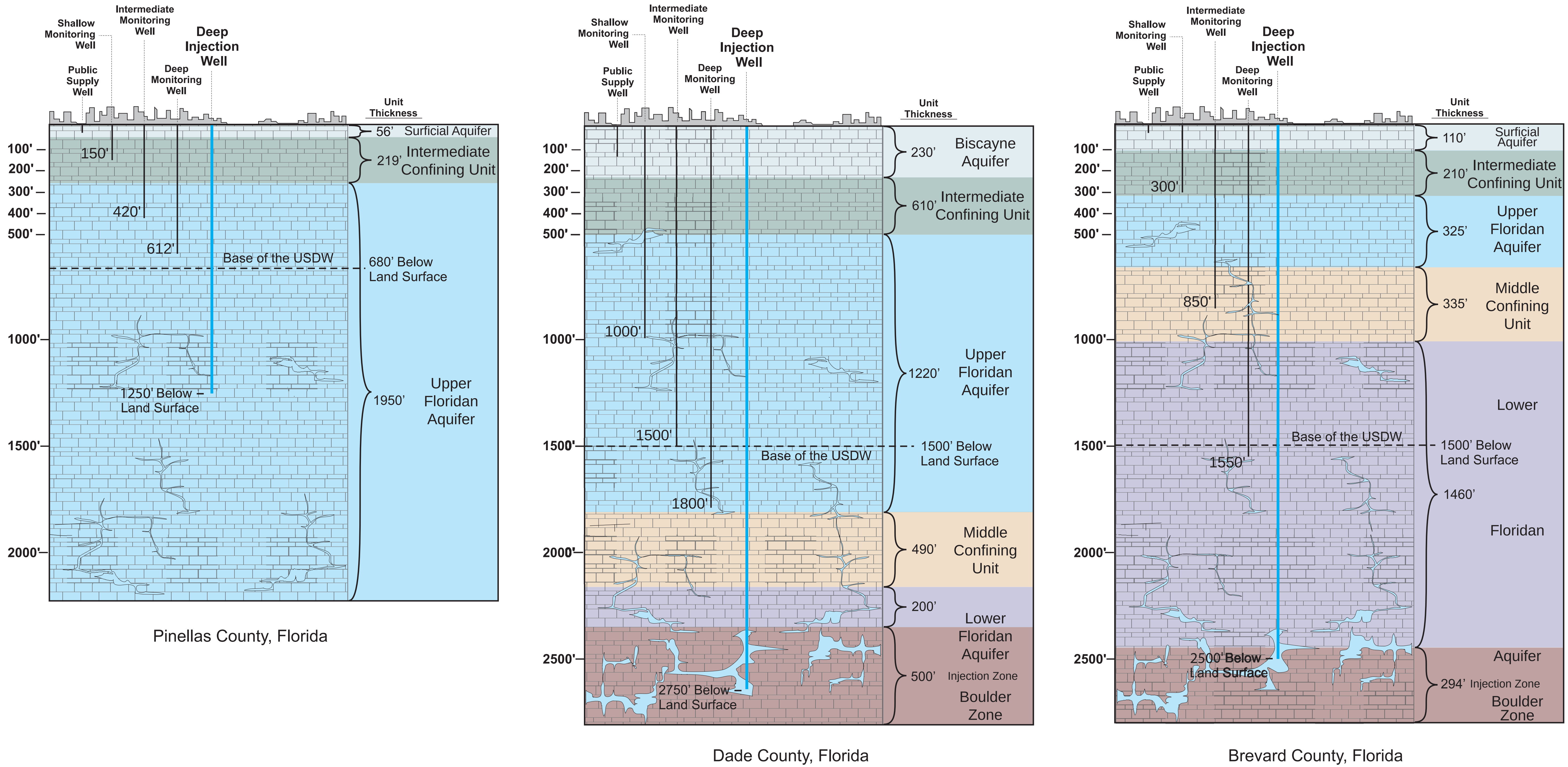

Figure 4-2. Representative Hydrogeologic Cross Sections

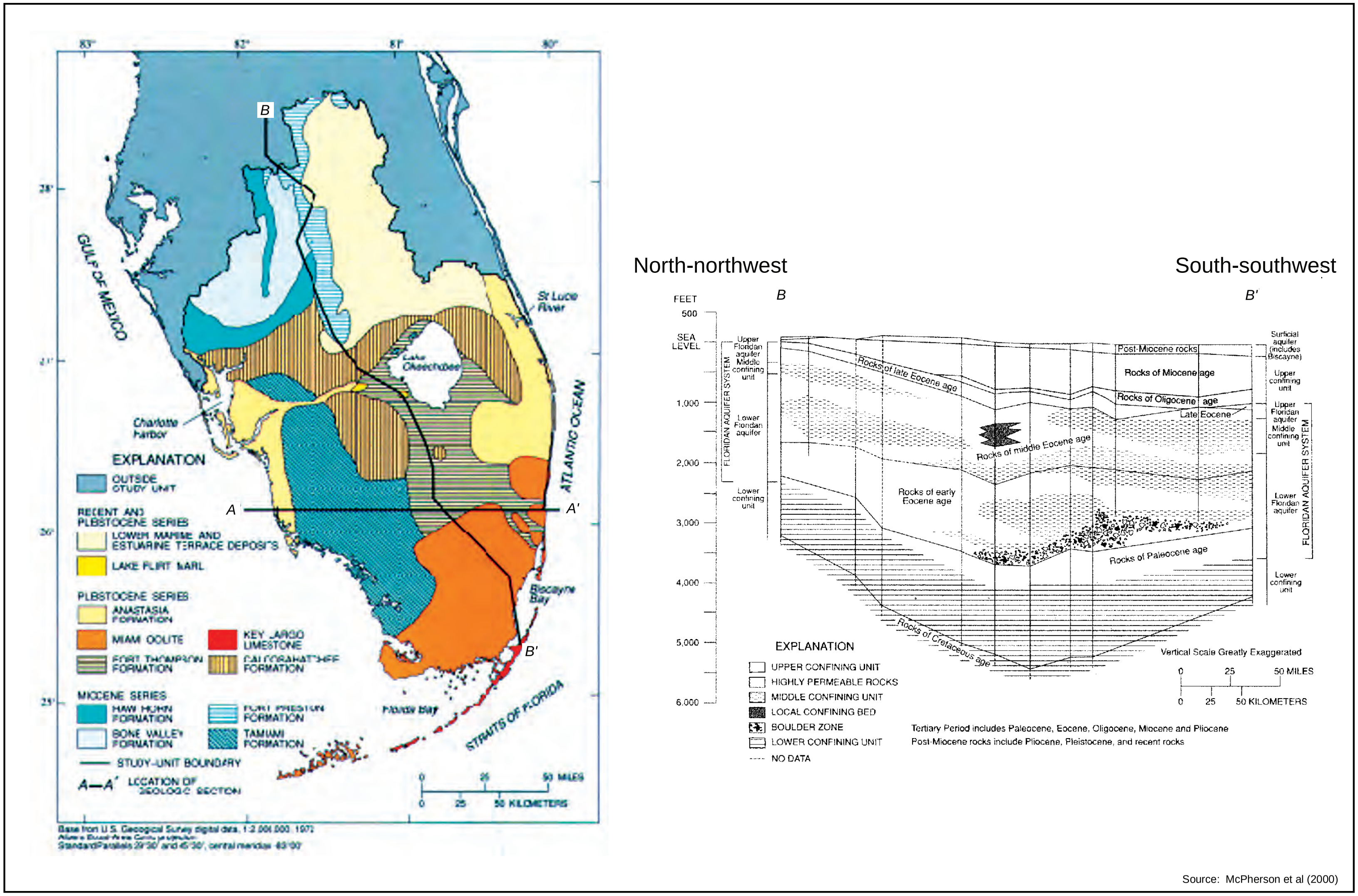

Figure 4-3. Geologic Profile of South Florida

Some reports indicate that groundwater flow in the Upper Oldsmar Formation is consistent with flow through porous media, with little or no channel flow (Meyer, 1989; Duncan et al., 1994; Maliva and Walker, 1998). This type of porous media flow through fine, interconnected pore spaces is typically less rapid than channel flow.

Representative values for hydraulic conductivity, porosity and thickness for each of the aquifer units in Dade, Brevard, and Pinellas counties are presented in the following sections. Mean (weighted) values are based on a statistical analysis of data reported in the scientific literature. Primary and secondary values of porosity and hydraulic conductivity are presented; these are used to examine flow through porous and fractured media, respectively.

4.3.2 Regional Conditions in Dade County

All documented deep-well injection in Dade County occurs within the Boulder Zone of the Lower Floridan Aquifer (Meyer, 1984, Duncan et al., 1994; Maliva and Walker, 1998). Typically, injection wells discharge within the top 250 to 300 feet of the Boulder Zone (FDEP, 1999a). In Dade County, this results in injection into saline groundwater at approximately 2,750 feet below the land surface. The base of the USDW is located approximately 990 feet above the injection zone, within the Upper Floridan Aquifer (Duerr, 1995) (Figure 4-2). Table 4-1 displays the representative values for hydraulic conductivity, porosity, and thickness for the aquifer units in Dade County.

Table 4-1. Dade County: Representative (Weighted Average) Hydraulic Conductivity, Porosity, and Thickness of Hydrologic Units

Hydrologic Unit or Subunit	**Hydraulic Conductivity (ft/day)**			**Porosity**		**Approx. Depth (ft below land surface)**	**Unit Thickness (ft)**
	Horizontal	Primary[1] Vertical	Secondary[2] Vertical	Primary[1]	Secondary[2]		
Biscayne Aquifer	1,524	15	15	0.31	0.31	0 – 230	230
Intermediate Confining Unit	90	0.1	2.38	0.31	0.1	230 – 840	610
Upper Floridan Aquifer	42	0.42	2.38	0.32	0.1	840 – 2,060	1,220
Middle Confining Unit	4.7	0.04	1.50	0.43	0.1	2,060 – 2,550	490
Lower Floridan Aquifer	0.01	0.1	0.1	0.4	0.1	2,550 – 2,750	200 [3]
Boulder Zone	6,540	65	65	0.2	0.2	2,750 – >3,250	500

Note: Descriptions of the statistical methods and literature-derived data are provided in Appendices 2 and 3.
[1] Primary values are used in scenario 1: flow through porous media.
[2] Secondary values are used in scenario 2: bulk flow through preferential flow paths.
[3] The Lower Floridan Aquifer extends below the Boulder Zone; this value for thickness represents only the portion *above* the Boulder Zone.

4.3.3 Regional Conditions in Pinellas County

Deep-well injection in Pinellas County is conducted in the Upper Floridan Aquifer, within the more permeable upper portion of the Avon Park Formation (Hickey, 1982; Hutchinson, 1991). Typically, injection wells discharge within the uppermost 100 to 300 feet of the Avon Park Formation (FDEP, 1989), approximately 1,250 feet below land surface (Figure 4-2). Wastewater is injected below the base of the USDW into moderately saline groundwater that has total dissolved solids (TDS) concentrations of 20,000 milligrams per liter (mg/L) (Hickey, 1982; Hutchinson, 1991). The base of the USDW is located approximately 570 feet above the injection zone, which is still within the Upper Floridan Aquifer (Duerr, 1995). Table 4-2 displays the representative values for hydraulic conductivity, porosity, and thickness for the aquifer units in Pinellas County.

Table 4-2. Pinellas County: Representative (Weighted Average) Hydraulic Conductivity, Porosity and Thickness of Hydrologic

Hydrologic Unit or Subunit	Hydraulic Conductivity (ft/day)			Porosity		Approx. Depth (ft below land surface)	Unit Thickness (ft)
	Horizontal	Primary[1] Vertical	Secondary[2] Vertical	Primary[1]	Secondary[2]		
Surficial Aquifer	29	7	7	0.31	0.31	0 – 56	56
Intermediate Confining Unit	4	1.2	1.5	0.31	0.1	56 – 275	219
Upper Floridan Aquifer	22	0.3	0.3	0.23	0.1	275 – 2,223	1,948

Note: Descriptions of the statistical methods and literature-derived data are provided in Appendices 2 and 3.
[1] Primary values are used in scenario 1: flow through porous media.
[2] Secondary values are used in scenario 2: bulk flow through preferential flow paths.

4.3.4 Regional Conditions in Brevard County

Deep-well injection in Brevard County occurs within the Lower Floridan Aquifer, approximately 2,500 feet below land surface. The base of the USDW is also located in the Lower Floridan Aquifer, approximately 1,500 feet below the land's surface and 950 feet above the injection zone (Duerr, 1995). The middle confining unit acts as a hydrologic barrier that separates and hydrologicly confines the Lower Floridan Aquifer from the Upper Floridan Aquifer (Figure 4-2). Table 4-3 displays the representative values for hydraulic conductivity, porosity, and thickness for the aquifer units in Brevard County.

Table 4-3. Brevard County: Representative (Weighted Average) Hydraulic Conductivity, Porosity and Thickness of Hydrologic Units

Hydrologic Unit or Subunit	Hydraulic Conductivity (ft/day)			Porosity		Approx. Depth (ft below land surface)	Unit Thickness (ft)
	Horizontal	Primary[1] Vertical	Secondary[2] Vertical	Primary[1]	Secondary[2]		
Surficial Aquifer	56	13	13	0.31	0.31	0 – 130	130
Intermediate Confining Unit	20	0.1	2.38	0.31	0.1	130 – 340	210
Upper Floridan Aquifer	20	0.2	2.38	0.26	0.1	340 – 665	325
Middle Confining Unit	0.8	0.04	1.50	0.43	0.1	665 – 1,000	335
Lower Floridan Aquifer	0.1	0.1	0.1	0.4	0.1	1,000 – 2,460	1,460[3]
Boulder Zone	650	65	65	0.2	0.2	2,460 – >2,754	294

Note: Descriptions of the statistical methods and literature-derived data are provided in Appendices 2 and 3.
[1] Primary values are used in scenario 1: flow through porous media.
[2] Secondary values are used in scenario 2: bulk flow through preferential flow paths.
[3] The Lower Floridan Aquifer extends below the Boulder Zone; this value for thickness represents only the portion *above* the Boulder Zone.

4.4 Groundwater Quality and Fluid Movement in South Florida

Deep-well injection facilities in South Florida conduct routine sampling and analysis of groundwater taken from units overlying injection zones. This information may be used to identify instances of apparent unintended movement of fluids from the injection zone, occurring now or in the past, although the monitoring wells are located near the injection wells and would not be capable of indicating the areal extent of the contamination.

There were few data collected to characterize the quality of deep groundwater resources in South Florida prior to construction and operation of injection wells. The U.S. Geological Service conducted a study of the water resources in Dade County prior to well completion and commencement of operations (Earle and Meyer, 1973). The study showed chloride concentrations between 15 and 14,500 mg/L.

Data are available for characterizing the quality of groundwater resources since injection-well construction and operation began. Englehardt et al. (2001) compiled a limited data set that includes information about the levels of inorganic contaminants present in lower and upper native (or ambient) groundwater monitoring zones (Appendix Table 1-1). Though it cannot be said conclusively that these data characterize preoperation

conditions, the data are sufficient for illustrating two points. First, deep native groundwater in southeast Florida does appear to exceed several primary or secondary drinking-water standards (maximum contaminant levels, or MCLs). Second, for some contaminants (for example, cadmium, lead, antimony, aluminum, iron), there is reason to conclude that these levels are of natural origin (resulting, for example, from the dissolution of the native aquifer matrix) and not attributable to any aspect of well construction or operation. For some other contaminants (for example, thallium, beryllium), it is less clear why there are slightly elevated levels present in upper and lower groundwater monitoring zones.

The Florida DEP has compiled groundwater monitoring information collected during construction and operation of deep-injection wells. Florida DEP has used this information to develop a map (reproduced as Figure 4-4) that depicts fluid movement associated with deep-injection wells throughout South Florida. This map identifies facilities where confirmed and probable fluid movement has occurred and specifies whether this movement is into a USDW or non-USDW (FDEP, 2002). Non-USDWs are used in this figure to depict wells with movement into aquifers containing groundwater of greater than 10,000 mg/L TDS concentration.

The Florida DEP has concluded that approximately three deep-well injection sites in Pinellas, Dade, and Palm Beach counties have caused confirmed fluid movement into USDWs (Figure 4-4). An additional six deep-well injection facilities in Pinellas and Brevard counties have caused probable fluid movement into USDWs. As many as nine additional facilities have caused fluid movement into non-USDWs, predominantly in Broward County (Figure 4-4).

Approximately 18 deep-well injection facilities appear to be associated with some form of unintended fluid movement from the injection zone. Deep-well injection facilities in many other parts of South Florida do not appear to have caused unintended fluid movement. Multiple facilities in each of several counties (Charlotte, Collier, Lee, Sarasota, and St. Lucie counties) have operated for years with no apparent fluid movement.

The sections that follow present data and information specific to Dade, Pinellas, and Brevard counties. These sections present information made available through exhaustive data collection efforts and the close cooperation of Florida DEP and water utilities in South Florida. These sections do not provide the same types and amounts of data for each county. The data and information do, however, serve as a means of better understanding what is known about the condition of groundwater resources, changes in water quality, and the occurrence of confirmed or probable fluid movement in South Florida.

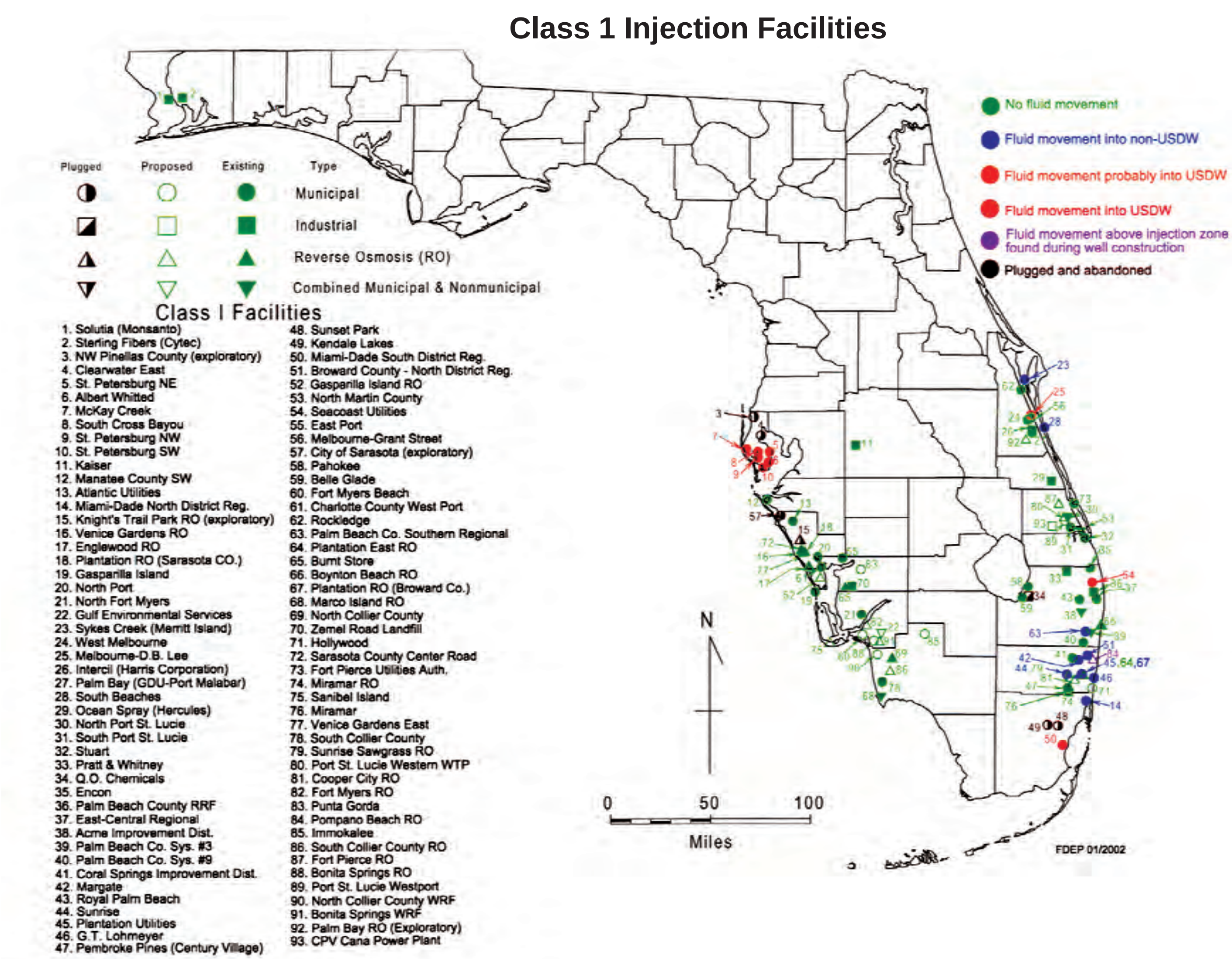

Figure 4-4. Fluid Movement Associated with Class I Deep Well Injection Facilities in South Florida

Analytical parameters widely used as indicators of fluid movement include dissolved ammonia, TDS, chloride, and fecal coliforms. Dissolved ammonia (or ammonium) is present in secondary-treated wastewater but is not typically found in native groundwater. Levels of chloride and TDS indicate if there has been a “freshening” of naturally saline native groundwater, which may suggest fluid migration of treated wastewater. Dissolved chloride is present at very low concentrations in treated wastewater but occurs at very high concentrations in Florida’s deep aquifers; reaching concentrations similar to seawater (20,000 to 30,000 mg/L). Detection of relatively “fresh” water (low chloride or TDS concentrations) in deep monitoring wells may be interpreted as evidence of fluid movement.

Fecal coliforms are present in secondary-treated wastewater at varying concentrations, depending upon whether or not the wastewater has undergone basic disinfection. (Secondary treated wastewater that has undergone basic disinfection may still contain concentrations of fecal coliforms; see Appendix 1.) Most fecal coliform strains are not pathogenic and are used only as indicators for the presence of other pathogenic microorganisms. Chapter 3 discusses pathogenic strains such as *E. coli* and examines some of the issues related to use of fecal coliforms as an indicator.

4.4.1 Dade County Groundwater Monitoring Information

Much of the groundwater monitoring information available for Dade County concerns the South District Wastewater Treatment Plant (SDWWTP), where there has been confirmed fluid movement into the USDW. Data and information obtained from monitoring wells at this facility provide evidence that upward migration of injected wastewater has occurred.

The SDWWTP uses 17 deep-injection wells, of which 13 are currently permitted for injection. Monitoring wells associated with each deep-injection well were constructed to monitor the Upper Floridan Aquifer, typically at two depths. Most monitoring wells at the site monitor zones at 1,500 feet and 1,800 feet below surface. The first of these zones represents the base of the lowermost USDW. Monitoring of the 1,800-foot zone provides an early warning of fluid movement and contamination below the base of the USDW.

Elevated concentrations of ammonia have been detected in monitoring wells at both the 1,500- and 1,800-foot zone. Elevated concentrations of dissolved chlorides have also been detected; these may indicate displacement of native formation water in an upward direction. Fecal coliforms have been detected in a number of monitoring wells.

In 1996, monitoring wells (FA-14 through FA-16) began to detect elevated ammonia concentrations in the 1,500-foot zone. Beginning in 1998, two of these wells, those nearest to a well suspected of mechanical failure (BZ-1), were purged of millions of gallons of water. This was initially accomplished by allowing them to flow freely by artesian pressure. Pumps were subsequently installed to increase the flow rate.

A purging report from December 1998 (SDWWTP, 1998) indicates that there was a slight decrease in the concentrations of ammonia detected by monitoring well FA-16 in

response to purging. In another well, FA-15, there was a larger drop in ammonia concentrations after purging but subsequently these concentrations stabilized at a lower, but still elevated, level. Detected levels of ammonia were higher than background levels for these depths, and as such, were interpreted as an indicator of potential contamination resulting from movement of injected fluids.

In 1994, around the time when chloride anomalies were first noticed in BZ-1, ammonia was detected in water taken from the 1,500-foot monitoring zone in newly constructed monitoring wells FA-5 though FA-8 (adjacent to newly constructed injection wells IW-13 through IW-16). The first samples taken from FA-5 through FA-8, soon after completion in 1994, showed elevated concentrations of ammonia.

Monitoring well FA-5 was purged between 1996 and 1998. Ammonia concentrations decreased by 43% during purging. When purging stopped, ammonia levels returned to approximately the same concentrations as were present before purging.

Elevated ammonia concentrations were detected in monitoring wells placed in the 1,800-foot zone (including wells FA-11 and FA-12) when these wells were first used to perform monitoring (February 1996). These wells were included in the purging program with little apparent impact to monitored ammonia concentrations. Monitoring has continued to detect elevated ammonia concentrations in these wells.

The authors of this report (SDWWTP, 1998) were unable to determine whether elevated ammonia levels existed as part of a finite volume of water or whether there was a continuous source. There has been no information to attribute elevated levels of ammonia in the areas surrounding FA-5 through FA-8 to conduits created by injection activities at the site. In 1994, there were no known anthropogenic conduits (“artificial penetrations”) between the Boulder Zone and the 1,500-foot zone close to these monitoring wells. In 1994, there were no wells in this part of the facility suspected of having faulty construction and no other operational problems.

An injection well, IW-2, near FA-11 and FA-12, may have contributed to movement of fluid from the injection zone to the 1,800-foot monitoring zone. However, periodic tests of this well (radioactive tracer surveys, a temperature survey, and television survey of inside the well bore) have failed repeatedly to identify any well construction problems above 2,500 ft.

The SDWWTP purging report also provides information on concentrations of fecal coliforms detected in groundwater between 1987 and 1995 (SDWWTP, 1998). For many wells and sampling dates, monitoring data indicate groundwater concentrations below the detect level (Appendix Table 1-5). Low concentrations of fecal coliform contamination (for example, tens of colonies per 100 milliliters (mL)) have been detected with roughly twice the frequency of higher concentrations. High concentrations (for example, several hundred colonies per 100 mL and, in one instance, greater than 2,000 colonies per 100 mL) were occasionally detected in groundwater, generally at depths of approximately 1,000 feet (Appendix Table 1-5).

Episodes of high fecal coliform contamination appear to have been most frequent during 1992 and, to a lesser extent, during 1993 and 1994 (Appendix Table 1-5). In 1995, the SDWWTP disinfected a number of monitoring wells. Following disinfection, there were fewer fecal coliform detections in groundwater, and only low concentrations were detected.

4.4.2 Pinellas County Groundwater Monitoring Information

Groundwater monitoring information is available in Pinellas County for the City of St. Petersburg facilities, where there has been probable fluid movement (and, in one case, confirmed fluid movement) into USDWs. Data and information obtained from monitoring wells at these facilities provide evidence that upward migration of injected wastewater has occurred. A review of this information follows.

The four St. Petersburg wastewater reclamation facilities (WWRFs) treat wastewater to reclaimed standards and provide high-level disinfection. Reclaimed wastewater that is not used by the reuse system (either because its volume exceeds current demands or because it does not meet stringent quality standards) is pumped into the middle and lower portions of the Upper Floridan Aquifer via 10 deep-injection wells. Injection zones in southern Pinellas County contain water with a high TDS content; these injection zones are not classified as USDWs.

The 2000 Annual Summary Report for St. Petersburg’s four injection facilities (CH2M Hill, 2001) provides evidence that upward migration of injected wastewater has occurred over the 20 years since injection operations first began. Monitoring data reveal that, at more than one of these facilities, there has been significant change in water quality both below and within USDWs.

At the Albert Whitted facility, the largest of the St. Petersburg facilities, water-quality profiles reveal significantly altered water quality above the injection zone. In 1989, background pre-injection TDS concentrations ranged from less than 2,700 mg/L at approximately 250 feet to 35,000 mg/L in the injection zone at 700 feet. (The 250-foot zone is both a USDW and part of the Upper Floridan Aquifer.) Once injection operations commenced, monitoring detected TDS concentrations greater than 7,400 mg/L within the USDW in 1993 before these concentrations declined to approximately 1,700 mg/L in 2000. At 375 feet, near the base of the USDW, TDS increased from 6,300 mg/L in 1986 to more than 15,000 mg/L in 1989. TDS then declined to 1,500 mg/L in 2000 (CH2M Hill, 2001). The most likely reason for these trends is that comparatively fresh and buoyant injectate has pushed highly saline formation waters upward into USDWs.

Ammonia concentrations detected within the 550-foot zone at the Albert Whitted facility have increased from as low as 0.4 mg/L in 1986 to as high as 17.8 mg/L in 1999 (CH2M Hill, 2001). These increases have coincided with observed decreases in TDS concentration.

A similar situation appears to have occurred at the Northeast WWRF. A single monitoring well completed into the USDW at approximately 150 feet has detected significant changes in TDS concentration. TDS levels increased from as low as 1,280 mg/L in 1980 to as high as 24,000 mg/L in 2000 data (CH2M Hill, 2001). Decreasing TDS levels have been detected in monitoring wells placed below the USDW.

At the Northwest WWRF, there is just one monitoring well, placed below the base of the lowermost USDW. Since 1985, monitored TDS levels have fluctuated widely. Concentrations decreased slightly from an initial concentration of 11,100 mg/L, then increased to over 20,000 mg/L, and finally decreasing to as low as 9,300 mg/L in 2000 (CH2M Hill, 2001). Data for this facility are sparse and difficult to interpret, but the trend appears to be consistent with data from the Northeast WWRF and the Albert Whitted facility.

At the Southwest WWRF, several wells that monitor non-USDWs have detected significant decreases in TDS concentration. One well that monitors water quality within the USDW at approximately 320 feet has detected increases in TDS concentration from 5,000 mg/L in 1979 to more than 11,000 mg/L in 2000 (CH2M Hill, 2001).

Data sets for the Northeast, Northwest, and Southwest facilities are not as complete as those available for the Albert Whitted facility. Nevertheless, it does appear that these WWRFs are experiencing a similar displacement of higher-salinity groundwater in an upwards direction by injected wastewater. This displacement may be occurring at a slower rate than has occurred at the Albert Whitted WWRF. There is some evidence at the Northeast, Northwest, and Southwest facilities that ammonia concentrations are increasing in the same zones that are experiencing declines in TDS concentration.

In 1993, the City of St. Petersburg initiated a program to identify and monitor offsite wells. Although most wells appear to be at shallow depths, private water-supply wells as deep as 200 feet have been identified near the facilities. It is believed that all wells are completed into a USDW and that these wells provide water primarily for irrigation. The 2000 Annual Summary Report indicates that monitored parameters (TDS, chlorides, sodium, conductivity) are within the range of unimpacted waters (CH2M Hill, 2001). No sampling data are included to substantiate these statements.

4.4.3 Brevard County Groundwater Monitoring Information

4.4.3.1 South Beaches

At the South Beaches facility in Brevard County, it is probable that there has been fluid movement into the overlying USDW. Data and information obtained from monitoring wells at this facility provide evidence that upward migration of injected wastewater into the USDW may have occurred.

A 2001 report prepared for the South Beaches facility (CDM, 2001) includes ground-water monitoring data for three monitoring wells at the site. A shallow well, MW-1,

monitors the Ocala formation from 300 to 350 feet. Well MW-3, placed at an intermediate depth, monitors the middle of the Upper Floridan Aquifer from 1,200 feet to 1,320 feet. A deep well, MW-2, monitors the lower part of the Upper Floridan Aquifer from 1,550 feet to 1,700 feet.

The deep well, MW-2, monitors below the lowermost USDW where significant changes in water quality occurred between 1987 and 2001. Conductivity and concentrations of chloride and TDS decreased rapidly for the first several years after commencement of injection operations. In recent years, these concentrations have stabilized (CDM, 2001).

Nitrate concentrations have remained fairly constant, just at the detectable level. Ammonia concentrations, initially at approximately 2 mg/L, increased slightly in 1991, but steadily decreased thereafter to 2001 levels at approximately 0.5 mg/L. Between 1987 and July of 1991, total Kjeldahl nitrogen (TKN) increased slightly to approximately 3 mg/L, at which time it began to decrease. Detected concentrations of TKN are now similar to the original ambient concentration of approximately 0.5 mg/L (CDM, 2001).

MW-3, the intermediate monitoring well, was constructed at a later date than the other two wells; monitoring began in 1990. Since 1991, detected concentrations of TDS have increased from approximately 3,500 mg/L to nearly 10,000 mg/L. Moderate increases in the concentration of chloride, increases in conductivity, and a slight increase in ammonia have also been observed. There has been no apparent change in the detected levels of nitrate and TKN.

Monitoring data from the shallow well, MW-1, indicate that groundwater quality has remained unchanged over the course of injection operations. This suggests that fluid movement has not reached these shallow depths (300 to 350 feet).

4.4.3.2 Palm Bay

The Port Malabar Wastewater Treatment Plant in Brevard County injects reclaimed wastewater at approximately 3,000 feet. Test wells monitor the Lower Floridan Aquifer at 1,534 to 1,650 feet and the shallower Upper Floridan Aquifer at 400 to 472 feet. Injection began in 1987; monitoring results were available for some parameters beginning in 1988 (HAI, 2000).

Monitoring performed in the deep interval reveals that nitrate and ammonia concentrations have varied widely, but not with any apparent increasing or decreasing trends. TDS concentrations have fallen from approximately 20,000 mg/L to approximately 15,000 mg/L. Chloride showed a slightly increasing trend from approximately 10,000 mg/L to 12,000 mg/L (HAI, 2000). No appreciable changes in TDS, chloride, nitrate, or ammonia have been detected in the shallow interval.

4.5 Regulations and Requirements for the Deep-Well Injection Option

The siting, construction, operation, and management of deep-injection wells are governed by a number of Federal and State regulations, which are summarized below.

Class I injection wells are prohibited from causing the movement of any fluid into USDWs. These are defined as aquifers, or portions of aquifers, having a sufficient quantity of groundwater to supply a public water system, and containing a TDS concentration of less than 10,000 mg/L (40 CFR 144.3, Florida Administrative Code (FAC) 62-520.410(1), and FAC 62-528.200(60)). However, this definition does not include aquifers, or portions of aquifers, that have been specifically exempted from this regulatory definition.

40 CFR 144.12 (b) and FAC 62-528.110(2) apply specifically to Class I injection and prohibit the movement of any contaminant into USDWs. This prohibition has been established as a means of ensuring that no Class I injection practices are allowed to endanger USDWs, as required by the Safe Drinking Water Act.

Criteria and standards for the construction, operation, and monitoring of nonhazardous Class I injection wells are given in 40 CFR Part 146 (Subpart B). 40 CFR 146.12 (b) and FAC 62-528.410(1) require that Class I wells be cased and cemented to prevent the movement of fluids into or between USDWs. 40 CFR 146.13(a)(1) and FAC 62-528.415(1) further state that injection pressures may not initiate fractures in the confining zone or cause the movement of injection or formation fluids into a USDW.

State of Florida permit requirements for Class I injection wells are defined by FAC Chapter 62-528, Underground Injection Control (FDEP, 1999b). Requirements include specifications for well construction, for defining hydrologic conditions relative to the site, for ensuring mechanical integrity of injection wells, and for proper well operation.

Construction requirements for Class I wells are set forth in 40 CFR 146.12 and FAC 62-528.410. State requirements, at FAC 62-528.425 and 62-528.300 (6), regulate mechanical integrity of injection wells (FDEP, 1999b). Operating requirements are set forth in 40 CFR 146.13(a) and FAC 62-528.415. Monitoring requirements are set forth in 40 CFR 146.13(b) and FAC 62-528.425.

Two additional sets of requirements apply to Class I nonhazardous wells in Florida. FAC 62-600.540(4) requires certain types of surface equipment at all injection-well facilities. Facilities must also comply with FAC 62-600, Domestic Wastewater Facilities (FDEP, 1996).

In Florida, Class I wells injecting treated wastewater into Class G-IV waters must provide secondary treatment, at a minimum, and must meet pH limitations. Class G-IV waters are defined as groundwater for nonpotable use or groundwater in confined aquifers, that has a TDS content of 10,000 mg/L or greater (FAC 62-520.410). Disinfection is not required,

but all Class I well permittees must maintain the capability to disinfect (FAC 62-600.540).

Secondary treatment requires an effluent contain not more than 20 mg/L 5-day biochemical oxygen demand (CBOD5) and 20 mg/L total suspended solids (TSS) or that 90% of CBOD5 and TSS be removed from the wastewater influent, whichever is more stringent. At a minimum, all facilities practicing Class I deep-well injection must meet the 20 mg/L effluent limitation. All facilities must be designed and operated to maintain effluent pH within the range of 6.0 to 8.5, taking into account background water quality (FAC 62-600).

4.6 Problem Formulation

Every day, hundreds of millions of gallons of treated wastewater is injected into deep-injection wells. Subsequent migration of this wastewater, and of any dissolved or entrained wastewater constituents, may result in exposure to receptors (including USDWs and water-supply wells). Migration of injected wastewater and the fate and transport of wastewater constituents from the point of injection to receptors serve an important focus for this option-specific risk analysis.

As has been described in Chapter 3, wastewater constituents that may act as stressors to human or ecological health can be grouped according to several broad categories (for example, pathogenic microorganisms or VOCs). Wastewater constituents (potential stressors) often exhibit unique physical, chemical, or biological behavior in the subsurface. Careful selection of representative stressors is meant to account for these differences in fate and transport. This analysis focuses on a limited number of representative stressors, each representing a larger category of stressor. Problem formulation, a process involving the collection and compilation of relevant sources of data and information, has served to identify the best available representative stressors for conducting this option-specific risk analysis.

The actions of large-scale physical, chemical, and biological processes in the subsurface are key considerations for this analysis. These processes define the exposure pathways that may be expected to bring injected wastewater (and stressors) into contact with receptors. Transport of injected wastewater is largely a physical process, dependent on patterns of advection or groundwater flow. Fate and transport of potential stressors, however, is dependent upon an entire suite of processes.

Injected wastewater that is completely and permanently confined within injection zones poses no risk to drinking water or ecological receptors; there is simply no exposure of receptors. Wastewater that does escape confinement and moves from the intended injection zone may pose a risk if receptors are exposed. The time of travel, which is the time that elapses between injection (or escape from confinement) and exposure of the receptor, is directly related to the risks that such exposure might introduce.

This analysis attempts to account for the complex physical phenomena that influence whether fluid movement from the injection zone will occur. Furthermore, this analysis is designed to investigate a number of critical questions about the nature of any such movement:

- What physical force components drive fluid movement (for example, buoyancy, pressure head)?
- How do differences between the characteristics of native groundwater and injected wastewater (for example, salinity, temperature, density) affect movement?
- What hydrogeologic units and unit properties most affect patterns of movement?
- How might features in the sequence of confining and overlying units (for example, fractured rock, solution channels), if they are present, result in changes in movement?
- Can the characteristics of injected wastewater and the properties of hydrogeologic units be quantified in a way that would allow them to be accurately depicted by modeling efforts?

This analysis produces modeled estimates of vertical time of travel that allow consideration of each question. However, accounting for the complexity at any single site is a challenge, and these challenges are greatly magnified by the broad scope of this analysis. Data gaps and remaining uncertainties are such that this analysis requires use of best professional judgment; these models are not field calibrated. However, this option-specific risk analysis, while depending in part upon fate and transport modeling, does not depend solely or entirely on this modeling. Model outputs are considered jointly with all other sources of information, including groundwater monitoring performed in geologic units above the injection zones.

Differences in fluid temperature and density between native and injected water affects relative buoyancy. Injected wastewater has fluid densities that are roughly equivalent to those of fresh water (FDEP, 1999a). This wastewater is injected at depths where the native groundwater is saline or hypersaline (Reese, 1994; Knochenmus and Bowman, 1998; Reese and Memburg, 1999). The comparatively lighter, less-dense wastewater responds to a buoyancy force component that promotes vertical movement.

Another factor influencing fluid movement in subsurface geology is injection pressure. In many settings where underground injection is practiced, increases in pressure head (resulting from injection pressure) play a crucial role in determining the movement of fluids. In parts of South Florida, where injection zones demonstrate a great capacity to accept injected fluid (for example, the Boulder Zone), this force component may be less significant. This analysis accounts for the injection-pressure force component, with attention to differences that exist between the injection zones typical of Dade, Brevard, and Pinellas counties.

The subsurface heterogeneity that is characteristic of South Florida introduces complexity. Unit properties (for example, hydraulic conductivity, porosity, effective porosity) vary from one unit to the next, within a given unit from one site to another, and even within a given unit at a given site. Accounting for this heterogeneity presents a

significant challenge in evaluating risk. In an effort to explore possibilities where available data are limited or inconclusive, this analysis relies on an exhaustive review of available data concerning unit properties and considers two different scenarios as it examines uncertainty.

One example of such uncertainty regards the presence or absence of fractures, fissures, and solution channels throughout some units in South Florida. Such conduits allow for rapid groundwater and wastewater movement. Although seismic techniques, well-bore imaging techniques, and other tools are available to help identify these features, such information is not generally or widely available.

The goal of this analysis is to determine the relative risk to potential receptors. To help evaluate this risk, this analysis uses estimated times of travel and basic information about the behavior of representative stressors and conditions in aquifer systems to translate initial concentrations at injection into final concentrations at receptors. An exposure analysis attempts to account for the various processes that attenuate and dilute stressors during the course of transport. However, as noted above, attenuation and dilution are exceedingly difficult to model in heterogeneous environments. Furthermore, the best available models (models that would more accurately describe three-dimensional fate and transport) have data requirements that, in this case, cannot be met, at least for the large study area. Necessarily, this analysis applies a number of conservative assumptions in describing the fate of stressors, and these assumptions are intentionally designed to overstate, rather than understate, exposure and risk.

Risk characterization is accomplished by comparing the anticipated final concentrations at receptors with assessment endpoints. Where assessment endpoints in the form of drinking-water-quality or other standards are not available, a weight-of-evidence approach is applied. The weight-of-evidence approach relies on the application of qualified professional judgment to use and apply findings from the scientific literature, especially information regarding dose response or ecological thresholds.

4.7 Conceptual Model of Potential Risks for the Deep-Well Injection Option

Figure 4-5 presents a generic conceptual model for the deep-well injection wastewater management option. The primary source of stressors is defined as the wastewater treatment plant from which treated effluent is pumped to one or more deep-injection wells. The rate of discharge varies, depending on the size and operational status of the facility but is generally measured in millions of gallons per day.

Wastewater discharged to the subsurface (injectate) enters geologic formations within the Floridan Aquifer System at a preselected elevation called the injection zone. Injection zones range from between 650 and 3,500 feet below the land surface. Injection zones are located at an elevation where one or more highly permeable zones have been identified (such as the Boulder Zone in the Lower Floridan Aquifer). Injection zones are saturated with groundwater of salinity similar to seawater.

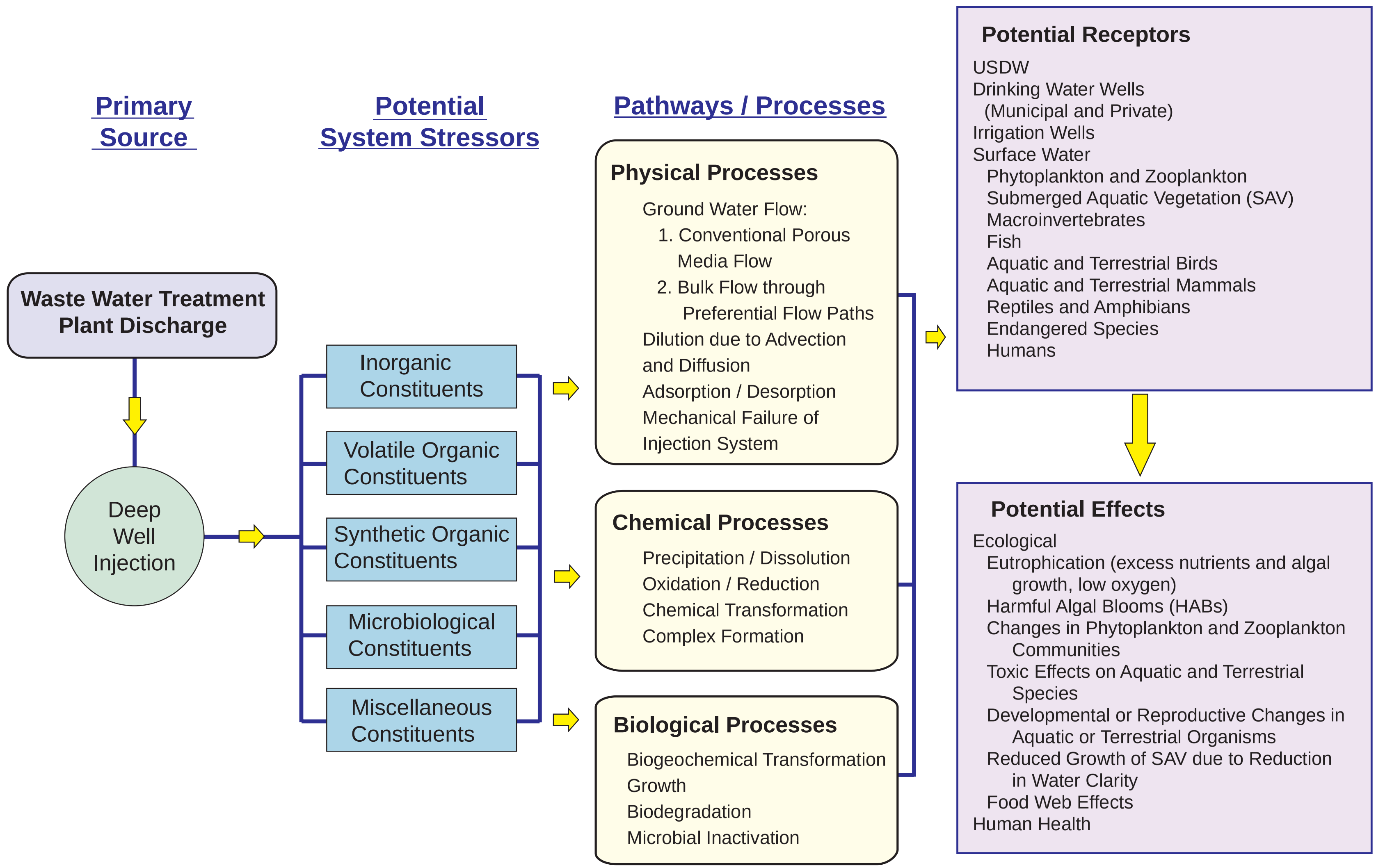

Figure 4-5. Conceptual Model of Potential Risks for the Deep Well Injection Option

4.7.1 Potential Stressors

Potential stressors include any dissolved or entrained wastewater constituents that may reach receptors in sufficient concentration to cause adverse human health or ecological effects. This may include pathogenic microorganisms, certain metals and inorganic substances, synthetic organic compounds and VOCs, and hormonally active agents.

Appendix 1 presents data to characterize the quality of treated wastewater. Appendix Table 1-1 presents data on a wide range of organic and inorganic wastewater constituents. Appendix Table 1-3 and Appendix Table 1-4 present data on microbial wastewater indicators that may be present in treated wastewater.

Several data sets included in Appendix Table 1-1 offer information to characterize injected wastewater in South Florida:

- Data obtained from the South Beaches Wastewater Treatment Facility in Brevard County describes the quality of wastewater treated to advanced wastewater treatment (AWT) standards.
- Data obtained from the Albert Whitted Water Reclamation Facility in Pinellas County describes the quality of reclaimed water (wastewater that has received advanced secondary treatment).
- Data obtained from a study sponsored by the South Florida Water Environment Association Utility Council (Englehardt et al., 2001). These three data sets describe wastewater treated by different means. In southeast Florida, where this study was conducted, secondary treatment is the norm for deep-well injection facilities.
- Data obtained from the SDWWTP in Dade County describes wastewater that has received secondary treatment.

These data reveal trends for the quality of injected wastewater. Very few wastewater constituents for which there are primary drinking-water standards (MCLs) have been found to exceed standards at the point of injection. There are no metals, synthetic organic compounds, or VOCs that appear to exceed primary drinking-water standards.

There are data to suggest that a small number of wastewater constituents may exceed primary drinking-water standards at injection. However, these constituents do not consistently exceed MCLs at the various facilities from which data have been collected. Secondary drinking-water standards for TDS, color, and odor do appear to be routinely exceeded at the point of injection.

Nitrate concentrations in excess of the MCL (10.0 mg/L) have been reported by the following facilities: South Port St. Lucie (11.0 mg/L), Gasparilla Island (11.99 mg/L), Seacoast Utilities (12.8 mg/L), Pahokee (14.0 mg/L), Miramar (27.0 mg/L), and North Fort Myers (36.0 mg/L). (Of these facilities, only Seacoast Utilities in Palm Beach County has detected any form of fluid movement from the injection zone; see Figure 4-3). No data collected from facilities in Dade, Pinellas, or Brevard counties indicate nitrate concentrations in excess of the MCL (Appendix Table 1-1).

At the South Beaches Water Treatment Facility in Brevard County, which provides advanced wastewater treatment, concentrations of total trihalomethanes in excess of the MCL (80.0 μg/L) have been reported. Presumably, wastewater chlorination is responsible for elevated concentrations (230 μg/L) of trihalomethanes, which are byproducts generated during the disinfection process.

Table 4-4 presents concentrations for those representative organic and inorganic stressors selected for further analysis and consideration. (All of this data may be found within Appendix Table 1-1.) For several of these stressors, there is no primary drinking-water standard. Some are of concern primarily because of their potential to act as ecological stressors (for example, copper, nitrogen, orthophosphate).

Table 4-4. Concentrations of Representative Organic and Inorganic Stressors

Wastewater Constituent	South Beaches WTF[1], Brevard (Advanced)	Albert Whitted WRF[1], Pinellas (Reclaimed)	Utility Council Report[1], SE FL (Secondary)
Arsenic (MCL of 0.05 mg/L)	<0.005 mg/L	<0.003 mg/L	0.003 mg/L
Copper (action level of 1.0 mg/L)	N/A	0.0086 mg/L	0.004 mg/L
Lead (MCL of 0.015 mg/L)	N/A	0.003 mg/L	0.004 mg/L
Total Trihalomethanes (MCL of 80.0 μg/L)	230 μg/L	6.7 μg/L	61.584 μg/L
Nitrate (MCL of 10.0 mg/L)	9.6 mg/L	0.28 mg/L	3.82 mg/L
Ammonia (lifetime health advisory of 30.0 mg/L)	N/A	18.0 mg/L	8.753 mg/L
Total nitrogen	N/A	18.3 mg/L	17.0 mg/L
TKN	N/A	17.9 mg/L	9.783 mg/L
Orthophosphate	N/A	2.18 mg/L	1.431 mg/L
Chlordane (MCL of 2.0 μg/L)	N/A	<0.64 μg/L	N/A
Tetrachloroethylene (PCE) (MCL of 3.0 μg/L)	N/A	<0.625 μg/L	N/A
Di(2-Ethylhexyl)phthalate (MCL of 6.0 μg/L)	N/A	<1.25 μg/L	N/A

Note: All data are extracted from complete data sets presented in Appendix 1.

Ammonia is an important potential human health stressor for which there is no MCL. The EPA has published a Lifetime Health Advisory for ammonia of 30 mg/L (US EPA, 2000). This Lifetime Health Advisory is an estimate of the acceptable level of ammonia in drinking water, based on health effects information. According to the advisory, at this concentration, a lifetime of exposure to ammonia is not expected to cause adverse health effects. Ammonia is not considered a suspected or human carcinogen. Ammonia and

other nitrogenous materials (as measured by the parameters total nitrogen and TKN) may also be of further significance to human health as sources of combined nitrogen that may be converted to nitrate.

Pathogenic microorganisms, which are often present in treated wastewater, are another potential human-health stressor. Appendix Tables 1-3 and 1-4 present data on a number of wastewater indictor microorganisms present in treated and injected wastewater. Table 4-5 presents concentrations for those pathogenic microorganisms selected as representative stressors for further analysis and consideration (see Appendix Tables 1-1, 1-3, and 1-4).

Table 4-5. Representative Pathogenic Stressors

Pathogenic Microorganism	Raw	Secondary Treated	Reclaimed	Advanced Treated
Total coliform, col/100ml (MCL of 1, 5% of samples)	2.2×10^7	0.0005 – 2100[a]	N/A	N/A
Fecal coliform, cfu/100ml (MCL of 0)	8×10^6	2 – 1.7×10^7 (397,814)[b]	1.0	0.125 – 1.15[c]
Cryptosporidium, oocysts/100 L (Risk-based criteria[d], 5.8 oocysts/100 L)	N/A	N/A	No Detect to 5.35 (0.75)	No Detect – 2.33
Giardia lamblia, cysts/100 L (Risk-based criteria[d], 1.4 cysts/100 L)	N/A	20 – 13,000 (88)[e]	No Detect to 3.3 (0.49)	No Detect
Enterovirus, pfu/100 L	N/A	N/A	No Detect to 0.133 (0.01)	N/A

Note: all data are extracted from complete data sets presented in Appendix 1.
[a] Range reflects single values and sampling means from various facilities.
[b] Range and mean acquired from data set for Miami-Dade, South District.
[c] Range reflects annual means (1999, 2001) from Cape Canaveral WWTP.
[d] York and Walker-Coleman, 1999; York et al., 2002.
[e] Rose et al., 1991; values converted from reported cysts/L.

One of these representative stressors is coliform bacteria. Levels of total coliform in secondary treated wastewater are highly variable. Data collected by the South Florida Utility Council indicate that secondary treated wastewater contains a mean concentration of 394 colonies per 100 mL (Appendix Table 1-1). Table 4-5 presents a range of total coliform levels that reflects the results of single-day sampling events from various facilities in South Florida.

An extensive data set for the Miami-Dade South District WWTP shows fecal coliform levels ranging over seven orders of magnitude. Levels of fecal coliform appear to be very substantially reduced in advanced treated and reclaimed wastewater (Table 4-5).

Data to describe concentrations of some representative pathogenic stressors (for example, rotaviruses, *Cryptosporidium parvum, Giardia lamblia*) are incomplete and not widely available. Rose et al. (1991) reported that secondary-treated wastewater contains

concentrations of *Giardia* ranging from 0.2 to 130 cysts/L (average 0.88 cysts/L). Levels of *Cryptosporidium* and *Giardia* in advanced treated and reclaimed wastewater compare favorably with risk-based criteria recommended by York and Walker-Coleman (1999) and York et al. (2002).

4.7.2 Potential Exposure Pathways

When human health or ecological receptors are exposed to wastewater constituents in sufficient concentration, these receptors may be at risk for potentially adverse health effects. Complex processes and interactions govern how wastewater discharged to the subsurface will move and behave. These processes and interactions define the pathways that may expose receptors to stressors present in treated wastewater.

Risk to receptors may arise from migration of wastewater constituents (stressors) with groundwater flow. Such migration may occur if groundwater is allowed to move vertically from the injection zone. Key factors influencing exposure and risk include the distances between injection zones and receptors such as the base of the overlying USDW and water-supply wells and times of travel to receptors. Stressors may be transported with groundwater through porous media flow or by means of bulk flow through preferential flow paths (for example, fractures, leaky wells).

Porous media flow, represented in this risk analysis as scenario 1**,** may be expected where there are aquifers set within layers of sedimentary rock, such as is found in South Florida. In the case of South Florida, there is a sequence of carbonate strata, both limestone and dolomite, within which the Upper Floridan Aquifer, middle confining unit, and Lower Floridan Aquifer are located. Porous media flow is characterized by relatively slow movement of fluid and by substantial dilution, especially over long distances. Dilution occurs as a result of advection and dispersion, physical processes that occur as water flows through interconnected pore spaces. Natural groundwater gradients, buoyancy, and injection pressures act to carry the plume away from the injection zone.

Groundwater monitoring data indicate that bulk flow through preferential flow paths may be occurring (and perhaps may be the dominant form of flow) in some portions of South Florida. This risk analysis represents bulk (channel or fracture) flow as scenario 2. Bulk flow differs from porous media flow; the flow is not through pore spaces in the rock matrix, but instead through natural or man-made conduits such as solution channels, fractures, or artificial penetrations (for example, wells with faulty construction). Bulk flow is more rapid than porous media flow and may result in little or no dilution. In some areas, porous media flow may be secondary to bulk flow through conduits.

4.7.3 Potential Receptors and Assessment Endpoints

Potential drinking-water receptors include USDWs overlying the injection zones, public and private water-supply wells, and surface waters. USDWs overlying the injection zones include the unnamed surficial aquifers, the Biscayne Aquifer, or potable portions of the

Floridan Aquifer System. Some portions of the deep groundwater resource are used for municipal water supplies; all USDWs represent a valuable resource for future use.

The surficial aquifers are important for private water supplies and for municipal supplies in central South Florida and along the east and west coasts (Randazzo and Jones, 1997). The Biscayne Aquifer is tapped by private wells and also supplies large public water systems in Dade, Broward, and Palm Beach counties.

Public and private water-supply wells are typically separated both vertically and horizontally from the injection zone and from the aquifer units directly overlying the injection zone. Water obtained through private wells is often used directly (without pretreatment). Community and municipal water systems generally do pretreat groundwater before distribution.

Utilities in South Florida make limited use of surface-water bodies as sources of drinking water. Nevertheless, migration of wastewater constituents to such sources of drinking water is a possibility, and therefore surface-water bodies are a potential drinking water receptor. Perhaps more significantly, surface-water bodies and the biological communities they support are potential ecological receptors. Surface-water ecosystems are particularly sensitive to some stressors present in treated wastewater (for example, nutrients).

Federal drinking-water standards and other health-based standards serve as the analysis endpoints for assessing risks to potential drinking-water receptors. State of Florida surface-water quality standards (for Class I waters), and known ecological dose-response thresholds, serve as the analysis endpoints for assessing risks to potential ecological receptors.

4.8 Risk Analysis of the Deep-Well Injection Option

In this section, site-specific data are integrated into the conceptual model for the deep-well injection option. Actual data on stressors, receptors, and exposure pathways were used to examine potential risks. For representative stressors (and stressor concentrations), information was obtained from Florida state requirements for wastewater treatment, from actual effluent quality sampling and analyses, and from a review of the scientific literature.

To describe the proximity and vulnerability of receptors, publicly available information was obtained regarding the locations of public water-supply intakes. A review of the scientific literature provided information about the locations and physical extent of aquifer units and USDWs in South Florida.

Information necessary to characterize possible exposure pathways was obtained from scientific literature describing the study area’s geology and aquifer unit properties, from well-bore log reports and other well completion reports, and from previous studies and investigations that have examined deep-well injection in South Florida.

This analysis incorporates a two-dimensional analytical description (model) of the fate and transport of injected wastewater and wastewater constituents. The analytical description is accompanied by uncertainty analyses that examine potential variations in time of travel. This analysis of deep-well injection also makes use of groundwater monitoring performed above some zones of injection. Monitoring information is incorporated as a means of analyzing the model outputs and of more fully exploring the various mechanisms that may allow for fluid and stressor movements in the subsurface.

Dade, Pinellas, and Brevard counties serve as three areas of focus for this risk analysis. Facilities with suspected or confirmed fluid movement are sited within each of these counties. However, these counties also present unique geologic environments and differences in injection system operation that may have a substantial bearing on risk.

This analysis examines, as broadly as possible, the fate and transport of injected wastewater within the South Florida study area. Data gaps and remaining uncertainties are significant, and this risk analysis provides only a generic description of the risks that may be associated with this wastewater management option. Findings are applicable, in a general way, to these counties and the region as a whole. Findings are not applicable, in a very specific way, to particular sites or facilities.

4.8.1 Application of the Analytical Transport Model

This analysis employs an analytical model that considers two different scenarios for fluid flow and migration of wastewater in the subsurface: conventional porous media flow and bulk flow through preferential flow paths. These scenarios represent two end-members of constraint upon fluid migration in the subsurface. Subject to data and model limitations, these scenarios provide estimates of what are likely to be the fastest and slowest rates of fluid flow and migration. Although these are analyzed and presented as separate scenarios, it is possible (perhaps even likely) that both types of flow occur simultaneously in some aquifer units (for example, fractures within, leading to, or leading from porous media).

Conventional porous media flow is a scenario where fluid flows through fine, interconnected pore spaces. This scenario is modeled under the assumption that aquifer units and geologic media do not have fractures or other major conduits that would permit rapid channel flow. Primary values of hydraulic conductivity and porosity are applied in modeling flow through porous media. (Tables 4-1, 4-2, and 4-3, presented earlier in this same chapter, report specific values.) Figure 4-6 illustrates movement of injectate where flow through porous media is the primary transport mechanism. Natural groundwater gradients, buoyancy, and injection pressures act to carry the plume away from the injection zone.

Bulk flow through preferential flow paths (channel or fracture flow) is a scenario where fluid flows through naturally occurring or man-made conduits. Naturally occurring conduits include fractures, solution channels, and fissures. Man-made conduits might include injection wells with faulty construction, monitoring wells with faulty

construction, abandoned wells, or fractures created because of well drilling or injection. Figure 4-7 illustrates the flow of injectate where bulk flow is the primary mechanism of plume migration. It is important to note that preferential flow pathways may result from the presence of naturally occurring solution channels or fractures in geologic strata or from mechanical problems associated with wells.

There are data to support the existence of naturally occurring features that could promote or allow for bulk flow. The Boulder Zone, a complex fracture zone with high hydraulic conductivity, is known in some locations to feature vertical fissures or solution channels. At the SDWWTP, small fractures have been detected by gamma ray and other surveys at depths ranging from 2,465 to 2,535 feet (CH2M Hill, 1977). This zone was originally thought to be part of the middle confining unit, but was later reassigned to the Lower Floridan Aquifer. Fractures appear to exist over a 70-foot interval within the confining unit and, if interconnected, could serve as preferential flow paths for injected wastewater.

Duerr (1995) and McNeill (2000) provide evidence to support the conclusion that natural fractures, pugs, or cavities may be common in South Florida. Duerr (1995) reports the findings from a study conducted by the U.S. Geological Survey in 1990. This study observed fractures of the Floridan Aquifer in at least three counties (Broward, Indian River, and Manatee counties). In contrast to these findings, other studies have found that groundwater movement in many aquifer units is consistent with flow through porous media, with little or no channel flow. Meyer (1989), Duncan et al. (1994), and Maliva and Walker (1998) have reported similar findings for groundwater flow in the Upper Oldsmar Formation (part of the middle confining unit).

This analysis applies a continuum approach to modeling groundwater flow through fractured rock (Freeze and Cherry, 1979). This approach reassigns values of hydraulic conductivity and porosity to represent fractured geologic media. Best professional judgment has been exercised in selecting and reassigning secondary porosities and hydraulic conductivities, based on an evaluation of the primary literature (Appendix 2). Many of the values employed for this analysis are reported in McNeil (2000). These values are consistent with what has been reported by other sources from the literature. Tables 4-1, 4-2, and 4-3 (presented earlier in this same chapter) report specific values applied in modeling transport for Dade, Pinellas, and Brevard counties, respectively.

For each scenario, the transport model estimates vertical times of travel to two receptors. The first of these is the base of the nearest overlying USDW. The vertical distance separating an injection zone from the nearest USDW is an important input to the model. These distances are similar for Dade and Brevard counties (roughly 1,000 ft.), but substantially shorter for Pinellas County.

The second receptor is defined as the depth of current water supplies. The model estimates vertical times of travel to a depth (in each county) that is typical of public water-supply intakes.

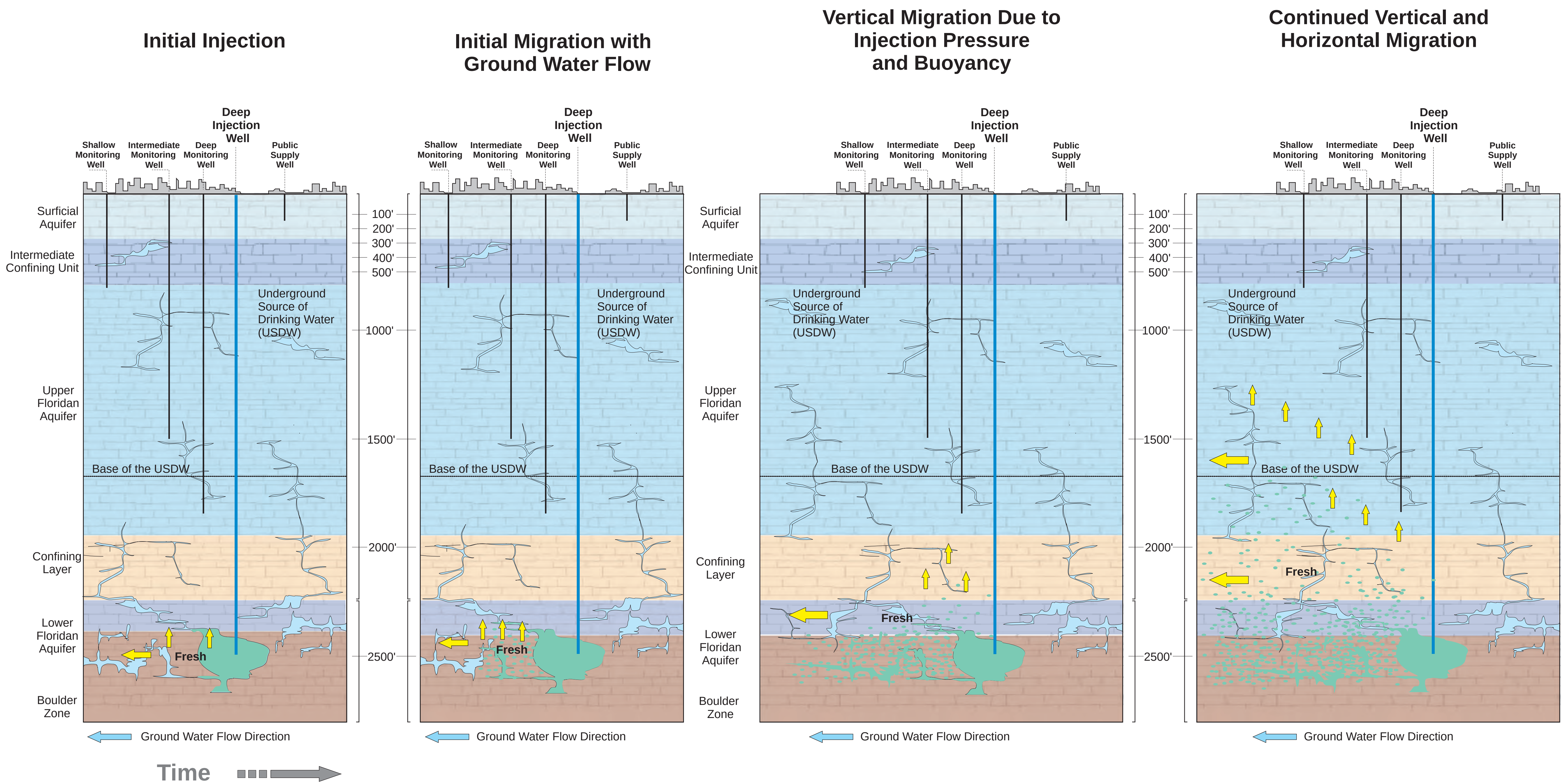

Figure 4-6. Migration Following Deep Well Injection; Fluid Flow Through Porous Media (Scenario 1)

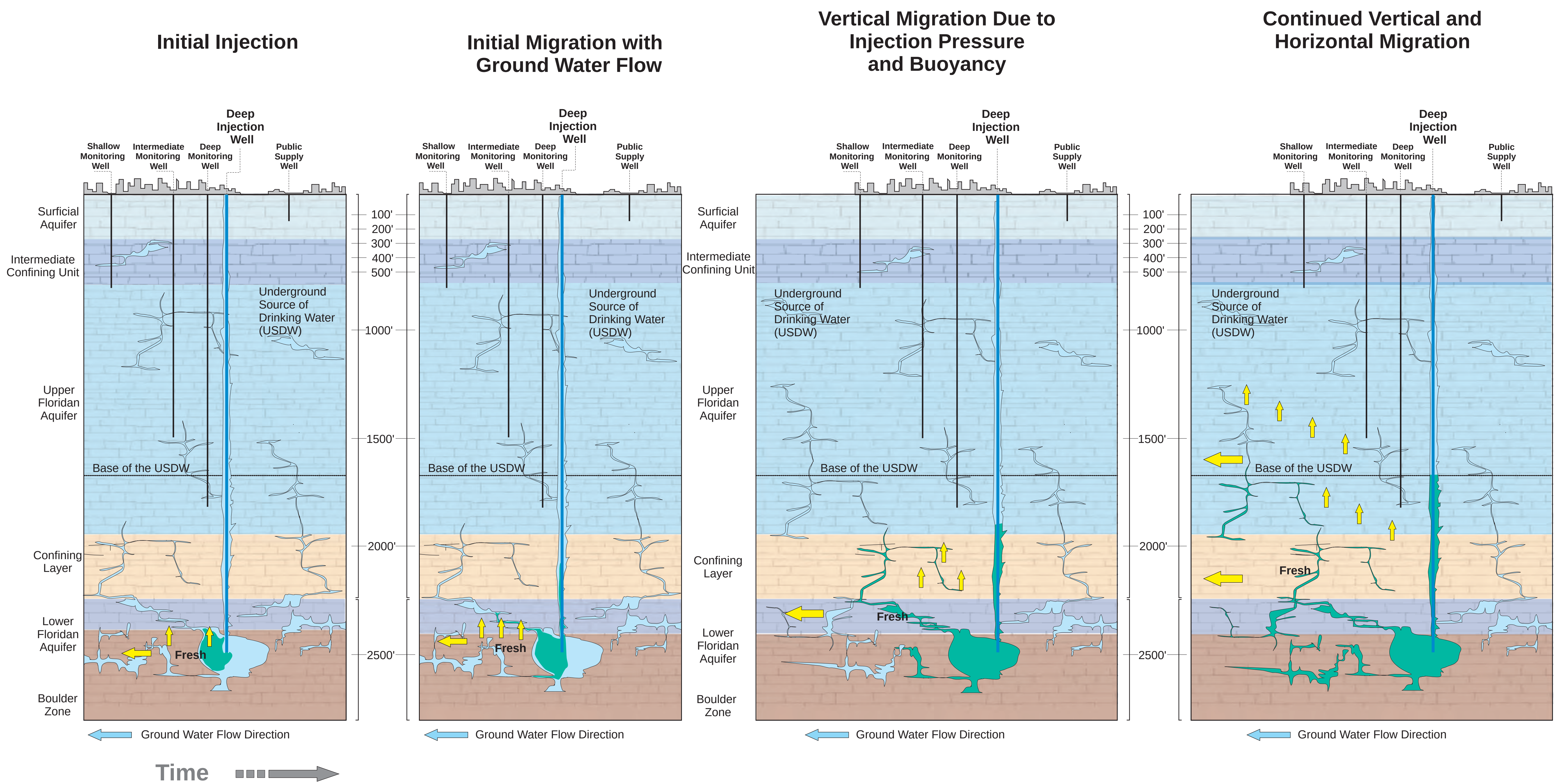

Figure 4-7. Migration Following Deep Well Injection; Bulk Flow Through Preferential Flow Paths (Scenario 2)

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This analysis estimates the extent of horizontal migration as a function of estimated vertical times of travel and hydrogeologic data (such as horizontal hydraulic conductivity and porosity, hydraulic gradients). This information provides for useful comparisons with the known real-world locations of public water-supply wells in Dade, Pinellas, and Brevard counties.

This analysis must contend with significant sources of uncertainty, especially regarding how key aquifer unit properties (for example, hydraulic conductivity, porosity) may vary throughout the study area. For each scenario, an uncertainty analysis examines how times of travel are influenced by the most important governing hydrogeologic parameters. The role and influence of primary hydraulic conductivity is analyzed for the conventional porous media scenario. The influence of secondary porosity is analyzed for the scenario that considers transport through preferential flow paths.

4.8.2 Vertical Times of Travel and Horizontal Migration

Injected wastewater moves both vertically and horizontally away from the point of injection. The rate of travel is influenced by properties of the aquifer, by the direction of prevailing groundwater flow, and by at least two separate force components (pressure head resulting from injection and pressure head resulting from buoyancy).

Groundwater flow equations may be used to estimate vertical times of travel through hydrologic units (Appendix 4). These equations take into account unit thickness, porosity, and vertical hydraulic conductivity. Tables 4-1 through 4-3 report representative values for these model parameters, specific to Dade, Pinellas, and Brevard counties. Mean (weighted) values are based on a statistical analysis of data reported in the scientific literature. A description of the statistical methods and literature-derived data are provided in Appendices 2 and 3.

Total pressure head, another input to the groundwater flow equations, is a composite of two force components. Pressure head from injection is the force component that results from the injection of treated wastewater and displacement of native groundwater. Pressure in the injection zone (and resistance to fluid emplacement) builds as a function of unit transmissivity and the injection rate (Appendix 4).

Pressure head from buoyancy results from differences in density between the injectate and native groundwater. Injected wastewater exhibits salinity and density comparable to freshwater (1.00 grams per milliliter), whereas the native groundwater has salinity and density comparable to seawater (1.025 grams per milliliter). The comparatively lighter, less dense wastewater responds to a buoyancy force component that promotes vertical movement (Appendix 4). A similar effect might result from temperature gradients. The temperature of injected wastewater is estimated to be 80º Fahrenheit, whereas native groundwater has a temperature far closer to 60º Fahrenheit. Warmer, less-dense injectate will tend to rise upward until it reaches fluids of a similar density (Appendix 4).

For Pinellas County, both force components are considered when estimating vertical times of travel to the overlying USDW and the depth of current water supplies. For Dade and Brevard counties, where substantial evidence indicates pressure from injection is negligible, only the effects of buoyancy are considered.

Horizontal migration of injected wastewater is assessed as the distance traveled laterally within each unit as function of estimated vertical time of travel. A set of groundwater flow equations (Appendix 5) estimates horizontal travel distance, taking into account porosity, horizontal conductivity, and hydraulic gradient.

4.8.2.1 Governing Assumptions for the Transport Model

The following are the governing assumptions for the transport model:

- Deep-well injection facilities are modeled as single-point sources of discharge. Volumes and rates of injection typical of whole facilities are modeled as single-point discharges within each injection zone. (Note that this is an abstraction; most facilities have more than one well.) This represents a conservative assumption about risk assessment, since it would tend to result in greater pressure heads from injection and shorter estimated times of travel.
- Pressure head from injection is estimated for the injection zone only. Pressure is attenuated as fluids pass through overlying units with differing hydraulic properties. Overlying units with lower relative hydraulic conductivity dampen and distribute pressure.
- In Dade and Brevard counties, pressure head from injection is regarded as negligible. The Boulder Zone is highly karstified with solution channels and wide fractures that do not constrain the flow of injected effluent; therefore, only negligible pressure buildup is likely to occur (Haberfeld, 1991).
- Estimated total pressure heads do not account for natural gradients that may occur at some sites.
- Changes in native groundwater temperature and salinity are assumed to be gradual.
- Calculations of pressure head because of buoyancy force assume no mixing of injected water and native fluid, dilution, or dispersion. This is a conservative approach; this assumption leads to higher buoyancy heads and shorter times of travel.

4.8.2.2 Vertical Time-of-Travel Results and Discussion

In Dade and Brevard counties, injection occurs within the Boulder Zone. Flow through the Boulder Zone is extremely rapid because of cavernous pores, fractures, and widened joints. Accordingly, pressure heads from injection are regarded as negligible in these counties (Table 4-6). In Pinellas County, injection occurs within the Upper Floridan Aquifer, a unit far less conductive than the Boulder Zone. As a means of comparison, consider the representative values for hydraulic conductivity of the UFA , (Pinellas County) and the Boulder Zone (Dade and Brevard counties); see tables 4-1, 4-2, and 4-3.

Table 4-6. Pressure Head from Buoyancy and Injection (Scenario 1)

	Components	To Receptor Well	To USDW
Dade County Injection rate = 112.5 mgd [1]	Buoyancy	73 ft	68 ft
	Injection	0 ft	0 ft
	Total Head [2]	**73 ft**	**68 ft**
Pinellas County Injection rate = 7 mgd	**Components**	**To Receptor Well**	**To USDW**
	Buoyancy	18 ft	16 ft
	Injection	533 ft	533 ft
	Total Head [2]	**551 ft**	**549 ft**
Brevard County Injection rate = 5 million mgd	**Components**	**To Receptor Well**	**To USDW**
	Buoyancy	111 ft	92 ft
	Injection	0 ft	0 ft
	Total Head [2]	**111 ft**	**92 ft**

Note: Scenario 1 assumes conventional porous media flow.
[1] Mgd = million gallons per day.
[2] Total pressure heads do not account for natural gradients that may be present at some sites.

In Pinellas County, pressure head from injection is a significant driving force, far more important than pressure head from buoyancy (Table 4-6). Pressure head from injection was evident during the course of injection-well testing performed in Pinellas County. Water levels in nearby monitoring wells increased in elevation during tests (CH2M Hill, 2001), indicating pressure head buildup from injection.

For Pinellas County, where pressure head from injection is significant, total pressure head is estimated a second time under the assumptions of scenario 2. This scenario examines behavior under an assumption that preferential flow paths (cracks, fissures, and so forth) exist. Applying representative secondary porosities and hydraulic conductivities, the estimated pressure head from injection is substantially reduced when compared to the estimate under scenario 1 (Table 4-7).

Table 4-7. Pressure Head from Buoyancy and Injection (Scenario 2)

	Components	To Receptor Well	To USDW
Pinellas County Injection rate = 7 mgd	Buoyancy	18 ft	16ft
	Injection	122 ft	122 ft
	Total Head [2]	**139 ft**	**137 ft**

Note: Scenario 2 assumes bulk flow through preferential flow paths.
[2] Total pressure head does not account for natural gradients that may be present at some sites.

Estimates of vertical time of travel under each scenario are presented in Table 4-8 for Dade, Pinellas, and Brevard counties. The full set of model inputs and outputs are included as part of Appendix 4. Table 4-8 also reports vertical distances (in feet) separating injection zones from the base of overlying USDWs and hypothetical water-supply wells. These distances and estimated times of travel reflect average conditions in

each county as a whole. Times of travel may vary across the injection facilities operating within each county.

Table 4-8. Times of Travel to USDWs and Hypothetical Receptor Wells

Location	Vertical Distance from Point of Injection (ft)	Estimated Time of Travel (scenario 1)[1]	Estimated Time of Travel (scenario 2)[2]
Dade County			
To base of USDW	1,500	421 years	14 years
To receptor well (100 ft below ground surface)	2,900	1,188 years	30 years
Pinellas County			
To base of USDW	570	2 years	170 days
To receptor well (30 ft below ground surface)	1,220	23 years	6 years
Brevard County			
To base of USDW	1,254	342 years	86 years
To receptor well (100 ft below ground surface)	2,650	1,118 years	136 years

Note: Travel time through each hydrologic unit is presented in Appendix Tables 4-1 through 4-4.
[1] Scenario 1 assumes conventional flow through porous media.
[2] Scenario 2 assumes bulk flow through preferential flow paths.

Under either scenario, Pinellas County has the shortest estimated times of travel to each receptor. Injection zones in Pinellas County are at significantly shallower depths relative to injection zones in Dade and Brevard counties; injectate has shorter distances to travel before reaching receptors. Hydrologic units in Pinellas County are also, in general, more permeable than in Dade and Brevard counties. In Dade and Brevard counties, there are confining units that serve to slow movement of fluid between injection zones and potential receptors (such as USDWs and hypothetical wells). The intermediate confining unit is completely absent in Pinellas County. Formations associated with the intermediate confining unit serve to slow transport to hypothetical receptor wells.

When bulk flow through preferential flow paths is assumed (scenario 2), estimated times of travel are significantly reduced in all three counties. In Dade and Brevard counties, times of travel are reduced by more than an order of magnitude (Table 4-8), from thousands of years to hundreds of years or less (scenario 1).

Dade County, exhibits the longest estimated times of travel: 421 years to the base of the USDW, 1,188 years to the hypothetical receptor well (under scenario 1). Since pressure head from injection is not an important factor in either Dade or Brevard County, differences in the rate of injection cannot account for the comparatively longer times of travel in Dade County. The comparatively longer estimated times of travel in Dade County are most attributable to differences in unit hydraulic properties.

Scenario 2 applies a set of very conservative assumptions regarding unit hydraulic properties and bulk flow. At no site where data have been collected is there sufficient evidence to conclude that bulk flow through preferential flow paths is characteristic of all hydrologic units. However, based on recent detection of treated effluent at certain wastewater treatment sites, bulk flow could contribute to the early detection of treated effluent. Accordingly, given the data and information that inform the present analysis, estimates obtained under scenario 2 are thought to represent the shortest possible times of travel.

Conservative assumptions are also implicit in the estimated times of travel to hypothetical receptor wells. These times of travel should be considered in light of the horizontal separation known to exist between injection wells and actual receptor wells.

4.8.2.3 Horizontal Migration

The ideal model, or set of models, would achieve multidimensional analysis. The data required to perform a multidimensional analysis of transport, particularly within heterogeneous environments, can be extensive. This requires a level of data specificity and field model calibration that is beyond the broad scales intended for this risk analysis. In the context of this regional-scale analysis, these data requirements proved prohibitive.

Table 4-9 presents estimates of horizontal travel distance for effluent in groundwater beneath the facilities in each county. These estimates take into account the estimated vertical times of travel and representative values for unit porosity, horizontal conductivity, and hydraulic gradient. Additional details and model inputs and outputs are described in Appendix 5.

Table 4-9. Estimated Horizontal Travel Distances

Scenario	Dade		Pinellas		Brevard	
	Time (years)	Distance (miles)	Time (years)	Distance (miles)	Time (years)	Distance (miles)
Scenario 1[1]	1,188	16	23	1.2	1,118	1.5
Scenario 2[2]	30	1.6	6	0.6	136	0.1

Note: Horizontal travel distance through each hydrologic unit is presented in Appendix 5.
[1] Scenario 1 assumes conventional porous media flow.
[2] Scenario 2 assumes bulk flow through preferential flow paths.

Horizontal travel distance is described analytically as a simple function of vertical time of travel. Accordingly, scenario 1 (conventional porous media flow) results in more substantial horizontal travel distances than does scenario 2 (bulk flow through preferential flow paths).

Assuming conventional porous media flow, horizontal travel distance was estimated at 16 miles for Dade County (Table 4-9). All other estimates (under either scenario) are less than 2 miles. The comparatively large horizontal travel distance estimated for Dade

County is most attributable to horizontal migration that occurs within the intermediate confining unit (Appendix 5). This retards vertical movement, but groundwater travel through this unit takes the greatest time.

Under a given set of hydraulic conditions, horizontal travel distance is a simple function of vertical time of travel. When travel distances are estimated under differing conditions, the significance of hydraulic gradient becomes apparent. Horizontal travel distances estimated for Pinellas County are comparable to those estimated for Brevard County, despite the great discrepancies in time of travel. This may be attributed to the fact that horizontal hydraulic gradient in the injection zone is estimated at 0.05 for Pinellas County and just 0.001 in Brevard County (Appendix Tables 5-1 and 5-2).

Estimates of horizontal travel through the Boulder Zone are relatively insignificant, when compared to total horizontal travel distances. The model predicts that injected wastewater moves quickly from the Boulder Zones, but primarily in a vertical direction. In Dade County and Brevard County, the estimated vertical times of travel through the Boulder Zone are 16 and 6 days, respectively. This allows for very limited horizontal transport within the Boulder Zone in the direction of prevailing groundwater flow (Appendix Tables 5-1 and 5-2). A numerical model used to simulate injection in Southwest Florida (Hutchinson and Trommer, 1992; Hutchinson et al., 1993) has described similarly short horizontal migration distances in the Boulder Zone.

4.8.2.4 Transport Model Limitations

As indicated in previous sections (especially sections 4.6 and 4.8.1), the analytical models applied in assessing vertical and horizontal transport are not ideal. It is critical, therefore, to recognize and acknowledge model limitations that may influence how risk is evaluated. These transport models are subject to two significant limitations:

- The presence and extent of preferential flow paths, or alternative wastewater migration pathways, is not adequately known. The significance of these pathways to both wastewater transport and risk can only be estimated.
- Substantial data gaps exist. There are limited data and information that may be used to develop and assign accurate values for some model input parameters. At present, this is an unavoidable source of remaining uncertainty.

Numerous studies and investigations offer evidence that indicate the presence of alternative wastewater-migration pathways, which are preferential flow paths that permit bulk flow of injected wastewater (CH2M Hill, 2001; McNeill, 2000; McKinley, 2000; MDWSAD, 1991; CH2M Hill, 1981; Miami-Dade Water and Sewer Authority, 1977; BC&E and CH2M Hill, 1977). Taken as a whole, these reports indicate that potential pathways may exist and that these pathways may short-circuit flow paths associated with conventional flow through porous media.

This analysis does not describe in a quantitative way the flow dynamics of particular types of alternative pathways (for example, fractured confining zones or wells with failed

mechanical integrity). Furthermore, it is beyond the scope of this analysis to determine what pathways may be responsible for bulk flows at particular sites or to evaluate the risks that may be associated with particular types of alternative pathways. For the purposes of this risk assessment, analysis of flow and transport through preferential flow paths (scenario 2) fairly and adequately describes these alternative pathways.

The permit process offers better opportunities to evaluate the suitability of specific well sites and injection zones. The permit process is also designed to anticipate and prevent potential problems related to well operation (and adverse impacts resulting from injection). State and federal underground injection control authorities are charged with ensuring that all necessary and appropriate measures are taken (that is, permit requirements established) to prevent endangerment of USDWs and adverse impacts to public health.

4.8.2.5 Uncertainty Analysis

Model accuracy is constrained by the completeness and accuracy of data used to assign values for model input parameters. This analysis employs values that are representative of each unit overlying injection zones in Dade, Pinellas, and Brevard counties. These values are based on a statistical analysis of data reported in the scientific literature (see Appendices 2 and 3). Inherently, however, there are site-specific variations in aquifer unit properties across each county and across the whole of the South Florida study area. As such, this transport analysis must contend with uncertainty, and the accuracy of estimated times of travel is somewhat constrained.

Uncertainty analyses may be conducted as a means of evaluating the range of expected times of travel under each scenario. These analyses focus on how times of travel are influenced by governing hydrogeologic parameters. Most important to this model are the assigned vertical hydraulic conductivity and porosity values. More specifically, the values assigned to those units that most significantly influence vertical time of travel (for example, the middle confining unit in Dade and Brevard counties and formations associated with the intermediate confining unit in Pinellas County).

Times of travel to hypothetical receptor wells, under the assumption of porous media flow (scenario 1), are estimated as employing a range of values for vertical hydraulic conductivity. Times of travel under the assumption of bulk flow through preferential flow paths (scenario 2) are estimated as employing a range of values for secondary porosity.

Table 4-10 reports results of the uncertainty analyses conducted for each scenario and county. Complete information to describe these analyses and the computed upper and lower bounds is included in Appendix 6. Appendix 6 also offers graphical representations of the uncertainty analyses for Dade, Pinellas, and Brevard counties (Appendix Figures 6-1, 6-2, and 6-3, respectively).

Table 4-10. Range of Travel Times to Hypothetical Receptor Wells

Effect of Hydraulic Conductivity (K_v) on Vertical Travel Times, in Years (scenario 1)[1]			
Location	**Lower Bound (High K_v)**	**Computed Mean (Representative K_v)**	**Upper Bound (Low K_v)**
Dade County	905	1,188	2,460
Pinellas County	20	23	38
Brevard County	1,023	1,294	2,515
Effect of Secondary Porosity on Vertical Travel Times (scenario 2)[2]			
Location	**Lower Bound (years)**	**Computed Mean (years)**	**Upper Bound (years)**
Dade County	28	30	32
Pinellas County	5.7	6.4	7.2
Brevard County	135	136	138

[1] Scenario 1 assumes conventional porous media flow.
[2] Scenario 2 assumes bulk flow through preferential flow paths.

Increases in vertical hydraulic conductivity, above the computed mean value (the representative value), do not result in very substantially decreased vertical times of travel. Decreases in vertical hydraulic conductivity, below the computed mean value (representative value), do result in substantially increased vertical times of travel. When values for vertical hydraulic conductivity in the confining unit falls below the representative value, the model parameter begins to exert a very strong and growing influence upon time of travel.

The effects of secondary porosity on vertical travel times are related linearly. As porosity decreases (less pore space), the vertical travel time decreases (faster travel time). Alternatively, as porosity increases, the vertical travel time increases.

The uncertainty analysis also shows how the model is more sensitive to varying vertical hydraulic conductivities relative to varying porosities. The range of travel times is greater when varying the hydraulic conductivity. Vertical travel times can vary by several hundred years using this range of hydraulic conductivity values.

4.8.3 Evaluation of Receptors and Analysis Endpoints

This section presents fate and transport analyses that examine the behavior of representative stressors in the subsurface. These analyses rely and build upon the vertical time of travel analysis presented in previous sections. These fate and transport analyses assess whether receptors are likely to be exposed to stressors; the analyses provide estimates of stressor concentrations that may be expected to reach potential receptors. This, in effect, is an exposure analysis focusing on those representative stressors believed to pose the greatest possible risk to human or ecological health. Risk characterization is accomplished by comparing anticipated final stressor concentrations at receptors (in Dade, Pinellas, and Brevard counties) with specific analytical endpoints.

For each county, these analyses estimate final concentrations of representative stressors anticipated to reach the base of the nearest overlying USDW and hypothetical water-supply well. Analyses are conducted under each of the scenarios developed in previous sections (conventional porous media flow and bulk flow through preferential flow paths) and apply mean times of travel estimated for each county.

These analyses attempt to account for the various processes that may attenuate and dilute stressors during the course of transport. Natural attenuation involves physical, chemical, and biological processes that result in reducing the mass, toxicity, mobility, volume, or concentration of contaminants in soil or groundwater (US EPA, 1999, cited in Suthersan, 2002). Processes that may contribute to stressor attenuation include biodegradation, hydrolysis, sorption, volatilization, radioactive decay, chemical or biological stabilization, and transformation.

Sorption processes cause stressors to adhere to geologic materials; this has the effect of slowing down migration and may increase the vertical time of travel for some representative stressors. Degradation is a biological process whereby organic materials are broken down under aerobic or anaerobic conditions. Hydrolysis occurs when organic or inorganic solutes react with water and transform to less mobile forms.

Modeling attenuation and dilution on these scales (particularly under heterogeneous conditions and with very limited data sets) is exceedingly difficult. These analyses apply a number of conservative assumptions that would tend to overstate, rather than understate, exposure and risk. Most importantly, these analyses only very crudely account for dilution as a result of advective transport and dispersion. Fluids that reach potential receptors because of injection activities (that is, wastewater and displaced native groundwater) may be more substantially diluted than predicted by these analyses.

Finally, because of model limitations and the general lack of needed data and information, quantitative fate and transport analyses are not provided for any of the pathogenic stressors. Rather, a weight-of-evidence approach applies information from the scientific literature to assess the likely behavior of these microorganisms and to characterize the risk posed to potential receptors.

4.8.3.1 Application of the Stressor Fate and Transport Model

The following stressors were selected for fate and transport analysis: ammonia, arsenic, chlordane, chloroform (measured as total trihalomethanes), di(2-ethylhexyl) phthalate (DEHP), nitrate, and tetrachloroethylene (PCE). Initial concentrations (concentrations at the point of injection) were assigned based on values reported in Appendix Table 1-1; these are summarized in Table 4-11.

Table 4-11. Concentrations of Representative Stressors at USDWs and Hypothetical Wells

Dade County	C_i at Injection	C_f at USDW (Scenario 1)[a]	C_f at Well (Scenario 1)[a]	C_f at USDW (Scenario 2)[b]	C_f at Well (Scenario 2)[b]	MCL
Ammonia (mg/L)	8.75 [c]	8.75	8.75	8.75	8.75	**NA**
Arsenic (mg/L)	0.01	0.01	0.01	0.01	0.01	**0.05**
Chlordane (µg/L)	0.01 [d]	0.000	0.000	0.000	0.000	**2.00**
DEHP (µg/L)	5.00 [d]	0.000	0.000	0.000	0.000	**6.00**
Nitrate (mg/L)	3.82 [c]	3.82	3.82	3.82	3.82	**10.00**
PCE (µg/L)	4.66	0.000	0.000	0.02	0.010	**5.00**
Trihalomethanes, total (µg/L)	61.58	0.000	0.000	7.24	5.32	**80.00**
Pinellas County						
Ammonia (mg/L)	18.00	18.00	18.00	18.00	18.00	**NA**
Arsenic (mg/L)	0.003 [d]	0.003	0.003	0.003	0.003	**0.05**
Chlordane (µg/L)	0.64 [d]	0.50	0.21	0.61	0.50	**2.00**
DEHP (µg/L)	1.25 [d]	0.22	0.00	0.86	0.22	**6.00**
Nitrate (mg/L)	0.28	0.28	0.28	0.28	0.28	**10.00**
PCE (µg/L)	0.63	0.27	0.02	0.52	0.27	**5.00**
Trihalomethanes, total (µg/L)	6.70	4.90	1.64	6.27	4.90	**80.00**
Brevard County						
Ammonia (mg/L)	8.75 [c]	8.75	8.75	8.75	8.75	**NA**
Arsenic (mg/L)	0.005 [d]	0.005	0.005	0.005	0.005	**0.05**
Chlordane (µg/L)	0.01 [d]	0.000	0.000	0.000	0.000	**2.00**
DEHP (µg/L)	5.00 [d]	0.000	0.000	0.000	0.000	**6.00**
Nitrate (mg/L)	9.60	9.60	9.60	9.60	9.60	**10.00**
PCE (µg/L)	1.00 [d]	0.000	0.000	0.000	0.000	**5.00**
Trihalomethanes, total (µg/L)	230	0.000	0.000	0.000	0.000	**80.00**

[a] Scenario 1 assumes conventional porous media flow.
[b] Scenario 2 assumes bulk flow through preferential flow paths.
[c] Limited site-specific data. Concentrations in secondary treated wastewater from various facilities in southeast Florida; reported by Englehardt et al., 2001.
[d] Detection limit.

Appendix 7 describes the fate and transport model used to estimate final stressor concentrations (concentrations at receptors). Times of travel specific to each representative stressor (excluding pathogenic microorganisms) are obtained by modifying the previously determined times of travel (section 4.8.2.2.) with retardation coefficients. (The fate and transport of pathogenic microorganisms are examined under a separate section, section 4.8.3.3.)

Retardation coefficients developed from referenced chemical sorption coefficients (Appendix 7) account for sorption processes that act to slow the movement of solutes as fluids move through hydrologic units. Ultimately, sorption processes produce differences between the velocity of groundwater flow and the velocities of dissolved or entrained stressors.

Biodegradation and hydrolysis are two processes that act to reduce the mass (or concentration) of organic stressors over the course of transport. Rates of biological degradation and hydrolysis may be expressed as a half-life for each organic compound. *Half-life* is the time required for a concentration of reactant to decrease to half of its initial concentration.

Time of travel directly affects how much attenuation will occur as a result of these processes prior to stressors reaching receptors. A first-order decay model is used to obtain final stressor concentrations that account for biodegradation and hydrolysis (Appendix 7). This model employs stressor-specific times of travel and published half-life values for organic stressors.

This model assumes conservative behavior for inorganic stressors. Final concentrations of inorganic stressors (for example, ammonia, arsenic, nitrate) are influenced by sorption processes but not by degradation, hydrolysis, or transformation. While these assumptions may be questioned, particularly in the case of ammonia, there is insufficient information with which to model the types of transformations that may occur (for example, oxidation of ammonia to other nitrogenous forms). Nevertheless, these assumptions do result in model outcomes that are conservative for exposure analysis and risk assessment.

4.8.3.2 Final Concentrations of Chemical Stressors

Four tables included in Appendix 7 (Appendix Tables 7-1 through 7-4) report, in their entirety, the model inputs and outputs. Table 4-11 provides a summary of the estimated final stressor concentrations that the model predicts may reach USDWs and hypothetical water supply wells under each scenario.

Under the assumptions of scenario 1 (conventional porous media flow) and scenario 2 (bulk flow through preferential flow paths), estimated final stressor concentrations for both receptors and in all three counties (Dade, Pinellas, and Brevard), are below primary drinking-water standards. This is despite the faster estimated times of travel that prevail where bulk flow through cracks, dissolution channels, and other conduits is assumed. Ammonia, for which there is no maximum contaminant level (only a Lifetime Health Advisory level), does not appear to exceed health-based criteria at either receptor, under any of the model conditions.

Time of travel plays a crucial role in determining the stressor concentrations to which potential receptors may be exposed. The clearest illustration of this role may be seen in the organic stressor concentrations estimated for receptors in Pinellas County. Section 4.8.2.2 demonstrates how bulk flow through preferential flow paths (scenario 2) may result in substantially shorter times of travel. Under the assumptions of scenario 2, organic stressors reach the base of the overlying USDW in Pinellas County only minimally reduced from the initial concentrations at injection (Table 4-11). In Dade and Brevard counties, where the times of travel are more than an order of magnitude greater than in Pinellas County, organic stressors are substantially reduced before reaching

USDWs. Under the assumptions of scenario 1, organic stressors in Pinellas County are more substantially reduced from attenuation that occurs prior to fluids reaching the base of the USDW.

Where this model is capable of describing attenuation processes (for example, for the organic stressors), results show very clearly the significance of time of travel. Furthermore, these results illustrate how the presence (or absence) of preferential flow paths can substantially influence the types of exposures that may be expected to occur. As was expected for the organic stressors, estimated final concentrations obtained under scenario 2 (bulk flow through preferential flow paths) are greater than the estimates obtained under scenario 1 (conventional porous media flow) for both receptors and in all three counties (Table 4-11).

There are important differences in the way that the various organic stressors behave in the subsurface. Variations in sorption characteristics and half-life translate into relatively more or less conservative behavior for individual organic stressors. Chlordane and DEHP have comparatively higher sorption and distribution coefficients that result in higher retardation coefficients and longer stressor-specific times of travel (Appendix Tables 7-1 through 7-4). Chlordane, and to a lesser extent trihalomethanes, have comparatively long half-lives and smaller decay coefficients; this has the effect of lessening (in a comparative sense) the amount of attenuation that occurs over time.

Among the organic wastewater constituents modeled as representative stressors, DEHP represents a relatively slow-moving compound and one that can be expected to significantly and quickly attenuate. Trihalomethanes represent a relatively fast-moving compound and one that can be expected to attenuate more slowly or incompletely.

Trihalomethanes, though present at varying concentrations in injected wastewater, do not under any of the model conditions pose a significant threat of violating drinking-water standards. For Pinellas County, where times of travel are comparatively short, this threat is mitigated by the fact that trihalomethanes appear to be present at only very low concentrations in the injected wastewater. For Brevard County, where some data indicate high trihalomethane concentrations at injection, this threat is mitigated by comparatively long travel times. Trihalomethanes injected at concentrations greater than twice the MCL are expected to reach receptors in Brevard County at below detection limits under either scenario.

This model assumes conservative behavior for the inorganic representative stressors (ammonia, arsenic, and nitrate). It is assumed that final concentrations of ammonia, arsenic, and nitrate will not be influenced by degradation, hydrolysis, or transformation processes. Accordingly, Table 4-11 reports final concentrations at each of the receptors (and under each scenario) that are identical to the concentrations at injection. These assumptions are conservative, as regards exposure analysis and risk assessment; they will tend to overestimate exposure and risk.

Under some geochemical conditions, metals such as arsenic may become immobilized in the aquifer matrix. Model estimates of the time of travel for arsenic, which does exhibit fairly strong sorption characteristics, are long by comparison to several of the other representative stressors. Only chlordane and DEHP have estimated stressor-specific times of travel that consistently exceed those estimated for arsenic (Appendix Tables 7-1 through 7-4). However, even under the conservative set of assumptions applied in examining the fate of arsenic, there appears to be no threat of drinking-water violations under any of the model conditions. Arsenic is often present in injected wastewater at very low concentrations and frequently at concentrations that cannot be detected.

Ammonia and nitrate both move far more readily with groundwater flow. It is unlikely that for either of these stressors that time of travel is significantly increased because of sorption processes (Appendix Tables 7-1 through 7-4). While there are processes that might cause attenuation of ammonia or nitrate in the subsurface, these processes are microbially mediated and very difficult to model with the present data limitations.

Under oxic conditions, dissolved ammonia (or ammonium) may be oxidized to nitrite and nitrate, as a result of a process called nitrification (Fenchel and Blackburn, 1979; Blackburn, 1983). Rates of growth for nitrifying bacteria are typically increased at temperatures between 30º and 35º Celsius; poor growth occurs at temperatures below 5 ºCelsius (Buswell et al., 1954; Deppe and Engel, 1960, summarized in Fenchel, 1983). Nitrifying bacteria can survive under anoxic conditions but experience high rates of mortality wherever hydrogen sulfide is produced by anaerobic sulfate-reducing bacteria (reviewed in Blackburn, 1983).

These findings from the literature imply that the conservative behavior assumed for ammonia may be more defensible with respect to estimated concentrations at the base of the USDW, than for estimated concentrations at hypothetical water-supply wells. Portions of aquifers lying below and including the base of the USDW are most certainly anoxic, allowing for comparatively less nitrification (conversion of ammonia to other nitrogenous forms). However, water-supply wells penetrate to shallow depths in most parts of South Florida. At these depths, oxic conditions may prevail and may lead to increased rates of nitrification and attenuation of ammonia.

Nitrate may be subject to microbial denitrification (conversion to nitrous oxide and ammonia) and to other forms biological uptake or conversion. The U.S. Geological Survey has reported significant rates of denitrification in shallow groundwater beneath Florida citrus groves (USGS, 2000). Denitrification in shallow groundwater has also been reported by a study of septic systems in areas bordering the Indian River Lagoon (Horsley & Witten, 2000). These findings suggest that completely conservative behavior of nitrate, at least in shallower aquifers, is unlikely.

4.8.3.3 Fate and Transport of Pathogenic Microorganisms

Assessing the potential human health risks from microbial pathogens in injected treated wastewater depends to a large extent on evaluating the fate and transport of pathogenic

microorganisms. A crucial step in risk assessment is determining whether pathogens can be transported in an infective form to drinking water receptors and to human receptors. Thus, there are four risk questions to address:

- Can pathogenic microorganisms be transported in groundwater through geologic media?
- Can pathogenic microorganisms survive and remain infective after a long period of time traveling in groundwater?
- What are regulatory standards or recommendations?
- What are infective doses and how do actual or predicted concentrations of microorganisms in effluent at the drinking-water receptor compare with infective doses and standards?

Assessment endpoints used in this microbial risk assessment include a 1 in 10,000 (1 x 10^{-4}) risk threshold used by the DEP and regulatory standards, where such standards exist (FDEP, 1998). If regulatory standards do not exist, then other human health advisory or illness doses or other state or federal recommendations are used.

Valuable information for this analysis of microbial risks was provided by the DEP, which published a risk assessment of reuse and reclaimed water based on a number of other Florida studies and its own risk assessment (FDEP, 1998). Although the objective of that study was evaluation of the risks of reclaimed water, the approaches and assumptions used are applicable for this study of deep-well injection. These are listed in Table 4-12.

Table 4-12. Assumptions Used for Florida DEP's Human Health Risk Assessment for Reuse

Parameter	Assumption
Daily human ingestion rate	2 L/day
Recreational contact dose	100 mL
Contact from residential irrigation (worst-case single ingestion)	100 mL
Residential irrigation, routine exposure	1 mL
Consumption of edible crops irrigated with water	10 mL
Irrigation of public-access areas such as golf courses, parks	1 mL
Exposure to aerosols	0.1 mL

Source: FDEP, 1998.

Microbial Standards or Guidelines

Fecal coliforms are often utilized by regulatory agencies as indicators of fecal wastes, effectiveness of disinfection, and water quality. Florida regulations for water quality and wastewater treatment and disinfection utilize fecal coliforms. Disinfection and water quality standards involving fecal coliforms are summarized in Table 4-13 (from FDEP, 1998).

Table 4-13. Coliform Standards

Fecal Coliform Limit (No./100 mL)	Application	Florida Administrative Code
200[a]	Basic disinfection (minimum required for surface-water discharge of treated wastewater and for reuse projects)	62-302.530, 62-600.440(4)
200[b]	Standard for Class I waters (drinking-water supplies)	62-302.530
200[b]	Standard for Class III waters (recreational waters)	62-302.530
200	Bathing beach standard	Department of Health regulates
14[a]	Intermediate disinfection (required for discharge to tributaries of Class II shellfish waters)	62-600.440(6)
14[b]	Standard for Class II shellfish waters	62-302.530
4[c]	Groundwater standard	62-520.420(1)
< Detection[d]	High-level disinfection required for reuse systems permitted under part III, Chapter 62-610, FAC	62-600.440(5)
< Detection[e]	Drinking-water standard	62-550.310(3)

Source: FDEP, 1998.

[a] Annual and monthly limits; higher limits apply for weekly and single sample limits.
[b] Monthly average limit; higher limits apply to a single sample. Total coliform limits also apply.
[c] In terms of total coliforms.
[d] At least 75% of all observations must be less than detection; no sample may exceed 25/100 mL.
[e] In terms of total coliforms; some excursions above detection are allowed.

Microbial Concentrations Needed to Cause Risk

The DEP risk assessment of reuse of reclaimed water relied upon results from several studies of potential microbial risks, in addition to its own risk analyses (Rose and Carnahan, 1992; Rose et al., 1996; FDEP, 1998). These studies concluded that in order to pose a 1 in 10,000 risk (also known as a 1 x 10^{-4} risk), pathogen concentrations in reclaimed water would have to be as shown in Table 4-14. This table presents concentrations of pathogens that would correspond to a risk of 1 in 10,000, for several doses (100 mL for recreation, 100 mL for residential irrigation, 1 mL for irrigation of public access areas, 0.1 mL for exposure to aerosols, converted to 1 liter and 100 liters for comparison).

Table 4-14. Pathogen Concentrations in Water Corresponding to 1 x 10-4 Risk

Microorganism	Units	Conc. Needed for 1 x10-4 Risk					
		0.1 mL	1 mL	10 mL	100 mL	1 liter	100 liters
Cryptosporidium	Oocysts	22,000	2,200	220	22	2.2	0.022
Giardia	Cysts	5,000	500	50	5	0.5	0.005
Rotavirus	PFU	165	16.5	1.65	0.165	0.0165	0.000165
Echovirus	PFU	50,000	5,000	500	50	5	0.05

Source: FDEP, 1998.
PFU = plaque-forming units

In this risk assessment of deep-well injection, the microbial concentrations that would cause a 1 in 10,000 risk can be used to evaluate possible concentrations of microbial pathogens at drinking-water receptors.

Microbial Transport in Groundwater

Transport of bacteria and viruses in groundwater has been documented by a number of studies in various countries (Rehmann et al., 1999; Yates et al., 1985) and in the Florida Keys (Paul et al., 1995). In such studies, microbial transport is generally assumed to be passive, whereby the microorganism is passively carried in a stream of water, rather than active, where the microorganism would actively move against an environmental gradient. The actual distances covered by viruses (including phages) and bacteria in groundwater moving through various geologic media are summarized in Table 4-15 (from Rehmann et al., 1999 and authors therein). Travel distances for viruses, the smallest microorganisms, range from 46 meters in gravel, sand, and silt to 1,600 meters in carbonate rocks in Missouri. Travel distances for bacteria range from approximately 122 meters for *Serratia marcescens*, *Enterobacter cloacae* in fractured chalk deposits to 900 meters for *Bacillus sterothermophilus* in gravel.

Table 4-15. Microbial Transport in Aquifers

Microorganism	Maximum travel distance (m)	Conditions	Hydraulic conductivity (m/day)	Mean pore velocity (m/day)	Reference
Phage T4	1,600	Carbonate rock, Missouri			Fletcher and Myers (1974)
Phages T4, 174	920	Gravel, New Zealand			Noonan and McNabb (1979)
Bacillus sterothermophilus	900	Gravel, New Zealand	10^4	164+ (colloid velocity is 200 m/day)	Martin and Noonan (1977)
E. coli	350–830	Sand with gravel, pebbles, 4–8 m thickness, Kazakhstan	10^5	160	Anan’ev and Demin (1971)
Type 2 *Aerobacter aerogenose* 243	680	Sandstone, Great Britain		36–180	Martin and Thomas (1974)
Coxsackie B3	408	Coarse sand with fine gravel, Babylon, New York			Vaughn and Landry (1977)
Unidentified phage	400	Fine sand with some gravel, coarse sand, Lake George, New York	4.6-19.5	3–12	Aulenbach (1979)
Serratia marcescens, Enterobacter cloacae	122–366	Fractured chalk, Great Britain			Skilton and Wheeler (1988)
Poliovirus 1, 2, 3	60–270	Sandstone, silt, clay, Dan region, Israel			Idelovitch et al. (1979)
Poliovirus, Coxsackie B3 and echovirus	250	Cohansey sand with coarse gravel, Vineland, New Jersey			Koerner and Haws (1979)
Coliphage f2, indigenous enteroviruses, fecal streptococcus	183	Silty sand and gravel, Fort Devens, Massachusetts	8.6		Schaub and Sorver (1977)
Echovirus 6, 21, 24, and 25 and unidentified viruses	45.7	Coarse sand with fine gravel, 1–2% silt, Holbrook, New York			Vaughn and Landry (1977)

Source: Rehmann et al., 1999, Table 1.

When these travel distances for microorganisms are compared with typical depths of injection wells in South Florida, which range from approximately 1,000 feet to more than 2,500 feet below the surface, it is apparent that microorganisms could be transported over such depths if a vertical transport mechanism exists. Probable mechanisms for vertical transport of effluent from injection pressure and buoyancy were described earlier. Thus, there is a mechanism for transporting microorganisms in South Florida, and there is information from other studies that microorganisms can be transported over distances in moving groundwater that are comparable to the deep-injection well vertical travel distances to drinking-water receptors.

Microbial Survival in Groundwater

A critical question is whether or not pathogenic microorganisms can survive long enough in groundwater to remain viable or infective over the estimated travel times calculated for effluent to reach the USDW and public water-supply wells. Under scenario 1 for porous media flow, characterized by slower effluent migration through small pore spaces, calculated travel times to the USDW range from 2 years in Pinellas County, to 342 years in Brevard County, to 421 years in Dade County. Estimated travel times to hypothetical public water-supply wells are even longer under scenario 1: 23 years in Pinellas County, 1,118 years in Brevard County, and 1,188 years in Dade County. Under scenario 2 for preferential flow, characterized by more rapid effluent migration through larger fissures, cracks, cavernous weathered voids, and channels, the travel times to the USDW range from 170 days in Pinellas County to 14 years in Dade County and 86 years in Brevard County. Estimated travel times to hypothetical public water-supply wells under scenario 2 are 6.4 years in Pinellas County, 30 years in Dade, and 136 years in Brevard.

Viability in particular is an important issue in risk assessment, because a number of pathogenic microorganisms may still remain viable (capable of causing disease) even if they can no longer reproduce or grow under laboratory culture conditions (Xu et al., 1982; Elliott and Colwell, 1985). Thus, a laboratory study that uses culturability of organisms alone as a measure of microbial risk, without a study of the viability or infective capacity of the microbial cells, would not necessarily paint a full picture of microbial risk. Studies of infective populations of microorganisms remaining after a period of time or some treatment would more accurately depict risk. Examples of such studies are given in Table 4-16, summarizing some values for time needed to inactivate infective microorganisms in water.

Table 4-16. Survival of Microorganisms in Water

Microorganism	Time elapsed	Inactivation	Reference
Cryptosporidium parvum	176 days	99% of infective populations in river water are inactivated	Robertson et al., 1992
	35 days	33% of infective populations are inactivated in sea water	Robertson et al., 1992
	24 hours	86% decrease in infective population after 24 hours of exposure to 0.149 M solution of ammonium	Bowman and Jenkins, 1996
E. coli S-2	13 days	85% of cells are not culturable in sterile estuarine water (salinity 11 ppt)*	Xu et al., 1982
E. coli	60 days +	Cells are not culturable*	Elliott and Colwell, 1985
Vibrio cholerae	9 days	No culturable cells remain in sterile estuarine water (salinity 11 ppt) at 4 to 6 ºC*	Xu et al., 1982
Enteric viruses (coxsackie viruses, Hepatitis A viruses and Norwalk-like virus)	> 2 months	Viability remained during this period; inactivation was not observed	Rose et al., 2000

* Results indicate that nonculturable bacterial cells may still be viable.

These results indicate that under some conditions approximating subsurface temperatures and other conditions, fecal coliforms (*E. coli*) can survive for at least 60 days (with some remaining viability), that a small percentage (1%) of *Cryptospororidium* can survive for 176 days, and that some viruses can remain viable for 2 months or more.

Interestingly, exposure to a 0.149 M solution of ammonium significantly increased the inactivation rate of *Cryptosporidium* after only 24 hours. This concentration of ammonium is at least two orders of magnitude greater than the concentrations of ammonium found in secondary-treated effluent. The effect of wastewater constituents on survival of pathogenic microorganisms poses an interesting, but probably largely unanswered, question for microbial risk assessment.

Another way to examine microbial survival in the environment is to look at microbial inactivation rates. Because microbiologists typically are studying large numbers of microorganisms rather than single cells, the rate of inactivation of a microorganism is often expressed on a logarithmic basis as the $\log_{10}$ decline in the viable or culturable organisms per day:

$$\text{Inactivation rate } r = -\log (N/ N_0) / \text{days}$$

Where r = inactivation rate in $\log_{10}$ /day
N = number of viable or culturable microorganisms at a given time
N_0 = initial number of microorganisms

The higher the inactivation rate, the fewer the numbers of microorganisms remaining after a period of time. Conversely, the lower the inactivation rate, the more microorganisms remain after a period of time. An alternate way of expressing the inactivation rate is in terms of the T_{90}, or the time needed to inactivate 1 log, or 90%, of the microbial population. A 2-log decrease in the microbial population would correspond to inactivation of 99% of the population.

Inactivation rates and T_{90}s for different microorganisms are given in Table 4-17. From these rates, it is apparent that *Cryptosporidium* survives relatively longer in the environment, with T_{90}s numbered in hundreds of days, than many pathogenic bacteria or viruses, whose T_{90}s are numbered in days or tens of days.

Table 4-17. Inactivation Rates for Microorganisms in Aquatic Media

Microorganism	**Inactivation Rate (log_{10}/day)**	**Corresponding T_{90} (days)**	**Conditions and days**	**Reference**
Cryptosporidium parvum	0.005	200		Robertson et al., 1992
Cryptosporidium parvum	0.01 to 0.024	100 to 41.7	From lamb wastes, incubated in raw water (35 days)	Medema et al., 1997
Fecal coliforms	0.03, 0.0384	33.3, 26.04	Florida groundwater sample at 22 ºC	Bitton et al., 1983
Fecal streptococci	0.0204	49.02	Florida groundwater sample at 22 ºC	Bitton et al., 1983
Fecal enterococci	0.025 to 0.233	40.0 to 4.29	From a sewage source, incubated in raw water (0 to 42 days)	Medema et al., 1997
Poliovirus	0.0456	21.93	Florida groundwater sample at 22 ºC, in laboratory	Bitton et al., 1983
E. coli	0.049 to 0.102	20.4 to 9.80	From a sewage source, incubated in raw water (0 to 42 days)	Medema et al., 1997
E. coli	0.1584	6.31	Florida groundwater sample at 22 ºC, in laboratory	Bitton et al., 1983
Poliovirus	0.035 to 0.667	28.6 to 1.50	Groundwater (unfiltered) incubated at native temperatures of 4 to 23 ºC (AZ, CA, NC, NY, TX, WI)	Yates et al., 1990
Echovirus	0.051 to 0.628	19.6 to 1.59	Groundwater (unfiltered) incubated at temperatures of 4 to 23 ºC (AZ, CA, NC, NY, TX, WI)	Yates et al., 1990

Reviewing the mean effluent travel times (Table 4-8) with microbial T_{90}s (as shown in Table 4-17) shows that, if *Cryptosporidium* were present in treated wastewater, Pinellas County has the potential to receive *Cryptosporidium* at its drinking-water receptors, because travel times for effluent are on the order of hundreds of days to several years. However, because Pinellas County treats injected wastewater to a higher standard than secondary and also employs filtration, it is not likely that concentrations of

Cryptosporidium in the treated effluent would be high enough to cause human health concerns.

Under the highest-risk scenario, scenario 2 (preferential flow along fractures), effluent travel times to drinking-water receptors in Dade County are about a decade or so (10 to 16 years) (Table 4-8). Ten years amounts to 3,650 days, or one order of magnitude longer than the T_{90} for *Cryptosporidium*, which is the time needed to inactivate 90% of the original *Cryptosporidium* population present.

These numbers suggest that the chances for *Cryptosporidium* to survive long enough to reach drinking-water receptors in Dade County are low. No data are available concerning *Cryptosporidium* or *Giardia* concentrations in secondary-treated wastewater from South Florida, and therefore assessment of the risk from pathogenic protozoans cannot be completed. However, the published literature values for inactivation rates and T_{90}s suggests that there may be a small chance that *Cryptosporidium* contamination could occur *if* initial concentrations in secondary-treated effluent were high to begin with.

Fecal coliforms and viruses pose concerns in deep-well injection. This is not because their survival times are long, but because their concentrations in unchlorinated effluent potentially may be high enough that, even if they become attenuated during transport, there may still be a significant number that survive the long transport distances. Also, virtually nothing is known concerning *in situ* growth of microorganisms in groundwater.

Monitoring of fecal coliforms and virus concentrations in discharged effluent indicates that, for the most part, secondary-treated effluent meets the fecal coliform standard of no more than 200 colonies per 100 mL for secondary treatment. However, discharged secondary-treated effluent does not always meet the drinking-water standard, which is nondetect (Appendix 9). Thus, bacteria and viruses may pose risks to water quality in the USDW and in public water-supply wells if secondary effluent is not disinfected to nondetect levels.

No data are available concerning concentrations of pathogenic protozoans in secondary-treated effluent from South Florida. However, because these microorganisms are not inactivated by chlorine but require filtration to be removed, neither of which is required for deep-well injection, they may be present in injected effluent in Dade and Brevard counties.

These data on microbial survival times, inactivation rates, and various times of travel for effluent migration suggest that, in some cases, particularly if scenario 2-type preferential flow is occurring, that longer-lived pathogenic microorganisms may pose a finite risk. Microorganisms capable of forming resistant or durable cysts or oocysts or spores that can survive longer periods of time are of particular concern. These include *Cryptosporidium*, *Microsporidium*, *Giardia*, *Clostridium*, and a number of other pathogenic microorganisms.

Another factor to consider in evaluating microbial risk is straining of microorganisms. Scenario 1 involves porous media flow through fine pore spaces, which is likely to strain or filter small particles or colloids such as microorganisms. If scenario 1 flow is the predominant or sole type of flow at an injection well site, then it is unlikely that pathogenic microorganisms could easily be transported through the subsurface.

Despite its short-modeled travel times for effluent migration, Pinellas County provides an example of low human-health risk from pathogenic microorganisms from deep-well injection. This is because Pinellas County treats wastewater to reclaimed-water standards before injecting it into deep-injection wells. Reclaimed-water standards require secondary treatment with basic disinfection, filtration, and high-level disinfection with chlorine. Such treatment would generally result in potable water. Filtration, if properly done, is effective at removing pathogenic protozoan cysts and oocysts (York et al., 2002). In Pinellas County, monitoring data indicate that, while *Cryptosporidium* concentrations may be higher than concentrations that pose a 1 in 10,000 risk (DEP, 1998), these concentrations generally are lower than the DEP's recommended limits of 5.8 oocysts per 100 liters and 1.4 cysts per 100 liters for *Cryptosporidium* and *Giardia*, respectively (York et al., 2002). Thus, Pinellas County has the lowest risks associated with microbial pathogens, because of its higher level of treatment, disinfection and filtration.

If migrating effluent that reaches drinking-water receptors does not meet drinking-water standards (for example, no detection of fecal coliforms), then actual risk would exist. However, this risk assessment does not take into account drinking-water treatment that would remove microbial pathogens.

4.9 Final Conceptual Model of Risk for Deep-Well Injection

Deep-well injection of treated municipal wastewater involves the injection of treated wastewater beneath a confining layer of rock and beneath a USDW. Deep-injection wells are regulated as Class I injection wells. In South Florida, injection is done at depths ranging from approximately 1,000 feet to more than 2,500 feet deep. These depths are below the shallow surficial aquifers (that is, the Biscayne Aquifer and an unnamed surficial aquifer) that extend to depths of approximately 20 to more than 800 feet and below the USDW.

Deep-well injection constitutes one of the most important and widely used methods of municipal wastewater management in South Florida, in terms of permitted discharge capacity. Overall, deep-well injection accounts for approximately 20%, or 0.44 billion gallons per day, of the total wastewater management capacity in the entire state.

Treatment of wastewater destined for deep-well injection in Dade and Brevard counties consists of secondary treatment with no disinfection, although backup disinfection capability is required. In Pinellas County, wastewater is treated to reclaimed water standards before being discharged into deep-injection wells. Reclaimed water standards include secondary treatment with basic disinfection, filtration, and higher-level disinfection.

This risk assessment and risk characterization is intended to provide a broad and representative picture of potential human health and ecological risks posed by deep injection of treated wastewater in different regions of South Florida. It is not intended to serve as a detailed risk assessment of specific sites. Therefore, for this risk assessment, three counties were selected for detailed risk analysis because they provide different and representative hydrogeologic conditions for their geographic areas: Dade County, Brevard County, and Pinellas County. These counties have significant wastewater management needs because of their populations.

A generic conceptual model of potential risk was developed to help evaluate risks. This model forms part of the generic risk analysis framework (GRAF) for evaluating risk, akin to a blueprint or conceptual plan for conducting a risk assessment. The generic conceptual model provides a set of guidelines for describing, analyzing, and understanding generalized or potential risks. The evaluation of the model involves use of specific information to examine whether the model is valid or not and to refine the model. This results in a final conceptual model that describes and characterizes risks based on specific information.

The generic conceptual model of potential human health and ecological risks was developed based upon the fate and transport of discharged treated effluent and its constituents in groundwater. A fate-and-transport approach to characterizing risk was selected because risk does not exist without exposure to stressors. Analysis of the fate and transport is an analysis of whether or not discharged effluent constituents can reach drinking-water supplies and pose risks to consumers. This involves an analysis and characterization of the pathways traveled by discharged effluent through the subsurface, analysis of the fate of chemical constituents and microorganisms as the effluent travels in groundwater, and characterization of the risks if effluent constituents were to reach drinking-water receptors (defined here as the USDW and public water-supply wells).

The analysis of groundwater transport evaluated two endpoints of possible transport pathways:

- Scenario 1, flow through porous media characterized by primary porosity
- Scenario 2, preferential flow through fractures, cracks, or other conduits, characterized by secondary porosity.

These two scenarios represent the two extremes of possible groundwater transport. Porous media flow involves groundwater movement through rocks or soil with many small pore spaces, or primary porosity; slow seepage through loamy soil is an example of porous media flow. Porous media flow typically occurs at slow rates. Conversely, preferential flow involves more rapid flow of water along preexisting fractures, cracks, channels, or other large conduits in rock, which constitutes secondary porosity **[?]**. (In this risk assessment, scenario 2 does not incorporate porous media flow, because evaluation of dual porosity is not feasible at this time).

Travel times for effluent water to travel through limestone to the USDW and to drinking-water wells were calculated. Different travel times were calculated, using primary porosity (scenario 1) and secondary porosity (scenario 2) and also based upon information on formation thickness, hydraulic conductivities, and other hydrogeologic parameters. Vertical travel times were used to calculate horizontal migration distances, which represent the horizontal distance that discharged effluent would travel in groundwater, given a vertical travel time.

Travel times for effluent constituents were also calculated; the latter may differ from travel times for effluent water if effluent constituents become attenuated (decrease in concentration) as the effluent migrates over time. If, on the other hand, effluent constituents behave conservatively, then they do not experience any change in concentration over time. Nitrate and ammonium were assumed to behave conservatively in the absence of information on microbiological transformation processes in the deep subsurface. Arsenic also was evaluated as a conservative constituent, based on its chemical behavior under reducing conditions.

The yardsticks used to measure risk, called assessment endpoints, include regulatory standards for water quality of treated effluent, groundwater, and drinking water MCLs. Other standards or recommended guidelines for water quality were also used, such as the DEP's guidelines for pathogenic microorganisms (FDEP, 1998; York et al., 2002). An assessment endpoint can be regarded as a concentration threshold or safe level above which there is a risk of an adverse effect.

The chemical constituents of wastewater selected as representative stressors for the analysis of fate of constituents included nutrients (nitrate, ammonium, phosphate), metals (arsenic, copper), VOCs (tetrachloroethene), synthetic organic compounds (chlordane, di(2-ethylhexyl)phthalate or DEPH), endocrine-disrupting compounds (DEPH), and chlorination by-products (trihalomethanes, including chloroform). Microbial pathogens or indicators of wastewater included representatives of bacteria, viruses, and pathogenic protozoans (*E. coli*, total coliform counts, rotaviruses, other enteric viruses, *Cryptosporidium parvum*, and *Giardia lamblia*).

These estimated fate and transport mechanisms were then compared with groundwater monitoring information from injection-well facilities.

The final conceptual model consists of the results of the evaluation of the conceptual model using site-specific, representative information wherever possible. The elements of the final conceptual model are described below.

4.9.1 Injection Pressure Head and Buoyancy Pressure

Vertical migration of effluent constituents depends on two major components: pressure head from injection and pressure head from buoyancy. Pressure head from injection is a result of injected effluent displacing native groundwater in the injection zone. Pressure head from buoyancy is a result of salinity and temperature differences between the

injectate and native groundwater. Fluids that are more saline tend to be denser than fluids that are less saline. Warmer fluids tend to be less dense relative to cooler fluids.

In each county (Dade, Pinellas and Brevard), the injection pressure head and pressure head from buoyancy was determined. Pressure head from injection is a governing component for vertical migration in Pinellas County. In Dade and Brevard counties, the pressure head from injection is considered to be negligible because of the hydrogeologic conditions (highly karstified) found in the Boulder Zone (injection zone). Therefore, in these counties, pressure head from buoyancy is the governing component for vertical migration.

4.9.2 Vertical Time of Travel

In scenario 1 (porous media flow), the total vertical travel times to receptor wells in Dade and Brevard counties are in the magnitude of more than 1,000 years (Table 4-8). In Dade County, it is estimated that discharged effluent will require more than 600 years to travel through the intermediate confining unit. In Brevard County, the discharged effluent will require more than 500 years to travel through the Lower Floridan because of the thickness of the aquifer (more than 1,400 feet). In Pinellas County, because of the injection pressure and the relatively short travel distance (and aquifer thickness) the total estimated time of travel to reach a hypothetical receptor well is 23 years.

Time to reach an USDW for scenario 1 is in the range of approximately 300 to 400 years in Brevard and Dade counties, respectively. In Pinellas County, the estimated travel time for effluent to reach the USDW is 2 years.

In scenario 2 (bulk flow through preferential flow paths), the vertical travel time was predicted to be 1 to 2 orders of magnitude shorter than travel times predicted for scenario 1 (Table 4-8). Scenario 2 represents flow through fractures or cracks and does not include primary porosity; such fractures can allow rising fluid to migrate through a confining unit. The travel times predicted to reach a receptor well in Dade, Brevard, and Pinellas counties are approximately 136, 30, and 6 years, respectively.

The time to reach the USDW in scenario 2 is approximately one order of magnitude shorter than in scenario 1. In Dade and Brevard counties, the travel times to the USDW under scenario 2 are 14 and 86 years, respectively. Travel time is 170 days in Pinellas County.

4.9.3 Horizontal Distance Traveled in a Given Travel Time

Based on horizontal hydrogeologic conditions and estimated vertical travel times, the extent of horizontal migration was estimated for each county. For scenario 1, the expected horizontal migration in Dade County is approximately 16 miles. Dade County has the furthest horizontal migration relative to Brevard and Pinellas counties, which have an expected horizontal migration of 1.5 and 1.2 miles, respectively. For scenario 2, as expected, Dade County has the furthest horizontal migration distance of 1.6 miles,

while Brevard and Pinellas counties have horizontal travel distances of 0.1 and 0.6 miles, respectively.

4.9.4 Fate of Chemical Constituents

For both scenarios 1 and 2, final concentrations of all chemical constituents were negligible or below drinking-water MCLs at representative USDWs and receptor wells. Figure 4-10 shows the rate of reduction of all nonconservative chemical constituents over a period of time. All nonconservative chemical constituents have negligible final concentrations after 40 years. Final concentrations of conservative chemical constituents, such as nitrate, ammonia, and arsenic, do not decrease, but because their initial concentrations in treated effluent are below MCL or Lifetime Health Advisory limits, their final concentrations are also below these limits. Therefore, they are not deemed to present significant human health risks, although there may still be cause for some concern because concentrations are occasionally near MCLs.

4.9.5 Comparison with Monitoring-Well Data

The scenarios described above represent two distinct scenarios of fluid flow occurring separately (that is, porous media or bulk flow only). In limited areas with minimal rock fracturing, porous media flow might occur alone. However, in general, flow through rock fractures would not occur without concurrent porous media flow.

The monitoring data are consistent with both types of flow. This relationship is expressed with slight differences in the different regions studied. In Pinellas County, steady and gradual changes in concentrations over 20 years of operation indicate that preferential pathways are present. These changes began to occur shortly after injection began, which is consistent with the model's bulk flow travel time for this region. In Brevard County, some changes have occurred more quickly than was predicted by the model, which is indicative of bulk flow. In Dade County, changes have also occurred with greater rapidity than predicted by the model. Instead of a steady concentration gradient like that detected in the other two studied regions, there are discontinuities in both the vertical and horizontal directions. Bulk flow through rock fractures may also be present, but it may be moving at slower rates, similar to those predicted by the model.

4.9.6 Mechanical Integrity as a Risk Factor

As discussed above, monitoring data indicate that upward migration of injectate is likely via both porous media and bulk flow in Pinellas and Brevard counties. Mechanical integrity of the injection and monitoring wells in these regions does not appear to be a significant risk.

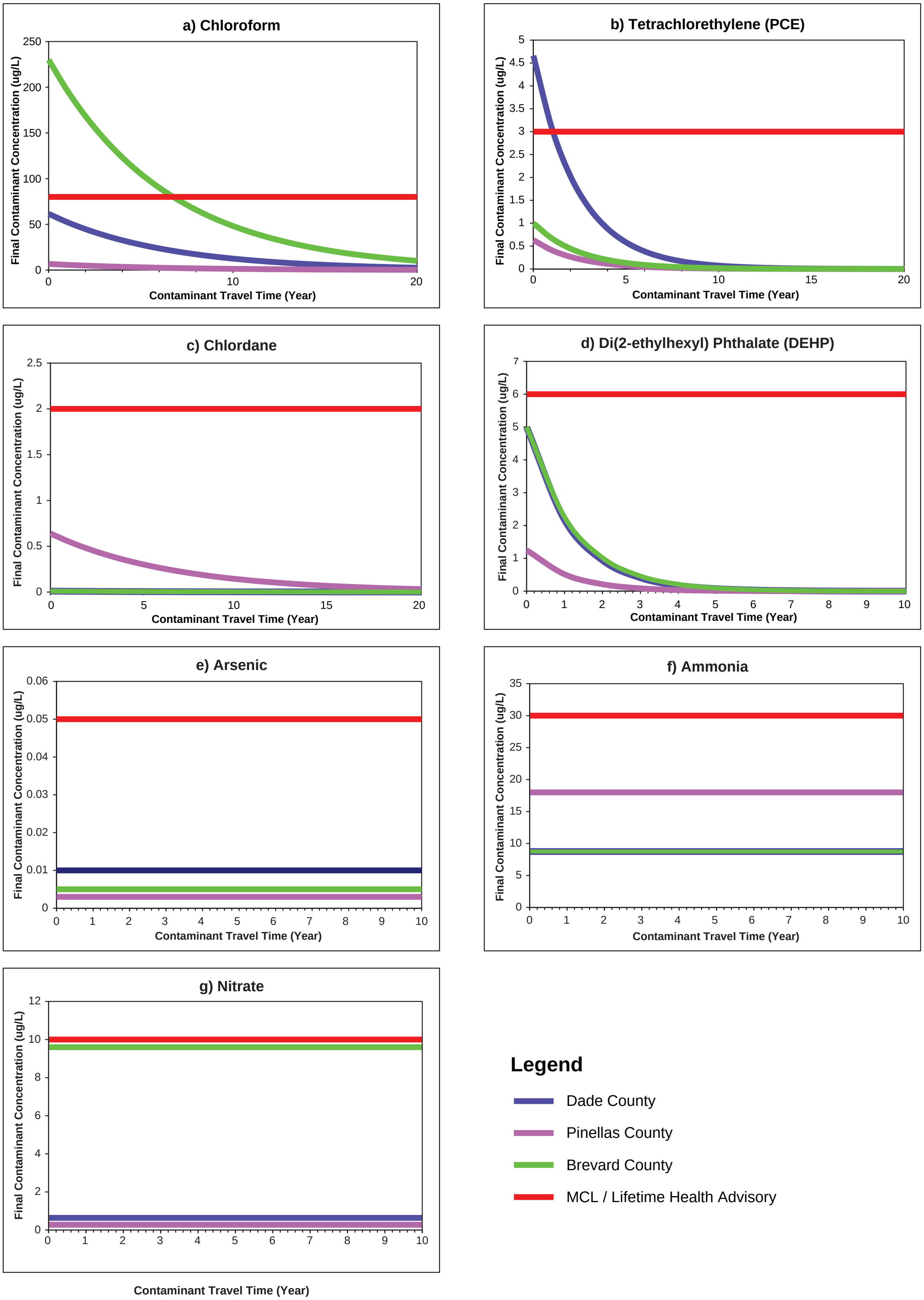

Figure 4-8. Final Concentrations of Representative Stressors Versus Time

4.9.7 Fate and Transport of Pathogenic Microorganisms

Because deep-well injection of wastewater does not require basic disinfection or filtration, there is a potential risk of microbial contamination of the USDW and possibly of public water-supply wells. Microorganisms (bacteria, viruses, protozoans) are capable of being transported in groundwater over distances comparable to the vertical and horizontal travel distances that effluent must travel in order to reach the USDW and wells.

Microbial inactivation rates for bacteria and viruses range from several days to tens of days for a 1 log reduction in microbial activity (equivalent to 90% inactivation). For injection wells that are experiencing fluid migration into the USDW because of rapid preferential flow, bacteria and viruses may pose some cause for concern.

Microbial inactivation rates for *Cryptosporidium*, one of the more resistant and long-lived pathogenic microorganism identified in water, are in the range of 200 days for a 1 log reduction, corresponding to 90% inactivation of the population present. This slow rate of inactivation means that chlorine-resistant pathogens like *Cryptosporidium* may be capable of surviving long enough to reach USDWs if travel times are on the order of months to several years.

The longer the vertical travel time, the more chance that natural inactivation of microbial activity will occur. Thus, Pinellas County, with its short travel times of several years, would appear to be at highest risk. However, Pinellas County employs basic disinfection, filtration, and high-level disinfection, in addition to secondary treatment. In Pinellas County, the quality of treated effluent is virtually that of drinking water. For these reasons, its risk from microbial pathogens is probably the lowest of the three counties evaluated.

Because basic disinfection and filtration are not done, Dade and Brevard counties, despite travel times of several decades or longer, may be at some risk from long-lived or especially resistant microorganisms or from those that can survive in an inactive state for long periods of time. Effluent quality from secondary treatment without basic disinfection or further disinfection would not meet drinking-water standards (no detection of fecal coliforms). No information is available concerning concentrations of *Cryptosporidium* or *Giardia* in such wastewater from South Florida, but it may be assumed that without disinfection and filtration, concentrations of these cyst-forming protozoans may be significant.

Scenario 2 (preferential flow) poses the highest potential human-health risk from microbial pathogens. Scenario 1 (porous media flow) poses low or very low potential human-health risk from microbial pathogens because of the long travel times, the fact that it is unlikely that microorganisms would survive long enough to reach receptors (unless there is *in situ* growth), and the fact that primary porosity may act to filter microorganisms and retain them. Fluid movement of effluent from injection wells with

mechanical integrity issues could also pose higher risks, because it would promote preferential flow.

4.9.8 Effects of Data Gaps

There are significant gaps in completeness of geographic coverage for monitoring-well data and effluent quality. Nevertheless, this risk assessment is useful on a regional basis, because values of parameters were selected to be representative of a wide range of possible values. There do not appear to be any monitoring wells in the Biscayne Aquifer, which represents a significant gap in information that would be useful for evaluating risks in the surficial aquifer from deep-well injection and aquifer recharge. There are no monitoring data on unregulated constituents of wastewater, such as endocrine-disrupting compounds.

The area of groundwater microbiology represents a scientific frontier in microbial ecology. This is to say, there is a severe shortage of information on microbial pathogens, other than fecal coliforms, in groundwater and in deeper aquifers in South Florida. This may be in part because monitoring for other types of microorganisms is not required, but it is also because *in situ* microbial ecological studies are difficult to conduct. Information that would be useful for a full and complete microbial risk assessment includes *in situ* rates of inactivation in groundwater; concentrations of pathogenic protozoans, viruses, and bacteria in groundwater and their viability; tracer studies to examine the sources of microbial contamination of groundwater; and time-series studies of microbially mediated chemical transformations *in situ*.

The lack of information on microbial biogeochemical processes in the deep subsurface also causes the analysis of fate of chemical constituents to be incomplete, at least for compounds that may undergo microbially mediated transformations. Examples of these include denitrification, nitrification, oxidation, reduction, volatilization, and other processes that can affect concentrations of metals, organic compounds, and nutrients. Indeed, weathering of rocks and soil is largely accomplished through such microbial transformations.

This risk assessment did not evaluate whether or not deep-injection fluids could be transported to coastal areas and to marine waters. Wastewater effluent appears to migrate from some shallow Class V injection wells and from onsite sewage-disposal systems (septic systems) into coastal ecosystems in the Florida Keys, based on tracer studies of nutrients. However, there is no corresponding tracer study of deep-injection fluids.

This risk assessment also did not account for cumulative risks from this wastewater management option and other sources of the same chemical and microbial stressors on the surface.

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5.0 AQUIFER RECHARGE

This section develops and presents information that has been incorporated into the conceptual model describing risks associated with the aquifer recharge wastewater management option.

5.1 Definition of Aquifer Recharge

Aquifer recharge in its broadest sense refers to the replenishment or recharge of a groundwater aquifer. In Florida, a number of practices involving use of reclaimed water may be termed aquifer recharge. *Reclaimed water* is wastewater that has received at least secondary treatment and basic disinfection or better and that is reused after leaving a municipal wastewater treatment facility. *Reuse* means the application of reclaimed water for a beneficial purpose (FDEP, 2001b). Reuse of reclaimed water is strongly supported and instituted in state law to encourage water conservation (FDEP, 2001c). Beneficial uses include irrigation, recharge of groundwater through rapid- or slow-rate land application, and enhancement or creation of wetland habitat. Reuse does not include direct consumption of water by humans.

The types of reuse allowed in Florida (FDEP, 1998) that involve aquifer recharge are listed below:

- Slow-rate land application systems (restricted public access)
- Rapid-rate land application systems
- Irrigation of public-access areas
- Rapid infiltration basins (RIBs)
- Unlined storage ponds
- Discharge to wetlands that percolate to groundwater
- Septic tanks
- Injection to groundwater
- Aquifer storage and retrieval
- Injection for salinity barriers
- Deep injection wells.

The first seven uses of reclaimed water involve application of treated water on or near the surface of the land, allowing percolation of the water to occur through soil. The last four uses of reclaimed water involve active injection of treated wastewater or other water into the ground at various depths. An example of the latter is aquifer storage and retrieval (ASR). ASR typically involves the storage of excess drinking-water-quality water in a subsurface aquifer for later recovery and use during periods when demand for drinking water exceeds availability. Although reclaimed water may be used, ASR typically is not used to dispose of treated wastewater but is instead aimed at temporarily storing drinking water. Reuse that involves discharges of reclaimed water to surface water is described in Chapter 7.

For this risk assessment, several types of reclaimed water reuse that may result in aquifer recharge were evaluated. These include slow-rate land application systems (including irrigation), rapid-rate land application systems (including RIBs and unlined storage ponds), and wetland treatment systems. These types of aquifer recharge are characterized by surface application of reclaimed water over an area and allowing the water to percolate downward and outward from the point of application.

Other practices involving reuse of reclaimed water or use of drinking-water-quality water were not evaluated in this risk assessment. These include Class V shallow-injection wells for disposal of treated wastewater, ASR systems, salinity barriers, and septic systems. Class V shallow-injection wells, which are regulated by federal and state regulations, are used for disposal of industrial, as well as treated, municipal wastewater and were not evaluated in this risk assessment. ASR was not evaluated because it often utilizes surface water rather than reclaimed water, as described above. Salinity barriers were not evaluated because they are not intended for disposal of wastewater. This risk assessment does not address on-site sewage disposal systems such as septic systems, a wastewater management option that serves about 25% of Florida's population. Nevertheless, where reclaimed water is used for such purposes, the risk analysis presented here may be applicable.

5.2 Use of Aquifer Recharge in South Florida

The Division of Water Resources Management of the Florida Department of Environmental Protection (DEP) conducts yearly inventories of all active domestic wastewater treatment facilities that provide reclaimed water for reuse. The DEP's *2000 Reuse Inventory* lists facilities having permitted capacities of at least 0.1 million gallons per day (mgd) or more and describes reuse activities throughout the state of Florida (FDEP, 2001a).

Types of reuse included in the DEP inventory are irrigation of public-access areas, landscape irrigation, agricultural irrigation, groundwater recharge, indirect potable reuse, industrial uses, wetlands, and other uses. Irrigation of public-access areas and landscapes includes irrigation of golf courses, residential areas, and other public-access areas. Agricultural irrigation includes irrigation of edible and inedible crops. Groundwater recharge and indirect potable reuse includes RIBs, absorption fields, surface-water augmentation, and injection. Industrial uses include those at the treatment plant or at other facilities. Wetland uses include discharge to wetlands and creation or enhancement of existing wetlands.

According to the *2000 Reuse Inventory* (FDEP, 2001a), the leading use of reclaimed water in Florida is irrigation of public-access areas and landscapes (Tables 5-1 and 5-2), totaling 107,123 acres, by far the largest area covered by any reuse activity. Agricultural irrigation accounts for the second-largest area receiving reclaimed water (35,282 acres). Groundwater recharge in Florida accounts for 7,418 acres, while wetland uses of reclaimed water account for 4,791 acres. Altogether, 154,954 acres receive reclaimed water through various types of reuse activities.

Table 5-1. Reclaimed Water Reuse Activities in Florida

Reuse Type	No. of Systems[1]	Capacity (mgd)	Flow (mgd)	Area (acres)
Public-access areas and landscape irrigation				
Golf course irrigation	179	241	108	46,730
Residential irrigation	82	163	95	39,896
Other public-access areas	98	99	44	20,497
Subtotal:[2]	359	503	247	107,123
Agricultural Irrigation				
Edible crops	21	54	35	14,414
Other crops	96	133	73	20,868
Subtotal:[2]	117	187	108	35,282
Groundwater recharge and indirect potable reuse				
Rapid infiltration basins	169	171	85	6,969
Absorption fields	20	8	3	449
Surface-water augmentation	0	0	0	NA
Injection	1	10	8	NA
Subtotal:[2]	190	189	96	7,418
Industrial				
At treatment plant	76	129	66	4
At other facilities	17	35	21	0
Subtotal:[2]	93	164	87	4
Toilet flushing	3	0	0	NA
Fire protection	0	0	0	NA
Wetlands	14	66	32	4,791
Other uses	10	7	5	336
Totals:[2]	**427**	**1,116**	**575**	**154,954**

[1]The numbers of facilities are not additive because a single facility may engage in one or more reuse activity.
[2] Discrepancies in column totals are from internal rounding associated with the development of this summary table.
Source: FDEP, 2001a.

Table 5-2. Reuse Flows for Reuse Types in Florida DEP Districts and Water Management Districts

Districts	Irrigation of Public-access Areas (mgd)	Agricultural Irrigation (mgd)	Ground-water Recharge (mgd)	Industrial (mgd)	Wetland Systems and Others (mgd)	Totals (mgd)
DEP Districts						
Southeast (West Palm Beach)	25.98	0.94	7.68	27.12	1.52	63.24
South (Fort Myers)	52.37	5.06	8.60	1.18	2.28	69.49
Southwest (Tampa)	79.89	21.50	15.44	30.80	6.64	154.27
Subtotal, DEP districts in South Florida study area	**158.24**	**27.5**	**31.72**	**59.1**	**10.44**	**287.00**
Central (Orlando)	71.69	43.90	50.17	15.96	21.84	203.56
Northeast (Jacksonville)	9.45	6.63	10.73	5.35	0.63	32.79
Northwest (Pensacola)	8.62	30.09	3.50	5.92	3.85	51.98
Totals, all DEP districts	**248.00**	**108.12**	**96.12**	**86.33**	**36.76**	**575.33**
Water Management Districts						
South Florida[1]	90.34	23.14	43.47	28.81	3.81	189.57
St. John's River[2]	67.16	25.05	31.11	20.64	22.37	166.33
Southwest Florida[2]	81.77	23.56	17.12	30.89	6.71	160.05
Northwest Florida	8.62	30.18	3.50	5.92	3.88	52.10
Suwannee River	0.11	6.19	0.93	0.06	0.00	7.29
Totals , all water management districts:	248.00	108.12	96.13	86.32	36.77	575.34

[1]The area covered by the South Florida Water Management District is smaller than the area of this study.
[2]Approximately half of these water management districts are outside of the area of this study.
Source: FDEP, 2001a.

As Table 5-2 indicates, use of reclaimed water for public-access areas accounts for the largest flows of reclaimed water in Florida (248 mgd), followed by agricultural irrigation (108.12 mgd), groundwater recharge (96.12 mgd), industrial use (86.33 mgd), and wetlands (36.76 mgd), based on DEP districts. In the South Florida study area, use of reclaimed water for public access is also the leading use (158.24 mgd), followed by industrial use (59.1 mgd), groundwater recharge (31.72 mgd), irrigation (27.5 mgd), and wetlands (10.44 mgd), based on DEP districts.

The DEP *2001 Reuse Inventory* states that Florida has 359 systems using reclaimed water for irrigation of public-access areas and landscape irrigation, of which approximately one-half (179) are golf-course irrigation systems. The other systems are nearly evenly divided among those serving other public-access areas (98) and residential irrigation (82).

According to the Florida DEP, reuse of reclaimed water on golf courses accounts for 42 percent of all reuse in Florida (FDEP, 2002). Agricultural irrigation systems using reclaimed water total 117. These two types of irrigation involve slow-rate land application. Industrial systems total 93. In the category of ground water recharge, there are 189 reuse systems utilizing rapid-rate land application (169 RIBs plus 20 absorption fields), out of a total of 427 reuse systems in the state. There are 14 wetlands systems using reclaimed water (see Table 5-1).

It is important to note that, to provide flexibility in meeting discharge requirements, a wastewater treatment facility may utilize more than one wastewater management option. Similarly, more than one type of reuse system may be used at a particular site (FDEP, 2001a).

5.3 Environment into Which Treated Wastewater is Discharged

Aquifer recharge involves surface infiltration and percolation of treated reclaimed wastewater through soils and geologic media overlying the surficial aquifer or the Biscayne Aquifer, depending on the location. In Dade County, the Biscayne Aquifer receives recharge. In Pinellas and Brevard counties, the unnamed surficial aquifer receives recharge. The Biscayne and surficial aquifers are described below. See chapters 2 and 4 for more detailed information on these aquifers.

5.3.1 Biscayne Aquifer System

The Biscayne Aquifer covers an area of approximately 4,000 square miles of South Florida (USGS, 2000). This aquifer extends along the eastern coast from southern Dade County into coastal Palm Beach County. It is located above the Floridan Aquifer, separated by approximately 1,000 feet of low-permeability clay deposits. The Biscayne Aquifer ranges in thickness from 50 to 830 feet and is composed of highly permeable limestone or calcareous sandstone (Meyer, 1989; Reese, 1994; Maliva and Walker, 1998; Reese and Memburg, 1999; Reese and Cunningham, 2000).

The Biscayne Aquifer system is the main source of water for Dade, Broward, and southeastern Palm Beach counties and serves the cities of Boca Raton, Pompano Beach, Fort Lauderdale, Hollywood, Hialeah, Miami, Miami Beach, and Homestead. According to the U.S. Geological Survey, this aquifer is the sole source of drinking water for 3 million people. Because the Biscayne Aquifer lies close to the surface and is highly permeable, it is highly susceptible to contamination.

5.3.2 Surficial Aquifer

In areas of South Florida outside the Biscayne Aquifer, the unnamed surficial aquifer is used locally for community and public water supply. The surficial aquifer is composed of relatively thin layers of sands and limestone. The surficial aquifer ranges in thickness from 20 to 800 feet, reaching its greatest thickness in southeastern Florida (Adams, 1992; Barr, 1996; Lukasiewiez and Adams, 1996; Reese and Cunningham, 2000). Although the

surficial aquifer yields relatively small volumes of water, it is an important source of private water supplies (Miller, 1997).

5.4 Regulations and Requirements for Aquifer Recharge

The level of wastewater treatment required for various reuse options is specified in state regulations, including chapters 62-600 of the Florida Administration Code (FAC) (Domestic Wastewater Facilities), 62-610 FAC (Reuse of Reclaimed Water and Land Applications), and 62-611 FAC (Wetland Applications).

In addition to required treatment levels, state regulations specify system design and operational requirements regarding facility capacity, monitoring requirements, backup systems, and setback distances. All potable and nonpotable water supply wells and monitoring wells within a 0.5-mile radius of reclaimed-water facilities must be identified in permit applications for reclaimed-water facilities. Engineering reports must demonstrate that reclaimed water or effluents will not violate water quality standards.

Reclaimed-water systems may be located in areas that have Class F-I, G-I, and G-II groundwaters for potable-water use, as defined by Rule 62-520 FAC (DEP 1996 Ground Water Standards and Exemptions). Reclaimed-water facilities are required by EPA Class I reliability regulations to provide backup treatment and wastewater-holding capability in the event that treatment is disrupted or interrupted. Redundant treatment, recirculation and retreatment, and the use of holding ponds with extra capacity are examples of backup treatment and retention methods.

Sampling for *Cryptosporidium* and *Giardia* is required for discharges that may potentially affect Class I surface waters and is also required for groundwater recharge or salinity-barrier-control discharges. Although there are no federal or state numerical standards for pathogenic protozoans in reclaimed water, the Florida DEP recommends that concentrations of *Cryptosporidium* and *Giardia* should not exceed 5.8 oocysts and 1.4 oocyst per 100 liters (L), respectively (York et al., 2002).

5.4.1 Slow-Rate Land Application Systems

Slow-rate land application involves the discharge of treated water to the land’s surface and the eventual percolation of this water through soils and rocks, leading to aquifer recharge. To prevent surface runoff or ponding of the applied reclaimed water, hydraulic loading rates are regulated. The loading rate is established after considering the ability of the plant and soil system to remove pollutants from the reclaimed water and the infiltration capacity and hydraulic conductivity of geologic materials underlying the system. Slow-rate land application systems typically are designed with hydraulic loading rates between 0.15 and 1.6 centimeters per day (cm/day) (US EPA, 1981; Metcalf and Eddy, 1991; Water Environment Federation, 1992; Kadlec and Knight, 1996).

Slow-rate land application systems must have backup disposal methods for wet weather conditions and when water quality treatment standards are not met. During wet weather,

effluent may be discharged to storage areas or discharged through an alternative permitted disposal system.

In restricted access areas, reclaimed water must be provided with secondary treatment and basic disinfection. In public-access areas, reclaimed water must receive secondary treatment with high-level disinfection, at a minimum. Concentrations of total suspended solids must be reduced through methods such as filtration or addition of substances that cause coagulation, such as polyelectrolytes. Filtration increases the effectiveness of disinfection, particularly for removing cyst-forming pathogenic protozoans such as *Cryptosporidium parvum* and *Giardia lamblia*. Because of the potential for public exposure to many reuse projects, particular care is necessary to minimize the spread of pathogens (FAC 62-610, Part III, Slow-Rate Land Application Systems: Public Access Areas, Residential Irrigation, and Edible Crops).

All land application systems, whether slow-rate or rapid-rate, must maintain setback distances to surface water and potable supply wells to protect water quality and ensure compliance with water quality and drinking-water standards. For example, RIBs, percolation ponds, basins, trench embankments, and absorptions fields must be set 500 feet from potable-water wells or Class I or II waters. The setback distance to potable-water wells can be reduced to 200 feet if high-level disinfection is provided, Class I reliability is provided, and if soils hydrology, well construction, hydraulic loading rates, reclaimed-water quality, and expected travel time of groundwater to the potable water supply provides reasonable assurance that water quality standards will be met at the well (FAC 62-610.521).

5.4.2 Rapid-Rate Land Application Systems

Rapid-rate land application also involves the discharge of treated water to the land's surface and the eventual recharge of the underlying aquifer. However, rapid-rate systems have a much faster percolation rate than slow-rate systems. Rapid-rate systems are typically designed with hydraulic loading rates between 1.6 and 25 cm/day over the area of the basins (Kadlec and Knight, 1996). No wet-weather backup system is required for rapid-rate land application. Rapid-rate land application systems are also required to meet groundwater quality criteria at the edge of a zone of discharge.

Because of the potential for faster migration of discharged water, treatment standards for rapid-rate systems are higher. For rapid-rate land application, Florida regulations require secondary treatment with high-level disinfection (FAC 62-610). The following standards of water quality must be met:

- Total suspended solids must be less than 5 milligrams per liter (mg/L) before disinfection
- Total nitrogen (total N) must be less than 10 mg/L
- Treatment must meet drinking-water standards.

High-level disinfection with filtration is effective at inactivating viruses, bacteria, and pathogenic protozoans in reclaimed water, especially if monitoring for removal of protozoans is conducted (York et al., 2002).

5.4.3 Wetland Systems

Florida's domestic wastewater-to-wetlands rule controls the quantity and quality of treated wastewater discharged to wetlands while protecting the type, nature, and function of wetlands. This is codified in chapter 62-611 FAC. The wastewater-to-wetlands rule regulates the quality of water discharged from wetlands to contiguous surface waters. It also provides standards for water quality, vegetation, and wildlife to protect wetland functions and values and establishes permitting and monitoring requirements for discharges of treated wastewater to wetlands. This rule allows the use of constructed wetlands and altered wetlands for discharge of treated wastewater to create and restore wetlands (FDEP, 2001e).

Reclaimed wastewater that is discharged to wetlands must undergo secondary treatment with nitrification to further reduce the concentration of nitrogen. The treated reclaimed wastewater must meet the following standards:

- Carbonaceous biochemical oxygen demand must be less than 5 mg/L
- Total suspended solids must be less than 5 mg/L
- Total nitrogen (as N) must be less than 3 mg/L
- Total phosphorus (as P) must be less than 1 mg/L.

Discharge to wetlands can be beneficial in several ways. Wetlands provide additional filtration to discharged waters, thereby improving effluent quality. Inputs of water help to maintain the wetland ecosystem. In some locations (for example, the Wakodahatchee Wetlands facility in Palm Beach County), rapid-rate land application systems have been converted to wetland treatment systems. The Wakodahatchee Wetlands receive approximately 2 mgd of highly treated reclaimed water. This water serves to maintain various types of wetland habitats for wildlife (FDEP, 2001e).

Treatment wetlands are prohibited within the boundaries of Class I or Class II waters (designated as Outstanding Florida Waters), or areas of critical state concern, or when the wetland is exclusively herbaceous. Groundwater and drinking-water quality standards are not specifically referenced in the wetland applications regulations. However, secondary treatment with nitrification generally assures that drinking-water standards will be met. According to a recent review of data from Florida reclaimed-water facilities, treatment systems that provide nitrification may also be more effective in removing pathogenic protozoans (York et al., 2002). Monitoring for fecal coliforms as an indicator of wastewater pathogens is required in treatment wetlands.

Disinfection of secondary-treated wastewater with chlorine (used in both basic disinfection and high-level disinfection) is highly effective at inactivating nearly all bacteria and viruses. Although there are no numerical water quality standards regulating

the concentrations of pathogenic protozoans in treated wastewater, the Florida DEP recommends that no more than 5.8 *Cryptosporidium* oocysts per 100 L and no more than 1.4 *Giardia* cysts per 100 L be allowed in reclaimed water. Filtration is the preferred method of removing pathogenic protozoans, although the DEP has found that filtration is not always effective (York et al., 2002).

5.5 Problem Formulation

In this section, the potential risks that may be associated with the aquifer recharge wastewater management option are described. In section 5.6, potential risks are analyzed.

In conducting the option-specific risk analysis for aquifer recharge, an effort was made to focus upon those reuse practices that best fit the broad definition of aquifer recharge and that are most widely used within the study area. Wetland systems, as well as rapid and slow-rate land application systems, are each used within the study area. However, for reasons outlined below, this option-specific risk analysis focused on rapid-rate land application systems (RIBs).

5.5.1 Slow-Rate Land Application Systems

Slow-rate land application systems often involve the use of reclaimed water to irrigate vegetated systems, which assist in wastewater polishing and disposal. Irrigation rates are generally low or intermittent, allowing aerobic soil conditions to become established, if not continually, at least intermittently. Aerobic conditions in turn allow the growth of upland vegetation, which removes nutrients, filters wastewater solids, and creates more permeable soils. Slow-rate land application of treated wastewater is used throughout the United States (Kadlec and Knight, 1996).

In South Florida, slow-rate land application nearly always means irrigation, including irrigation of public-access areas and landscape areas (for example, golf courses, parks, highway medians, and cemeteries), and agricultural irrigation. In addition to plant uptake and evapotranspiration (water loss to the atmosphere because of plant respiration), a portion of the applied water may percolate to groundwater.

Following treatment, reclaimed water may still contain nutrients such as nitrogen, phosphorus, and other substances that act as nutrients. If such reclaimed water is applied to vegetated areas, additional nutrient removal can be expected because of uptake by vegetation. Vegetation is often used as a “polishing” agent to help remove nutrients in wastewater treatment, and there are some wastewater treatment approaches that are based largely upon the use of plants to remove nearly all pollutants. Wetland treatment systems in particular rely heavily upon vegetation to remove or reduce pollutants.

The efficacy of removal of nutrients and other substances by plants depends upon many factors, such as the rate of application, concentration of nutrients in the treated water being applied to vegetation, plant species used, rate of nutrient uptake by plants, microbial processes that may further affect uptake rates, soil type, moisture, pH,

temperature, whether other sources of nutrients also happen to be present, and length of exposure time (Kadlec and Knight, 1996).

If the rate of nutrient application equals the total rate of uptake by vegetation and all other uptake processes, then there should be little or no excess nutrients. Similarly, if irrigation with reclaimed water does not occur at a rate that exceeds the rate of uptake by vegetation and all other uptake processes, there will be little or no recharge of groundwater. Reuse systems that involve application to vegetated areas are typically operated so as to take into account a specific water budget and assimilative capacity. However, if the plants' capacity for water and nutrient uptake is less than the rate of application, excess water and nutrients will percolate without the beneficial functions of nutrient removal and water reuse that plants may provide.

Biodegradation of many wastewater constituents in soils and vegetation can also be expected. Biodegradation processes in soil include microbial uptake and transformation, microbially mediated decomposition of organic matter, microbial volatilization or solubilization, and further transformations as the breakdown products pass through the food chain to higher organisms (Brock et al., 1984; Kadlec and Knight, 1996). Microorganisms are important in the biogeochemical cycling of biologically important elements, including carbon, nitrogen, phosphorus, sulfur, iron, manganese, and silica, and play an important role in the decomposition of rocks and soils (Krumbein et al., 1983). Biological degradation of pesticides, petroleum products, metals, and other pollutants is often accomplished through microbial processes (Kadlec and Knight, 1996).

Facilities operating slow-rate land application systems are required to balance the application of reclaimed water with evapotranspiration rates. Therefore, these facilities do not typically operate their land application systems during periods of wet weather. Slow-rate land application systems are not likely to provide significant recharge to groundwater. Risks are expected to be very low to nonexistent.

5.5.2 Rapid-Rate Land Application Systems

Rapid-rate land application systems discharge treated wastewater to RIBs and absorption fields with highly permeable soils. RIBs involve a series of basins that may include subsurface drains, which are designed to receive and distribute reclaimed water. Absorption fields include subsurface absorption systems that may include leaching trenches, pipes, or other conduits to receive and disperse water underground. They are typically covered with soil and vegetation.

Rapid-rate application systems are typically loaded at hydraulic loading rates between 1.6 and 25 cm/day over the area of the basins (Kadlec and Knight, 1996). Absorption fields must be designed and operated to avoid saturated conditions at the ground surface. Projects proposed in areas with unfavorable hydrogeology (for example, karst) or other unfavorable characteristics must meet additional levels of treatment, as described below.

The use of rapid-rate land application may result in significant volumes of reclaimed water directly recharging the surficial aquifer. There is little potential for reduction in volume or additional removal of stressors by in situ natural attenuation processes, because of the large volumes applied and the rapid application rate. Because larger volumes of reclaimed water are applied and only an intermediate level of treatment is used, this form of aquifer recharge may pose the highest risks. Therefore, this option-specific risk analysis and risk assessment focuses on rapid-rate land application.

5.5.3 Wetland Systems

Wetlands, which are wet or inundated during part or all of the year, are often transitional areas between uplands and permanently flooded aquatic basins, such as lakes, ponds, lagoons, or coastal embayments. Wetlands are characterized by vegetation that has adapted to living under wet or occasionally inundated conditions and by hydric soils that develop chemical and physical characteristics related to low oxygen and frequent or constant exposure to water (US Army Corps of Engineers, 1987; Dennison and Berry, 1993; Cowardin et al., 1979). Wetlands are characterized by high rates of biological activity and productivity relative to upland ecosystems, making them capable of transforming and neutralizing many of the constituents found in treated wastewater (Kadlec and Knight, 1996).

Wetland systems or wetland treatment systems involve the application of reclaimed water to existing wetlands for the purpose of restoring wetlands and providing further treatment of water. Wetland reuse systems may provide more significant amounts of recharge to groundwater, particularly where there are direct hydrologic connections between the wetland and groundwater systems.

However, where perched wetlands exist because of the presence of a relatively impermeable soil layer (for example, clays, organic matter) that slows or prevents direct hydrologic connection with the underlying aquifer, a wetland may actually retard recharge of groundwater. The major difference between wetland systems receiving reclaimed water and all other types of aquifer recharge is that wetlands, particularly natural wetlands, will typically contain more ecological receptors than human receptors. Because discharge to wetlands is analogous to surface-water discharge of treated wastewater, the evaluation of risks from wetlands discharge is discussed in Chapter 7.

5.5.4 Florida DEP Study of Relative Risks of Reuse

In this risk assessment, information from a Florida DEP study of the risks of reclaimed water was integrated into the fate and transport analysis (FDEP, 1998). The Florida DEP risk study provided a qualitative ranking of the relative human health risks of reuse of reclaimed water that involves release to surface water or groundwater used for drinking-water supplies. The DEP study was intended to support state rulemaking. The qualitative ranking of various reuse options was based on the best professional judgment of professionals in regulatory agencies and other groups and on the 1×10^{-4} threshold for risk (that is, there is a 1-in-10,000 chance of a stressor causing illness or other adverse effect

in consumers). However, according the DEP, the $1x10^{-4}$ risk threshold may not be appropriate for defining microbial risk thresholds.

The DEP’s relative-risk ranking assigns a relative risk from 1 (high) to low (25) for various reuse activities using reclaimed wastewater. Injection of reclaimed water to aquifers, aquifer storage and retrieval using reclaimed water, discharge to Class I surface waters (drinking-water sources), and injection for salinity barriers were rated as the six highest-risk activities. Rapid-rate infiltration systems in karst (RIBs) ranked 7th, discharge to surface waters hydrologically connected to groundwaters ranked 11th, discharge to wetlands ranked 14th, rapid-rate infiltration systems in suitable geology ranked 15th, slow-rate systems ranked 17th, and irrigation of public-access areas ranked 18th. The lowest risk ranking was assigned to lined storage ponds.

Based on the DEP’s relative-risk ranking of various reuse options for reclaimed wastewater, rapid-rate infiltration systems were selected as a higher-risk form of aquifer recharge (excluding injection, ASR using reclaimed water, and salinity barriers) for this risk assessment. Selection of a higher-risk form of aquifer recharge provides a conservative or protective approach to risk assessment.

5.5.5 Potential Stressors

Potential stressors entrained or dissolved in the reclaimed water are discharged to RIBs. Wastewater constituents that may act as stressors to human or ecological health include pathogenic microorganisms, certain metals and inorganic substances, synthetic and volatile organic compounds, and hormonally active agents.

Rapid-rate land application systems are required to meet groundwater quality criteria at the lower edge of a discharge zone. Accordingly, most systems that utilize RIBs are operated in such a way that concentrations of stressors are substantially reduced before reclaimed water reaches and recharges the underlying aquifers.

The primary source of potential stressors is the effluent from wastewater treatment plants (that is, reclaimed water) that is discharged through one or more aquifer recharge facilities and eventually percolates to reach the underground surficial aquifer, a formation containing underground sources of drinking water (USDWs). Stressors include reclaimed water constituents such as metals and other inorganic elements; compounds such as inorganic nutrients (nitrate, ammonium, and phosphate); volatile and synthetic organic compounds; microorganisms that survive basic or high-level disinfection or are resistant to disinfection, such as pathogenic protozoans; and miscellaneous constituents. Chlorination, and especially high-level disinfection, is effective at inactivating bacteria and viruses; however, cyst-forming pathogenic protozoans, such as *Cryptosporidium parvum*, *Giardia lamblia,* are only removed through filtration designed for their removal (York et al., 2002).

Potential risks associated with the use of emergency ponds to receive wastewater during upset bypass conditions, such as storms or other events resulting in large volumes of

wastewater, can also be characterized using this conceptual model. Exposure pathways, receptors, and assessment endpoints are similar; concentrations and types of stressors may differ.

5.5.6 Potential Receptors and Assessment Endpoints

Potential drinking-water receptors include USDWs beneath the RIB, other USDWs to which groundwater flow may carry potential stressors, public and private water-supply wells, and surface waters. Federal drinking-water standards (maximum contaminant levels (MCLs)) and other health-based standards serve as the analysis endpoints for assessing risks to each of these potential drinking water receptors.

The USDWs that may be recharged by RIBs include the unnamed surficial aquifers and the Biscayne Aquifer. The surficial aquifers are used for domestic private water supplies and for municipal water supplies in central South Florida and along the east and west coasts (Randazzo and Jones, 1997). The Biscayne Aquifer is tapped by private wells and also supplies large public water systems in Dade, Broward, and Palm Beach counties. Water obtained through private wells is often used directly (without pretreatment). Community and municipal water systems generally do pretreat groundwater before distribution.

Utilities in South Florida make limited use of surface water bodies as sources of drinking water. Nevertheless, migration of wastewater constituents to these sources of drinking water is a possibility; surface water bodies are potential drinking-water receptors.

Potential ecological receptors include surface water bodies and the biological communities they support. The state of Florida surface-water quality standards for Class I waters and known ecological dose-response thresholds serve as the assessment endpoints for assessing risks to potential ecological receptors.

5.5.7 Potential Exposure Pathways

When drinking-water or ecological receptors are exposed to wastewater constituents in sufficient concentration, these receptors may be at risk for potentially adverse health effects. The complex set of processes and interactions that govern how reclaimed water will move and behave in the subsurface define the pathways that may expose receptors to such concentrations.

Dissolved and entrained wastewater constituents move through soils and geologic media under the influence of physical, chemical, and biological processes. These processes govern the movement of water and the fate and transport of stressors present in the water. Pathways of reclaimed-water migration, and the processes that may modify its constituents, are dependent upon both the hydrogeologic system into which the reclaimed water has been recharged and the nature of the constituents themselves.

Conservative (nonreactive) constituents will move through the hydrogeologic system unaffected by chemical or biological processes. Concentrations of conservative constituents are diluted in groundwater through advection (groundwater flow) or diffusion. On the other hand, concentrations of wastewater constituents that are subject to chemical and biological transformation will be influenced by abiotic processes (that is, ion exchange, adsorption), by biological degradation or transformation, and by dilution in the subsurface.

The highly permeable limestone formations of the Biscayne Aquifer and the less permeable formations of the surficial aquifers provide pathways for migration of reclaimed-water and wastewater constituents. Groundwater transport of these constituents may result in migration from the point of recharge to a receptor well or surface water body.

Following recharge, inorganic and organic wastewater constituents that are not removed by the treatment process will be entrained in the effluent. As the effluent moves through the subsurface soil and rocks during advection, these constituents will be subject to a number of physical, chemical, and biological processes such as dilution, absorption, chemical transformation, volatilization, and other processes.

5.5.8 Conceptual Model of Potential Risks of Aquifer Recharge

A generic conceptual model for the aquifer recharge wastewater-management option is presented in Figure 5-1. The primary source of potential stressors is defined as the wastewater treatment plant from which reclaimed water is distributed to one or more rapid-rate land application systems.

Reclaimed water is discharged to RIBs located directly above surficial aquifers. RIBs are generally located tens of feet (not hundreds or thousands of feet) above the water tables receiving the recharge. Underlying surficial aquifers are typically USDWs of potable-water quality (less than 1,500 mg/L total dissolved solids content).

For aquifer recharge, the expected principal exposure pathway is migration of reclaimed water from the point of recharge by rapid-rate land application systems to the USDW. Groundwater may also carry reclaimed-water constituents to areas where groundwater discharges to surface water, potentially affecting ecological receptors.

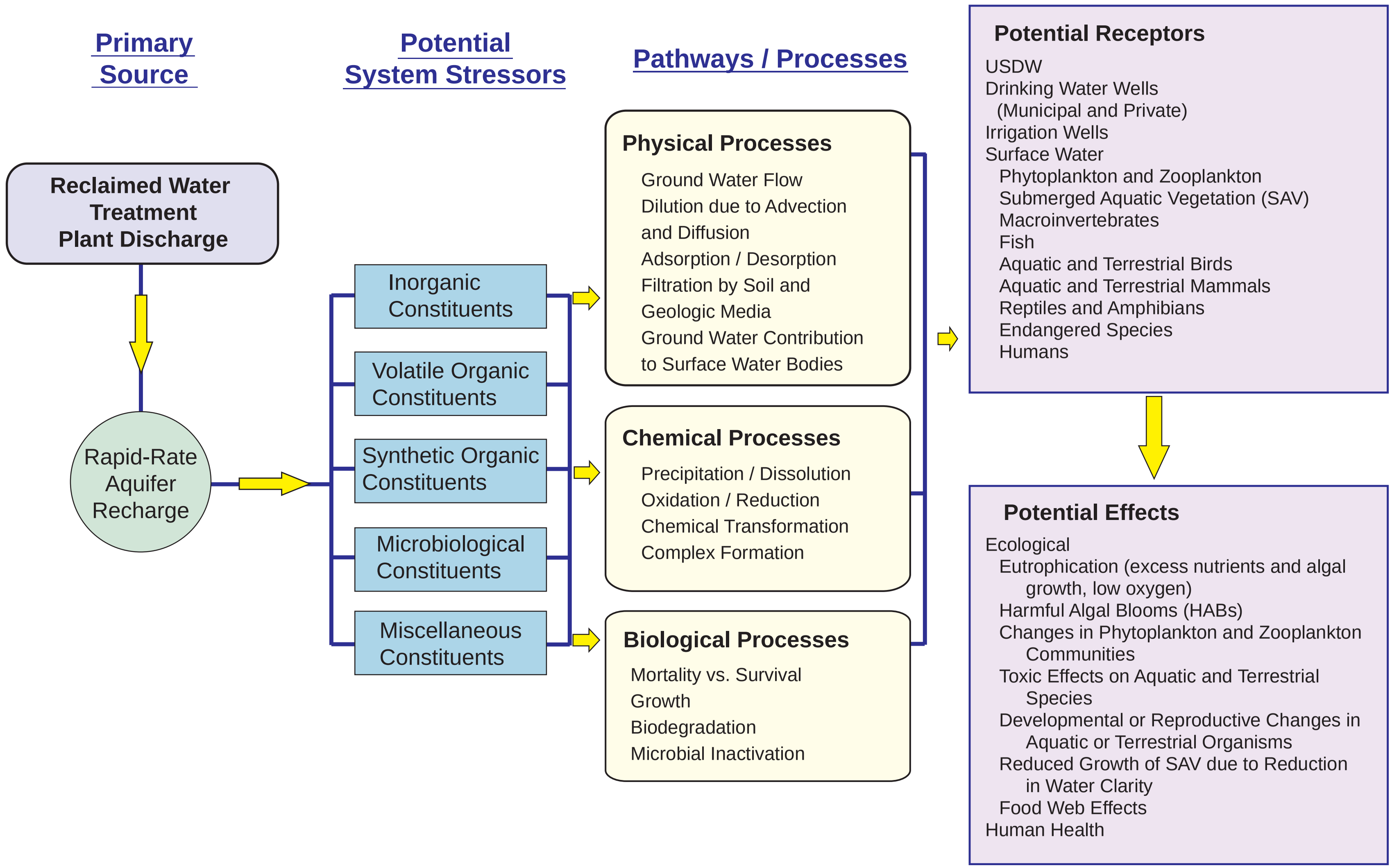

Figure 5-1. Conceptual Model of Potential Risks for the Aquifer Recharge Option

The dissolved and entrained constituents move through the geologic media under the influence of physical, chemical, and biological processes governing water movement and the fate and transport of the stressors in groundwater. The surficial aquifer may also act as a secondary source of dissolved and entrained stressors that may be carried to other parts of the aquifer where receptors may be exposed.

5.6 Risk Analysis of the Aquifer Recharge Option

In this section, information on stressors, receptors, and exposure pathways are used to examine potential risks and evaluate the conceptual model for aquifer recharge.

This analysis evaluates how reclaimed water may be transported horizontally within USDWs away from the point of recharge. Estimated times of travel are used to characterize the fate and transport of wastewater constituents (stressors) present in the reclaimed water. The fate and transport equations used in chapter 4 for evaluation of deep injection-well disposal are valid for aquifer recharge as well.

Information concerning potential stressors was obtained from effluent water quality monitoring reports required by the state of Florida and from a review of the scientific literature. To describe the proximity and vulnerability of receptors, publicly available information was obtained regarding the locations of public water-supply intakes. A review of the scientific literature provided information regarding the locations and physical extent of USDWs in South Florida. Information necessary to characterize possible exposure pathways was obtained from scientific literature describing the study area's soils, geology, and hydrology.

5.6.1 Vertical and Horizontal Times of Travel

Analyzing the transport of discharged effluent involves the analysis of the time of travel, which is the time needed for discharged effluent to move in groundwater over a specified distance to a drinking-water receptor. In aquifer recharge, typically the discharge location is directly above the surficial aquifer, and therefore the migration pathway will be downward and outward from the point of application. The potential for migration will be affected by site-specific factors, including the following:

- Required setback distances
- Locations of potential receptors (water-supply wells)
- Local direction of groundwater flow
- The distance to potential receptor wells
- Surficial aquifer characteristics that govern groundwater flow velocity.

Required setback distances vary depending on facility operations and range from 200, 500, and 2,640 feet. Engineering reports for new facilities must identify all potable water supplies within 0.5 mile of the facility.

Representative hydrogeologic parameters for Dade, Brevard, and Pinellas counties were used to estimate the potential groundwater flow velocity and associated time for groundwater to travel 200 feet, 500 feet, and 0.5 mile (2,640 feet) in the surficial aquifer (Adams, 1992; Barr, 1996; Lukasiewiez and Adams, 1996; Reese and Cunningham, 2000). Assumptions, calculations, and results are provided in appendix 8 and are summarized in table 5–3. Since local hydrogeologic conditions in the surficial aquifer may vary significantly, these travel times are intended only to provide representative values.

Table 5-3. Effluent Travel Times in the Surficial Aquifer

Surficial Aquifer Location	Horizontal Distance (ft)	Travel Time	
		Days	Years
Dade County: horizontal hydraulic conductivity: 1,524 ft/day	200	41	0.11
	500	102	0.28
	2,640	537	1.47
Brevard County: horizontal hydraulic conductivity: 56 ft/day	200	1,107	3.03
	500	2,768	7.58
	2,640	14,614	40.01
Pinellas County: horizontal hydraulic conductivity: 29 ft/day	200	2,138	5.85
	500	5,345	14.63
	2,640	28,221	77.26

Note: hydraulic gradient = 0.001; porosity = 0.32.

The results of these calculations (table 5–3) indicate that the shortest estimated travel times for effluent to travel 200, 500, and 2,640 feet are predicted for Dade County, where the Biscayne Aquifer has a high hydraulic conductivity. Horizontal travel time is significantly longer, by approximately 2 orders of magnitude, in Brevard County. Pinellas County has the longest horizontal travel times. These estimates are based on constant porosity and constant hydraulic gradient, but varying hydraulic conductivity from region to region. Again, site-specific conditions may differ substantially from the values used.

These results indicate that, solely in terms of transport of effluent, the highest risks for aquifer recharge may be found in Dade County, where the time of travel is the lowest, and the lowest risks for aquifer recharge may occur in Pinellas County, where the time of travel is the highest.

5.6.2 Evaluation of Stressors

Monitoring data indicates that concentrations of wastewater constituents in reclaimed water used in aquifer recharge generally meet drinking-water standards for reclaimed water. Also, treated effluent generally meets or is better than standards for reclaimed water or advanced wastewater treatment effluent (see Appendix Table 1-1).

Several representative chemical elements and compounds, potentially found in reclaimed water recharged via rapid-rate systems, were chosen for fate and transport analysis. The analysis is designed to estimate the final concentration of these wastewater constituents by taking into account calculated travel times in groundwater, biodegradation, hydrolysis, and sorption processes. These natural attenuation processes will reduce the overall concentration of chemicals during transport in groundwater.

Examples of natural attenuation processes include sorption, biological degradation, and chemical transformation. Compounds and elements dissolved in groundwater are removed from solution by sorption onto geologic material. Such sorption-desorption reactions result in a slowing of movement of the compound or element in groundwater. Sorption may be reversible, however. Biological activity by microorganisms may also result in the degradation of organic material and may also mediate transformations of inorganic materials, resulting in decreasing concentrations over time. Hydrolysis is another process whereby organic and inorganic solutes react with water, resulting in degradation and transformation. Rates of biological degradation and hydrolysis reactions may be expressed as a half-life for specific compounds (that is, the time it takes the concentration of the compound or element to decrease to one-half of its original concentration).

Selected representative stressors included arsenic (As), chloroform ($CHCl_3$) (representing trihalomethanes), nitrate (NO_3), and di (2-ethyl) phthalate (DEPH). Chloroform and several other similar compounds known as trihalomethanes may be present in reclaimed water as a result of the chlorination process. The fate and transport characteristics of chloroform were selected to represent the potential for migration of all trihalomethanes. DEPH, a synthetic organic compound used as a plasticizer for polyvinylchloride (PVC) and in consumer products, is a suspected endocrine disruptor (ASTDR, 1993).

Concentrations of representative compounds were based on typical values for reclaimed water (presented in Table 5-4); these were obtained from a large data set of monitoring results for treated effluent (see Appendix Table 1-1). The concentration of chloroform was used as a representative of total trihalomethanes, a group of compounds that includes chloroform. Chloroform was selected for the analysis based on the availability of fate and transport information. All initial stressor concentrations in the data sets available met drinking-water standards. The selected concentration for DEPH was the detection limit reported for wastewater analyses.

Table 5-4. Initial Concentration of Representative Stressors in Reclaimed Water

Compound	Initial Concentration
Arsenic	0.003 mg/L
Chloroform	26.85[1] (µg/L)
Di (2-ethylhexyl) Phthalate (DEPH)	5.0[2] (µg/L)
Nitrate	3.69 (mg/L)

[1]Concentration of total trihalomethanes.
[2]DEPH detection limit.

In addition to chemical stressors, the pathogenic protozoans *Cryptosporidium parvum* and *Giardia lamblia* were selected for evaluation of biological stressors that may be present in reclaimed water (York et al., 2002).

Florida's reuse rules have required monitoring for pathogenic protozoans since 1999. Results of monitoring through September 2001 were reviewed by York et al. (2002). Based on 48 observations, *Cryptosporidium* was detected in 23% of observations, with 8.3 % (3 observations) having more than 5 oocysts per 100 L. *Giardia* was detected in 58% of observations, with 46% of observations having more than 5 cysts per 100 L. Although there are no specific reclaimed water standards for pathogenic protozoans, the Florida DEP encourages improvements in the filtration process at facilities where greater than 5.8 *Cryptosporidium* oocysts or cysts per 100 L are detected or greater than 1.4 *Giardia* cysts are found per 100 L (York et al., 2002).

5.6.3 Evaluation of Receptors and Assessment Endpoints

Based on required treatment levels and review of data from wastewater treatment facilities utilizing aquifer recharge for wastewater management, representative concentrations of chemical stressors were selected. These stressor concentrations were used in fate and transport analyses based on travel distances of 200 feet, 500 feet, and 0.5 mile (2,640 feet), which were selected based on required setback distances and reporting requirements. The procedures described in section 4.3 for fate and transport of stressors in effluent injected to deep wells were applied to aquifer recharge. Referenced soil sorption coefficients and half-lives for representative stressors used in chapter 4 were used in this analysis to calculate attenuation of stressors during transport. Results of the fate and transport analysis are presented in Table 5-5.

Table 5-5. Contaminant Transport and Fate in the Surficial Aquifer

	Chloroform (μg/L)	Arsenic (mg/L)	Di(2-ethylhexyl) Phthalate (DEPH) (μg/L)	Nitrate (mg/L)
Dade County (effluent travels 200 feet in 0.11 years; 500 feet, 0.28 years; 2,640 feet in 1.47 years)				
Contaminant travel time	For 200 ft., 0 yrs. For 500 ft., 0 yrs. For 2,640 ft., 2 yrs.	For 200 ft., 0 yrs. For 500 ft., 0 yrs. For 2,640 ft., 2 yrs.	For 200 ft., 0 yrs. For 500 ft., 0 yrs. For 2,640 ft., 2 yrs.	N/A
Concentration at injection	7.18	0.01	5.00	N/A
Concentration at 200 feet	7.06	0.01	4.56	0.64
Concentration at 500 feet	6.88	0.01	3.97	0.64
Concentration at 2,640 feet	5.73	0.01	1.48	0.64
MCL	80 (as trihalomethane)	0.05	6	10
Brevard County (effluent travels 200 feet in 3.03 years; 500 feet, 7.58 years; 2,640 feet in 40.01 years)				
Contaminant travel time	For 200 ft., 3 yrs. For 500 ft., 8 yrs. For 2,640 ft., 43 yrs.	For 200 ft., 3 yrs. For 500 ft., 9 yrs. For 2,640 ft., 45 yrs.	For 200 ft., 4yrs. For 500 ft., 9 yrs. For 2,640 ft., 48 yrs.	N/A
Concentration at injection	230	0.005	5.00	9.60
Concentration at 200 feet	146	0.005	0.5	9.60
Concentration at 500 feet	73.7	0.005	0.0	9.60
Concentration at 2,640 feet	0.6	0.005	0.0	9.60
MCL	80 (as trihalomethane)	0.05	6	10
Pinellas County (effluent travels 200 feet in 5.85 years; 500 feet, 14.63 years; 2,640 feet in 77.26 years)				
Contaminant travel time	For 200 ft., 6.5 yrs. For 500 ft., 16.3 yrs. For 2,640 ft., 86.1 yrs.	For 200 ft., 7.12 yrs. For 500 ft., 17.80 yrs. For 2,640 ft., 93.97 yrs.	For 200 ft., 9.9yrs. For 500 ft., 19.8 yrs. For 2,640 ft., 104.6 yrs.	N/A
Concentration at injection	6.7	0.003	1.25	0.28
Concentration at 200 feet	2.68	0.003	0.01	0.28
Concentration at 500 feet	0.68	0.003	0.00	0.28
Concentration at 2,640 feet	0.00	0.003	0.00	0.28
MCL	80 (as trihalomethane)	0.05	6	10

Dilution and dispersion of stressors in groundwater were not considered in this analysis. These groundwater processes could result in lower concentrations at the 1,000-foot distance. Local hydrologic conditions may result in longer or shorter travel times.

The shortest estimated travel times for effluent to reach receptor wells in the surficial aquifer were in Dade County, where effluent travel times to reach wells at 200 feet, 500 feet, and 2,640 feet were 0.11, 0.28, and 1.47 years, respectively. Such short travel times pose relatively higher risks than longer travel times found elsewhere in South Florida. However, because concentrations of representative chemical stressors in discharged effluent were below their respective drinking-water MCLs, the final concentrations of representative stressors at the receptor wells were also below MCLs. Therefore the human health risks do not appear to be significant for these stressors and these travel times.

In Dade County, some stressors (for example, chloroform, DEPH) underwent further reduction as they traveled in the migrating effluent and decreased in concentration during their migration. However, the reduction amounts to less than a full order of magnitude reduction. Some other stressors (for example, arsenic, nitrate) did not undergo any decrease in concentration as they traveled through the shallow aquifer.

In Brevard County, estimated travel times for effluent in groundwater were intermediate in value. Effluent travel times to reach 200, 500, or 2,640 feet were 3.03 years, 7.58 years, and 40.01 years, respectively. For chloroform, effluent quality was elevated at injection (230 μg/L), but reduced to below the MCL at 500 feet. Like Dade County, final concentrations of all stressors, whether nonconservative or conservative, were below their MCLs. The modeled final concentration of one stressor, DEPH, fell to 0.00 at a distance of 500 feet, after an estimated travel time of 9 years. Again, like Dade County, the human health risks do not appear to be significant for these stressors and travel times.

The longest estimated travel times for effluent were found in Pinellas County. Estimated effluent travel times to reach 200, 500, and 2,640 feet were 5.85, 14.63, and 77.26 years, respectively. Initial concentrations of all stressors evaluated were below MCLs. The modeled final concentration of chloroform fell to 0.00 at a distance of 2,640 feet and a travel time of 86 years. The modeled final concentration of DEPH fell to 0.00 at a distance of 500 feet and a travel time of 19.8 years. Long travel times represent the lowest risk. Again, like Dade and Brevard counties, there do not appear to be any human health risks for the compounds and substances regulated.

Because reclaimed water treatment involves both basic disinfection and high-level disinfection using chlorine, which effectively inactivates most viruses and bacteria, reclaimed wastewater does not appear to pose any significant human health risk in terms of pathogenic bacteria or viruses (York et al., 2002).

However, pathogenic protozoans that are not inactivated by chlorine may pose concerns, particularly if reclaimed water is not filtered adequately. Pathogenic protozoans such as *Cryptosporidium parvum* and *Giardia lamblia* oocysts may be capable of surviving for

relatively long periods of time in groundwater and surface water, based on laboratory studies (There are very few in situ studies of oocyst inactivation). The most complete review of survival of *Cryptosporidum* is that by Walker et al. (1998). This review describes studies by Mawdsley et al. (1996a), who concluded that runoff contaminated with oocysts posed a more significant threat to water quality than infiltration through the soil profile, because of straining that tends to slow the transport of microorganisms (McDonald and Kay, 1981). For these reasons, the Florida DEP recommends that reclaimed wastewater should not contain more than 5.8 *Cryptosporidium* oocysts per 100 L or more than 1.4 *Giardia* cysts per 100 L (York et al., 2002). However, this is not yet a regulatory requirement.

Cryptosporidium and *Giardia* also occur in groundwater and surface water in South Florida (Rose et al., 2001; York et. al., 2002). The potential for aquifer recharge practices to remobilize *Cryptosporidium* or *Giardia* cysts derived from other sources cannot be evaluated in this study because of the lack of information concerning site-specific monitoring for pathogenic protozoans.

In summary, pathogenic protozoans that are not removed by chlorination pose the highest health risks associated with this wastewater management option. However, it should be pointed out that pathogenic protozoans are widespread in many natural surface water bodies and in groundwater, from a variety of sources (agricultural runoff, domestic animals, and, in particular, calves) (York et al., 2002; Walker et al., 1998). These concentrations in natural surface waters frequently exceed the amounts typically found in reclaimed water (see Table 5-6).

Other chemical constituents of treated reclaimed wastewater appear to generally meet or are lower than drinking-water standards.

Concentrations of nitrate and other nutrients that may remain in reclaimed water even after removal of nitrogen may pose ecological concerns, because most natural aquatic systems do not contain nitrate concentrations above the range from a few tenths of a ppm to several ppm

Table 5-6. Comparison of *Cryptosporidium* Concentrations in the Environment

Water Type (and Location)	Average (oocysts/100 L)	Range (oocysts/100 L)	Notes
Reclaimed water (St. Petersburg)[1]	0.75	ND – 5.35	12 samples
Phillippi Creek (FL)[2]	16	ND – 158	16 samples from urban stream in Sarasota
Five streams (FL)[2]	6.6	ND – 157	24 samples near Sarasota
Sarasota Bay (FL)[2]	ND	ND	4 samples from high-quality estuary
Tampa Bypass Canal (FL)[3]	3.1	ND – 11	7 samples
Filtered drinking water[4]	1.52	ND – 48	66 water-treatment plants in 14 states and 1 Canadian province (85 samples)
Treated drinking water[5]	3.3	ND – 57	1991–1993, 262 samples at 72 water plants
Surface-water supplies for drinking-water plants[5]	240	ND – 6,510	1991–1993, 262 samples at 72 water plants
Groundwaters[6]	41	__	74 samples
Springs[7]	4	__	7 samples
Lakes (pristine)[7]	9.3	ND – 307	34 samples
Rivers (pristine)[7]	29	ND – 24,000	59 samples
Surface waters (all categories)[7]	43	ND – 29,000	181 samples in 17 states
Irrigation canals (AZ)[8]	555,000	530,000–580,000	2 samples
Rivers in protected watershed[9]	2	ND – 13	6 samples, western United States

[1]Rose and Carnahan, 1992.
[2]Rose and Lipp, 1997.
[3]Rose, 1993.
[4]LeChevallier et al., 1991.
[5]LeChevallier and Norton, 1995.
[6]Rose, 1997.
[7]Rose et al., 1991.
[8]Madore et al., 1987.
[9]Rose, 1988.
ND = nondetectible
Source: Florida DEP, 1998.

5.7 Final Conceptual Model of Probable Risk

A final conceptual model of probable risk was developed as described below.

Aquifer recharge is broadly defined in this risk assessment as the replenishment or recharge of a groundwater aquifer through a variety of application methods, including rapid-rate land application, slow-rate land application, irrigation, and discharge to wetlands that are hydrologically connected to groundwater. The aquifers of concern in South Florida are the Biscayne and surficial aquifers, which are highly permeable and are susceptible to contamination from a large variety of point and nonpoint sources. In South Florida, the leading use of reclaimed wastewater is for irrigation of public-access areas (158.24 mgd), followed by industrial uses (59.1 mgd), groundwater recharge (31.72 mgd), irrigation of restricted access areas (27.5 mgd), and discharge to wetland systems (10.44 mgd).

Aquifer recharge using wastewater treated to reclaimed-water standards is called reuse in the state of Florida and is regulated under Florida's reuse regulations. Beneficial uses of reclaimed water includes aquifer recharge to restore or maintain aquifers, creation or restoration of wetlands that have been adversely affected by human activities, and creation of barriers to saltwater intrusion in coastal areas where withdrawal of fresh groundwater has exceeded natural recharge rates. Beneficial uses also include the use of reclaimed water for irrigation, which helps to conserve high-quality drinking-water resources.

Although ASR can be conducted with reclaimed water, most ASR being discussed in Florida involves the injection of high-quality water into aquifers for storage and later retrieval. Therefore, ASR is not considered in this risk assessment.

Reuse regulations require that reclaimed wastewater be treated with secondary treatment with basic disinfection if reclaimed water is intended for use in restricted-access locations. In public-access areas, slow-rate application systems must use wastewater treated to secondary levels with high-level disinfection, at a minimum. Nitrification, which helps to remove nitrogen from the wastewater, generally ensures that drinking-water standards for nitrogen are met. Disinfection with chlorine, particularly high-level disinfection, is highly effective at inactivating viruses and bacteria. Monitoring for fecal coliforms as an indicator of wastewater pathogens is required in treatment wetlands.

Filtration, which is required to reduce concentrations of total suspended solids, also reduces concentrations of pathogenic oocyst-forming protozoans, such as *Cryptosporidium parvum* and *Giardia lamblia*. Although there are no numerical water-quality standards regulating the concentrations of pathogenic protozoans in treated wastewater, the Florida DEP recommends that no more than 5.8 *Cryptosporidium* oocysts per 100 L and no more than 1.4 *Giardia* cysts per 100 L be allowed in reclaimed water. Filtration is the preferred method of removing pathogenic protozoans, although the DEP has found that filtration is not always effective (York et al., 2002).

Reuse regulations also require setbacks for aquifer recharge from public water-supply wells, surface-water supplies, and public-access areas. These setback distances vary, depending on the particular reuse option, from 75 feet to 500 feet or more. Such setbacks help to protect public water supplies from potential contaminants in surface-water runoff and in groundwater.

Figure 5-1 presents the generic conceptual model for the aquifer recharge wastewater management option. The primary source of potential stressors was defined as rapid-rate land application systems using reclaimed wastewater. In this conceptual model, reclaimed water is discharged to RIBs located directly above surficial aquifers. RIBs are generally located tens of feet (not hundreds or thousands of feet) above the water table. The principal exposure pathway in aquifer recharge was postulated to be migration of reclaimed water from the discharge point to the USDW. Groundwater may also carry reclaimed water constituents to areas where groundwater discharges to surface water, potentially affecting ecological receptors.

This option-specific risk assessment used an analysis of fate and transport of discharged reclaimed wastewater and representative chemical and microbiological constituents of wastewater, applied to rapid-rate land application. The fate-and-transport analysis was based on an analysis of the movement of discharged effluent in groundwater, estimation of the time of travel needed for effluent water to reach a drinking-water receptor such as a water supply well, and estimation of the fate of chemical constituents within the time of travel, using half-lives of chemical compounds and other characteristics. The approach used is the same as that used in chapter 4 for the fate-and-transport analysis of effluent discharged from Class I deep injection wells, except that the discharged effluent in aquifer recharge is moving down towards the aquifer rather than migrating upward towards the aquifer. Porous media flow is assumed for aquifer recharge.

The analysis of estimated travel times for rapid-rate land application indicated that Dade County may have the shortest travel times for effluent and hence the highest risk of contaminating the aquifer. These travel times ranged from 0.11 years to 0.28 years and 1.47 years for effluent to travel 200 feet, 500 feet, and 0.5 miles, respectively. However, the fact that reclaimed water is treated to relatively high standards, and because attenuation further reduces the concentrations of constituents along the path of travel, means that the actual risk to human health is most likely nonexistent to very low. The only possible exception is where filtration is not done or is ineffective at removing pathogenic protozoans, as described below).

In Brevard County, effluent travel times ranged from 3.03 years to 7.58 years to over 40 years for effluent to travel 200 feet, 500 feet, and 0.5 miles, respectively. As in Dade County, concentrations of chemical constituents in reclaimed water meet drinking-water standards before discharge. Concentrations of nonconservative constituents decrease further over this time period, while concentrations of conservative constituents remain the same over time. For these reasons, aquifer recharge using reclaimed water is not expected to pose significant human health risks in Brevard County, with the possible exception of pathogenic protozoans, as described below.

Pinellas County has the longest estimated effluent travel times and hence the lowest relative risk of the three areas evaluated. Estimated effluent travel were 5.85 years, 14.63 years, and 77.26 years to travel 200 feet, 500 feet, and 0.5 mile, respectively. Initial concentrations of all wastewater constituents were below their MCLs, and the final concentrations of conservative constituents remained the same. Concentrations of nonconservative constituents decreased even further over these time periods. Again, there do not appear to be any human health risks posed by the chemical constituents of reclaimed water.

Of all possible wastewater constituents remaining after treatment, oocyst-forming pathogenic protozans, such as *Giardia lamblia* and *Cryptosporidium parvum*, probably pose the greatest risks to human health, particularly if filtration is not effective at removing these oocyst-forming protozoans below DEP-recommended levels of 1.4 and 5.8 oocysts per 100 L, respectively. However, even if filtration is not this effective, the risks would be roughly comparable to ingesting untreated water from other natural surface-water sources that are considered pristine or relatively unimpacted by human activities or animal wastes.

Since reclaimed water may contain higher concentrations of nutrients than those found in ambient surface waters, there could potentially be ecological effects in nearby surface water bodies that receive reclaimed water. Chapter 7 provides a full discussion of water-quality criteria for unimpacted natural surface water bodies.

5.8 Potential Effects of Data Gaps

Because of the variable nature of geology and soils across the study area and the relative lack of site-specific information regarding groundwater flow and times of travel, actual conditions may differ from those expected. These differences may affect the risk assessment of the aquifer recharge methods in important ways. Data gaps occur in the groundwater information used for modeling fate and transport and in data on the water quality of discharged effluent and groundwater monitoring. Some of the potential effects of such data gaps are the following:

- Local variations in geologic and hydrologic conditions may result in differences in travel time from recharge locations to receptor wells and surface water bodies.
- Because of the lack of monitoring wells in the Biscayne Aquifer, there is no ability to predict or foresee potential adverse effects on public water supplies, whether risks arise from this wastewater management options or other activities.
- If hydrologic connections between groundwater and surface water bodies exist, then that provides another exposure or transport pathway whereby surface waters may be affected by aquifer recharge. The information reviewed in this study did not permit such detailed conclusions to be made, and this is an aspect of aquifer recharge that should be investigated on a site-specific basis. Site-specific monitoring of movement and water quality of groundwater and surface water should be used to determine whether there is a direct hydrologic connection

between the groundwater that receives discharged reclaimed water and surface water bodies or wetlands.

- The fate and transport of preexisting contaminants in groundwater and soils beneath the recharge site are unknown. There is a possibility that such preexisting contaminants may become remobilized by application of reclaimed water from above, but there is no specific monitoring information to indicate whether this might actually occur.

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6.0 OCEAN OUTFALLS

In this chapter, the potential ecological and human health risks associated with management of treated municipal wastewater via discharge to ocean outfalls are described and evaluated.

6.1 Definition of Ocean Outfalls

Management of treated municipal wastewater using ocean outfalls involves discharging treated wastewater directly to the ocean via outfall pipes. Wastewater receives secondary treatment, including basic disinfection with chlorine.

6.2 Capacity and Use in South Florida

South Florida has six publicly owned wastewater treatment facilities that discharge treated municipal wastewater to the ocean. These six facilities are the Miami-Dade Central District, Miami-Dade North District, City of Hollywood, Broward County, Boca Raton, and Delray Beach facilities (Figure 6-1). All six facilities discharge secondary-treated wastewater effluent into the western portion of the north-flowing Florida Current. Table 6-1 displays the distance from shore and the depth at which treated wastewater is discharged from these six facilities.

Table 6-1. Characteristics of Southeast Florida Ocean Outfalls

Parameter	Miami-Dade Central District	Miami-Dade North District	City of Hollywood	Broward County	Delray Beach	Boca Raton
Approximate volume discharged, million gallons per day (mgd)	133* (both Central and North)	100*	42*	66* - 80**	16.55**	13.66**
Discharge depth, meters (m)	28.2	29.0	28.5	32.5	29	27.3
Distance offshore (mi)	3.56	2.08	1.90	1.32	0.99	0.94
Number of ports	5	12	1	1	1	1
Diameter of ports (m)	1.22	0.61	1.52	1.37	0.76	0.91
Port orientation	Vertical	Horizontal	Horizontal	Horizontal	Horizontal	Up 45 degrees from horizontal

*Source: NOAA, 2002a
**Source: Marella, 1999
Source: Hazen and Sawyer, 1994.

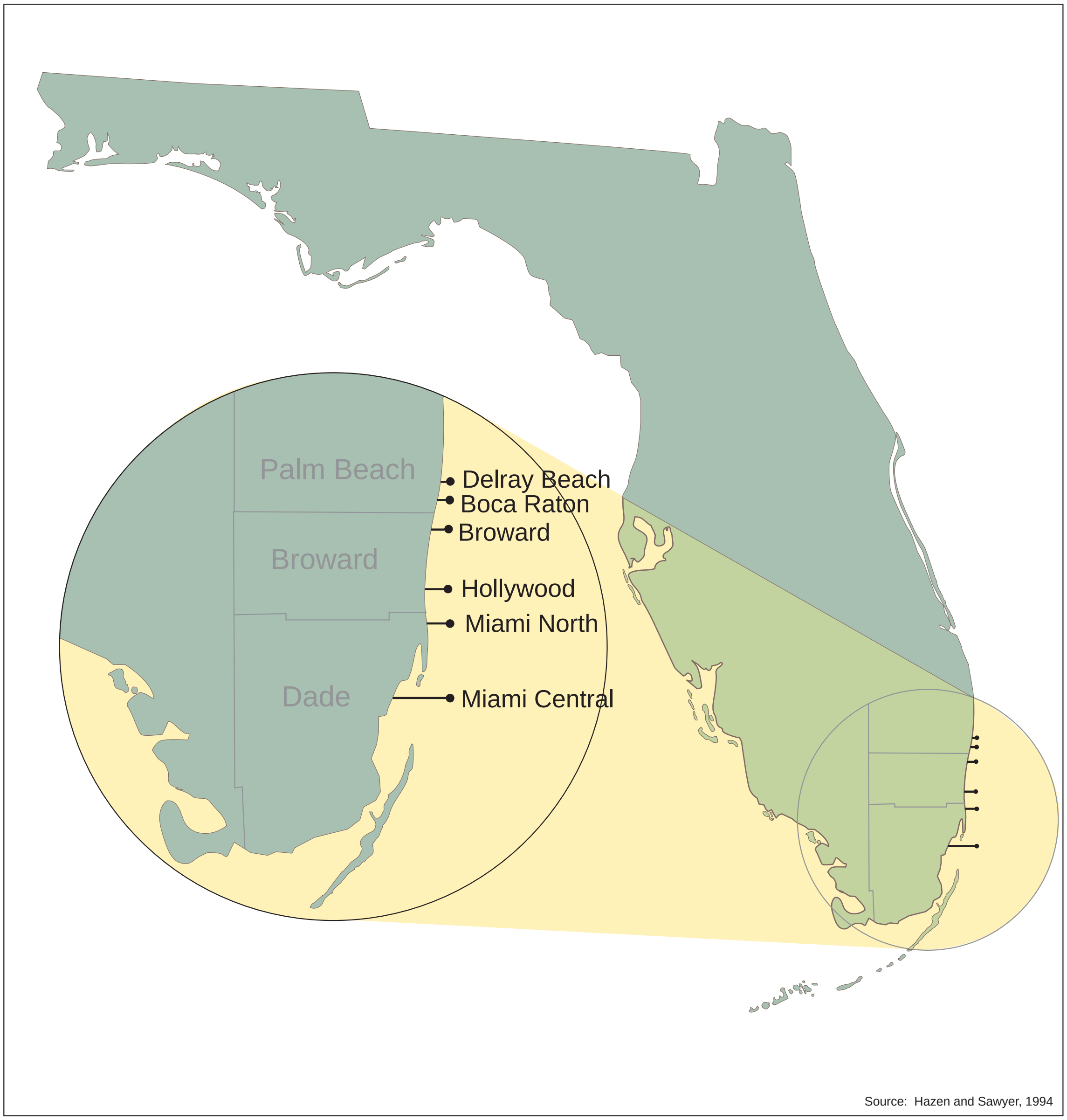


Figure 6-1. Locations of Ocean Outfalls in Southern Florida

The two outfalls with the highest flow rates (Miami-Dade North and Miami-Dade Central) have multiport diffusers, while the other four outfalls with lower flow rates outfalls discharge through single ports. The Miami-Dade Central Outfall discharges beyond the 3-mile state jurisdiction into federal waters. All six treatment facilities provide secondary treatment and basic disinfection, using chlorine.

The physical behavior of effluent plumes in the ocean is well understood, based on studies at a number of ocean outfalls worldwide (Wood et al., 1993). The physical behavior of the effluent plumes from the Florida ocean outfalls has also been extensively studied. When treated wastewater is discharged into the ocean from an outfall pipe, a plume of effluent is formed that tends to rise in seawater because the effluent is less saline and more buoyant than seawater. The speed and orientation of the ocean currents are the primary factors governing plume dispersion.

Figure 6-2 illustrates the behavior of an effluent plume discharging into the Florida Current. Water column stratification; determined by water inputs, precipitation, temperature, and advection caused by winds (Wood et al., 1993), may also play a role. For example, the thermocline, (a horizontal plane at which a distinct change in water temperature occurs) may present some barrier to mixing. Off the east coast of Florida, although the plume feature may remain relatively intact near the outfall pipe, the Florida Current rapidly disperses the effluent water and constituents, diluting it and mixing it with the surrounding water.

When evaluating the potential impacts of the southeast Florida ocean outfall discharges on the marine environment, South Florida wastewater utilities and regulatory agencies recognized that additional information was needed in order to develop conditions for outfall permitting. Understanding how discharged effluent undergoes dispersion, mixing, and dilution in the ocean is particularly important for risk assessment of ocean outfalls. While earlier studies of circulation and mixing provided critical knowledge concerning the large-scale behavior of the Florida Current, they did not provide the extensive amount of detail needed to thoroughly understand and predict effluent dispersion and dilution at all six of the outfall sites.

The Southeast Florida Outfall Experiment (SEFLOE) studies were initiated in the early 1990s. The SEFLOE studies were undertaken by the wastewater treatment facilities, working closely with the Ocean Acoustics Division of the Atlantic Oceanographic and Meteorological Laboratory of the National Oceanic and Atmospheric Administration (NOAA), the Florida Department of Environmental Protection (DEP), and the U.S. Environmental Protection Agency (EPA). These studies provide a significant amount of information concerning the mixing, dispersion, and dilution of wastewater plumes originating from these six ocean outfalls, the environmental characteristics of the outfall sites, and the chemical characteristics of both treated wastewater and receiving waters. This information was used to develop recommendations for the width of mixing zones that are required under state regulations. These mixing zones are necessary to allow discharged effluent to meet water-quality standards through dispersion and dilution.

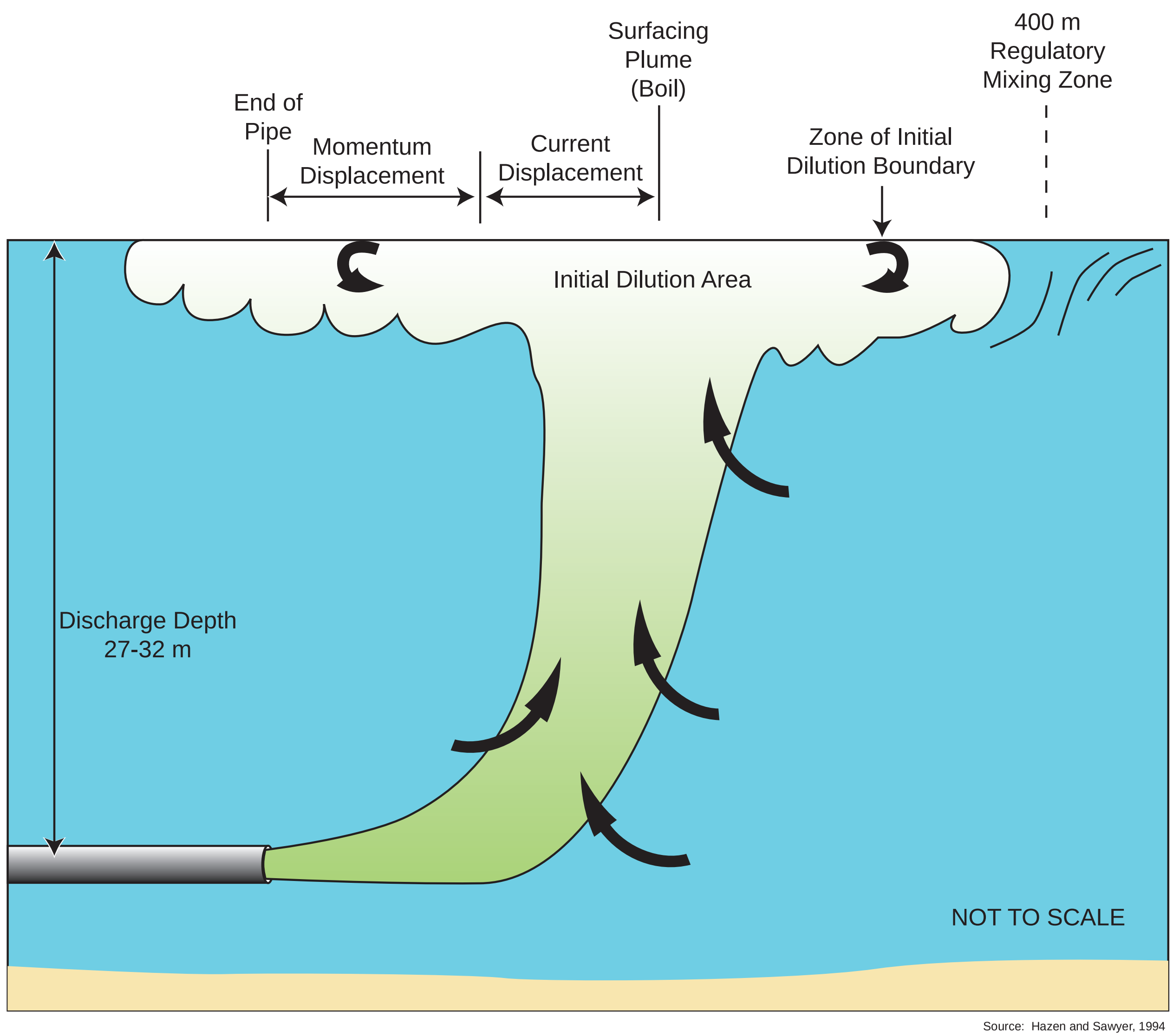


Figure 6-2. Effluent Plume Characteristics for Ocean Outfalls

The SEFLOE studies began with several physical oceanographic studies of effluent plume dispersion, mixing, and dilution. Effluent plumes were tracked and monitored using acoustical backscatter techniques, in one of the most extensive applications of acoustics to wastewater effluent studies in the United States (Proni, 2000; Proni and Williams, 1997; Proni et al., 1995; Williams and Proni, 1994; Proni and Dammann, 1989). Mixing zones for the southeast Florida outfall plumes were modeled using three different models that incorporated field data: CORMIX, PLUMES, and OMZA. All three models predicted realistic initial dilutions for outfalls with only minor exceptions (Huang et al., 1998). The results of these studies were used to develop wastewater treatment recommendations aimed at meeting water-quality standards within a 400-m-radius mixing zone.

Biotoxicity testing of secondary-treated wastewater and diluted effluent were conducted as well (Commons et al., 1994a). Many of these studies are summarized in the comprehensive report assembled by Hazen and Sawyer (1994). According to these studies, toxicity testing on marine organisms indicated that diluted effluent did not cause toxic effects in marine test organisms.

The initial SEFLOE I study focused on characterizing initial and farfield dilution properties of the ocean outfall plumes using acoustical backscatter techniques, determining the nutrient and bacterial content of the effluent and receiving waters, characterizing marine conditions, and evaluating concerns about nondegradable substances in the discharged treated effluent.

The SEFLOE II study continued to improve understanding of year-round physical oceanographic conditions at four of the outfalls, defining rapid dilution and mixing zones through modeling of near-field and farfield conditions. SEFLOE II also continued monitoring of nutrient concentrations in the effluent plumes. The SEFLOE II study examined the toxic characteristics of the receiving water/effluent mixture with and without chlorination, using bioassay techniques. Finally, the study examined whether the diluted wastewater met water-quality standards for priority pollutants, bacteria, and oil and grease.

6.3 Environment into Which Treated Wastewater is Discharged

Two major current systems dominate marine circulation along the western and eastern coasts of South Florida: the Loop Current, which flows out of the Gulf of Mexico in a southeasterly direction, passing the Dry Tortugas, and the Florida Current, which is the extension of the Loop Current as it flows east towards the Florida Keys and then north along the east coast of South Florida, until it joins the northward-flowing Gulf Stream (Lee et al., 1995). Smaller countercurrents, flowing west from the Florida Keys and Florida Bay, and southerly currents from the southwest Florida shelf meet the Loop Current in the area near the Dry Tortugas to form the Tortugas Gyre (Lee et al., 1995), another major eddy system. The Pourtales Gyre exists to the east of the Tortugas Gyre.

Understanding the movements of the Florida Current, particularly in its northern reaches off the east coast of Florida, is important for this risk analysis because the six ocean outfalls located in southeast Florida discharge treated wastewater effluent to the Florida Current. The Florida Current is made up in roughly equal parts of waters originating in the south Atlantic and north Atlantic subtropical gyres, connecting the Loop Current's flow out of the eastern Gulf of Mexico with the north Atlantic or Gulf Stream (Schmitz and Richardson, 1991; Lee et al., 1995). In the southern Straits of Florida, the presence of at least two gyre systems and variations in the flow of the Loop Current can cause the Florida Current to meander before it turns northward in the Santaren Channel (Lee et al., 1995).

As the Florida Current travels northward off the east coast of Florida, spin-off eddies are created (Lee, 1975; Lee et al., 1995). These eddies include several components, including northerly flows associated with western meanders of the Florida Current, southerly flows, and rotary flows, composed of groups of rotations interspersed between northerly and southerly flows. Rotary flow involves water flows that move in a roughly circular manner, much as a whirlpool does. As the Florida Current moves north to join the Gulf Stream, these rotary flows also move, or are translated, in a northerly direction. These eddy and rotary flow systems were studied extensively during SEFLOE. Figure 6-3, from Hazen and Sawyer (1994), depicts the three different current regimes and their circulation characteristics, as the current moves or translates from time t_1 to a later time t_2.

The eddies and rotary flows occurring along the western boundary of the Florida Current impart a variability to the circulation system that is important for understanding potential ecological or human health risks that may be associated with ocean outfalls in this area. The variability of the Florida Current's western boundary is important because the Florida Current represents a major source of nutrients for primary productivity in the area. Incursions of the Florida Current onto the continental shelf are reflected in enhanced phytoplankton and zooplankton growth from Cape Canaveral to Cape Hatteras (Atkinson, 1985). Shorter incursions of Florida Current water onto the continental shelf, lasting days to weeks, have been recorded from Miami to Pompano (Lee, 1975; Lee and Mayer, 1977).

6.4 Regulations and Requirements Concerning Ocean Outfalls

6.4.1 General Requirements

Ocean outfalls in South Florida are required to provide secondary treatment of municipal wastewater and disinfection with the minimal amount of chlorine necessary to achieve water-quality standards. Overchlorination of wastewater containing organic materials can result in creation of organochlorine compounds such as trihalomethanes, which are associated with human health risks.

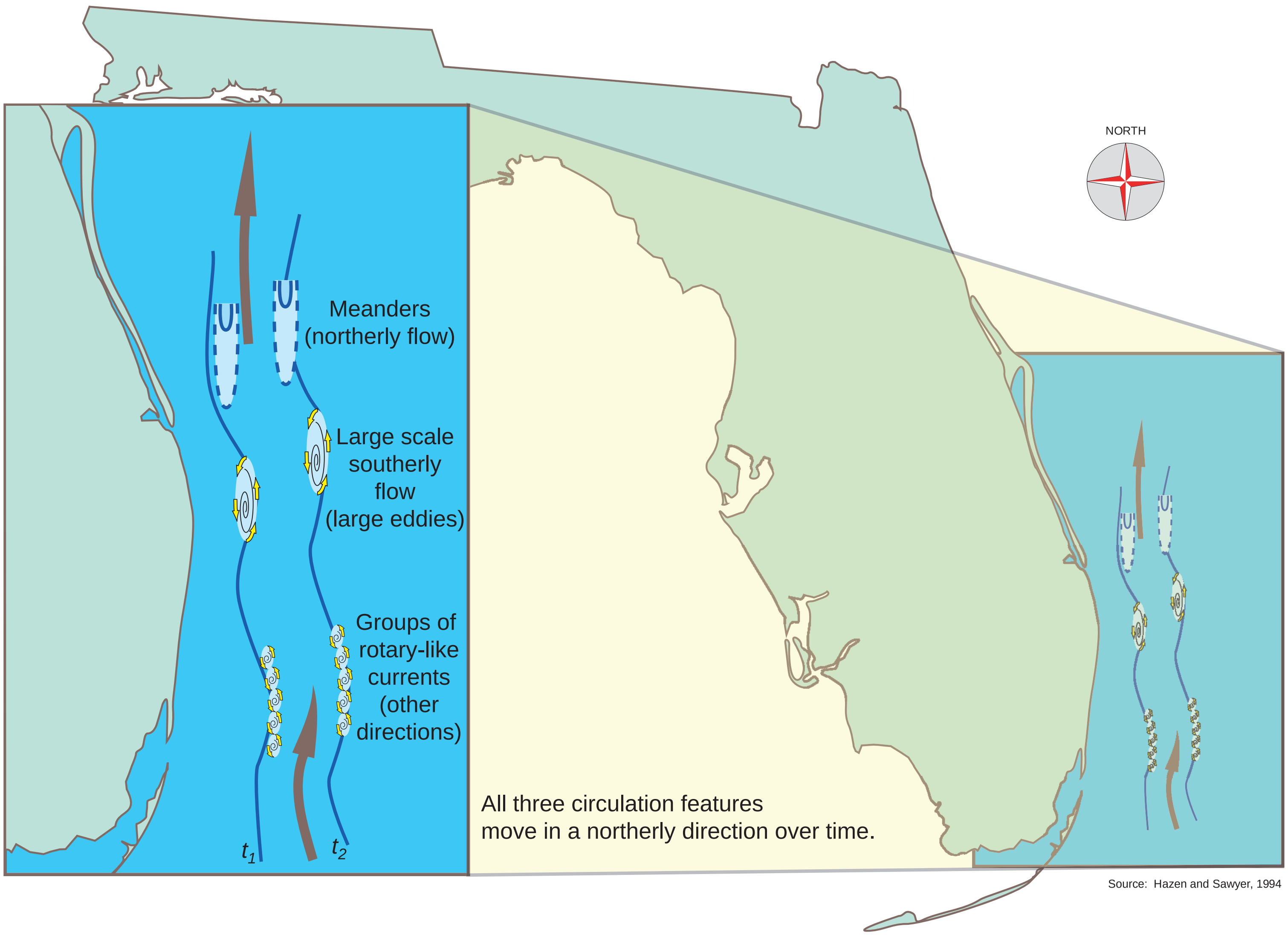


Figure 6-3. Circulation Characteristics of the Western Boundary Region of the Florida Current.

The federal Clean Water Act (33 USC 1251 et seq.) prohibits discharge of any waste to any waters of a state unless the waste is first treated to protect the beneficial uses of such water (see also Florida Administrative Code (FAC) 62-650). At a minimum, sewage treatment plants discharging to the ocean or other surface waters must provide secondary treatment in order to meet this pollution reduction standard.

The Florida Air and Water Pollution Control Act (Title 19, Chapter 403, Part I, Florida Statutes) also prohibits discharge of any untreated wastes to any waters of the state (FAC 62-650). In the state of Florida, waters used for recreation, propagation, and maintenance of a healthy, well-balanced population of fish and wildlife are classified as Class III Waters (FAC 62-302.400(1)). In such waters, state regulations require that, prior to discharge and after disinfection, wastewater effluent meet the most stringent of the following two standards: either (1) effluent must not exceed 20 milligrams per liter (mg/L) $CBOD_5$ and 20 mg/L of total suspended solids (TSS), or (2) 90% of $CBOD_5$ and TSS must be removed from the wastewater influent (FAC 62-600.420(1)(a) and 62-600.420(b)(1)). All wastewater treatment facilities, whether new or existing, must achieve at a minimum the specified effluent limitations (20 mg/L) and must also maintain safe pH and disinfect (FAC 62-600.420(b)(2)). The Florida DEP has also established technology-based effluent limits (TBELs), which include requirements for secondary treatment, pH levels, and disinfection.

6.4.2 Secondary Treatment of Wastewater

Secondary treatment for the state of Florida removes biodegradable organic matter and suspended solids and includes basic disinfection. Secondary treatment plants are designed to produce effluents that contain no more than 30 mg/L $CBOD_5$ and 30 mg/L TSS. The plants must also remove 85% of $CBOD_5$ and TSS from wastewater. State regulations require that, after basic disinfection, secondary-treated wastewater cannot exceed 20 mg/L of $CBOD_5$ and 20 mg/L of TSS or that 90% of $CBOD_5$ and TSS must be removed from the wastewater influent, whichever is more stringent (FAC 62-600.420(1)(a)). The effluent pH, after disinfection, must be within the range of 6.0 to 8.5 (FAC 62-600.420).

6.4.3 Basic Disinfection

Basic disinfection of wastewater must result in effluent with not more than 200 fecal coliforms per 100 milliliters (mL), at a minimum (FAC 62-600.445, 62-600.520(2), 62-600.420). When chlorine is used as the disinfection agent, the facility must provide for rapid and uniform mixing, with a total chlorine residual of at least 0.5 mg/L after at least 15 minutes contact time at the peak hourly flow (FAC 62-600.440(4)). In addition, wastewater must be disinfected so as to achieve Class III microbiological standards at the edge of the mixing zone or the level of disinfection deemed appropriate (FAC 62-600.520(2) and (3)). If the discharge is to Class III coastal waters, the disinfected effluent cannot contain more than 20 mg/L $CBOD_5$ and 20 mg/L TSS, or 90% of these pollutants must be removed from the wastewater, whichever is more stringent. In addition to these standards, bioassay toxicity tests must be conducted to ensure that aquatic organisms do not experience toxic effects from the effluent.

6.4.4 Water Quality Standards for Receiving Waters

Section 403(c) of the Clean Water Act, Ocean Discharge Criteria, applies to point-source discharges to ocean waters. Point-source discharges to ocean waters must not cause unreasonable degradation of the marine environment. Standards for receiving waters are generally more stringent than end-of-pipe limits, and thus there are regulations that pertain to the water quality of the discharge at the end of the pipe, within the mixing zone, and at the edge of the mixing zone. The Florida DEP has also established water-quality-based effluent limits to carry out the goals of the Florida statute. These limits are applied when additional treatment is necessary to ensure that the available assimilative capacity of a water body will be protected (FAC 62-650.)

Within the mixing zone, the EPA addresses acute toxicity by establishing criteria for the maximum concentrations (CMC). The CMC is approximately one-half of the acute concentration of the parameter of interest for the most sensitive species. A facility can meet these criteria by any one of the four following methods:

- Demonstrate that the CMC level is not exceeded at the end-of-pipe
- Provide rapid mixing with a high-velocity discharge so that the CMC is met a short distance from the outfall
- Meet the CMC within 10% of the distance to the edge of the mixing zone or 5 times the concentration of the parameter in local waters (Florida DEP)
- Demonstrate that a drifting organism is not exposed to average concentrations exceeding the CMC for a 1-hour time interval.

The federal, state, and local regulations require compliance with surface-water quality standards at the edge of the mixing zone. A mixing zone range is the distance needed for the effluent plume to become sufficiently diluted. The dilution occurs when the effluent plume mixes with ambient seawater to the point where the concentration of indicator bacteria reaches Class III water quality standards. The FAC allows a maximum mixing zone area of up to 502,655 square meters (m^2) for open-ocean outfalls (FAC 62-4.244(1)(h)). Water quality must meet Class III microbiological standards at the edge of the mixing zone, or the level of disinfection deemed appropriate, as described in Table 6-2 (see FAC 62-4.244 regarding mixing zones and see 62-600.520(2)). Although the mixing radius need not be circular in shape, the area required is equivalent to that of a 400-m-radius circle, which can be more easily visualized and incorporated into a conceptual model. The actual mixing zone will never be exactly circular.

Table 6-2. Federal and Florida Class III Water Quality Criteria and Guidance Values for Indicator Bacteria Groups

Group	Monthly Geometric Mean (colonies per 100 mL)	Percent	Maximum Single Value (colonies per 100 mL)
Fecal coliform	200	not more than 10% over 400	≤800
Total coliform	1,000	not more than 20% over 1,000	≤2,400
*Enterococcus**	35	not more than 10% over 70	≤140

*Guidance values
Source: Hazen and Sawyer, 1994.

6.5 Problem Formulation

In this section, general information concerning potential stressors, receptors, and exposure pathways is used to develop a conceptual model that depicts potential risk that may be associated with ocean outfalls. Section 6.6 presents an evaluation of actual risk.

6.5.1 Potential Stressors

Potential ecological stressors that may be present in secondary-treated wastewater include the following:

- Nutrients (nitrogen, phosphorus, iron) that could promote primary productivity and growth of harmful algal blooms
- Metals
- Volatile organic compounds
- Synthetic organic compounds (for example, organochlorine compounds such as trihalomethanes and chlorinated hydrocarbons)
- Other substances suspected of causing adverse effects on aquatic organisms (for example, endocrine-disrupting compounds)
- Substances whose ecological and biological effects are not yet well studied (for example, detergents, surfactants).

Potential human health stressors include the following:

- Pathogenic enteric microorganisms (bacteria, viruses, and protozoans) capable of surviving basic disinfection
- Metals
- Organic compounds
- Endocrine-disrupting compounds
- Nutrients such as nitrate and nitrite that can cause human health effects at higher concentrations.

Basic disinfection will deactivate most of the viruses and pathogens (see treatment requirements, above), but will not deactivate protozoans such as *Cryptosporidium* or *Giardia*, which must be filtered out.

6.5.1.1 Nutrients and Eutrophication

Nutrients act as potential stressors when they stimulate primary production that results in eutrophication. In coastal waters such as those of southeast Florida, as in large areas of the world's oceans, coastal, and estuarine waters, primary production is usually limited by nitrogen (Dugdale, 1967; Ryther and Dunstan, 1971; Codispoti, 1989; Paerl, 1997). However, phosphorus can be limiting under some conditions, particularly in coastal waters where there may be varying salinities. On geologic time scales, phosphorus is believed to limit marine productivity (Howarth, 1988; Holland, 1978; Smith, 1984; Codispoti, 1989; Ruttenberg, 1993). Some marine cyanobacteria, Sargasso Sea phytoplankton, and some Caribbean macroalgae are phosphorus-limited (LaPointe, 1997; Sellner, 1997; Cotner et al., 1997).

A recent National Academy review of the causes of eutrophication of coastal waters found that nutrient overenrichment of coastal marine waters have resulted in the following adverse effects (National Research Council, 2000):

- Increased primary productivity
- Increased oxygen demand and hypoxia
- Shifts in community structure caused by anoxia and hypoxia
- Changes in phytoplankton community structure
- Harmful algal blooms
- Degradation of seagrass and algal beds and formation of nuisance algal mats
- Coral reef destruction.

The National Research Council review concluded that, while nitrogen is important in controlling primary production in coastal waters and phosphorus is important in fresh water systems, both need to be managed to avoid one or the other becoming the limiting nutrient (National Research Council, 2000). The differences in causes of eutrophication between fresh and marine ecosystems stem from a variety of ecological and biogeochemical factors, including the relative inputs of nitrogen versus phosphorus within the ecosystem and the extent to which nitrogen fixation can alleviate nitrogen shortages. In addition, eutrophication of coastal systems is often accompanied by decreased silica availability and increased iron availability, both of which may promote the formation of harmful algal blooms (National Research Council, 2000).

There are exceptions to the general principle that nitrogen is limiting in coastal ecosystems. For instance, the Apalachicola estuarine system on the Gulf coast of Florida appears to be phosphorus-limited (Myers and Iverson, 1981). Howarth (1988) and Billen et al. (1991) postulate that this is related to the relatively high ratio of nitrogen to phosphorus inputs. However, in this case, the ratio may also reflect the relatively small amount of human disturbance in the watershed and the relatively low nutrient inputs

overall. Howarth et al. (1995) suggests that there is a tendency for estuaries to become more nitrogen-limited as they become more affected by humans and as nutrient inputs increase overall. This is because productivity is a function of the availability of nutrients to phytoplankton.

In nearshore tropical marine systems, phosphorus appears to be more limiting for primary production (Howarth et al., 1995), while the tropical open ocean is nitrogen-limited (Corredor et al., 1999). Nutrient limitation switches seasonally between nitrogen and phosphorus in some major estuaries such as the Chesapeake Bay (Malone et al., 1996) and in portions of the Gulf of Mexico, including the so-called “dead zone” (Rabalais et al., 1999).

There are approximately 300 species of algae known to produce “red tides,” including flagellates, dinoflagellates, diatoms, silicoflagellates, prymnesiopytes, and raphidophytes. Of these 300 species, approximately 60 to 80 species are actually harmful or toxic as a result of their biotoxins, nutritional unsuitability, and ability to cause physical damage or anoxia, reduce irradiance, and so forth. (Smayda, 1997). In Florida, problematic harmful algae bloom (HAB) species include *Pfiesteria* species, *Cryptoperidiniopsoid*, *Alexandrium monilatum*, *Chattonella subsalsa*, *Dinophysis* spp., *Gambierdiscus toxicus*, *Gymnodinium pulchellum*, *Gyrodinium galatheanum*, *Gymnodinium breve*, *Karenia brevis* (said to be the most common cause of red tide on the Florida coast), *Karenia mikimotoi*, and the benthic genus *Prorocentrum* spp. The Gulf coast of Florida has been typically more affected by HABs, particularly of *Gymnodinium breve*, often during the summer and fall when seasonal changes in the wind and sea surface temperature occur (FFWCC, 2001).

Toxic symptoms of HABs can affect both humans and animals and include paralytic shellfish poisoning (PSP), diarrheic shellfish poisoning (DSP), amnesic shellfish poisoning (ASP), ciguatera fish poisoning (CFP), and neurotoxic shellfish poisoning (NSP). The effects range from discomfort to incapacitation to mortality (FFWCC, 2002a).

Environmental changes that may stimulate HABs include a variety of physical, chemical, and biological factors, such as climate change, increased pollution and nutrient inputs, habitat degradation through dredging, resource harvesting and regulation of water flows, and the failure of grazing organisms to control algal growth. The two primary algal groups that produce blooms in response to nutrient inputs are the cyanobacteria and macroalgae, as well as other species from different groups (NOAA, 2002b). Even nontoxic HABs can disrupt other organisms through biofouling, clogging of gills, or smothering of coral reefs and seagrass beds in South Florida (LaPointe, 1997).

HABs can also be caused by marine cyanobacteria, commonly called blue-green algae. Marine cyanobacterial species responsible for HABs include only a few taxa, such as *Trichodesmium*, *Richelia*, *Nodularia*, and *Aphanizomenon*. *Trichodesmium*, which is nitrogen-fixing, is found in low- and mid-latitude oceans and seas of the Atlantic, Pacific, and Indian oceans. Marine cyanobacterial blooms can occur in warm stratified areas in

the ocean and in embayments and estuaries where nitrogen concentrations are often low, salinities are reduced, and where phosphorus becomes enriched through upwelling, eddies, mixing, or other sources. Phosphorus limitation appears to be more important than nitrogen limitation, since some of these species are nitrogen-fixing and inhabit nitrogen-poor waters (Sellner, 1997).

Human and animal health can be affected by ingestion of the toxins created by cyanobacteria such as *Trichodesmium*, *Nodularia* and *Aphanizomenon*, as documented by livestock, canine, and human cases (Sellner, 1997; Nehring, 1993; Edler et al., 1985). Other adverse effects of *Trichodesmium* blooms include mortality of mice, brine shrimp, and copepods; asphyxiation of fish, crabs, and bivalves; retreat of zooplankton to deeper waters free of the algae; and food-chain effects (reviewed in Sellner, 1997).

In Florida, extensive blooms of cyanobacteria, involving the cyanobacteria *Lyngbya majuscula*, a species that occurs worldwide, were documented in Tampa Bay in 1999 and from Sarasota Bay to Tampa Bay in 2000. Although this species is not toxic, it can produce large slimy brown floating mats and emit a foul odor (FFWCC, 1999). The causes of these blooms are unknown, although they are not believed to be related to sewage releases.

6.5.1.2 Pathogenic Microorganisms

Potential microbial stressors in treated wastewater include pathogenic enteric bacteria, protozoans, and viruses associated with human or animal wastes. Untreated raw sewage typically contains fecal indicator bacteria (such as fecal coliforms, total coliforms, and fecal streptococci) in concentrations ranging from several colonies to tens of millions of colonies per 100 mL (see Table 6-3). Other pathogens that are potentially present include other bacteria (*Campylobacteria jejuni*, *Legionella pneumophila*, *Salmonella typhi*, *Shigella*, or *Vibrio cholerae*), helminthes (such as hookworm, roundworm, or tapeworm), viruses (adenovirus, enteroviruses, hepatitis A, rotavirus, Norwalk agent, parvovirus, and others), and protozoa (*Cryptosporidium parvum*, *Giardia lamblia*, *Balantidium coli*, *Entamoeba histolytica*) (York et al., 2002).

Table 6-3. Typical Concentrations of Fecal Indicator Bacteria in Raw Untreated Sewage

Wastewater Source	Total Coliforms (colonies per 100 mL)	Fecal Coliforms (colonies per 100 mL)	Fecal Streptococci (colonies per 100 mL)
Raw sewage	22×10^6	8×10^6	1.6×10^6

Source: Wood et al., 1993, based on data from Geldreich, 1978, for communities in the United States.

For comparison, basic disinfection of secondary-treated wastewater must achieve the microbial standards of 200 and 2,000 colonies per 100 mL of wastewater for fecal coliforms and total coliforms, respectively, depending on the type of bacteria involved. Disinfection to these levels represents reductions of 10^4 or more.

Although secondary-treated wastewater destined for ocean outfalls is treated with chlorination, the minimal amount of chlorination needed to meet Class III water quality standards after dilution is generally used, in order to avoid the adverse effects of overchlorination. Pathogenic microorganisms that are not affected by secondary treatment or chlorination include the protozoans *Giardia* and *Cryptosporidium*, which are resistant because they form cysts that can remain dormant for periods of time and can be removed only through filtration. Filtration followed by disinfection is effective at removing viruses, while secondary treatment and chlorination is effective at removing helminthes (Rose and Carnahan, 1992).

Microbial contamination from enteric viruses, bacteria, and protozoans is a chronic problem in the Tampa Bay, Sarasota Bay, and Florida Keys coastal environments. This is probably because of high concentrations of onsite sewage disposal systems, porous sandy karst soils, and hydrologic connections between groundwater and coastal embayments and estuaries (Lipp et al., 2001; Paul et al., 1995). Survival of microorganisms in water is affected by a number of physical and biological factors, such as ultraviolet radiation and predation by grazers (Wood et al., 1993). Field measurements around the world provide a range of values of the time needed for reduction of enteric bacterial populations in seawater to 90 percent of their original concentrations (that is, t_{90}). These values for t_{90} range from 0.6 to 24 hours in daylight to 60 to 100 hours at night (reviewed in Wood et al., 1993). Enteric viruses tend to survive longer in seawater than do enteric bacteria: at 20 ºC, if the t_{90} for bacteria was 0.6 to 8 hours, the t_{90} for enteric viruses was 16 to 24 hours (Feacham et al., 1983). Fecal streptococci tend to be more persistent than fecal coliforms in seawater (Wood et al., 1993).

The initial SEFLOE experiments involved the monitoring of plumes of unchlorinated treated effluent in the ocean to determine how dilution and natural attenuation processes would affect microbial concentrations of fecal coliforms, total coliforms, and enterococci. To provide guidance on the level of chlorination needed, these data were then used to calculate what the maximum bacterial concentrations in chlorinated effluent should be to achieve a given dilution at a given distance from the outfall. Southeast Florida

wastewater treatment plants routinely provide secondary treatment and chlorination of wastewater to meet these standards (Hazen and Sawyer, 1994).

Because secondary effluent discharged through ocean outfalls is not filtered to remove protozoans such as *Giardia* or *Cryptosporidium*, these protozoans may pose potential human health risks that need to be evaluated.

6.5.1.3 Priority Pollutant Metals

Metals found in wastewater may constitute potential stressors because of potential human health risks and ecological risks. Metals are normally present in trace amounts in seawater (Bruland, 1984) and in higher amounts in sediments (Holland, 1978), but their concentrations are commonly elevated in wastewater because of the many anthropogenic uses of metals. As a consequence, metals are frequently used as tracers of wastewater in the ocean (Matthai and Birch, 2000; Flegal et al., 1995; Hershelman et al., 1981; Ravizza and Bothner, 1996; Morel et al., 1975). Marine disposal of untreated sewage or sewage sludge typically results in elevated concentrations of metals (typically chromium, copper, nickel, lead, silver, zinc, and iron) and other contaminants on the seafloor (Zdanowicz et al., 1991; Zdanowicz et al., 1995). Other sources of anthropogenic and natural metals to the ocean include stormwater runoff, inputs from surface water (rivers, streams) and groundwater, and atmospheric dust (Burnett and Schaeffer, 1980; Finney and Huh, 1989; Forstner and Wittman, 1979; Huh et al., 1992; Huntzicker et al., 1975; Klein and Goldberg, 1970).

Information on metal concentrations in marine organisms from this area includes the Mussel Watch Program, which is part of NOAA’s National Status and Trends Program (NSTP). The NSTP found elevated concentrations of arsenic in oysters extending from the Florida panhandle in the eastern Gulf of Mexico, South Florida (Biscayne Bay and Miami River), and up the east coast of Florida to North Carolina. Potential sources of arsenic include both natural sources (phosphorite rocks) and anthropogenic sources (for example, anthropogenic inputs to Biscayne Bay from pesticides in agricultural runoff and phosphate mining). Oysters are a food source for humans, birds, and other organisms, thus there is a potential for secondary uptake of arsenic (Valette-Silver et al., 1999). Because oysters are typically found in nearshore environments and not in deeper shelf waters, it is probable that the arsenic found in these studies originates from more nearshore or terrestrial sources, whether anthropogenic or natural. This information indicates, however, that such bioaccumulation is common and needs to be taken into consideration when examining the potential effects of secondary effluent discharge into the ocean.

6.5.1.4 Organic Compounds

Potential organic stressors that may be present in secondary-treated wastewater include EPA priority pollutant organic compounds, including volatile organic compounds (VOCs), synthetic organic compounds (pesticides, herbicides), organochlorine

compounds such as trihalomethanes, and a variety of other unregulated compounds, such as endocrine disruptors, surfactants, and organic matter.

6.5.2 Potential Receptors

Potential receptors of ocean outfall effluent constituents include any organism that may be exposed to seawater containing effluent constituents. Because seawater is not used for drinking water (unless it is treated through desalination), potential receptors mainly considered in this risk assessment are those that may be *directly* exposed to seawater containing effluent constituents. Such potential receptors in the South Florida marine environment include a wide variety of animals and plants living in or near brackish coastal waters or marine waters, including marine mammals, reptiles, fish, birds, marine invertebrates, and aquatic vegetation. Humans also use the ocean for recreation, fishing, and other activities and can be exposed by eating contaminated seafood.

6.5.2.1 Ecological Receptors

Marine mammals that may be found in the South Florida coastal and marine environment include Florida manatees, whales (right, Sei, finback, humpback, sperm), and dolphins. In coastal brackish and freshwater environments such as estuaries and rivers, river otters also occur. The U.S. Fish and Wildlife Services and NOAA list all of these marine mammals except dolphins as endangered species (FFWCC, 1997).

Reptiles known to occur in marine or brackish South Florida waters include the American crocodile (endangered), Atlantic salt marsh snake (threatened), gray salt marsh snake, Atlantic green turtle (endangered), Atlantic hawksbill turtle (endangered), Atlantic loggerhead turtle (threatened), Atlantic Ridley turtle (endangered), and the leatherback turtle (endangered) (Carmichael and Williams, 1991; FFWCC, 1997).

The South Florida shelf environment is host to a wide variety of subtropical marine invertebrates, including mollusks (clams, conchs, snails, octopi, squid), annelids (worms), arthropods (crabs, lobster, shrimp), coelenterates (corals, sea anemones, echinoderms, starfish, sea urchins), sponges, bryozoans, and many others (Alevizon, 1994; FFWCC, 1997). These marine organisms feed in a number of ways, including predation, scavenging, filter-feeding, grazing, and feeding on organic detritus. Predatory invertebrates include octopi, many snails such as conchs, starfish, and squid. Filter-feeding organisms include corals, sponges, bryozoans, and bivalves such as clams and mussels. Some filter-feeding organisms, like certain corals, have symbiotic algae that help the host animal to survive. Grazing organisms include sea urchins and mollusks. Detritus feeders and scavengers include many worms, crabs, lobsters, shrimp, and snails.

The most extensive reefs of South Florida are primarily associated with the Florida Keys, but reef-forming organisms such as corals, sponges, and bryozoans may be found along the South Florida coast. Associated with these reef-forming animals may be found coralline and encrusting algae, which require solid substrates for attachment. In the Florida Keys, coral reefs have declined from a combination of factors, not all of which

may be manmade. An epidemic disease occurred in the early 1980s, affecting the longspined black sea urchins that graze on the macroalgae that compete with corals for space. The absence of urchins may account for increased growth of seaweed on the reefs. Groundwater nutrient inputs from onsite sewage disposal systems may also account for the growth of macroalgal blooms, such as *Codium isthmocladum* in southeast Florida and the Caribbean (LaPointe, 1997; NOAA 2002c).

Fish species found in Florida waters include yellowtail snapper, grouper, barracuda, stingray, parrotfish, porcupine fish, Key blenny (endangered), angelfish, butterflyfish, damselfish, goby, trumpetfish, and wrasse, among many others (FFWCC, 1997).

Birds that may be found in brackish and marine waters include the brown pelican, American oystercatcher, frigatebird, piping plover (threatened), roseate spoonbill, roseate tern (threatened), cormorant, least tern, and southeastern snowy plover (threatened). Many other birds found in more inland brackish to fresh waters include the flamingo, heron, kingfisher, little blue heron, osprey, reddish egret, snowy egret, tricolored heron, white ibis, whooping crane, bald eagle, and others (FFWCC, 1997; Williams, 1983).

6.5.2.2 Human Receptors

Potential human receptors who may be exposed to ocean outfall effluent include recreational and industrial fishermen, boaters, workers associated with ocean outfall operations or wastewater treatment and, if the exposure pathways exist, recreational swimmers.

6.5.3 Potential Exposure Pathways

For nonpotable water, the primary potential exposure pathways are related to direct exposure of humans to water containing stressors and ingestion of seafood with elevated levels of contaminants. There is also a possibility of airborne exposure if water droplets containing effluent constituents somehow are formed through turbulence or aerosolization. Potential primary human exposure pathways for waterborne stressors in discharged effluent include ingestion of stressors (followed by bioaccumulation or excretion), dermal contact with stressors, and inhalation of water vapor containing chemical or microbiological stressors. Recreational or fishing activities in or near the ocean outfall could bring humans into a situation where exposure could occur.

Potential exposure pathways for marine mammals, reptiles, and fish are similar to the above-named pathways (that is, ingestion, dermal contact, and inhalation). Predation or scavenging of other organisms feeding upon contaminated organisms or algae that contain elevated tissue concentrations of effluent constituents could also cause bioaccumulation of these constituents.

Potential exposure pathways for marine invertebrates include ingestion of particles or dissolved materials containing effluent constituents. Examples include filter-feeding or detrital-feeding organisms feeding on organic particles containing adsorbed metals or

organic constituents or ingesting water containing dissolved effluent constituents. Such organisms may be feeding upon the fecal pellets of other marine organisms that may have ingested effluent constituents. Predators may feed on other organisms that have already ingested or bioaccumulated constituents such as metals or organic compounds.

Settling organic and inorganic particles in the ocean represent a significant mass transport mechanism for the cycling of particles from the surface of the ocean to the seafloor. Such settling particles can scavenge other materials in the water column by adsorption or other complexation processes (Honjo et al., 1982). Fecal pellets produced by zooplankton settle to the sea floor as organic detritus, thereby providing a conduit for the rapid removal of nutrients and other substances from the upper layers of the ocean to the deeper layers of the ocean (Pilskaln and Honjo, 1987). Much of the organic matter found on the seafloor ultimately derives from primary and secondary production in the photic zone, which is typically 10 m deep (Parsons et al., 1984).

Unlikely exposure pathways include direct exposure of shallow shelf or photic zone organisms to discharged effluent. Receptors could be exposed to stressors from the physical transport of stressors towards the coast. For example, if the Florida Current were to move nearshore or if an eddy of the Florida Current were to transport effluent constituents, then nearshore or onshore receptors could be exposed to effluent constituents.

The question of whether exposure and uptake pathways exist is crucial for risk assessment. The primary risk questions to be asked are these:

- Do these actual exposure pathways exist?
- If they do exist, is there actual uptake?
- If there is uptake, are there adverse effects upon humans or biota?

Unless seawater is used for desalination for a drinking-water source, the primary type of human risk that might occur would be related to recreational or occupational exposures to seawater and consumption of seafood.

6.5.4 Conceptual Model of Potential Risk for Ocean Outfalls

A conceptual risk model is a generic model of potential risks that may result from management of treated municipal wastewater using ocean outfalls. Such a model lists all potential exposure pathways and processes that control whether a receptor is actually exposed to a stressor or not. This conceptual model of potential risk represents the risk model to be tested using specific data. Section 6.6 describes the data and the testing of the model. It contains an evaluation of how realistic the potential risks are. A conceptual model for evaluating potential risks associated with ocean outfalls is shown in Figure 6-4.

The model components that control the fate and transport of wastewater discharged into the open ocean environment were adapted from a 1984 National Academy of Science study entitled “Disposal of Industrial and Domestic Wastes: Land and Sea Alternatives”

and the Waquoit Bay National Estuarine Research Reserve Watershed risk assessment model that provides a method for identifying valued natural resources and evaluating the risk to those resources (Bowen et al., 2001).

In this conceptual model, the source of stressors is the wastewater treatment plant providing secondary treatment and basic disinfection of municipal wastewater derived from industrial and domestic sources. The potential stressors are inorganic compounds (for example, metals, salts), organic compounds, nutrients, and pathogenic microorganisms.

The physical pathways and processes that occur when treated wastewater is discharged into any water body, either open ocean or surface water (such as rivers, lagoons, or estuaries), are extremely important in determining large-scale exposure pathways. In the vicinity of the outfall, the ways in which ocean currents affect dispersion and dilution of the effluent plume are extremely important. Farther away from the outfall, as dilution occurs, it is important to determine whether ocean circulation and mixing could vary enough to expose terrestrial or nearshore receptors.

Physical processes refers to the transport process that moves suspended or dissolved materials from one place to another (National Academy of Sciences, 1984). Examples include advection of a plume through current movement, dilution or dispersion of the plume through mixing with surrounding waters, density-driven advection, sedimentation of solids from the plume to the benthos, resuspension of sediment through turbulence or bioturbation, adsorption, and volatilization to the atmosphere.

Potential chemical processes are chemical reactions that wastewater constituents can undergo when discharged into the aquatic environment. These processes include adsorption and desorption, changes in oxidation state, precipitation and dissolution, photodegradation, transformation, and complex formation.

Potential biological processes affecting the fate and transport of stressors include uptake, bioconcentration and accumulation of stressors, inactivation of pathogenic microorganisms, biochemical transformation or degradation of stressors, photosynthesis, and the formation of organic marine particles such as zooplankton fecal pellets that transport stressors to benthic habitats. Both chemical and biological processes determine the fate and effect of a particular constituent.

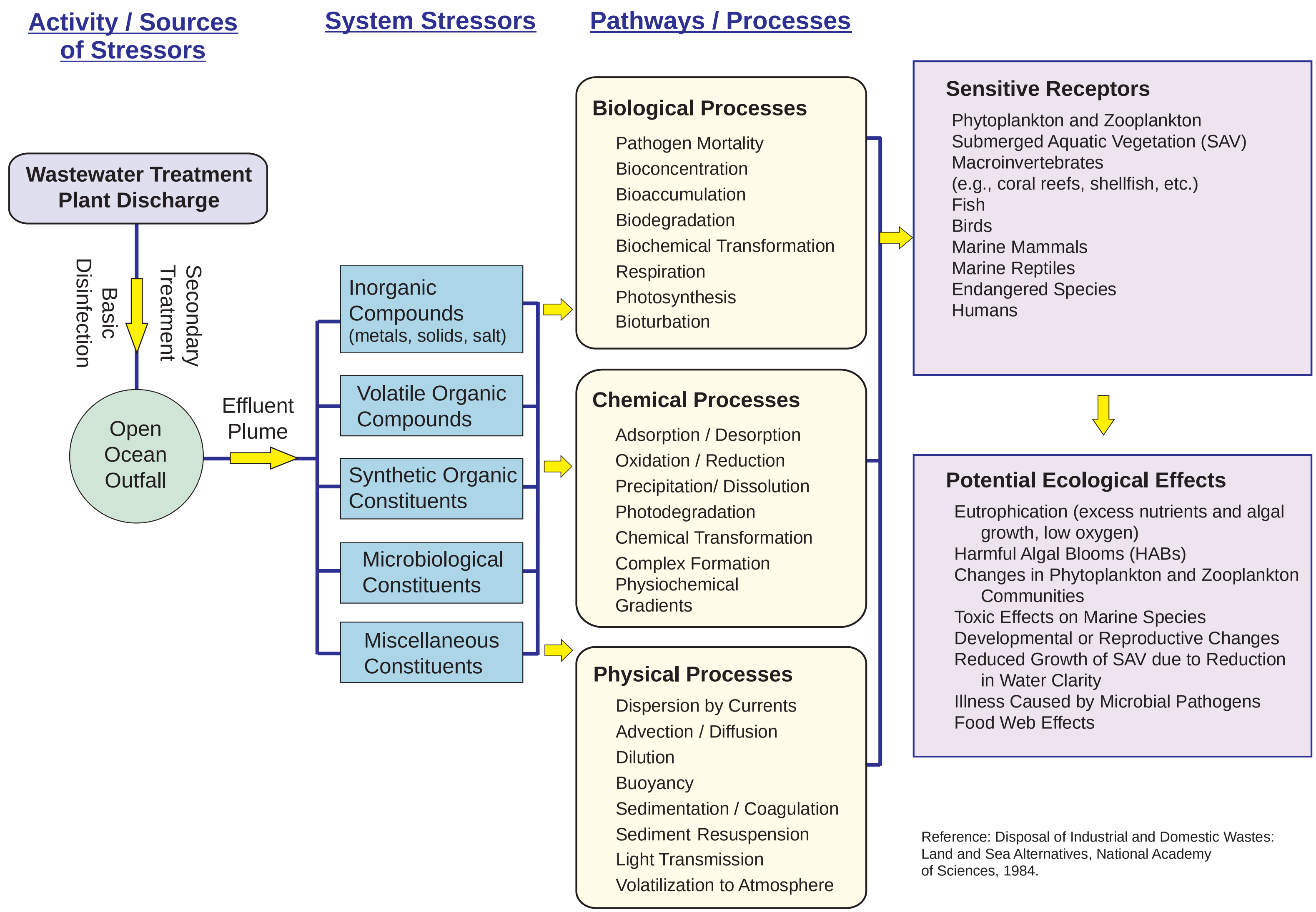


Figure 6-4. Conceptual Model of Potential Risks for the Ocean Outfall Option

Potential receptors include submerged aquatic vegetation, plankton (phytoplankton, zooplankton), larger aquatic organisms (invertebrates, fish, marine mammals, and reptiles), birds, and humans. There are no drinking-water receptors in this conceptual model. If seawater were to be used for a drinking-water source through desalinization, which is being considered in South Florida, then this potential receptor would be added to the conceptual model. However, seawater in coastal areas would contain many of the same stressors derived from other sources on land.

6.6 Risk Analysis of Ocean Outfalls

In this section, the potential risk model expressed by the conceptual model is tested using actual data from existing ocean outfalls or the SEFLOE studies. As part of the risk analysis, the following questions will be answered:

- Do plausible exposure pathways exist for receptors to be exposed to stressors?
- Are concentrations of stressors high enough to potentially cause adverse effects?
- Is there evidence for adverse effects in receptors caused by exposure to stressors derived from treated wastewater effluent?

6.6.1 Evaluation of Physical Transport

In order to appreciate the large-scale risk setting, it is important to understand physical environmental risk factors. These are the physical features of the environment that play a significant role in the risk of a particular wastewater management option. A thorough understanding of physical oceanography, circulation, mixing, dispersion, and dilution of the discharged effluent plume at the ocean outfalls is necessary for evaluating the physical environmental risk factors associated with ocean outfalls.

The SEFLOE studies and other related studies provide much of the information needed to assess such risk factors. Intensive cruises were conducted to each outfall during winter and summer to detect and track, using acoustic measurements, the initial plume and to develop two-dimensional models of the effluent plumes. To map and model current velocities and water column structure, moored current monitors were deployed near outfall discharge sites for several periods between August 1991 and October 1992. An acoustic Doppler current profiler was deployed at the Miami-Central outfall diffuser in the summer of 1992 to obtain more information on current regimes and depth variations in currents. Dye and salinity tracking were also used to map the distribution and movements of water masses. Water-column characteristics (conductivity, temperature, and depth, or CTD) were measured using CTD water-column profiling on a semimonthly basis at each outfall from July 1991 to October 1992 (except months when intensive cruises were underway). Physical characteristics of the surfacing effluent plumes were monitored using towed CTDs. Initial and subsequent dilutions were estimated, using differences in salinity between the effluent and ambient seawater as a tracer.

The SEFLOE studies also collected water-quality information from the effluent plume and from ambient seawater. Parameters measured included bacteria (total coliforms, fecal

coliforms, and *Enterococcus* bacteria), nutrients (ammonia, total Kjeldahl nitrogen (TKN), total phosphorus, nitrate, nitrite), oil and grease, 126 priority pollutants, and total suspended solids (TSS). Effluent samples from the six wastewater treatment facilities were also analyzed for salinity, bacteria, nutrients, priority pollutants, oil and grease, $CBOD_5$, and TSS.

6.6.1.1 Transport, Dispersion, and Dilution by Currents

Transport, dispersion, and dilution of effluent plumes by ocean currents and circulation are critical risk factors for evaluating potential risks of ocean outfalls. The direction and speed of current flow, which together determine current velocity, are critically important risk factors. The faster the current speed is at the outfall, the greater the rate at which the plume is dispersed and diluted by ambient seawater, and the lower the concentration of stressors. Conversely, the slower the current speed is at the outfall, the lower the rate at which the plume is dispersed and diluted by ambient seawater, and the higher the concentration of stressors remaining in the area. The direction of current flow, away from or towards human or ecological receptors, is also important to characterize. Current flow towards the coast will increase the likelihood that coastal receptors (human or ecological) will be exposed to effluent constituents, while current flow away from the coast will decrease the likelihood of exposure.

The SEFLOE II study provided an extensive set of current measurements and water-column density profiles, using a combination of acoustical backscatter and direct sampling methods. Information on water quality of the effluent plume and ambient seawater also was obtained. Analysis of the current data from the four outfall locations indicated that three major current regimes, characterized by different flow directions, were present at all outfall sites:

- *Current Regime i* = Northerly flows, thought to be associated with western meanders of the Florida Current
- *Current Regime ii* = Southerly flows, which are part of an extensive eddy current
- *Current Regime iii* = Rotary-like flow, which consists of groups of rotations interspersed between northerly and southerly flows. The rotations were irregular and temporally fleeting, with durations of 5 to 8 hours.

Current Regime *i* predominates, displaying rapid current flow in a northerly direction throughout the entire water column. The SEFLOE II study reports that this current flow occurs approximately 60% of the time. Current Regime *ii*, representing southerly flow, occurs approximately 30% of the time. Current Regime *iii*, representing flow in other directions (easterly, westerly) occurs irregularly and less than 10% of the time, and the duration of such flows is very short (5 to 8 hours).

To estimate the percentage of time that Current Regime *iii* flows to the west, data points reflecting current direction at a depth of 16.8 m at the Miami-Dade Central District wastewater treatment plant (Hazen and Sawyer, 1994) were visually analyzed. It was assumed that, as described in the SEFLOE II report, easterly and westerly flows occur a

total of 10% of the time. This analysis indicates that westerly flows occur approximately 4% percent of the time, while easterly flows occur approximately 6% percent of the time. Data sets using an Aanderaa current meter and using an acoustic Doppler current profiler yielded equivalent results.

The Florida Current in the vicinity of the ocean outfalls can be characterized as a fast-flowing current, with speeds ranging from approximately less than 5 centimeters per second (cm/sec) to maximum speeds of over 60 to 70 cm/sec. In general, the mean current velocity observed during Regime *i* northerly current flow is greater than any other current regime, while the mean current velocity of Regime *iii* rotary-like flow is the lowest (Hazen and Sawyer, 1994). Because the dilution of the effluent plumes is a function of current velocity, the Regime *iii* rotary-like flow will result in the higher concentration of stressors.

For the purposes of evaluating plume dispersal, the SEFLOE II report used the lowest 4-day average current speeds and the lowest 10th-percentile average current speeds as conservative (that is, protective) estimates of average current speeds. These current speeds are shown in Table 6-4 (Hazen and Sawyer, 1994). The maximum current speeds recorded during the study are shown for comparison. In general, the average current speeds are highest at the Broward outfall.

Table 6-4. Average Current Speeds (cm/sec)

Outfall	Lowest 4-Day Average Current Speed	Lowest 10th-Percentile Average Current Speed	Maximum Current Speed
Broward	15.7	12.3	> 70
Hollywood	13.7	7.8	> 60
Miami-Dade North	13.2	7.7	> 70
Miami-Dade Central	13.6	11.6	> 70

Source: Hazen and Sawyer, 1994.

Irrespective of which current regime was predominant, current direction was generally the same at all depths, based on water column profiles. Slight variations in current speed occurred throughout the water column, with higher speeds occurring near the ocean surface.

6.6.1.2 Dilution of the Effluent Plume

The SEFLOE I study characterized dilution of the effluent plumes at all six of the ocean outfalls, using dye and salinity data and acoustic backscattering methods. Based on these studies, the effluent plume typically has three distinct phases:

1. The initial dilution phase commences when the effluent leaves the outfall pipe and lasts until the effluent reaches the surface of the ocean.
2. The near-field dilution phase commences when the plume reaches the surface and undergoes radial dispersion because of the momentum of the rising effluent

within the upper 3 m of the ocean surface. This phase is visible on the surface of the ocean as a feature that is often called a "boil."

3. The farfield dilution phase is characterized by an effluent plume that has undergone dilution during the initial and near-field dilution phases and is further dispersed by surface currents.

The characteristics of each of these dilution phases are discussed below.

Initial and Near-Field Dilution

Because the water samples are collected at or near the boil, within 1 m of the surface, the sampling actually is conducted within both the initial and near-field dilution phases, as defined above in the SEFLOE study. Therefore, these two dilution phases are discussed together in this section.

Initial dilution using tracer dye methods and chlorine was defined in the SEFLOE study as the ratio of measured concentrations of the dye in the effluent boil to the initial concentration of the dye in the effluent at the treatment facility. The initial dilution that occurs over a 4-day period at a conservative current speed (worst-case scenario with the lowest average current speed) is described in Table 6-5 below as the flux-averaged initial dilution factor. The greater this factor is, the higher the dilution.

Table 6-5. Flux-Averaged Initial Dilution of Effluent Plume.

Ocean Outfall	Lowest 4-day Average Current Speed (cm/s)	Flux-Averaged Initial Dilution Factor
Broward	15.7	43.3
Hollywood	13.7	28.4
Miami-Dade North	13.2	50.1
Miami-Dade Central	13.6	28.3

Source: Hazen and Sawyer, 1994, Table III-5.

The initial dilution factors from the SEFLOE studies indicate that initial dilutions were highest for the Miami-Dade North ocean outfall and lowest for the Miami-Dade Central and Hollywood outfalls. Yet, according to Table 6-5, Miami-Dade North outfall had the lowest 4-day average current speed. The high initial dilution at this outfall may explained by the use of multiport diffusers at the Miami-Dade North outfall. The use of multiport diffusers at the Miami-Dade North outfall appears to aid in dispersal of the effluent plume over a wider area, thereby decreasing potential risk. However, these effluent plumes were diluted at slower rates than the effluent plumes from the Hollywood and Broward outfall plumes, according to Englehardt et al. (2001).

The rate of initial dilution of the effluent also is largely dependent upon the current speed and the rate of discharge of effluent from the outfall. As current speed increases, dilution also increases. As the rate of effluent discharged from the outfall increases, the rate of dilution increases. These relationships are shown in Figure 6-5, which shows flux-

averaged dilution vs. current speed for effluent discharged from the Miami-Dade Central ocean outfall (from Hazen and Sawyer, 1994).

At a lower current speed of 10 cm/sec at a 253 mgd discharge rate, the dilution factor is approximately 20; at a higher current speed of, say, 60 cm/sec, the dilution factor is over 40. Also, for a given current speed, at higher discharge rates (that is, 253 mgd), the dilution is lower than if effluent is discharged at a lower rate (that is, 115 mgd).

The SEFLOE study found that normally surfacing plumes were present at all outfalls throughout the year, even in summer months when density stratification of the water column was weak. It is noteworthy, however, that during several strong stratification events, portions of rising plumes were trapped and prevented from freely dispersing, based on acoustic profiling conducted by John Proni and colleagues (Proni et al., 1996, 1994; and Proni and Williams, 1997). In such areas of plume trapping, effluent constituents were present at relatively higher concentrations than in areas in which there was no such trapping and the effluent was freely dispersed. Concentrations of effluent constituents were, however, quite low, but their existence is quite significant. Definitive measurements of dilution in trapped plumes are planned for an upcoming SEFLOE III study (John Proni, personal communication). Plume trapping during strong stratification events therefore represents one potential risk factor.

<u>Farfield Dilution</u>

The SEFLOE II report indicates that measurements of farfield dilutions were the most difficult field measurements to obtain. Measurements of salinity, dye concentration, and acoustic backscatter intensity were used simultaneously for dilution calculations and to guide sampling for biological and chemical parameters for subsequent dilution determinations. Subsequent dilution is defined in the SEFLOE report as dilution that occurs as plume elements move away from the boil location, which represents the initial dilution and near-field dilution phases.

Average subsequent dilutions in the near-field and farfield for the four ocean outfalls are compared in Figure 6-6, which plots the inverse of total physical dilution (1/total physical dilution) of the plume on a logarithmic scale against the distance from the boil in meters (from Hazen and Sawyer, 1994, Figure III-77). On this plot, as one moves away from the boil, dilution increases. Figure 6-6 shows the following:

- Treated effluent discharged from the Broward and Hollywood outfalls experiences more rapid dilution in the 0- to 100-m range than the effluent discharged from the two Miami-Dade outfalls. These two outfalls have larger diameter ports than the other ocean outfalls (see Table 6-1).
- Between 100 and 200 m from the plume boil, there is a change in the rate of dilution, suggesting that buoyancy spreading and positive buoyancy of the plumes is still occurring at this distance from the outfall.

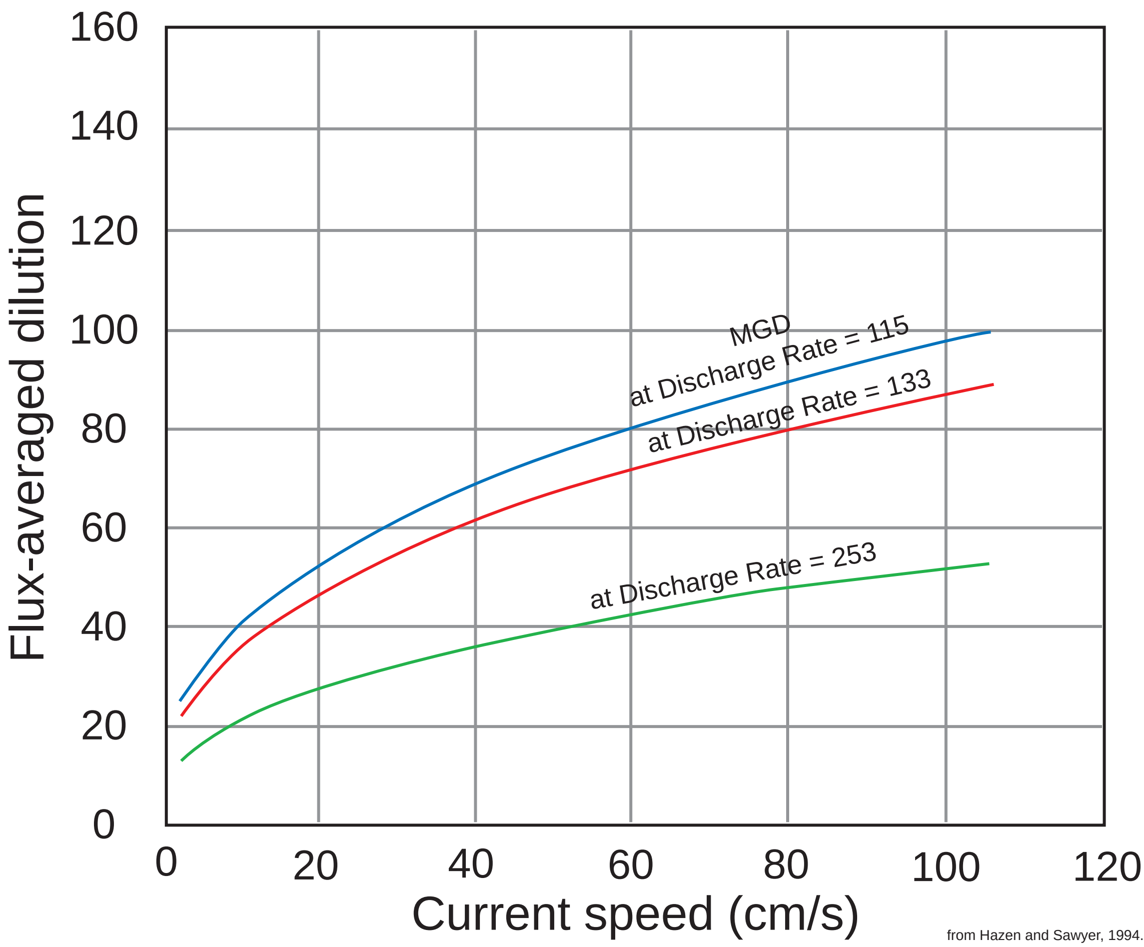
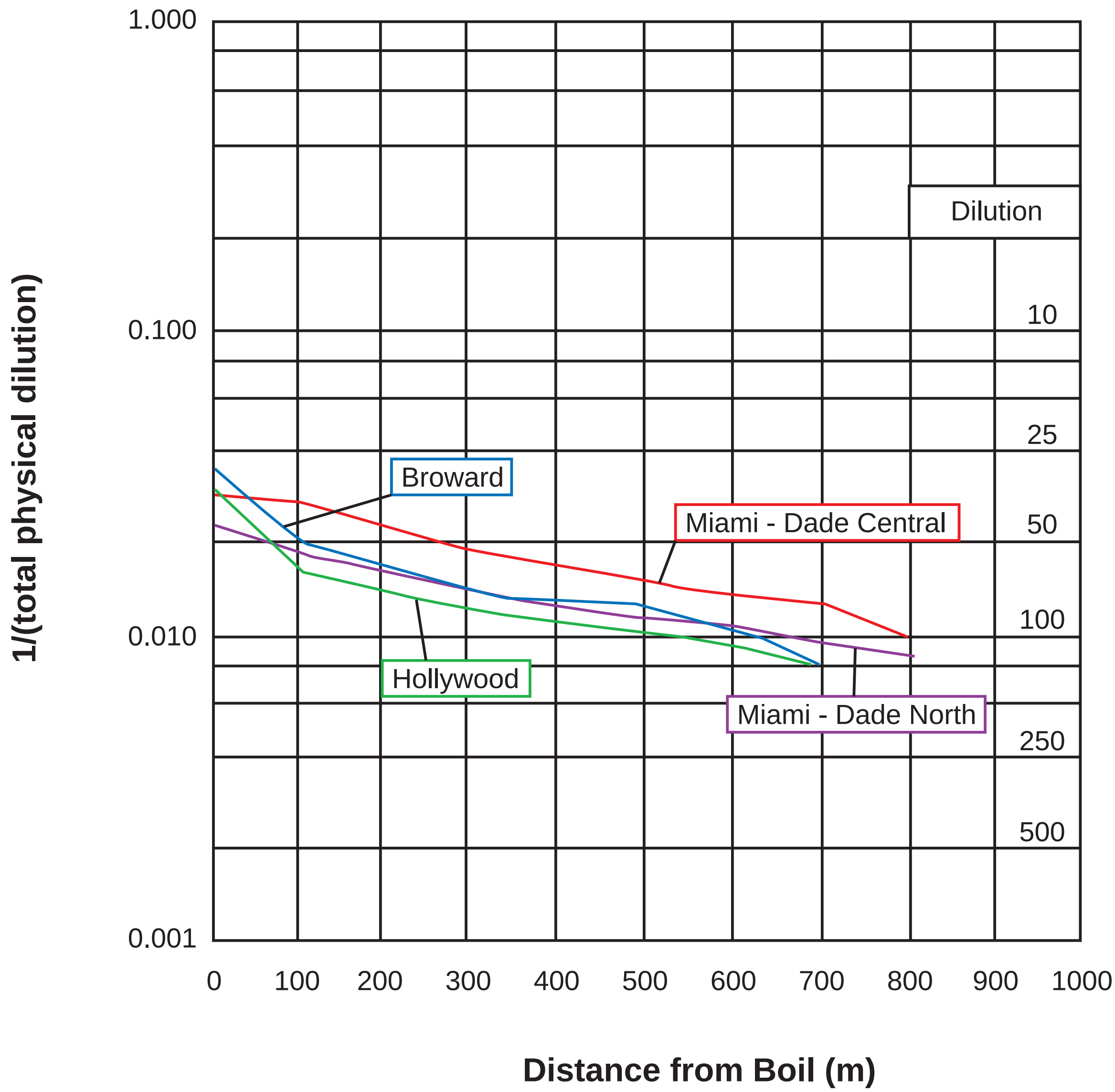


Figure 6-5. Initial Dilution as a Function of Current Speed and Discharge Rate (Miami-Dade Central Outfall)



Source: Hazen and Sawyer, 1994

Figure 6-6. Total Physical Dilution as a Function of Distance from the Boil (Four Ocean Outfalls)

- Between 100 m and 400 m from the boil, total physical dilution with distance curves are approximately similar for both current Regimes *i* (northerly flow) and *ii* (southerly flow), with average dilutions ranging from approximately 60:1 to 90:1 at a distance of 400 m from the boil location.
- Dilution rate increases slightly at 500 m from the boil for Broward, 700 m from the boil for Miami-Dade Central, and 600 m from the boil for Hollywood outfalls. The effluent plume from the Miami-Dade North outfall shows steady dilution throughout nearly the entire distance sampled.

Note that the 400-m mixing zone equates to the 502,655 m^2 maximum mixing zone size for open ocean outfalls regulated by the state of Florida. Dilutions ranging from 60:1 to 90:1 were used to evaluate the concentrations of the measured constituents of concern in wastewater and were compared to the Class III standards.

The information from the SEFLOE studies indicates that overall dispersal and dilution of the discharged effluent occurs rapidly, within hours to days, and that the mixture of effluent and receiving water rapidly achieves background or near-background concentrations of tracer dyes and salinity within 400 to 600+ m of the outfall. Rapid dispersal results in dilution of the effluent and therefore reduces the risk of exposure to undiluted effluent.

6.6.2 Evaluation of Stressors, Exposure Pathways, and Receptors

In this section, information from the SEFLOE studies and other studies are used to evaluate the following risk questions posed by the conceptual model:

- Do concentrations of stressors exceed standards intended to protect human health or ecological systems?
- Is there evidence that human or ecological receptors are exposed to or take up stressors derived from the treated effluent or secondary stressors that are created by discharge of effluent?
- Is there evidence of adverse effects on human or ecological receptors in the vicinity of the outfalls?
- If there are adverse effects that can be attributed to the use of ocean outfalls, are these effects reversible?

6.6.2.1 Pathogenic Microorganisms

Pathogenic Microorganisms in Unchlorinated Effluent as a Worst-Case Scenario

The SEFLOE study measured three types of bacteria indicative of mammalian wastes: total coliforms, fecal coliforms and *Enterococcus*. Samples for microbiological analysis were taken from both secondary treated unchlorinated effluent and from within the effluent plume itself (Hazen and Sawyer, 1994). Based on these measurements and the effluent plume characteristics, SEFLOE provided recommendations to regulators

concerning the width of the mixing zones that would have to be defined to allow dilution of the effluent to meet Florida water-quality criteria for bacteria.

Recommendations on the maximum allowable concentrations of bacteria in effluent were also provided to help guide treatment plant operators in determining the correct amount of chlorine to use in disinfecting effluent. Florida regulations require basic disinfection to meet a standard of 200 fecal coliforms per 100 mL of treated wastewater. However, because chlorine disinfection itself can create unwanted chlorinated byproducts (for example, trihalomehanes) and pose potential health or environmental risks, the regulations also allow for an effluent mixing zone range. This allows dilution of effluent, reducing the amount of chlorine used, while still meeting water-quality standards. SEFLOE's recommended widths of mixing zones of unchlorinated effluent to achieve Class III bacterial water quality standards are summarized in Table 6-6.

Table 6-6. Recommended Mixing Zone Ranges for Unchlorinated Effluent, Using Different Methods of Calculating Bacterial Concentrations

Facility	Radial Distance (m)			
	Maximum Single Requirement	Percent Not Greater Than Requirement	Geometric Mean Requirement	Range of Controlling Criterion
Broward County	900	900	400	900
Total coliform	800	800	400	800
Fecal coliform	800	800	400	800
Enterococci	900	900	400	900
City of Hollywood				
Total coliform	500	800	0	80
Fecal coliform	200	700	0	700
Enterococci	1,000	800	400	1,000
Miami-Dade North				
Total coliform	900	900	400	900
Fecal coliform	900	900	400	900
Enterococci	1,000	1,000	800+	1,000
Miami-Dade Central				
Total coliform	Uncertain	Uncertain	800	Uncertain
Fecal coliform	900	900	800+	900
Enterococci	Uncertain	Uncertain	800+	Uncertain

Note: Data from Miami-Dade Central are shown as uncertain because of suspected high background concentrations of indicator bacteria from the Miami River.
Source: Hazen and Sawyer, 1994.

Because these values represent distances that the unchlorinated effluent would have to travel before the concentration of bacteria became diluted to background levels, they provide information for an evaluation of one potential worst-case risk scenario, which is failure of chlorination to treat secondary effluent to meet Class III water-quality standards. In general, the results show that even if unchlorinated effluent were discharged, it would become dilute enough to meet Class III bacteriological water quality standards within 800 to 900 m of the outfall or, in some cases, much closer.

Disinfection To Achieve Microbial Standards

The Florida regulations require a mixing zone area of up to 502,655 m^2 to allow dilution of the effluent to Class III water-quality standards. Although the Florida regulations do not require that a circular mixing zone be established, and in fact do not specify a shape, the use of a circular mixing zone for evaluating whether dilution achieves the standards makes it easier to compare actual versus expected concentrations of effluent constituents. Such a circular mixing zone would have a radius of 400 m. It is worth noting, however, that a circular mixing zone would occur only in an environment where there is no current flow.

To assist facility operators in determining how to manage bacteria to meet Class III regulatory standards, SEFLOE provided calculations of the maximum allowable numbers of indicator bacteria in effluent within 400 m of the outfall. These calculated concentrations are shown in Table 6-7. They include assumptions concerning microbial attenuation processes that are not based solely on physical dilution alone (John Proni, personal communication). These bacterial numbers provide wastewater treatment facility operators with specific bacterial concentration goals to meet, using chlorination of effluent in order to meet Class III water-quality standards. The 800-m mixing zone is included because, as stated above, the mixing zone is not required to be a circular mixing zone but instead is an area, and the effluent plume may well extend outside of the 400-m-radius zone.

Table 6-7. Maximum Allowable Concentrations of Indicator Bacteria in Effluent within Different Mixing Zones

Facility	0 m Initial Dilution Zone	400 m Mixing Zone	800 m Mixing Zone
Broward County			
Total coliform	302	3,388	10,471
Fecal coliform	72.6	437.6	935
Enterococci	--	284	53
City of Hollywood			
Total coliform	575	2,884	3,631
Fecal coliform	296	324	1626
Enterococci	7.3	38.4	106
Miami-Dade North District			
Total coliform	3,715	14,454	28,840
Fecal coliform	1,517	20,465	7,962
Enterococci	153	840	879
Miami-Dade Central District			
Total coliform	186	417	11,000
Fecal coliform	68	252	1,910
Enterococci	2.4	29.0	334.0

Note: All bacterial numbers = 100 per 1000 mL
Source: Hazen and Sawyer, 1994, Table III-17.

Proximity of Effluent Plume to Coastal Receptors

One significant microbiological risk factor is the proximity of the ocean outfalls to land and to potential terrestrial and nearshore receptors. If one assumes that the required mixing zone area of 502,655 m^2 is translated into a circle of radius 400 m centered on the outfall, one can compare this with the actual distance of the outfall from land (Table 6-8). This table indicates that the highest risk outfalls, solely in terms of distance from shore, are the Del Ray Beach and Boca Raton outfalls, while the lowest risk outfall in terms of distance is the Miami-Dade Central outfall.

Table 6-8. Comparison of Circular Mixing Radii for Effluent and Outfall Distance from Shore (m)

Parameter	Miami-Dade Central District	Miami-Miami-Dade North District	City of Hollywood	Broward County	Delray Beach	Boca Raton
Distance offshore	5,730	3,350	3,050	2,130	1,600	1,515
Distance from 400 m circle to land	5,330	2,950	2,650	1,703	1,200	1,115
Distance from 800 m circle to land	4,930	2,550	2,250	1,330	800	715

Note: A 400-m mixing radius is required for chlorinated effluent to meet Class III bacteriological standards. If the effluent is unchlorinated, an 800-m mixing radius is required.
Source: Hazen and Sawyer, 1994.

It is important to note that in reality the effluent plumes do not disperse equally over a circular area, as implied by the circular mixing zone calculations used by the SEFLOE study, but are instead dispersed by the strong Florida Current to form an extended plume, whose longest dimension is aligned with the northerly flowing Florida Current. It is not known what would happen if the northerly current flow were to weaken or disappear. It is probable that, for such a major change in the Florida Current to occur, there would have to be major changes in ocean circulation elsewhere as well.

There are a number of gaps in information concerning human health and ecological risks from pathogenic microorganisms remaining in treated effluent. The SEFLOE studies of enteric microorganisms in effluent and the dilute effluent plume did not include measurements of *Cryptosporidium* or *Giardia*. Other enteric viruses and bacteria were not measured. Ecological risks posed by effluent microorganisms could not be evaluated in this report because of the lack of long-term monitoring studies of benthic organisms in the effluent plume track or adjacent waters.

Human health risks posed by effluent microorganisms also could not be evaluated directly because of the lack of information on pathogenic microorganisms in coastal waters adjacent to the outfalls and derived from the outfalls. Beach water-quality information would provide information on microbial concentrations, but would not distinguish between onshore versus offshore sources of pathogenic microorganisms. Many onshore sources of pathogenic microorganisms undoubtedly exist in southeast

Florida, from a combination of intensive urban and agricultural activities. To distinguish between these different sources, a tracer study involving microbial tracers or combined microbial/chemical/biochemical tracers would have to be conducted. However, it remains clear that there is a risk from pathogenic protozoans such as *Cryptosporidium*, which is not addressed by chlorination, and that the risk is highest during the westward-flowing current phase, which occurs approximately 4% of the time.

Nevertheless, the SEFLOE studies provide a significant body of knowledge for risk managers to understand the processes that affect microbiological risks to human health. They also provide specific recommendations concerning the level of dilution and disinfection of treated wastewater needed to achieve Class III water-quality standards for microbial indicators of wastewater (fecal coliforms, *Enterococcus*, total coliforms) at a hypothetical 400 –m-radius mixing zone. Although the SEFLOE studies do not provide follow-up monitoring to confirm that these standards are indeed met all of the time, monitoring of chlorinated treated wastewater at treatment facilities suggests that these microbial standards for regulated pathogens and indicator bacteria are nearly always met.

6.6.2.2 Nutrients

There are three questions that must be addressed in order to evaluate potential risks from nutrients in the secondary treated effluent:

- Are nutrient levels in the effluent higher than ambient water or applicable marine water-quality standards to protect ecological health?
- Is there evidence that nutrients from the treated effluent are taken up by phytoplankton and microalgae and then converted to biomass?
- Is there evidence of ecological effects from nutrient inputs from the effluent plume?

To evaluate ecological risks associated with nutrient discharge, information on effluent nutrient concentrations was compared with Florida water-quality standards designed to protect aquatic ecosystems. The Class III Florida water-quality standards state, “In no case shall nutrient concentrations of a body of water be altered so as to cause an imbalance in natural populations of aquatic flora or fauna.” Therefore, it is also valuable to compare nutrient concentrations in secondary treated effluent with nutrient concentrations in ambient seawater at the site, because natural populations of organisms will be adapted to the ambient concentrations and may experience changes if nutrient concentrations change.

Table 6-9 compares the nutrient concentrations in secondary effluent, ambient seawater, and 400 m and 800 m mixing zones at three ocean outfalls (from Hazen and Sawyer, 1994, Tables III-18, III-19 and III-20). All values in the table are from Hazen and Sawyer (1994) unless otherwise noted as calculated for this report. The 800 m mixing zone is included as a conservative approach.

Table 6-9. Nutrient Concentrations in Secondary Treated Effluent, Ambient Water, and 400 m and 800 m Mixing Zones for Three Ocean Outfalls

Parameter	Ammonia (mg/L)				Nitrate (mg/L)				Total Phosphorus (mg/L)			
	Mean	Max.	No. of Samples	Other Values	Mean	Max.	No. of Samples	Other Values	Mean	Max.	No. of Samples	Other Values
Broward												
Ambient water	0.09	0.5	28		0.16	0.16	1		0.08	0.16	17	
Effluent	12.48	20.0	42	8.7 [d]	0.42	1.08	7	0.64 [a], 9.6 [b], 0.28 [c] 3.8 [d]	1.66	2.45	43	1.33 [d]
Boil	0.35	0.9	55		0.12	0.46	9		0.18	0.84	41	
400 m	0.13	0.35	32		0.02	0.02	1		0.10	0.3	25	
800 m	0.11	0.5	26		0.01	0.01	1		0.10	0.13	21	
Dilution at 400 m**	96x				21x				16.6x			
Dilution at 800 m**	113x				42x				16.6x			
Hollywood												
Ambient water	0.09	0.14	8		0.11	0.11	9		0.09	0.11	6	
Effluent	5.96	14.00	31		1.70*	1.60*	31		0.97	1.60	32	
Boil	0.25	0.80	15		0.16	0.52	15		0.11	0.40	12	
400 m	0.09	0.12	4		0.12	0.30	4		0.10	0.10	2	
800 m	0.09	0.12	4		0.11	1.40	3		0.10	0.10	3	
Dilution at 400 m**	66x				14x				9.7x			
Dilution at 800 m**	66x				15.5x				9.7x			
Parameter	**Ammonia (mg/L)**				**TKN (mg/L)**							
	Mean	**Max.**	**No. of Samples**	**Other Values**	**Mean**	**Max.**	**No. of Samples**	**Other Values**				
Miami-Dade North												
Ambient water	0.66	1.96	4		0.72	2.24	14		--	--	--	
Effluent	10.46	13.7	11		13.4	17.4	11		--	--	--	
Boil	0.56	2.24	10		0.6	3.64	28		--	--	--	
400 m	0.10	0.15	2		0.19	0.4	9		--	--	--	
800 m	0.38	0.84	4		0.61	1.96	13		--	--	--	
Dilution at 400 m**	105x				71x							
Dilution at 800 m**	28x				22x							

* These values are listed here as reported in the SEFLOE II report (Hazen and Sawyer, 1994).

** Calculated for this report using the ratio of observed concentration at the distance indicated to the initial concentration in effluent.

[a] = Miami-Dade North District, 1999. See Appendix Table 1-2.
[b] = Brevard County (South Beaches WWTF), 2000. See Appendix Table 1-2.
[c] = Albert Whitted WRF, St. Petersburg. See Appendix Table 1-2.
[d] = Mean value from Englehardt et al. (2001), compiled from several sources. See Appendix Table 1-2.

These data indicate that there are site-specific differences in whether or not the effluent nutrients become diluted to background levels by the time the effluent water reaches a distance of 800 m from the outfall. For example, at Broward, the average ammonia concentration in the plume did not reach ambient levels at 800 m. The average nitrate concentrations in the boil did not exceed background concentrations at the boil, although there are individual boil values which exceed background (John Proni, personal communication). Average total phosphorus in the plume did not reach background concentrations at 800 m.

There is also considerable variability in the concentrations and data as well; for example, the background value for nitrate is 0.16 mg/L based on 1 measurement, while the mean concentration of nitrate at the boil is 0.12 mg/L (9 measurements) and the concentration of nitrate at 400 m is 0.02 mg/L (1 measurement). Differences in time of sampling could account for these differences, as well as natural variability.

At the Hollywood outfall, average ammonia concentrations in the effluent plume reached background levels at 400 m, perhaps partly because the initial effluent concentration of ammonia was lower than at Broward. Nitrate concentrations in the plume did not reach background concentrations until 400 m out from the outfall. Total phosphorus did not reach background concentrations even at 800 m, similar to the Broward outfall.

At the Miami-Dade North outfall, ammonia at the boil did not exceed background concentrations. Concentrations of nitrate and phosphorus at this location were not reported in the SEFLOE study.

The calculated dilutions at 400 m and 800 m indicate that there are differences in dilution of ammonia, nitrate, and total phosphorus as the effluent plume becomes dispersed. Ammonia appears to dissipate most rapidly, nitrate may or may not become diluted to background concentrations at a distance of 400 m, and total phosphorus may not become diluted to background concentrations even at a distance of 800 m from the outfall. These differences in dilution may be from the differences in chemical behavior, natural variability in concentrations, differences in sampling time or location, or a combination of all of these factors.

The variability in dilution factors calculated using measurements of nutrient concentrations do provide an illustration of how the actual behavior of wastewater constituents (for example, nutrients) as measured *in situ* at a given time may deviate from the ideal modeled dilution factors, even if the modeled dilutions are based on the use of actual data on distributions of conservative tracers such as salinity, dyes, or density. To do a detailed analysis of nutrient dilution as the effluent plume moves further from the outfall, a specific study designed to track nutrient concentrations and composition would need to be conducted. Such a study should examine all inorganic and organic phases of nitrogen and phosphorus, as well as use stable isotope tracers to track effluent nitrogen and organic matter. The same is true for dissolved organic matter, which is not addressed in the SEFLOE study. The physical oceanographic conditions present during such a study would have to be documented as well, since it would be highly possible for dynamic

changes in local current flow to disrupt an otherwise orderly plume tracking experiment. It should be noted that nutrient fate and transport was not a focus of the SEFLOE studies as reported in Hazen and Sawyer. A study of nutrient fate and transport, based on the use of stable isotope tracers, is described below (Hoch et al., 1995).

At the time of the SEFLOE II studies, there was concern that nutrients from the discharge of wastewater into the open ocean was causing enhanced growth of *Codium*, an algae observed in 1991 and 1992 on several southeast Florida coral reefs. The SEFLOE II reports gives several reasons as to why the nutrients discharged from the outfalls are not likely to cause increases in *Codium* (Hazen and Sawyer, 1994), as summarized below:

- *Codium* plants require attachment to a solid substrate in order to grow, while the outfall plume rises. Thus, *Codium* habitat is not exposed directly to the effluent plume.
- Attached *Codium* plants were not present near the outfall sites where nutrient levels are above background-seawater levels.
- *Codium* attaches to solid substrate in deeper waters, outside of the nutrient dispersal area associated with the outfalls. The effluent nutrient levels quickly reach background concentrations within a short distance of the outfall (typically several hundred meters).
- Natural cycles of *Codium* growth have been reported in the literature prior to the discharge of wastewater effluent to the open ocean.
- The sporadic occurrences of the algal blooms are not consistent with the uniform discharge of the effluent, indicating no significant relationship.

The SEFLOE summary states that, "While the introduction of nitrogen into the marine environment can have significant impacts on water quality and wildlife, in this case the impacts to the open ocean appear to be mitigated by the vast reservoir of water available for dilution, the speed with which dilution occurs due to the currents at the Floridian outfalls, and the uptake and removal of nitrate by phytoplankton which entrains the nitrogen into the food chain, thereby removing it from the area where it was first emitted. The rapid dilution and removal of nitrate from the area immediately surrounding the boil quickly decreases any measurable ecological risks associated with the discharge of nitrate into the open ocean at the point where the effluent meets the receiving waters."

The stable isotopic composition of nitrogen in organic matter, called $\delta^{15}N$, can be useful in distinguishing the sources of organic matter and nutrients and the trophic level of the organisms producing the organic matter (Hansson et al., 1997; Peterson, 1999). Wastewater nitrogen tends to be isotopically enriched in the heavier isotopes of nitrogen relative to the atmospheric nitrogen standard, which represents a pristine source of nitrogen. Sewage effluent nitrogen is often isotopically heavier (more positive numbers) because of isotopic fractionation along the food chain that results in higher trophic levels, producing isotopically heavier nitrogen in wastes (LaPointe, 1997; Densmore and Bohlke, 2000; Rau et al., 1981; Schroeder et al., 1993; Spies et al., 1981; Spies, 1984; Van Dover et al., 1992). Wastewater nitrogen has been implicated as a source of isotopically heavier nitrate in the Florida Keys (LaPointe, 1997). Carbon ($\delta^{13}C$) and sulfur

($\delta^{32}S$) also provide useful isotopic tracers for organic matter (Fry et al., 1998; Gearing, 1988; Gearing et al., 1991; Wainwright and Fry, 1994; Peterson et al., 1996; Peterson, 1999).

LaPointe et al. (1992) suggests that phosphorus may be of greater importance than nitrogen as a limiting nutrient in macroalgal growth in carbonate-rich tropical waters, while nitrogen is more important in siliciclastic systems. More recently, work by LaPointe (1997) found that, at four sites located between West Palm Beach and Hobe Sound approximately 2 to 3 kilometers offshore, waters enriched with dissolved inorganic nitrogen (DIN) increased the photosynthetic efficiency of *Codium isthmocladum* in southeast Florida waters. In addition, elevated $\delta^{15}N$ values of *C. isthmocladum* tissue indicated that wastewater dissolved in DIN was a source of nitrogen to blooms in southeast Florida. LaPointe found that increases in $\delta^{15}N$ values in *Codium* tissue of more than 10 parts per thousand (*o/oo*) occurred with the onset of the rainy season, suggesting that discharges during the rainy season provided a significant nitrogen source.

A different study of the fate of sewage effluent-derived nitrogen and carbon using stable isotope tracers was conducted by researchers from EPA and Texas A&M University (Hoch et al., 1995). This study examined suspended particulate organic matter, sediment organic matter, filter-feeding organisms (sponges, soft, or gorgonian corals), settling particle fluxes, and dissolved nutrients (ammonia, nitrate and nitrite, phosphorus, and organic carbon) in the vicinity of the six southeast Florida ocean outfalls and one small outfall located in the Florida Keys. The study hypothesized that pelagic suspended organic matter composed of phytoplankton is a source of organic matter to benthic ecosystems and sediments and that the isotopic composition of these phytoplankton sources (and the nutrients they utilize) would be reflected in the isotopic composition of organic matter in sediments.

Hoch et al. (1995) found that sewage effluent ammonia from the southeast Florida outfalls had $\delta^{15}N$s ranging from 4.4*o/oo* at the Central Miami-Dade outfall, to 8.6*o/oo* at Broward and 15.4*o/oo* at Key West. Sewage effluent DIN ranged from 4.3*o/oo* to 8.1*o/oo* to 12.7*o/oo*, respectively. Nitrate and nitrite together had $\delta^{15}N$s ranging from –1.6*o/oo* to –5.7*o/oo* to 10.5*o/oo*, respectively. In comparison, suspended particulate organic matter (including phytoplankton that take up nutrients) had $\delta^{15}N$s that were more negative than effluent DIN and more similar to ambient marine organic matter (2*o/oo* to 4*o/oo*). This suggests that the effluent plume nitrogen was being diluted with ambient marine suspended particulate organic matter. In general, the nitrogen isotopic composition of ammonia and DIN at the Central Miami-Dade outfall was not very different from that of ambient marine organic matter and DIN, while ammonia and DIN from the Broward plant had isotopic signatures significantly different from that of ambient marine organic matter.

The results for the six southeast Florida ocean outfalls indicated that phytoplankton uptake of effluent-derived nitrogen into biota was not clearly demonstrated at any of the southeast Florida outfalls, including the largest outfalls (Broward and Central Miami-

Dade). At these outfalls, there appears to be little coupling between the pelagic and benthic ecosystems, even though loading of sewage effluent-derived nitrogen to coastal environments was significant (about 6 x 10^6 kg of total N per year, of which more than 97 percent is derived from the six southeast Florida outfalls). Furthermore, the measured rates of primary production were less than production estimated from the nitrogen load. Hoch and colleagues concluded that the strong currents and rapid dilution at the southeast Florida outfalls may have caused rapid dilution of sewage effluent nitrogen prior to uptake by plankton. An alternate explanation for the observed isotopic values of organic matter is that phytoplankton may have taken up a form of nitrogen not measured isotopically (for example, organic nitrogen) (Hoch et al., 1995).

In contrast, the same study found that, at the Key West outfall, a conservative estimate of the amount of effluent particulate carbon contributing to the diet of soft corals immediately adjacent to the outfall is about 40%, based on the use of both carbon and nitrogen stable isotopes. These different results suggest strongly that the physical dispersion and dilution of effluent along the southeast Florida coast plays a major role in reducing the ecological significance of effluent nitrogen. However, it also suggests that the use of stable isotopes may not be an extremely sensitive tracer if the sewage effluent isotopic composition is not significantly different from ambient marine organic matter to begin with (Hoch et al., 1995).

6.6.2.3 Metals and Organic Compounds

As with nutrients, there are three basic questions concerning potential effects of metals and organic priority pollutants remaining in effluent following secondary treatment:

- Are concentrations of priority pollutants in effluent or diluted effluent higher than water-quality standards for protection of ecological health?
- Can biological uptake of priority pollutants be demonstrated for any ecological component?
- Is there evidence of adverse effects because of exposure to or uptake of priority pollutants?

Metals

To address the question of whether metals in undiluted and diluted effluent meet water-quality standards, the SEFLOE studies measured priority pollutant metals and detected several (copper, arsenic, silver, lead) and cyanide in undiluted effluent. Concentrations sometimes exceeded Class III marine water-quality standards (Table 6-10). None of the metals tested in undiluted effluent exceeded the Florida Maximum Allowable Effluent Levels (Hazen and Sawyer, 1994).

Table 6-10. Priority Pollutant Metals Detected in Treated Wastewater Effluent Exceeding Class III Marine Water-Quality Standards

Metal	Concentration in Treated Effluent (μg/L)	Background Concentration in Oceans[a] (μg/L)	Florida Maximum Allowable Effluent Level (μg/L)	Florida Class III Marine Water Standards (μg/L)	EPA Saltwater Criteria (μg/L)	Dilution To Meet Most Stringent Criteria
Broward						
Arsenic, total	BDL, **124**, <1.70, 2.3	1.77	N/A	≤ 50	36	3.6
Copper, total	2.1, 20, **111.3**, 14.4	0.261	N/A	≤ 2.9	2.9	42.1
Lead, total	BDL, 5.0, 4.8, **6.7**	0.0021	≤ 500	≤ 5.6	8.5	1.2
Silver, total	BDL, 0.5, **0.9**, 0.5	0.0028	N/A	≤ 0.05	N/A	19.0
Miami-Dade North						
Arsenic, total	**0.83**, BDL, <10.0[c]	1.77	N/A	≤ 50	36	N/A
Copper, total	**19.0**, 16.0, <10.0[c]	0.261	N/A	≤ 2.9	2.9	7.1
Lead, total	**20.2**, BDL, <5.0[c]	0.0021	≤ 500	≤ 5.6	8.5	3.6
Cyanide	**8.41**, 8.0, <4.0[c]	N/A	N/A	≤ 1	1	8.41
Miami-Dade Central						
Copper, total	**35**, 10	0.261	N/A	≤ 2.9	2.9	13.2
Lead, total	**40**, BDL	0.0021	≤ 500	≤ 5.6	8.5	7.2
Silver, total	**14**[d], BDL	0.0028	N/A	≤ 0.05	N/A	296.6
Cyanide	9.6, BDL	N/A	N/A	≤ 1	1	9.6

Note: Data are from Hazen and Sawyer, 1994, unless indicated otherwise by superscripts.
[a] From Bruland, 1983.
[b] Values shown in boldface represent the highest sample values. The dilutions to meet most stringent criteria are calculated in this report based on these highest sample values.
[c] Miami-Dade North District, 1999. See Appendix Table 1-2.
[d] Questionable value, according to Hazen and Sawyer, 1994.
BDL Below detection limits.
N/A Not available.

Note that with the possible exception of silver at the Miami-Dade Central plant (where the value may be incorrect), the dilution required to meet the most stringent water quality standard varies from 1.2 to 42, depending on the metal and the effluent concentration. The 400 m to 800 m mixing zones required under the Florida regulations are intended to provide dilutions ranging from 60:1 to 90:1 or more, based on modeling of the effluent plume. Also, the concentrations of metals in effluent are measured in the parts per billion range, which is low for industrial effluent.

Both the regulatory criteria for Class III marine water and the effluent studies of South Florida ocean outfalls address total metals concentrations rather than dissolved metals. Since dissolved metals are the most bioavailable, they have the most potential to cause ecosystem toxicity effects. Therefore, the values in Table 6-10 can only be used for a general estimate of risks.

The SEFLOE studies did not specifically report on biota in the vicinity of the outfalls, although Hazen and Sawyer (1994) report that a healthy ecosystem appeared to be present. Thus, there is no information concerning potential effects of metals or other stressors on benthic populations of organisms in the outfall areas.

It is therefore not possible to answer the third question concerning evidence of adverse effects of priority pollutants on marine ecosystems in the area, but it is also not possible to rule out adverse effects. No long-term ecological monitoring studies of possible ecological effects were done following the conclusion of the SEFLOE studies in 1994.

Volatile Organic Compounds

Monitoring data were very limited for volatile organic compounds (VOCs); the only detected compound originates from the Miami-Dade Water Sewer North District, which reported a one-time measurement of tetrachloroethene of 4.66 ug/L on March 19, 1999. The Florida Class III marine water-quality standard for tetrachloroethene is ≤8.85 ug/L on an annual average. Although the SEFLOE report sampled for 126 EPA priority pollutants, including tetrachloroethene and many other organic compounds, there were no other reported detections of tetrachloroethene.

The one data point for VOC concentration in effluent is less than the regulatory standard for VOCs in Class III marine waters, and it is less than the reported literature toxicity values (see Section II). VOCs are highly volatile and would be expected to volatilize as the effluent rises to the upper ocean layer. There is little or no evidence concerning VOCs in ecological receptors. Unfortunately, there are not enough data available to offer firm conclusions on this point. Again, while the effluent toxicity testing suggests that there is no short-term acute toxicity, there are no long-term ecological monitoring studies to examine long-term or cumulative ecological changes that might occur as a result of the discharge of effluent containing trace amounts of VOCs. Thus, for VOCs, the small amount of data available from the SEFLOE report suggests that the amounts of VOCs present in treated discharged effluent are very low and becomes even lower when rapid dilution by currents occurs. The toxicity testing indicates no toxic effects for chronic short-term testing or acute toxicity testing.

Synthetic Organic Compounds

Very little data were available concerning linear alkylbenzenesulfonates (a detergent component used as a representative detergent compound in this study) in Florida wastewater effluent. Effluent data from the Miami-Dade North District detected a concentration of methylene blue anionic surfactant (MBAS) surfactant of 0.063 mg/L in

the effluent prior to discharge (Table 6-11 from Hazen and Sawyer, 1999). This concentration is lower than the regulated Class III standard of ≤0.5 mg/L for detergents. More information on occurrence and levels of surfactants in treated effluent and in receiving waters and their biological effects is needed to adequately evaluate ecological risks posed by this category of compound.

Table 6-11. MBAS Concentrations in Effluent and Calculated Dilution Concentration at 400 m from the Boil

	MBAS in Effluent (mg/L)	MBAS in Effluent (mg/L), 60:1 Dilution	MBAS in Effluent (mg/L), 90:1 Dilution	Background Seawater (mg/L)	Class III Standard for Detergents (mg/L)
MBAS surfactant	0.063[1]	0.001	0.0007	0	≤0.5

[1] Data from Miami-Dade Water/Sewer, North District. 1999. Submission #9903001041, pp. 47-52. Screen effluent collected 3/19/99.

No information is available on monitoring of detergents or other synthetic organic compounds in ecological receptors at or near the effluent outfall.

<u>Hormonally Active Agents</u>

Estrogen equivalences were measured from two grab samples at the Gulfgate and Southgate treatment plants in Sarasota, Florida. Both of these plants treat to advanced wastewater treatment levels and discharge to surface-water creeks. The average concentration of estrogen substances in the treated wastewater effluent was 3.253 nanograms per liter (Frederic Bloettscher, Consulting Professional Engineer, personal communication). At this point, this information only indicates that these substances may be present in treated wastewater effluent intended for discharge into surface water. The literature suggests that, while these concentrations may not induce toxic effects in aquatic organisms, more study is needed concerning the concentrations at which endocrine disruption may occur because of biodegradation byproducts.

No information is available concerning concentrations of estrogen-like compounds in ambient seawater at the southeast Florida ocean outfall sites, nor in ecological receptors at or near the ocean outfall sites. Ongoing and future research should provide a better framework for discussing these compounds and evaluating their risks. Having monitoring data for these constituents in effluent would allow risk to be better evaluated.

6.6.2.4 Toxicity Testing of Effluent

One way to address the question of whether there could be adverse effects from effluent is to conduct toxicity testing of effluent using marine organisms. In order to comply with Florida standards, biological toxicity testing of the diluted and undiluted treated effluent was conducted as part of the SEFLOE studies (Commons et al., 1994a, 1994b) and is

summarized in the Hazen and Sawyer (1994) report. A total of 1,727 acute bioassay toxicity tests and 109 short-term chronic bioassays were performed, using diluted effluent water from four ocean outfall wastewater treatment plants and effluent plume samples. Acute toxicity was assessed using the mysid shrimp *Mysidopsis bahia* and the estuarine fish *Menidia beryllina*. Short-term chronic toxicity testing was assessed using those organisms, the sea urchin *Arbacia punctulata*, and the macroalga *Champia parvula*. The bioassay results were compared with current velocities to determine initial and farfield dilutions and to calculate actual exposure times. This allowed researchers to determine potential toxicity of the undiluted effluent, initial dilution, and mixing-zone effluent/seawater mixture.

In all ocean bioassay tests, no potential acute toxicity of effluent or diluted effluent was demonstrated. The bioassays are believed to be conservative: during the tests using diluted effluent, organisms are exposed to the effluent longer and at concentrations that greatly exceed actual measured concentrations of effluent constituents in the ocean outfall area (Commons et al., 1994a, 1994b; Hazen and Sawyer, 1994).

While toxicity testing indicates that there are no acute toxic effects to biological organisms, long-term low-dose chronic toxicity testing was not conducted. Toxicity testing also does not address effects of nutrient enrichment on ecological processes of production, organic cycling, or microhabitats where nutrients may remain more concentrated. Ecological processes that are not addressed by toxicity testing include nutrient-stimulated primary production and respiration, production of organic matter for consumers and detrital feeders, decomposition of organic matter, and the effects of these processes on water quality and biological communities.

6.6.3 Final Conceptual Model of Probable Risk for Ocean Outfalls

The SEFLOE studies provide a risk assessment and a prediction that there should not be any adverse effects resulting from ocean discharge of secondary-treated effluent. This prediction is based largely on the rapid dispersal and dilution of the effluent plumes by the Florida Current and that the treated effluent has relatively low concentrations of stressors to begin with. Prevailing current directions and fast current speeds of the Florida Current are major factors that decrease risk for the six ocean outfalls that discharge into the Florida Current. Current speeds can be more than 60 or 70 cm/sec for the Florida Current, while speeds of 20 to 40 cm/sec commonly occur. Northerly flow with the fastest speeds occurs approximately 60% of time. Southerly flow with similar or lesser speeds occurs about 30% of time. Flow in other directions (easterly, westerly) exhibits the lowest current speeds and occurs less than 10% of the time. Westerly flow towards the east coast of Florida, which represents the highest risk, is estimated to occur less than approximately 4% of the time, while easterly flow is estimated to occur less than approximately 6% of the time.

Other factors that decrease risk are the distance of the outfalls from land. The lowest risk outfalls are farthest from land (Miami-Dade Central outfall), while the highest risk outfalls are closest to land (Boca Raton, Del Ray Beach). The use of multiport diffusers,

compared to the use of single-port diffusers, appears to aid in dispersal of the effluent plume over a wider area, decreasing potential risk. Discharging the effluent at a faster initial speed also appears to increase the rate of dispersal and dilution of the effluent plume.

Based on toxicity testing of marine organisms, there is no evidence that the diluted effluent causes acute toxic effects or short-term chronic effects.

Based on nitrogen isotope studies of organic matter in sediments and nutrients in the water column, it does not appear that the nitrogen in outfall effluent is taken up in significant amounts by phytoplankton in the area. This may be because of the rapid dilution of the effluent nitrogen by the Florida Current.

The state of Florida requires that Class III water quality standards be met outside a mixing zone of 502,655 m^2 around the outfall. This mixing zone allows for dispersal, mixing, and dilution of the effluent plume. A mixing zone with a circular radius of 400 m measured from the outfall was used by the utilities in the SEFLOE study. This circle would cover an area equivalent to 502,655 m^2. The use of a circular mixing zone is not required by Florida, but is used for ease of defining an area to monitor.

Concentrations of pathogens are controlled at the treatment plant through chlorination to meet water-quality standards within the required mixing zone; viruses and most bacteria are expected to be adequately inactivated by chlorine. However, there is no filtration to remove *Cryptosporidium* and *Giardia*. Lack of treatment to remove pathogenic protozoans probably constitutes the greatest human health risk posed by this wastewater management option.

Pathogenic protozoans may also pose significant ecological risks related to infections of marine mammals. The effects of pathogenic protozoans on aquatic organisms need to be further investigated.

Concentrations of priority pollutant metals in undiluted effluent may exceed marine water-quality standards (but meet effluent standards), but there is no information on actual receptors or exposure pathways because there were no benthic tissue monitoring studies, benthic ecology studies, or studies of trace metals in the water column as part of the SEFLOE studies. The results of the SEFLOE study for metals monitoring indicates that, in general, water-quality standards are met at 400 m or 800 m.

In coastal areas from North Carolina south to Florida, oysters, other shellfish, and sediments have elevated concentrations of arsenic, although not at levels that would pose a threat to humans or to marine life, according to a NOAA National Status and Trends Program report (Valette-Silver et al., 1999). Postulated sources of arsenic include pesticides, mining of arsenic-containing phosphate rocks, atmospheric dust, river and groundwater inputs, and ocean upwelling. The NOAA study did not examine ocean outfalls as potential sources of metals. Since oysters are a nearshore intertidal species, it

is most likely that the arsenic is derived from terrestrial and coastal sources close at hand, rather than the ocean outfalls.

Concentrations of priority pollutant organic compounds in treated wastewater are generally very low. Monitoring data were very limited for volatile organic compounds; the only data available originates from the Miami-Dade Water Sewer North District, which reported a one time measurement of tetrachloroethene of 4.66 μg/L, which meets the Florida Class III annual average marine water-quality standard for tetrachloroethene of ≤8.85 μg/L. There were no reported detections of tetrachloroethene in the SEFLOE study.

Concentration of a surfactant, MBAS, of 0.063 mg/L in the effluent is lower than the regulated Class III standard for detergents of ≤0.5 mg/L. The effects of low concentrations of surfactants on aquatic organisms in natural settings are not well understood or documented. The lack of knowledge concerning effects of surfactants on the tissues and physiologic functions of aquatic organisms is not cause to eliminate this as a potential stressor. Surfactants act to decrease surface tension and reduce adhesion, which may affect microorganisms or for other functions in higher organisms.

Despite the lack of information on effects of endocrine disruptors in South Florida marine waters, effluent discharged to marine waters typically contains such compounds. Endocrine disruptors may pose a concern because they can cause effects in aquatic organisms at very low concentrations and because they are typically present in wastewater and not removed by existing wastewater treatment technology. However, better information on the concentrations of these substances in Florida wastewater, coastal waters, and in aquatic organisms is needed. A better understanding of their effects is also needed.

In summary, the chlorinated discharged effluent largely meets Class III water-quality standards for all regulated wastewater constituents within 400 m of the outfalls, with exceptions as noted.

The lack of long-term ecological, microbial pathogen, and chemical monitoring studies makes it difficult to evaluate whether the conclusions of the SEFLOE studies will continue to hold true in the future. It is not possible to evaluate whether long-term, cumulative, chronic risks exist or not. There are no ongoing monitoring studies downcurrent of any of the effluent plumes or within the footprint of the effluent plume. An initial project to formulate a long-term study to address issues concerning nutrients, growth of nuisance macroalgae (*Codium*), productivity, and the benthic community had begun in the early 1990s, but this project did not go forward at that time. A long-term extensive program is now being contemplated that will examine long-term monitoring of the outfalls and adjacent areas and examine sources of nutrient loading (personal communication, John Proni).

Potential long-term ecological risks may exist, particularly within the 400-m mixing zone, but also outside it. Nutrients, including both nitrogen and phosphorus, may

constitute the most important ecological stressors resulting from ocean outfalls. Nutrient dispersal poses concerns because coastal water quality throughout Florida is already impacted by a variety of human activities on land, such as agriculture, septic systems, urbanization, and channelization of wetlands. The cumulative ecological risks associated with continually discharging nutrients into the Florida Current, and ultimately the Gulf Stream, are not known. The same is true of other effluent constituents, such as metals and organic compounds.

Information needed to assess whether or not there is a long-term, chronic, or cumulative adverse effect on marine organisms would include the following:

- Monitoring of benthic communities in the plume track and adjacent areas
- Tissue studies of bioaccumulation in the food chain
- Monitoring of primary production and nutrient uptake and cycling
- Tracer studies of the sources of nitrogen and phosphorus being utilized by phytoplankton
- Marine particle fluxes of metals in the plume track and adjacent areas to determine whether metals discharged in the effluent adsorb onto marine snow particles or precipitate as solid particles or not
- Related studies of the ecology and chemistry of the ocean within the plume footprint and adjacent to it.

Human health risks are of some concern, both within the 400-m mixing zone and outside of it, primarily because treatment of effluent prior to discharge via ocean outfalls does not include filtration to remove *Cryptosporidium* and *Giardia*. The most probable human exposure pathways include fishermen, swimmers, and boaters who venture out into the Florida Current and experience direct contact, accidental ingestion of water, or ingest fish or shellfish exposed to effluent. Otherwise, there is a very small, but not nonzero, chance for onshore or nearshore recreational or occupational users to be exposed to effluent constituents, since there is a small (10%) chance that currents will change direction to east or west.

Finally, there is the question of whether any adverse effects, if they exist, are reversible. Monitoring studies of Tampa Bay, where tertiary treatment of effluent is now required instead of secondary treatment (see Chapter 7, Surface Water Discharge) indicates that water quality and benthic ecological conditions will improve upon upgrading treatment (Lipp et al., 2001). Even at highly affected marine disposal sites where sewage sludge has been disposed of, cessation of disposal has resulted in improvement of the benthic communities and water and sediment quality (Studholme et al., 1995, 1989). Because the existing southeast Florida ocean outfalls discharge to the Florida Current, recovery from any adverse effects, if they exist, would probably be rapid because of the rapid flushing by the Florida Current.

6.7 Potential Effects of Data Gaps

Because of the relatively short term of the SEFLOE studies (several years), the long term or cumulative ecological risks of nutrient loading and loading of other effluent constituents cannot be evaluated. Some of the specific questions that cannot be answered at this time include:

- Effects of adding nitrogen and phosphorus to the Gulf Stream nutrient budget and its potential to affect primary productivity in the open ocean
- Effects on productivity and marine organisms within the plume where nutrient concentrations are higher than background concentrations
- Potential changes in the ratio of nitrogen to phosphorus and effects on phytoplankton diversity
- Frequency of harmful algal blooms in the vicinity of the outfalls
- Bioaccumulation of effluent constituents by marine organisms in the vicinity of the outfall and its plume footprint
- Changes in trophic structure and potential food-web effects
- Effect of global climate change or other factors on the Florida Current that would cause changes in current speed, direction, or position and affect dilution of the effluent plume, affecting risk
- Long-term, chronic effects of exposure of benthic or nektonic marine organisms to effluent constituents in the vicinity of the effluent plume.

Regarding potential human health risk issues, there are also significant data gaps. Some examples of questions that remain unanswered include the following:

- Are *Cryptosporidium* and *Giardia* present in nearshore waters that are used by humans, are their concentrations within safe limits, <u>and if not</u>, can their sources be determined (for example, onshore sources versus ocean outfalls)?
- Are pathogenic *E. coli*, enteric viruses, and other enteric pathogens present in the treated effluent in numbers high enough to be of concern for human health?
- What is the relative contribution of enteric pathogens and other stressors from existing onsite septic disposal systems and other sources versus ocean outfalls to water quality near the outfalls?

These are just a few of the issues that remain to be addressed if long-term risk from ocean outfalls is to be fully assessed.

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7.0 DISCHARGE TO SURFACE WATERS

In this chapter, the potential risks associated with discharge of treated municipal wastewater into surface-water bodies are evaluated for South Florida.

7.1 Definition of Discharge to Surface Waters

In South Florida, treated wastewater managed by this option is discharged into canals, creeks, and estuaries. At a minimum, wastewater discharged to surface waters must receive secondary treatment with basic disinfection. However, wastewater discharged to some water bodies (for example, Tampa Bay, Indian River Lagoon) must first receive advanced treatment, including nutrient removal.

Florida's Anti-Degradation Policy, which prohibits surface-water resources from being degraded, discourages discharge to surface waters because of the high cost of treatment and the ecological risks, which are generally perceived as high. Even treatment plants that use this option generally do so infrequently, as a backup when other options (for example, reuse) are not available.

7.2 Use of Discharge-to-Surface-Waters Option in South Florida

The discharge-to-surface-waters option is used to varying degrees throughout South Florida. As described in Chapter 2, Figure 2-2, facilities in Brevard, Hillsborough, and Sarasota counties make significant use of this option. Facilities in Hillsborough County rely on this option (roughly 75% of total design capacity) to a greater extent than do facilities in most other counties in South Florida. In Pinellas and Collier counties, treatment facilities use a combination of options, including discharge to surface waters. In Collier County, discharge to surface waters accounts for an insignificant portion (1%) of the total design capacity. Facilities in Broward, Palm Beach, and Dade counties rely primarily upon ocean outfalls and underground injection and do not discharge to surface-water bodies (see Figure 2-2).

The treatment facilities reviewed in this study that discharge to surface waters either discharge directly to estuaries with brackish water, coastal embayments, or to freshwater creeks or canals that eventually discharge to embayments. In Brevard County, the South Beaches and Cape Canaveral wastewater treatment facilities discharge to the Indian River Lagoon only when no other practical alternative exists. The Indian River Lagoon System and Basin Act of 1990, contained in Chapter 90-262, Laws of Florida, "prohibits new discharges or increased loadings from domestic wastewater treatment facilities into surface waters…." (FDEP, 2002a). Exceptions are made if the applicant can meet the following conditions:

- The permit applicant conclusively demonstrates that no other practical alternative exists and that the discharge will be treated to advanced treatment levels or higher

- The applicant conclusively demonstrates that the discharge will not cause or contribute to water-quality violations and will not hinder efforts to restore water quality in the Indian River Lagoon System
- The discharge is an intermittent discharge to surface waters occurring during wet weather conditions, subject to the requirements of applicable Florida Department of Environmental Protection (DEP) rules.

The Act also requires facilities to investigate the feasibility of using reclaimed water to promote reuse and reduce nutrient loadings. Based on these requirements, the Cape Canaveral treatment plant was upgraded in the mid-1990s to provide advanced wastewater treatment (AWT). The new AWT plant is part of a reclaimed water system that supplements the City of Cocoa Beach’s reclaimed water supply. Discharge to the Banana River, a segment of the Indian River Lagoon, is allowed during periods of wet weather or when demand for reclaimed water is low (FDEP, 20021; Cape Canaveral Wastewater Treatment facility, personal communication).

In Hillsborough County, the Howard F. Curren AWT plant serves the city of Tampa. In 2000, the plant managed 48.5 million gallons per day (mgd) using a combination of discharges to Hillsborough Bay (a portion of Tampa Bay) and reuse of reclaimed water for cooling and irrigation (City of Tampa, Florida, 2001). In Sarasota County, the Gulfgate and Southgate treatment plants discharge into two freshwater creeks, Phillippi Creek and Methany Creek. These eventually drain to Roberts Bay (Marella, 1999). Gulfgate has a permit capacity of 1.80 mgd and no reuse capacity. Southgate has a permit capacity of 1.36 mgd and very limited reuse capacity. Both facilities discharge approximately 70% to 80% of their permitted capacity, and each is planning for expanded reuse (Joseph Squitieri, Florida Southwest DEP, personal communication).

7.3 Environment Into Which Treated Wastewater Is Discharged

7.3.1 Estuarine Environments

An estuary is defined as “a semi-enclosed coastal body of water that is connected to the sea and within which seawater is measurably diluted with fresh water from land drainage” (Pritchard, 1967). Estuaries are some of the most productive, diverse, and complex ecosystems on earth. They exhibit tremendous temporal and spatial variability in their physical, chemical, and biological characteristics.

Lagoons are considered a type of estuary. They are produced by wave action and are typically found behind a barrier beach or spit. Lagoons are characterized as being less well drained and are uniformly shallow, often less than 2 meters deep. Physical processes of mixing and circulation in lagoons are mostly wind-dominated, whereas freshwater inflow (from surface water and groundwater) tends to drive mixing and circulation in salt-marsh estuaries.

The Tampa, Sarasota, and Florida bays are representative of estuarine coastal embayments in South Florida. The Indian River Lagoon is an example of a lagoon

system. Tampa Bay, Sarasota Bay, and Indian River Lagoon each receive effluent discharges treated to AWT standards. Although Florida Bay does not receive known or permitted discharges of treated wastewater, there are a number of relevant concerns regarding its water quality and aquatic habitat. These concerns establish a useful context in which to consider risks associated with the discharge-to-surface-waters option.

Potential human health and ecological risks associated with discharges to these environments would be greatly influenced by site-specific flushing rates and the depths of water bodies.

7.3.1.1 Tampa Bay

Tampa Bay is located on the west coast of the Florida peninsula and is part of the Gulf of Mexico. This extremely shallow bay (average depth of 4 meters) is the largest open-water estuary in Florida, encompassing over 400 square miles and with over 100 freshwater tributaries (Pribble et al., 1999). Dominant habitats in the Tampa Bay estuary include sea-grass beds, mangrove forests, salt marshes, and oyster bars. Wildlife is abundant; over 40,000 breeding pairs of birds, such as the brown pelican and roseate spoonbill, nest in Tampa Bay every year. The bay is also home to dolphins, sea turtles, and manatees.

Tampa Bay was heavily polluted before 1979. This pollution largely resulted from discharges of primary-treated wastewater from the Hooker's Point Wastewater Treatment Facility (now the Howard F. Curren Plant) into Hillsborough Bay, a subembayment of Tampa Bay. Since the state of Florida began requiring advanced treatment to remove nitrogen, the bay has been recovering. Water clarity and the health of benthic communities have improved, and sea grasses have reappeared (City of Tampa Bay Study Group, 2001a, 2001b). While the adverse effects of discharged wastewater have been reduced, the bay is still suffering from other pollution sources, particularly atmospheric and nonpoint source loading of nutrients. Sediment quality in Hillsborough Bay remains impaired; 33% of sediments are of marginal quality with respect to metals, and 8% of sediments are of poorer quality (Pribble et al., 1999).

7.3.1.2 Sarasota Bay

Sarasota Bay, located on the Gulf of Mexico in southwest Florida, is another coastal embayment that receives discharges of treated municipal wastewater. The bay is composed of two major embayments, Sarasota Bay and Little Sarasota Bay, and many smaller embayments. The bay is 56 miles long and ranges in width from 300 feet to 4.5 miles. Average depth throughout much of the bay ranges from 8 to 10 feet (Roat and Alderson, 1990). Sarasota Bay exhibits wildlife and habitat that are very similar to Tampa Bay, including mangroves, sea grasses, marine mammals, and waterfowl.

Since 1990, nitrogen discharges from wastewater treatment plants have been reduced by 80% because of the implementation of AWT and reuse programs (Sarasota Bay National Estuary Program, 1993). As a result, water quality and habitat quality have improved.

Sea-grass coverage in the bay has increased by 18% since 1988 (Sarasota Bay National Estuary Program, 2000).

7.3.1.3 Indian River Lagoon

The Indian River Lagoon is located on the east coast of Florida, stretching 156 miles from Ponce de Leon Inlet, south of Daytona Beach, to Jupiter Inlet near West Palm Beach (Adams et al., 1996). The Indian River Lagoon is a lagoonal estuary composed of several water bodies, including the Indian River, the Banana River, and Mosquito Lagoon. The lagoon system receives inputs of salt water via inlets from the ocean. Fresh water is received in the form of direct precipitation, groundwater seepage, surface runoff (discharges from creeks, streams, and drainage systems), and point sources such as wastewater treatment plants. The long narrow shape and shallow waters of the lagoon result in sluggish circulation patterns in many places. Circulation is primarily wind-driven, and tidal exchange is limited to six widely separated inlets with restricted tidal flushing (Adams et al., 1996).

In some areas, habitat loss and alteration have been significant. Portions of the Banana, North Indian, and South Indian rivers have experienced the greatest long-term declines in sea-grass cover within the lagoon system (Adams et al., 1996). Approximately 27% of the mangrove acreage in the Fort Pierce area was lost between 1940 and 1987 (Hoffman and Haddad, 1998). Many salt marshes and mangrove swamps were impounded and flooded to control mosquito breeding.

7.3.1.4 Florida Bay

Florida Bay is located at the southernmost tip of Florida, bounded by the mainland and the Keys. It is a semi-enclosed, shallow, oligotrophic bay, with depths ranging from 6 to 30 feet. The watershed, which discharges to the bay, includes all of the freshwater wetlands south of Lake Okeechobee. This vast area slopes gently and drains towards Florida Bay and the Gulf of Mexico (NOAA, 1999).

Although there are no known discharges to surface waters of municipal wastewater into Florida Bay, conditions in Florida Bay provide examples of many of the natural resource issues confronting wastewater and water managers in South Florida. The Florida Bay hydrologic system has been highly altered, largely through the construction of a complex canal and levee system to control flooding and provide fresh water for agriculture. The U.S. Geological Service (USGS) has been investigating environmental changes that have occurred over the past 150 years within Florida Bay and the surrounding South Florida ecosystem (McPherson et al., 2000; McPherson and Halley, 1996). Recent studies (Boyer et al., 1997, Brewster-Wingard and Ishman, 1999; Brewster-Wingard et al., 1996) have focused on describing temporal and spatial variability within the bay ecosystem. These studies show the following:

- Salinity in the bay has increased since the 1950s

- Before 1940, fluctuations in salinity and sea-grass distribution matched a natural cycle; since 1940, fluctuations have been greater and no longer match a natural cycle
- Sea grass and macrobenthic algae were much less abundant in the 1800s (and early 1900s) and have increased in the last half of the 20th century
- Invasive plants (for example, cattails) have increased in number and are slowly displacing the native saw grass communities along the canals that form part of the drainage system to Florida Bay
- Regional ecosystem disturbances occurring in the late 20th century have been accelerated by human activities
- Between 1991 and 1994, in the central region, nitrate, ammonia, and chlorophyll *a* increased
- Over the past 7 years, concentrations of phosphate and total phosphorus decreased dramatically throughout the bay
- The bay is becoming more phosphorus-limited from west to east.

In recent times, the bay has experienced sea grass die-offs, algal blooms, and declines in the populations of shellfish and sponges (USGS, 1996a). In western Florida Bay, a massive sea grass die-off began in 1987. Since then, some recovery of sea grasses has occurred, while other areas have been slow to revegetate. Algal blooms have been reported in the last few years across western Florida Bay, extending to the Florida Keys (NOAA, 1999).

7.3.2 Freshwater Environments

Much of the information that informs this analysis of the discharge-to-surface-waters option was obtained from treatment facilities located in Brevard, Hillsborough, and Sarasota counties. These facilities discharge directly to estuaries or to creeks or canals that discharge to an estuarine environment. This study did not reveal any effluent discharges to freshwater lakes or ponds in South Florida.

Florida's surface-water features include extensive wetlands and numerous lakes, streams, and canals. Streams and wetlands in South Florida have direct hydrologic connections to the surficial aquifer (Randazzo and Jones, 1997). Much of South Florida was originally covered with wetlands. Canals, which are a prominent surface-water feature in South Florida, were dug to drain these wetlands and make the land useable. Canals are the major surface-water drainage feature in South Florida outside of the Everglades (Englehardt et al., 2001). Many canals that receive effluent discharges subsequently empty into saltwater bodies.

Canals are generally man-made waterways or artificially improved rivers; they serve various uses such as irrigation, shipping, recreation, and flood control (Kapadia and Swain, 1996). They vary in size from a few feet wide and deep, to several hundred feet wide and 12 to 15 feet deep. Some canal banks are earthen, while others are encased in concrete.

Surface-water quality throughout large areas of South Florida has already been degraded by human activities, as summarized in two USGS reports on the National Water Quality Assessment (NAWQA) Program Study of South Florida. The USGS made several major findings concerning surface-water quality in South Florida (McPherson et al., 2000; McPherson and Halley, 1996):

- Concentrations of total phosphorus at NAWQA sites in South Florida exceeded the Environmental Protection Agency's (EPA's) Everglades water-quality standard of 0.01 milligram per liter (mg/L) and were above Everglades background levels. A major source of the phosphorus is fertilizer from agriculture.
- Dissolved organic carbon (DOC) concentrations were relatively high when compared with those in other waters of the United States. High DOC concentrations provide food for microorganisms to grow, reduce light penetration in water, and enhance transport and cycling of pesticides and trace elements, such as mercury.
- Pesticides were detected in almost all South Florida NAWQA samples. Most concentrations were below aquatic-life criteria, but the criteria do not address cumulative effects of mixtures of pesticides or their degradation products, which were common in the samples. Organochlorine pesticides, such as DDT and its degradation products, are still prevalent in bottom sediment and fish tissue at South Florida NAWQA sites, even though use of these pesticides has been discontinued in recent decades.
- Exotic plants and animals pose a threat to native biota, and herbicides that were used to control exotic plants were detected in surface water at NAWQA sites.
- Of 21 NAWQA areas nationwide, the Everglades has the second highest enrichment of methylmercury relative to mercury in sediments; methylmercury is highly biologically active and can be taken up by biota.
- The frequency of external anomalies (lesions, ulcers, and tumors) on fish collected at two NAWQA sites in South Florida places these sites among the top 25% of 144 NAWQA sites sampled nationwide. Such anomalies may indicate that fish are stressed by contamination.

The NAWQA study found that major causes for degradation of surface-water quality include modification of drainage patterns, wetland destruction, runoff from agricultural and urban areas, high concentrations of DOC and its effects on mercury transport and light transmission, and release of exotic species.

The USGS also collected water-quality samples between 1996 and 1997 within selected southeast canals that show increases in nutrient concentration corresponding to patterns of land use. For example, nitrate concentrations were highest in agricultural areas; ammonia and total and inorganic phosphorus concentrations were highest in urban areas; total organic nitrogen was highest in wetlands (Lietz, 2000).

In summary, surface-water quality in South Florida shows significant degradation as an apparent result of urban and agricultural activities. Canals in areas of urban and

agricultural land use commonly contain water with high concentrations of nutrients, coliform bacteria, metals, and organic compounds when compared to water taken from areas that are remote from these canals. Wildlife has been stressed by human alteration of the hydrologic regime and by the addition of nutrients, sediment, and other pollutants to surface-water bodies (McPherson et al., 2000; McPherson and Halley, 1996).

7.4 Option-Specific Regulations and Requirements

This section describes regulations concerning treatment and discharge of wastewater to surface-water environments.

7.4.1 Treatment and Disinfection Requirements

At a minimum, treatment prior to discharge to surface water must include secondary treatment with basic disinfection (Florida Administrative Code [FAC] 62-600.510(1)). When discharges to surface waters is used as a backup to reuse systems, wastewater is frequently treated to reclaimed-water standards before being discharged. Discharge to Class I drinking waters requires principal treatment, which consists of secondary treatment and high-level disinfection (see Chapter 2). Discharge to waters contiguous to Class I waters requires review of the travel time of effluent to the drinking-water intake; the discharge must also meet Technology Based Effluent Limits (TBEL) or Water Quality Based Effluent Limits (WBEL), as established by the permit. The Florida DEP may require that a facility meet additional water-quality-based effluent limits; these provide and enforce more stringent requirements for effluent quality. TBELs and WBELs are based on the characteristics of the discharge, the receiving-water characteristics, and the criteria and standards of FAC 62-302.

Effluent discharge must not exceed 10 mg/L total nitrogen (FAC 62-600.420(2)(a)(2)), and effluent must contain maximum pollutant levels less than those specified for community water systems in FAC 62-550. These facilities must be designed to reduce total suspended solids to 5.0 mg/L or less before the application of disinfectant (FAC 62-600.540(5)(e)).

In order to be permitted to discharge to either Tampa Bay or the Indian River Lagoon, wastewater treatment plants must treat using AWT. Typically, AWT includes secondary treatment, basic disinfection, nutrient removal (nitrification, denitrification, and phosphorus removal), additional removal of metals and organic compounds, and filtration. Dechlorination is also required (see Appendix Table 1-1). AWT standards must be met on an average annual basis. AWT standards are summarized as follows:

- Carbonaceous biological oxygen demand ($CBOD_5$) must be less than 5 mg/L
- Total suspended solids must be less than 5 mg/L
- Total nitrogen (as N) must be less than 3 mg/L
- Total phosphorus (as P) must be less than 1 mg/L
- Discharge to a treatment or receiving wetland may not exceed 2 mg/L total ammonia (as N) on a monthly average.

Some treatment plants utilize wetland treatment before discharge into surface-water bodies; this provides further reductions in nutrient concentrations prior to discharge.

Basic disinfection (no more than 200 fecal coliform colonies per 100 milliliters (mL)) is a minimum requirement for all discharges to surface waters in Florida. High-level disinfection (fecal coliform removal below detectable limits per 100 mL) is required of all facilities discharging to Class I surface waters. Intermediate-level disinfection may be allowed, if discharge is to wetlands with restricted public access (FAC 62-600.440(5)g) or to surface waters that serve as backup to a reuse system and provided that there is no discharge to Class I waters or their tributaries (FAC 62-600.440(5)(h)). Dechlorination of chlorinated wastewater before discharge to surface waters is also required (see Tables 2-4 and 2-5).

Currently, there are no federal or state limits for concentrations of the pathogens *Giardia lamblia* or *Cryptosporidium* in treated wastewater. However, on January 1, 2002, the EPA did establish drinking-water treatment requirements for these pathogenic microorganisms. The EPA mandated drinking-water treatment to remove 99.9% of *Giardia lamblia* and 99% of *Cryptosporidium* from raw water sources (National Primary Drinking Water Standards, CFR 141). Florida DEP applies a numerical standard (no more than 5.8 cysts or oocysts per 100 L, which corresponds to a 1 in 10^{-4} human illness risk) for *Cryptosporidium* and no more than 1.4 cysts per 100 L for *Giardia* in reclaimed water (York et al., 2002). These recommended limits address the significant human health risks that may be associated with ingestion of viable pathogenic protozoans present in unfiltered or inadequately filtered treated wastewater.

7.4.2 Standards for Surface-Water Quality

In addition to discharge standards, Florida has use and classification standards for surface-water bodies (FAC 62-302.530). The standards are meant to protect the designated use of the water bodies. Table 7-1 summarizes the uses and criteria for some of the relevant stressors reviewed in this study (FAC 62-302.530).

Table 7-1. Criteria for Surface-Water Quality Classifications

Parameter	Units	Class I: Potable-Water Supply	Class II: Shellfish Propagation or Harvesting	Class III: Recreation, Propagation, and Maintenance of a Healthy Well-Balanced Population of Fish and Wildlife	
				Fresh	***Marine***
Fecal coliform bacteria	Numbers per 100 mL	MPN or MF counts cannot exceed monthly average of 200, nor exceed 400 in 10% of samples, nor exceed 800 on any day. Monthly averages must be based on minimum of 5 samples taken over a 30-day period.	MPN shall not exceed a median value of 14, with not more than 10% of the samples exceeding 43, nor exceed 800 on any day.	MPN or MF cannot exceed monthly average of 200, nor exceed 400 in 10% of samples, nor exceed 800 on any day. Monthly averages must be based on minimum of 5 samples taken over a 30-day period	MPN or MF counts shall not exceed monthly average of 200, nor exceed 400 in 10% of samples, nor exceed 800 on any day. Monthly averages must be based on minimum of 5 samples taken over a 30-day period
*Copper	µg/L	Cu ($_e$(0.8545[lnH]-1.465)	2.9	Cu ($_e$(0.8545[lnH]-1.465)	0.9
Nitrate	mg/L	10, or concentration that exceeds nutrient criteria.			
Nutrients		Discharge of nutrients is limited as needed to prevent violations of other standards. Man-induced nutrient enrichment (total nitrogen or total phosphorus) is considered degradation (Section 62-302.300, 62-302.700, and 62-4.242 FAC). Nutrient concentrations in a body of water cannot be altered so as to cause an imbalance in natural populations of aquatic flora and fauna.			
Phosphorus	µg/L		0.1		0.1

*Florida surface-water quality standards for metals were used as assessment endpoints. The standard for copper in Class I and Class III freshwater bodies takes into account water hardness ($CaCO_3$) and provides a range from 0.00361 mg/L to 0.036 mg/L (corresponding to a range in $CaCO_3$ from 25 to 400 mg/L).
MPN = most probable number
MF = membrane filter

In addition to the above classes of water bodies, Florida has a category for Outstanding Florida Waters and Outstanding National Resource Waters. This generally refers to waters of exceptional recreational or ecological significance that are found within national and state parks and wildlife preserves. A complete listing is available under 62-302 and includes the waters of the Everglades National Park. These waters fall under Florida's Antidegradation Policy and are afforded the highest protection.

In December 2000, the EPA published recommendations for ambient freshwater quality criteria for different regions around the country. These water-quality goals or recommendations are intended to assist states and tribes in establishing nutrient limits for water bodies that are consistent with Section 303(c) of the Clean Water Act. These criteria are recommended, not required.

Using historical data and reference sites, the EPA determined that the unimpacted lakes and reservoirs of South Florida (Ecoregion XIII) had a mean background predevelopment total nitrogen concentration of 1.27 mg/L (US EPA, 2000a). The 3 mg/L standard for treating nitrogen before discharge represents a concentration that is 2.4 times higher than this background.

Similar mean background predevelopment nitrogen concentrations for rivers and streams in South Florida are not currently available. In Ecoregion XII, which includes central and northern Florida (as well as portions of Alabama, Georgia, and Mississippi), the EPA recommends a background total nitrogen concentration of 0.9 mg/L in streams and rivers (US EPA, 2000b). The 3 mg/L standard for treatment before discharge represents a concentration that is approximately 3.3 times higher than this background level.

Total phosphorus includes all forms of phosphorus, both inorganic and organic. For streams and rivers in nearby Ecoregion XII, the EPA recommends a total background phosphorus water-quality criterion of 40.0 μg/L, or 0.040 mg/L (US EPA, 2000b). This is two orders of magnitude lower than the AWT treatment standard. Florida regulations require that plants that discharge to surface-water bodies treat wastewater so that the final concentration of total phosphorus in the discharged effluent is 1 mg/L. The EPA has determined that the unimpacted lakes and reservoirs of South Florida (Ecoregion XIII) had a mean background predevelopment total phosphorus concentration of 17.50 μg/L, or 0.0175 mg/L (US EPA, 2000a). The standard for AWT-treated wastewater, 1 mg/L, represents a concentration 57 times larger than this recommended background level for lakes and reservoirs.

7.5 Problem Formulation

Human health and ecological risks that may be associated with the discharge-to-surface-waters option are expected to be highly site-specific. There may be substantial differences of scale in important physical processes and variations in the assimilative capacity of individual water bodies. Therefore, this option-specific risk analysis focuses on whether surface-water quality standards are likely to be exceeded by actual discharges. This is coupled with a review of the types of adverse effects that might be anticipated where surface-water quality standards are exceeded. Implicit in this approach is an assumption that surface-water quality standards are adequately protective of human and ecological health. For one area where this assumption may be suspect (standards for nutrient discharges), a set of surface-water quality recommendations serve to expand this analysis to include additional considerations.

7.5.1 Potential Stressors

Potential stressors entrained or dissolved in treated wastewater are discharged to surface-water outfalls located in canals, creeks, or estuaries. Wastewater constituents that may act as stressors to human or ecological health include nutrients (nitrogen and phosphorus), certain metals, organic compounds, pathogenic microorganisms, and hormonally active agents. A group of potential “secondary stressors” (for example, shifts in community

structure and productivity) may at the same time be caused by the presence of wastewater constituents and, in turn, be the cause for additional adverse effects. Secondary stressors include such things as changes to plant, invertebrate, and fish community structure; growth of invasive species; reduction in oxygen levels; and harmful algal bloom.

7.5.1.1 Nutrient Stressors

Because most, if not all, of the permitted discharges to surface waters eventually reach coastal embayments, the risk assessment of these discharges resembles the risk assessment of the ocean outfall option in many ways. Nutrient stressors are an example. Nutrients act as ecological stressors when present in surface waters at sufficient concentration to overstimulate primary production (leading to eutrophic conditions) or otherwise cause adverse changes in ecosystem health or structure (for example, loss of native species, growth of invasive species).

Nitrogen limitation in coastal and ocean waters was reviewed in Chapter 6 (see Paerl, 1997; Dugdale, 1967; Ryther and Dunstan, 1971; Codispoti, 1989; Eppley, et. al., 1979). Freshwater ecosystems are typically characterized by phosphorus limitation (Schindler, 1977, 1978; Smith, 1982). Phosphorus limitation is generally stems from low levels of naturally occurring dissolved inorganic phosphorus. However, ecosystem responses to additions of phosphorus will depend on both the levels of additional phosphorus made available and the levels of nitrogen that are latent in the ecosystem, often as a result of human activity (such as agricultural inputs). In Florida, natural ambient levels of phosphorus may be higher than in other areas of the country because of high phosphorus content in the regional geology (Valette-Silver et. al., 1999).

The National Research Council concluded that, while nitrogen is important in controlling primary production in coastal waters and phosphorus is important in freshwater systems, both need to be managed to avoid overproduction (National Research Council, 2000). The causes of eutrophication in fresh and marine ecosystems are not identical but do reflect ecological and biogeochemical processes. In either case, the relative inputs of nitrogen and phosphorus and the extent to which nitrogen fixation can alleviate limitation play a crucial role in determining the limiting nutrient to production in aquatic ecosystems. The limiting nutrient is the nutrient in shortest supply in a natural system. In marine waters, nitrogen is generally present in low concentrations, while in fresh water, phosphorus is present in low concentrations.

While phosphorus limitation in fresh water seems universal, there are exceptions to the general principle that nitrogen is limiting in coastal ecosystems. For example, the Apalachicola estuarine system on the Gulf coast of Florida appears to be phosphorus-limited (Myers and Iverson, 1981). Howarth (1988) and Billen et al. (1991) suggest that this is related to the relatively high ratio of nitrogen to phosphorus inputs. Howarth et al. (1995) suggests that there is a tendency for estuaries to become more nitrogen-limited as they become more affected by humans and as nutrient inputs increase overall.

In nearshore tropical marine systems, phosphorus tends to be more limiting for primary production (Howarth et al., 1995). In some major estuaries, nutrient limitation switches seasonally between nitrogen and phosphorus. Examples of such seasonally varying nutrient limitation include the Chesapeake Bay (Malone et al., 1996) and portions of the Gulf of Mexico, including the so-called “dead zone” (Rabalais et al., 1999). Tampa Bay has become a nitrogen-limited system instead of a phosphorus-limited system because of the long-term mining of phosphorus. In contrast, Florida Bay is phosphorus-limited (Bianchi et al., 1999).

7.5.1.2 Metals

Trace metals in wastewater are potential stressors because they may cause adverse human health and ecological effects at high concentrations. Trace metals are frequently elevated in wastewater as a result of common industrial usage. Levels in treated wastewater are, in general, greatly reduced, but trace metals are still frequently used as tracers of wastewater in the aquatic environment (Matthai and Birch, 2000; Flegal et al., 1995; Hershelman et al., 1981; Ravizza and Bothner, 1996; Morel et al., 1975). Additional sources of metals that may contribute to levels present in surface-water bodies include combustion of fossil fuels, mining activities, stormwater runoff, atmospheric deposition, and other surface-water and groundwater sources (Burnett et al., 1980; Finney and Huh, 1989; Forstner and Wittman, 1979; Huh et al., 1992; Huntzicker et al., 1975; Klein and Goldberg, 1970).

Metals can bioaccumulate in the food chain, thus having adverse secondary impacts on an ecosystem. For example, arsenic may bioaccumulate in aquatic organisms. However, there is considerable variability in aquatic food-web bioaccumulation (Penrose et al., 1977; Woolson, 1977). See Chapter 3, Methodology, for further description of metals as a potential stressor in the environment.

7.5.1.3 Organic Compounds

Potential organic stressors that may be present in treated wastewater include volatile organic compounds (VOCs), synthetic organic compounds (such as pesticides, herbicides, surfactants), trihalomethanes, and some hormonally active agents (endocrine disruptors). See Chapter 3, Methodology, for a further description of organic compounds as potential stressors in the environment.

Hormonally active agents may have potentially adverse effects on aquatic organisms, based on the scientific literature. A study conducted in the United Kingdom found that wastewater induced vitellogenin synthesis in caged and wild fish several kilometers downstream of points of discharge (Rodgers-Gray et al., 2000); vitellogenin is a protein important to yolk production. These effects were induced at dilutions of treated wastewater ranging from 9.4% to 37.9%. Similar studies were conducted in the United States. However, there was no apparent vitellogenin induction in fathead minnow (*Pimephales promelas*) in response to exposure to treated wastewater (Nichols et al., 1998).

Studies in Florida have documented potential adverse effects from exposure to hormonally active agents in upland and freshwater organisms, including the Florida panther (Facemire, et al., 1995) and American alligator (Guillette, 1994, Semenza, 1997). However, these studies do not document the sources of these agents.

These studies indicate that hormonally active agents may be capable of causing potentially adverse health effects in aquatic organisms. However, more information is needed to determine how these compounds cause adverse reactions.

7.5.1.4 Pathogenic Microorganisms

Pathogenic stressors that may be present in treated wastewater include enteric bacteria, protozoans, and viruses associated with human or animal wastes. Secondary treatment, chlorination, and filtration generally remove all viruses, helminthes, and pathogenic bacteria. However, the protozoans *Giardia* and *Cryptosporidium* form cysts that are resistant to chlorination and that can only be removed through careful filtration. The Florida DEP has evaluated monitoring data from reclaimed-water treatment facilities that treat wastewater intended for reuse or discharge to surface waters. Wastewater treated at some facilities still contains levels of *Cryptosporidium* and *Giardia* that may pose human health risks, despite chlorination and filtration (York et al., 2002).

Much of the information concerning survival and transport of pathogenic protozoans discussed in Chapter 4 applies to discharges to surface waters. *Cryptosporidium* oocysts, for example, have a T_{90} (that is, the time needed to inactivate 90% of the population) of approximately 200 days (Robertson et al., 1992). This time frame is long enough that discharged effluent traveling over short distances and short travel times may still contain some pathogenic protozoans.

Contamination of Florida's coastal environments with enteric viruses, bacteria, or protozoans is a widespread and chronic problem. This is notably the case for Tampa Bay, Sarasota Bay, and the marine environment surrounding the Florida Keys. There are a number of potential causes for this. They include the prevalence and high density of onsite sewage-disposal systems (such as septic systems), the presence of predominantly porous and sandy soils, and karst topography and the hydrologic connection between groundwater and coastal embayments and estuaries (Lipp et al., 2001; Paul et al., 1995).

7.5.1.5 Secondary Stressors

Secondary stressors are the result of exposure to the potential stressors discussed above and include the following:

- Increased primary productivity
- Increased oxygen demand and hypoxia
- Shifts in community structure caused by anoxia and hypoxia
- Changes in phytoplankton community structure
- Harmful algal blooms

- Marine mammals and human impacts from harmful algal blooms
- Degradation of sea-grass and algal beds and formation of nuisance algal mats
- Coral reef destruction
- Trophic impacts.

Sea-grass degradation in Tampa Bay, Sarasota Bay, and Indian River Lagoon has been attributed to nutrient loading, from both point and nonpoint sources. Sea grass serves as a valuable habit for juvenile fish, some marine mammals, and shellfish as it provides food, oxygen, and refuge. In addition, sea grass stabilizes the bottom substrate, keeping sediment out of the water column. The loss of sea grass can also cause secondary effects by adversely affecting other species that utilize this habitat. Nutrient loading that increases phytoplankton populations can damage sea grass; this in turn decreases light transmission to the substrate.

The increase in production can also result in increased organic loading that, upon decomposition, utilizes oxygen, thus creating hypoxic or anoxic conditions. These conditions can result in fish kills or a decrease in available fish habitat.

Changes in nutrient concentrations in the water column can alter the phytoplankton community structure. This may result in increased nuisance or harmful algal blooms. In addition, the availability of silica and iron appears to play a role in coastal eutrophication and may promote the formation of harmful algal blooms (National Research Council, 2000).

Harmful algal blooms (HABs) pose particular concerns in brackish, coastal, and estuarine environments. Harmful algal blooms taxa and associated problems in coastal or estuarine environments are described in the Chapter 6. The causes of harmful algal blooms are still controversial. They include a variety of physical, chemical and biological changes, such as climate change, increased pollution and nutrient inputs, habitat degradation through dredging, resource harvesting and regulation of water flows, failure of grazers to control algal growth, and better monitoring. It is uncertain whether higher numbers of harmful algal bloom reports in recent years are a result of an actual increase in harmful algal blooms or better water-quality monitoring.

Harmful cyanobacterial ("blue-green") algal blooms can occur in warm stratified areas in embayments and estuaries, where nitrogen concentrations are low, salinities are reduced, and phosphorus is enriched through upwelling, eddies, or mixing. Phosphorus limitation is generally more important than nitrogen limitation (Sellner, 1997). In Florida, extensive blooms of the cyanobacterium *Lyngbya majuscula* were documented in Tampa Bay in 1999 and from Sarasota Bay to Tampa Bay in 2000. Although this species is not toxic, it is a nuisance alga because it produces large, slimy, brown odorous floating mats (Florida Fish and Wildlife Conservation Commission, 1999). The causes for this bloom are unknown; it is not believed that discharges of treated effluent played a significant role.

Harmful algal blooms of *Gymnodinium breve* occur frequently off the southwest coast of Florida, especially from Clearwater to Sanibel Island, occurring in 21 of the last 22 years

(Boesch, et al., 1997). Blooms move inshore and can have impacts on the health of humans or wildlife. In 1996, more than 150 manatees died from exposure to brevetoxin during prolonged red tides along the southwest coast of Florida (Steidinger et al., 1996). There is some evidence that dense blooms of *Gymnodinium* rely on new nutrient inputs; human impacts to watersheds may be responsible for extending the duration and adverse effects of red tides once they enter nearshore areas (Boesch et al., 1997).

Effects of secondary stressors also include changes in trophic processing of organic matter, uptake and bioaccumulation, biodiversity and populations, and growth of invasive species displacing native species.

7.5.2 Potential Receptors and Assessment Endpoints

Assessment endpoints represent discrete natural resource values or functions deemed important to local ecology or natural communities. Water-quality standards are set based upon such endpoints. For example, maintenance and protection of aquatic life might be one such endpoint. Other endpoints might be fishable and swimmable waters. Water-quality criteria then would be set, based on reaching that goal. As discussed in section 7.4.2, Florida uses a class system to designate uses of water bodies and applies water-quality standards to meet those uses.

The water-quality standards are set based upon the best science available and are conservative. Still, there are many unknowns and uncertainties, particularly when setting standards related to protecting complex ecosystems. For example, many times numerical standards are not set for nutrients in water bodies because the ecoystem effects are very site-specific.

Canals, which are a frequent receptor for discharge of treated wastewater into surface-water bodies, are often hydrologicly connected to groundwater and are recharged by groundwater. Adams (1991) examined water in the surficial aquifer and canals in Martin and Northern Palm Beach counties and concluded that groundwater quality did not seem to be affected by canal water, probably because the aquifer is discharging to the canal rather than the canal recharging the aquifer. However, water from canals may enter the surficial aquifer when canals are used as an irrigation source. Drinking-water receptors (underground sources of drinking water (USDWs) or water-supply wells) may be exposed where surface waters have a direct hydrologic connection to the groundwater resource

7.5.3 Potential Exposure Pathways

When human health or ecological receptors are exposed to wastewater constituents in sufficient concentration, these receptors may be at risk for potentially adverse health effects. Complex processes and interactions govern how wastewater discharged to surface waters will move and behave. These processes and interactions define the pathways that may expose receptors to stressors present in treated wastewater.

Potential transport processes include advective transport in stream and nearshore currents, and estuarine and tidally driven circulation. The action of these transport processes varies substantially over time and space. Patterns and mechanisms of transport are often quite different in water bodies of different sizes, shapes, and orientations. Transport processes can also vary substantially within water bodies, over the course of time, and in response to localized changes in depth, currents, temperature, and many other factors.

The capacity of water bodies to dilute or assimilate wastewater constituents is fundamentally important to the fate of potential stressors in surface-water ecosystems and to the risks that may be posed by such stressors. In this respect, the rate of flow through a canal or creek and the rate of flushing for an embayment or lagoon are key parameters that influence both fate and risk. In general, adverse effects are expected to be greater in smaller surface-water bodies that flush slowly than in larger water bodies that are well flushed.

Sedimentation and flocculation are important physical and chemical processes that can act to take wastewater constituents out of the water column. Turbulent mixing and resuspension frequently act to counteract these processes, setting up a dynamic equilibrium in which materials are exchanged (over time and space) between the water column and sediment layer. Where conditions are conducive to sedimentation or flocculation, the sediment layer can become a sink, potentially affecting local flora and fauna at the sediment interface.

Potential exposure pathways for ecological receptors include direct ingestion of water or sediments, dermal contact and other forms of uptake (for example, diffusion into submerged plants and soft-bodied invertebrates), and bioaccumulation or food-chain bioconcentration. Ecological receptors are exposed to secondary stressors, such as the disappearance of favorite prey items or reduced levels of available oxygen, through their trophic relationships and position within the larger biological community.

Potential human exposure pathways include direct ingestion or dermal contact with surface water and ingestion of contaminated fish, shellfish, or other plants and animals exposed to treated wastewater. Drinking-water receptors may be exposed where surface waters have a direct hydrologic connection to the groundwater resource.

7.5.4 Conceptual Model of Potential Risk for the Discharge-to-Surface Waters Option

Figure 7-1 presents a generic conceptual model for the discharge-to-surface-waters wastewater management option. The primary source of potential stressors is defined as the wastewater treatment plant from which treated effluent is routed to one or more surface-water outfalls. The rate of discharge may vary, depending on the size and operational status of the facility. but is generally measured in millions of gallons per day.

Treated wastewater is discharged directly to surface-water bodies. These are predominantly small, flowing, fresh-to-brackish bodies of water (canals, creeks, and estuaries). According to the Florida DEP, discharge to closed bodies (ponds and lakes) is no longer practiced in South Florida. Wastewater is typically treated to a higher level than effluent discharged through ocean outfalls. Treatment includes secondary treatment and basic disinfection, followed by filtration and, in some cases, nutrient reduction and dechlorination to remove harmful chlorination by-products. In the model, nutrient limitation varies, depending on whether disposal into freshwater, estuarine, or coastal marine waters is conducted.

Potential ecological receptors include the wildlife, waterfowl, fish, and invertebrates that are dependent on canals, estuaries, and other surface-water ecosystems for food and habitat.

Potential human receptors include recreational fishermen, swimmers, agricultural workers, and others whose work or recreation brings them into close proximity or contact with surface-water bodies that receive effluent discharges. Waters classified as fishable and swimmable are assessment endpoints meant to protect these ecological receptors.

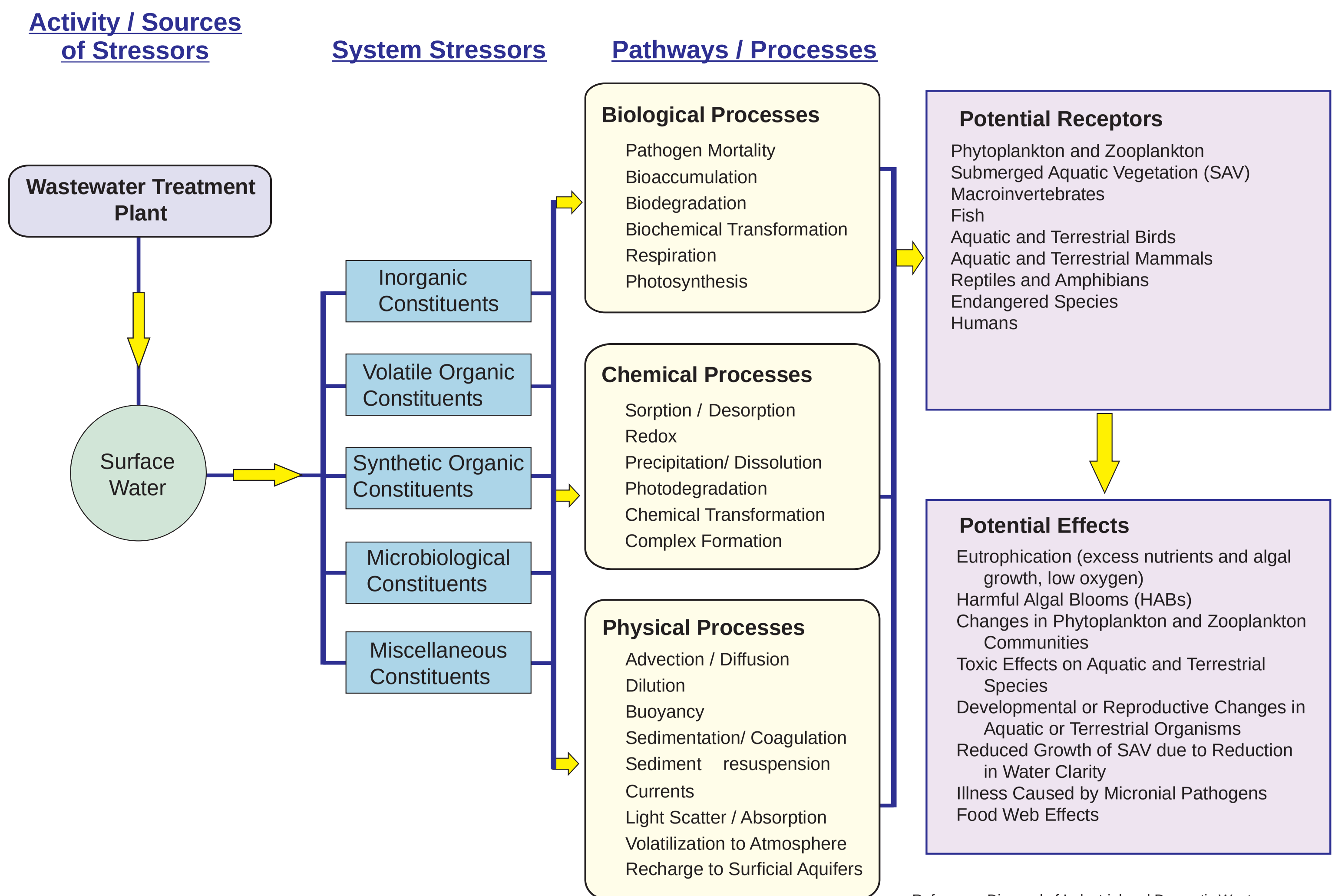
Drinking-water receptors may be exposed to wastewater when surface waters have direct hydrologic connection to the groundwater resource. While this study did not find any evidence of wastewater discharging to surface waters in direct connection to groundwater wells in South Florida, it is a consideration when analyzing potential receptors.

7.6 Risk Analysis of the Discharge-to-Surface-Waters Option

In this section, data are integrated into the conceptual model for the discharge-to-surface-waters option. Actual data on stressors, receptors, and exposure pathways are used to examine potential risks.

Discharge monitoring data from several public treatment facilities, as well as a database provided by the Florida DEP (2002b), were used to examine where (and to what extent) the discharge-to-surface-waters option is used in South Florida. Staff from Florida DEP assisted in determining which options are utilized by specific treatment facilities (personal communication, Kathryn Muldoon, February, 2002).

Information to describe the volume and quality of treated wastewater discharged to surface waters was limited. In order to characterize potential stressors and stressor concentrations, data were obtained from three AWT plants that discharge to surface waters (the City of Cape Canaveral and South Beaches treatment facilities in Brevard County and the Howard F. Curren treatment plant in Hillsborough County). In addition, information on AWT effluent managed at two wastewater treatment plants in Sarasota County (Gulfgate and Southgate Wastewater Treatment Plants) was obtained from the report by Englehardt et al. (2001) (Appendix Table 1-1). No data were available to characterize discharges to surface waters treated to less-than-AWT standards.



Reference: Disposal of Industrial and Domestic Wastes: Land and Sea Alternatives, National Academy of Sciences, 1984.

Figure 7-1. Conceptual Model of Potential Risks for the Surface Water Option

To describe the proximity and vulnerability of receptors, information was obtained regarding biological communities present in the receiving water bodies, particularly sensitive or vulnerable populations. A review of the scientific literature provided information about potential exposure pathways, adverse impacts, and risks. Wherever available, previous studies and investigations were used to appropriately expand the scope of this analysis.

7.6.1 Evaluation of Stressors and Assessment Endpoints

7.6.1.1 Nutrients

Annual average concentrations of total nitrogen in treated wastewater for 1999 and 2001 were calculated from monthly monitoring report averages for the City of Cape Canaveral's AWT wastewater treatment plant (Appendix Table 1-1). The annual average concentration of total nitrogen during this period ranged from 0.752 to 0.970 mg/L; the maximum monthly average was 1.353 mg/L, and the minimum monthly average was 0.321 mg/L of total nitrogen. These values are well below the 3 mg/L AWT standard for treatment. Background concentrations of nitrate (a component of total nitrogen) at two ocean locations off the east coast of Florida reported in Hazen and Sawyer (1994) were 0.11 mg/L and 0.16 mg/L. One monitoring result for nitrate for the City of Cape Canaveral's wastewater treatment plant revealed a nitrate concentration in treated effluent of 0.062 mg/L. This is an order of magnitude lower than background concentrations of nitrate reported for the SEFLOE studies in an open ocean environment (summarized in Hazen and Sawyer, 1994).

Annual average concentrations of total phosphorus for 1999 and 2001 were calculated from monthly monitoring report averages for the City of Cape Canaveral's AWT wastewater treatment plant (Appendix Table 1-1). The annual average concentration of total phosphorus during this period ranged from 0.119 to 0.152 mg/L; the maximum monthly average was 0.273 mg/L, and the minimum monthly average was 0.064 mg/L total phosphorus. The annual average concentrations of total phosphorus are higher than recommended background levels for total phosphorus in fresh water. Thus, the excess phosphorus may pose some ecological risks.

Permitted concentrations of nitrogen and phosphorus (3 and 1 mg/L, respectively) in AWT-treated effluent discharged to surface waters are often greater than background concentrations in unimpacted water bodies. Phosphorus concentrations in AWT effluent were generally significantly higher than recommended background concentrations for fresh waters. However, as indicated above, actual nitrate concentrations in AWT effluent can be lower than background oceanic nitrate concentrations.

Long-term water-quality and biological monitoring in Hillsborough Bay indicates that water quality and clarity have improved and shoal grass (*Halodule wrightii*) has recovered since AWT was implemented at the Howard F. Curren Wastewater Treatment Plant (City of Tampa Bay Study Group, 2001b).

Given the limited use of this disposal option and limited data on actual discharged effluent, it is difficult to estimate risk for this option except in these more general terms relating to water-quality standards. Nevertheless, nutrient loading is one of the top reasons for impairment of surface-water bodies in Florida. It is likely that point sources are part of this larger problem. Rivers, streams, and canals typically empty into other water bodies that can be impacted by nutrient enrichment. In some instances, treatment plants discharge to a wetland before ultimately discharging to surface waters; when this occurs, the nutrient load decreases and thus the risk from this type of disposal may be diminished.

7.6.1.2 Metals

Concentrations of all inorganic and secondary analysis metals in AWT effluent reviewed for this study were below standards for drinking water quality (Appendix Table 1-1). Copper concentrations in AWT effluent were similar to concentrations found in secondary-treated effluent (Englehardt et al., 2001). Total copper in advanced treated wastewater was 0.003 mg/L. This is below copper water-quality standards in Florida. Because the concentrations of copper in wastewater effluent reported by utilities in this study were below water-quality standards, it is unlikely that this constituent poses significant risks to human or ecological health. For the Cape Canaveral plant, copper concentrations were below detection limits (<0.0005 mg/L).

7.6.1.3 Organic Compounds

Concentrations of trihalomethanes, synthetic organics, and volatile organics were below drinking-water standards (Appendix Table 1-1). Compared to the Florida standards for surface water quality, all trihalomethanes in AWT wastewater were below Class II and Class III standards for fresh and marine surface-water quality. Class I standards, which apply to surface waters used as drinking-water supplies, were not met by the AWT effluent monitoring results reviewed for this study. However, none of the AWT plants surveyed in this report discharge treated effluent to Class I surface-water drinking supplies.

All synthetic and VOCs that were analyzed from one monitoring sample of treated effluent by the City of Cape Canaveral wastewater treatment facility were below detection limits (Appendix Table 1-1).

The representative contaminant chosen to evaluate potential risk include a number of estrogenic and estrogen-like substances. Estrogen equivalence is a measure of the response of breast cancer cells to exposure to strongly estrogenic substances, such as hormone replacement and birth-control pills (Frederic Bloettscher, personal communication). Estrogen equivalence was measured from two grab samples at the Gulfgate and Southgate treatment plants in Sarasota, Florida. Both of these plants treat to AWT levels and discharge to surface-water creeks. The average concentration of estrogen-equivalence substances in the treated wastewater effluent was 3.253 nanograms per liter (ng/L) (Frederic Bloettscher, personal communication).

At this point, this information only indicates that these substances may be present in treated wastewater intended for disposal into surface water. Recent literature suggests that concentrations below 1 ng/L can cause vitellogenin levels to increase in aquatic organisms (Sadik and Witt, 1999; Larsson, et al., 1999). The literature suggests that more study is needed concerning the concentrations at which endocrine disruption may occur from biodegradation byproducts.

No information is available concerning concentrations of estrogen-like compounds in ambient surface waters near the outfall sites, nor in ecological receptors at or near the outfall sites. Ongoing and future research should provide a better framework for discussing these compounds and evaluating their risks.

7.6.1.4 Pathogenic Microorganisms

Monitoring data reported by the city of Cape Canaveral to the Florida DEP for its National Pollutant Discharge Elimination System permit indicate that, between 1999 and 2001, the maximum concentration of fecal coliforms in treated effluent (measured monthly) ranged from 0 to 8 colonies per 100 mL (Cape Canaveral NPDES Database, 1999–2001). As noted above, a certain number of fecal coliforms are permitted, up to a limit of 200 fecal coliforms per 100 mL of effluent, for all but Class I surface waters. These concentrations do not meet drinking-water standards.

The Howard F. Curren Wastewater Treatment Plant in Tampa Bay reported annual sampling results in 2000 and 2001 for *Giardia lamblia* and *Cryptosporidium*, pathogenic protozoans that can cause gastrointestinal illness in humans when ingested (David York, pers. comm.). In 2000, the concentration of *Giardia lamblia* and *Cryptosporidium* were each less than 0.7 cysts per 100 L of effluent. In 2001, the concentration of *Giardia lamblia* was less than 0.29 cysts per 100 L of effluent, and the concentration of *Cryptosporidium* was 2.33 oocysts per 100 L of effluent. These numbers are below the DEP's recommended limit of 5.8 per 100 L for both *Cryptosporidium* and *Giardia*. Monitoring of other wastewater treatment facilities in Florida indicates that a few facilities do not meet the informal standard of 5.8 per 100 L, despite the fact that the effluent is filtered (York et al., 2002).

7.6.2 Evaluation of Receptors and Exposure Pathways

Some potential ecological receptors in water bodies in Florida that receive treated wastewater are described below. Water-quality problems that have arisen or been corrected through the implementation of improved wastewater treatment are noted.

- **Submerged aquatic vegetation** (such as sea grasses) populations are abundant in the nearshore areas surrounding South Florida. In recent years, there have been documented changes in the abundance of sea grass in the nearshore environment. For example, in Tampa Bay, there have been recent declines in sea-grass populations, but this has occurred after several years of sea-grass expansion throughout the bay. In the late 1980s and early 1990s, sea grasses were returning at the rate of 500 acres a year as Tampa Bay responded to improvements in water

quality resulting from improvements in wastewater treatment. The sea-grass expansion rate slowed to about 350 acres in the mid-1990s. The latest figures show an overall cumulative loss of sea grass to pre-1990 levels (Coastlines, Issue 11.4).

- **Bordering habitats** (such as mangroves and salt marshes) are located throughout the nearshore estuarine environment in South Florida. Like sea-grass habitats, these areas offer food and refuge to many aquatic species and are affected by increased nutrients.

- The Indian River Lagoon supports one of the most diverse **bird populations** in the United States, with 125 breeding species and 172 species that over-winter in the area (Adams et al., 1996). Many bird species in the region are impacted by human activities, especially activities that contribute to habitat loss and fragmentation. In 1987, the dusky seaside sparrow became extinct in the Indian River Lagoon because of alterations to coastal marsh habitat (marsh impoundment)**.** Avian communities are also susceptible to overexploitation (primarily hunting) and to the adverse effects of widespread use of chemicals (especially DDT).
- **Marine mammals,** such as the West Indian manatee and the Atlantic bottlenose dolphin, inhabit lagoons and estuaries along the Florida coast. One-third of the endangered Floridian population of West Indian manatee (*Trichechus manatus*) resides in the Indian River lagoon. Collisions with boats pose the most significant threat to these populations, at least from human activities. However, *Cryptosporidium* and *Microsporidium* infections have been implicated in recent manatee deaths along the Gulf Coast of Florida, according to biologists at Tampa's Lowry Park Zoo (Grossfield, 2002). Dolphin (*Tursiops truncatus*) populations are believed to be stable. Approximately 20 dolphin fatalities are reported annually; 8% to 12% of these fatalities are believed to be related to boat accidents or fishnet entanglement. A fungal skin disease that affects approximately 12% of the dolphin population may be linked to water quality, as documented by the Treasure Coastal Dolphin Project conducted in 1994 (Adams et al., 1996).

- Both **green and loggerhead turtles** are on the U.S. Fish and Wildlife Service list of threatened and endangered species (Adams et al., 1996; Gilmore, 1995; Gilmore et al., 1981). The green turtle (*Chelonia mydas mydas*), a state and federally endangered species, inhabits the Indian River Lagoon. Boat collisions and fishing line entanglement are believed to be the principal causes of sea turtle mortality. However, 40% to 60% of green turtles surveyed in the Indian River Lagoon were found to be infected with fibropapillomatosis; this disease may be linked to water quality (Ehrhart and Redfoot, 1995).

- As of January 1994, 782 **fish species** were documented in the east-central Florida region. At least half of these species use estuaries and lagoons, such as the Indian

River Lagoon, at some point in their life histories (Gilmore, 1995; Gilmore et al., 1981).

Toxicity testing results from the city of Cape Canaveral AWT Wastewater Treatment Plant in June 2001 (City of Cape Canaveral, 2001) revealed that the survival rate of *Ceriodaphnia dubia* ranged from 85% to 95% for undiluted treated wastewater. The survival rate for *C. leedsi* was 100% for all tests. While the data were limited, this indicates that the AWT-treated wastewater is not acutely toxic.

There is no direct evidence (such as the use of tracer studies) that indicates that constituents in AWT-treated wastewater are taken up by aquatic biota or human receptors in the coastal embayments or canals reviewed. However, although there is no direct evidence, indirect evidence indicates that discharges of treated wastewater do affect water quality on a regional scale. Zhou and Rose (1995) and City of Tampa Bay Study Group (2000b) reported that water quality in Sarasota Bay and Hillsborough Bay (Tampa Bay) improved after wastewater treatment plants that discharged to rivers or the bay itself upgraded their wastewater treatment to meet tertiary or advanced standards. This suggests that the high nutrient levels previously measured in the bay were at least partly the result of discharges of secondary-treated effluent.

Some potential ecological receptors, such as endangered species, may be more susceptible to harm and may be at risk from concentrations less than the applicable standards. Additionally, eutrophication is site-specific as it is greatly influenced by physical and biological processes. Addition of nutrients and, indeed, any constituents that may be present in treated effluent needs to be examined in a site-specific context to truly evaluate risk.

Little information was found on ecological receptors in canals that may be receiving wastewater effluent. However, estuaries examined in this study that are receiving treated wastewater contain marine mammals, fish, and birds that are known to be at risk from other effects of human development.

In terms of the applicable water-quality standards, surface waters receiving discharges of treated wastewater reviewed in this report were designated as Class III waters. Class III water-quality standards are meant to protect a healthy population of fish and wildlife and provide recreational uses. Compared to these standards, the quality of AWT effluent was often well below the required minimum concentrations.

Physical mixing and dilution are important large-scale processes that will act to decrease concentrations of stressors in a water body. This is especially true for streams, rivers, estuaries, and coastal embayments that are well mixed. Such dispersion and dilution will decrease the risks to human and ecological receptors.

There is a strong coupling of groundwater and surface water in South Florida. At present, there are few estimates of the hydrologic fluxes between groundwater and surface water in south Florida. However, in recent studies in the Everglades, it was found that extensive

human manipulation of the natural drainage system in southern Florida has altered hydrology that has led to increased recharge and discharge in the north-central Everglades (USGS, 2002). Additional evidence of interaction between groundwater and surface waters in the Everglades was provided when mercury was found to be recharged from surface water to groundwater and stored in the surficial aquifer. Indeed two-way exchange of surface water and groundwater may be a localized phenomenon, as was found in Taylor Slough (USGS, 2002).

Canals, which are a frequent receptor for discharge of treated wastewater into surface-water bodies, are often hydrologicly connected to groundwater and are recharged by groundwater. Adams (1991) examined water in the surficial aquifer and canals in Martin and Palm Beach counties and concluded that groundwater quality did not seem to be affected by canal water. This suggested that the aquifer is discharging to the canal rather than the canal recharging the aquifer. However, water from canals may enter the surficial aquifer when canals are used as an irrigation source. Drinking-water receptors may be exposed where surface waters have a direct hydrologic connection to the groundwater resource.

7.7 Final Conceptual Model of the Discharge-to-Surface-Waters Option

This disposal option presents limited risks, because the volumes of treated effluent discharged to surface water are much smaller than volumes discharged via ocean outfalls or Class I injection wells and because the discharges are typically discharged intermittently.

- The degree and kind of treatment of wastewater is an important factor determining effluent quality and therefore risk. To discharge to surface waters in the state of Florida, wastewater treatment plants are likely to treat using AWT. AWT treats wastewater to a higher standard than secondary treatment, removing additional nutrients, organic compounds, and total suspended solids from the effluent.
- Several of the AWT standards (for example, nutrients) are elevated when compared to natural background levels of these compounds in unimpacted surface waters and when compared to the EPA's recommended standards for unimpacted surface waters, which are based on monitoring of more pristine water bodies. Nutrients, both nitrogen and phosphorus, pose ecological risks for the aquatic environment as they may increase primary production, alter phytoplankton communities, and encourage or exacerbate the growth of harmful algal blooms. The data available reveal that wastewater treatment facilities often have the ability to remove nitrogen to well below the standard required, which would reduce risk. While phosphorus met treatment standards, the concentrations that remain in treated wastewater are often higher than recommended water-quality standards, based on unimpaired waters.
- There is a lack of water-quality monitoring data and tracer studies that would show whether effluent constituents are taken up by receptors.

- There are no effluent or surface-water quality standards for *Cryptosporidium* and *Giardia*, although the Florida DEP has recommended that numerical standards corresponding to a 1 in 10^{-4} human illness risk be adopted for *Cryptosporidium* and *Giardia* in reclaimed water (York et al., 2002). These recommendations are 5.8 oocysts per 100 L and 1.4 cysts per 100 L for *Cryptosporidium* and *Giardia*, respectively. For comparison, background concentrations of *Cryptosporidium* oocysts in North American water bodies, such as lakes, rivers, springs, and groundwater, averaged 44, 43, 4, and 0.3 oocysts per 100 L, respectively (York et al., 2002).
- Concentrations of pathogenic microorganisms in treated wastewater from the Howard F. Curren facility were well below the standards for discharges to surface waters for Class III waters. Concentrations of the pathogenic protozoans *Giardia* and *Cryptosporidium* in effluent from the Howard F. Curren AWT plant were very low.
- Monitoring of pathogenic protozoans at other wastewater treatment facilities in Florida indicates that a few facilities do not meet the recommended limit of 5.8 per 100 L, despite the fact that filtration is done (York et al., 2002). While human health risks from pathogenic protozoans are generally very low, they are not zero.
- Facilities that nitrify appear to be better at removing *Giardia* than facilities that do not nitrify (York et al., 2002).
- All inorganic compounds, including nutrients and metals, measured in AWT effluent were below drinking-water-quality standards. Copper was used as a surrogate because of its known toxicity in the aquatic environment. Copper concentrations in treated wastewater met Florida water-quality standards.
- Measured organic compounds, which include trihalomethanes, synthetic organics, and volatile organics, were below drinking-water standards. All synthetic and VOCs were below detection limits for the data reviewed in this study. Two grab samples for estrogen equivalence (hormonally active agents) revealed that these constituents are present in the effluent in relatively small concentrations (on the order of ng/L). Despite the lack of information on *in situ* concentrations, hormonally active agents pose ecological risks for aquatic ecosystems because of information from studies of their effects on other aquatic organisms elsewhere and because the effects are observed at very low concentrations.
- Toxicity testing of AWT effluent revealed no toxicity to aquatic organisms. The limited data available suggests that AWT effluent poses little or no ecological or human health risks.
- The relative risk of AWT-treated wastewater is lower than the risks posed by lesser-treated wastewater, based on improvement of water quality in Tampa Bay after AWT was required.
- Despite the relative lack of monitoring information from surface-water disposal outfalls and lack of evidence of adverse effects, it is reasonable to assume that, given the already-impacted nature of many surface-water bodies in South Florida, further discharge of nutrients in treated wastewater poses some ecological risks. The potential effects of nutrients on surface-water bodies will vary, depending on site-specific characteristics and the existing nitrogen loading from other sources. Preferably, a water-quality-based effluent limit (such as total maximum daily

loading) would be established that takes into account these site-specific characteristics and the carrying capacity of an individual surface-water body.

- In some areas, depending on existing impairment of water quality, it may be worthwhile to consider whether discharge of treated wastewater could help restore hydrology or water quality.

7.8 Gaps in Knowledge

Possible gaps in knowledge and their possible effects on this risk analysis are summarized below.

- The benefits or detriments of discharging AWT-treated wastewater into natural systems have yet to be proven.
- One of the most important gaps in knowledge concerns the numbers and significance of unpermitted, inadvertent, or occasional unplanned discharges of untreated or secondary-treated wastewater to surface-water bodies. Such discharges may occur at treatment facilities when storms or other causes combine to produce wastewater volumes that cannot be treated rapidly enough to keep up with incoming volumes. Rapid infiltration basins receiving untreated or secondary treated wastewater that overflow to nearby surface-water bodies, such as canals or creeks, provide examples of such untreated or minimally treated discharges. Such discharges are believed to occur at a number of South Florida facilities, including those at Miami-Dade South Treatment Facility. Although such discharges are outside the scope of this study because they are not a permitted form of wastewater management, they nonetheless pose high risks.
- The potential and actual human health and ecological health effects of exposure to AWT-treated effluent that has not been filtered to remove pathogenic protozoans to the levels recommended by the Florida DEP have yet to be determined. The ecological effects of pathogenic protozoans are only beginning to be documented; the latest example involves the implication of *Crytosporidium* and *Microsporidium* in mortality of manatees along the Gulf coast of Florida.
- Distinguishing between other sources of wastewater stressors and those derived directly from AWT-treated wastewater will be difficult unless specific tracers are utilized in studies designed specifically to distinguish different sources. Many other sources of stressors already have adversely affected Florida's surface waters and coastal waters.
- The effects of discharging wastewater treated to AWT standards into water bodies that are already adversely affected have not been explored or documented, according to available information. Comparing AWT-treated wastewater with water-quality recommendations based on pristine or unaffected ambient Florida waters also raises water-management questions that can only be answered through a combination of public process and scientific studies of the fate of these stressors and the capacity of the watershed or embayment to assimilate stressors without experiencing adverse effects.

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8.0 RELATIVE RISK ASSESSMENT

This chapter presents the findings of EPA's relative risk assessment for the South Florida municipal wastewater management options: deep-well injection, aquifer recharge, discharge to ocean outfalls, and discharge to surface-water bodies. The preceding chapters outlined the overall framework for the risk assessment and the application of that framework to the individual assessments of the South Florida municipal wastewater treatment options. The main issues that were considered when assessing the risk are summarized in this chapter, followed by an examination of the human health risks and the ecological health risks.

Although the term *option*, used to describe the wastewater treatment methods, suggests any of these are available for use by the wastewater treatment plants in South Florida, in fact most facilities are limited by local conditions as to possible treatment methods. However, most wastewater treatment facilities do not rely solely on one method but combine management options to meet the current demands and local conditions.

8.1 Identified Risk Issues

Although all four disposal options deal with municipal wastewater, they differ from each other in almost every aspect. The option used depends on geographic location, the underlying geology, final injection point, type of treatment, disinfection level, site-specific conditions, local needs and constraints, the opportunities for water reuse, and, in some instances, weather conditions. Because of this variation, each disposal option has its own specific stressors (hazards), exposure pathways, receptors, and effects. Also, parameters that are relevant to one particular disposal option are not necessarily relevant to the remaining three. As a result, it is not feasible to present strictly quantitative data for all parameters associated with all options.

Table 8-1 identifies the major issues relevant to assessing risk associated with each of the four options. This information and data is a summary of the findings from the option-specific risk assessments that were discussed in detail in Chapters 4 through 7. Although overall quantitative comparisons are not feasible, the information in the table identifies key issues and allows the reader to relate these issues between the four wastewater treatment options. The issues are central to managing wastewater treatment in a way that limits risk to people and the environment.

Table 8-1. Relevant Risk Assessment Issues for the Four Wastewater Management Options

Issue	Deep-Well Injection	Aquifer Recharge (RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Type of Treatment and Level of Disinfection	Secondary treatment; treatment plants must maintain basic disinfection capability. (Exceptions such as Pinellas County use secondary treatment, high-level disinfection and filtration.)	Secondary treatment, including high-level disinfection. Meets Florida's reclaimed-water standards.	Secondary treatment, including basic disinfection.	Secondary treatment, including basic disinfection. Discharge to Class I waters requires high-level disinfection. Discharge to sensitive waters (such as Tampa Bay) requires advanced wastewater treatment (AWT) with nitrogen removal.
Wastewater Constituents Remaining After Treatment (Stressors)	Moderate levels of nutrients (phosphorus, nitrogen); concentrations typically meet maximum contaminant levels (MCLs), but may exceed surface-water quality standards (for example, AWT standards). Small amounts of metals and organic compounds; concentrations typically meet MCLs (trihalomethanes may occasionally exceed the MCL). Pathogenic protozoans are not removed; infective bacteria and viruses remain.	Low levels of nutrients (phosphorus, nitrogen); concentrations frequently exceed AWT standards and EPA recommendations for ambient surface-water quality. Trace amounts of metals and organic compounds; concentrations typically meet MCLs. Low mean numbers of pathogenic protozoans (occasional instances of higher numbers); bacteria and viruses are effectively inactivated.	Moderate levels of nutrients (phosphorus, nitrogen). Trace amounts of metals and organic compounds; concentrations typically meet MCLs. Metals frequently exceed ambient seawater concentrations. Pathogenic protozoans are not removed; small numbers of infective bacteria and viruses may remain.	Low levels of nutrients. Nutrient concentrations typically meet standards specific to water bodies, but may exceed EPA recommendations for ambient surface-water quality. Trace amounts of metals and organic compounds; concentrations typically meet MCLs. Low numbers of pathogenic protozoans; bacteria and viruses are effectively inactivated.

Table 8-1. Relevant Risk Assessment Issues for the Four Wastewater Management Options

Issue	Deep-Well Injection	Aquifer Recharge (RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Large-Scale Transport Mechanisms	Simultaneous upward and horizontal migration. Vertical transport occurs as a result of injection pressure and fluid buoyancy. Horizontal transport in the direction of groundwater flow.	Initial downward migration (infiltration, percolation). Horizontal transport in the direction of groundwater flow. Potential for recharge to surface waters.	Initial upward migration into the ocean water column. Horizontal transport within the Florida Current (northward). Occasional transport towards the coast.	Downstream (horizontal) transport in canals. Turbulent mixing in estuaries and bays. Potential for recharge where water body is hydrologically connected to groundwater.
Distance Between Point of Discharge and Potential Receptors *(Note: Depending on the particular option, receptors may be USDWs and drinking-water supplies, or they may be human or ecological.)*	Injection occurs between 1,000 and 3,000 feet below ground surface. Vertical distance to the nearest overlying USDW varies geographically: • Dade Co.: approx. 1,000 ft. • Brevard Co.: approx. 950 ft. • Pinellas Co.: approx. 570 ft. Thousands of feet to water-supply wells or potential ecological receptors.	The distances range from tens of feet to hundreds of feet.	Discharge occurs between roughly 1 and 3.5 miles offshore. No drinking-water receptors exist at the ocean outfall discharge points. Tens of feet (or more) to ecological receptors in the vicinity of outfalls.	Tens of feet to receptors at discharge point; hundreds of feet to other receptors.

Table 8-1. Relevant Risk Assessment Issues for the Four Wastewater Management Options

Issue	Deep-Well Injection	Aquifer Recharge (RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Time of Travel to Potential Receptors *(Note: Depending on the particular option, receptors may be USDWs and drinking-water supplies, or they may be human or ecological.)*	Potential receptors are deep USDWs and current drinking-water supplies. Vertical times of travel to these receptors vary geographically: • Dade Co.: Between 14 and 420 years to deep USDWs; 30 to >1,100 years to the depth of current water supplies • Brevard Co.: Between 86 and 340 years to deep USDWs; 136 to >1,100 years to the depth of current water supplies • Pinellas Co.: Between 170 days and 2 years to deep USDWs; 6 to 23 years to the depth of current water supplies. Fluid movement into deep USDWs confirmed at 3 facilities; probable movement into USDWs at an additional 6 facilities.	Horizontal times of travel within the surficial aquifers vary with site-specific characteristics and with mandatory setback distances: • Dade Co.: Approx. 40 days to travel 200 feet; 1.5 years to travel ½ mile • Brevard Co.: Approx. 3 years to travel 200 feet; 40 years to travel ½ mile • Pinellas Co.: Approx. 6 years to travel 200 feet; 75 years to travel ½ mile.	There are no drinking-water receptors. Immediate transport (minutes) to receptors that may occur around the discharge points. Rapid transport to downstream ecological receptors (hours to days); however, there is rapid attenuation by dilution in the ocean.	Immediate transport to receptors around surface-water outfalls. Rapid transport to downstream human and ecological receptors (hours to days). Delayed and variable recharge to surficial USDWs.

Table 8-1. Relevant Risk Assessment Issues for the Four Wastewater Management Options

Issue	Deep-Well Injection	Aquifer Recharge (RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Attenuation Processes	Dilution; filtration by porous geologic media; sorption onto media; other chemical degradation processes.	Filtration by soils and by porous geologic media; sorption onto soils and media; dilution; microbial degradation; other chemical degradation processes.	Dilution; settling; sorption onto sediments; biological uptake and degradation; photo-oxidation; other processes.	Dilution; settling; sorption onto sediments; biological uptake and degradation; photo-oxidation; other processes.
Anticipated Reduction in Stressor Concentration *(Note: Depending on the particular option, receptors may be USDWs and drinking-water supplies, or they may be human or ecological.)*	Minimally to substantially reduced before reaching deep USDWs. Minimal reduction where estimated times of travel are short (for example, Pinellas Co.) or where groundwater monitoring indicates rapid vertical fluid movement (for example, Miami-Dade, South District). Moderate to substantial reduction where estimated times of travel to USDWs are long. Substantially reduced before reaching the depth of current water supplies or potential ecological receptors.	Minimally reduced before reaching USDWs. Moderately to substantially reduced before reaching other potential receptors.	Minimally to moderately reduced before reaching receptors that may occur or be near points of discharge; mean dilutions between 60:1 and 90:1 are achieved within 400 meters of the discharge point. Substantially reduced before reaching receptors that may occur or be at greater distances from points of discharge.	Minimally reduced before reaching receptors near outfalls. Moderately to substantially reduced before reaching receptors at further distances from outfalls.

Table 8-1. Relevant Risk Assessment Issues for the Four Wastewater Management Options

Issue	Deep-Well Injection	Aquifer Recharge (RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Factors That May Increase Risk	The potential for long-term impacts to USDWs. Long time frames for recovery. The difficulty in performing remediation in the deep subsurface. The lack of attenuation where conduit flow is a major fluid movement mechanism. Instances where there is little natural straining or filtering of particulates or microorganisms.	The potential for long-term impacts to USDWs and current water supplies. The proximity to drinking-water and ecological receptors.	The proximity to ecological receptors. The potential for shifts in current toward shore and human receptors. This is currently estimated to occur approximately 4% of the time. The lack of natural straining (filtration) of particulates or microorganisms.	The potential for recharge to surficial USDWs. The proximity to ecological receptors. The potential for long-term impacts to surface-water quality. The lack of natural straining (filtration) of particulates or microorganisms.
Factors That May Decrease Risk	Appropriate siting, construction, and operation of wastewater treatment plants and outfalls.	Use of a high level of wastewater treatment and disinfection (results in high-quality wastewater).	The absence of drinking-water receptors (resulting from off-shore location for discharge points). Rapid, significant dilution achieved by siting in fast-moving currents and perhaps by the use of multiport diffusers.	Use of a high level of wastewater treatment and disinfection. The absence of drinking-water receptors (resulting from little reliance on surface-water bodies as sources of drinking water).

Table 8-1. Relevant Risk Assessment Issues for the Four Wastewater Management Options

Issue	Deep-Well Injection	Aquifer Recharge (RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Data and Knowledge Gaps	Site-specific mechanisms of transport (for example, porous media flow vs. conduit flow); locations and connectivity of natural conduits such as solution channels. The fate and transport of pathogenic microorganisms; rates of die-off and natural attenuation. The extent of, if any, reduction in inorganic stressor concentration resulting from local geochemical conditions (for example, rate of biologically mediated transformation of ammonia). Groundwater monitoring data to describe transport to (or within) the Biscayne and surficial aquifers.	Site-specific hydrologic data (for example, horizontal hydraulic conductivities); site-specific estimates of horizontal time-of-travel. Groundwater monitoring data to describe transport within the Biscayne and surficial aquifers. Geospatial data to describe proximity to water-supply wells (especially private wells). Fate and transport of pathogenic micro-organisms still present after disinfection; rates and die-off.	The potential for adverse ecological effects near outfalls. The potential for bioaccumulation (such as metals, persistent organic compounds) through food chains. Water-quality and ecological monitoring downcurrent of outfalls (beyond mixing zones). The potential for changes in ocean currents, sea level, or climate that might affect changes in circulation and transportation patterns or exposure.	The potential for adverse ecological effects near points of discharge. The potential for bioaccumulation (such as metals, persistent organic compounds) through food chains. Water-quality and ecological monitoring data for specific potentially impacted water bodies. The nature and extent of recharge to surficial USDWs.

8.1.1 Wastewater Treatment and Disinfection

The four disposal options are generally associated with four different types of treatment and disinfection levels (Table 8-1). The type of treatment and level of disinfection given the wastewater before disposal, discharge, or recharge are the most important issues that affect risk. The treatment and disinfection determine the constituents that remain after treatment and therefore the potential stressors in the wastewater to be discharged.

The type of treatment and level of disinfection are factors that can be prescribed and controlled through management. This is in contrast to factors that are related to physical setting and natural processes and that are largely beyond the control of plant operators and risk managers. State and Federal laws require different minimum types of treatment, depending on the final disposal method. Although plant operators can opt to provide treatment beyond the minimum required, it is not usually practical.

Advanced wastewater treatment (AWT) is the highest level of wastewater treatment conducted in South Florida and poses the fewest risks to human health or ecological values. It combines several treatments and results in water that meets water-quality standards for receiving water bodies and also, for the most part, meets drinking-water standards. AWT includes secondary treatment, basic disinfection, filtration, high-level disinfection, nutrient removal, and removal of toxic compounds. Wastewater discharged to Tampa Bay, Sarasota Bay, and other already-impaired surface-water bodies must be treated to AWT levels (Table 8-1).

Treated wastewater bound for aquifer recharge and for discharge to Class I surface waters undergoes **secondary treatment and high-level disinfection**. This reclaimed water may contain small amounts of nitrogen and phosphorus and trace amounts of other inorganic and organic constituents.

Secondary treatment with basic disinfection represents a third and lower level of treatment (Table 8-1). This type of treatment and level of disinfection represent the minimum standard required of most wastewater treatment facilities in South Florida. Secondary treatment generally results in water of a quality that may often meet drinking-water standards in terms of chemical constituents but that still contains moderate amounts of nutrients (nitrogen and phosphorus) and small amounts of inorganic and organic compounds. However, basic disinfection may not achieve drinking-water standards for fecal coliform bacteria (nondetection). Because filtration is not provided, pathogenic protozoans, such as *Cryptosporidium*, *Giardia,* and other chlorine-resistant microorganisms may remain in the treated wastewater. Secondary treatment with basic disinfection is provided for wastewater destined for ocean outfalls.

In Florida, **secondary treatment without disinfection** is used when wastewater is discharged to deep-injection wells. This lowest level of treatment poses the highest potential risks (Table 8-1). Moderate amounts of nutrients and microorganisms may remain in this treated wastewater.

8.1.2 Large-scale Transport Processes

Large-scale transport processes represent another important factor in assessing risk. They include the physical processes of advection (large-scale mixing) and of dispersion and diffusion (small-scale movements of water and diluted constituents). Dilution occurs as a result of dispersion, advection, and diffusion. Concentrations of wastewater constituents decrease as dispersion and dilution occur in the receiving water body. The receiving water may be groundwater (deep-well injection and aquifer recharge), the ocean (discharge to the ocean), or surface-water bodies (discharge to surface waters) (Table 8-1). The relative effects of large-scale transport are likely more significant for discharges to the ocean, where there is rapid dilution in the Florida Current, than to deep-well injection and aquifer recharge, where transport is through porous rock media. However, regardless of the medium, large-scale transport processes have a role in the level of risk associated with each of the four wastewater treatment processes.

8.1.3 Distance and Time Separating Discharge Points and Potential Receptors

The physical separation (distance between the point at which effluent is discharged into the environment and the potential human or ecological receptors or drinking-water receptors) is another important factor when assessing risk. Like the type of treatment and level of disinfection used, the physical separation of discharge points from the potential receptors is under the control of risk managers and can be adjusted through careful planning and siting of treatment plants and of the associated discharge points. However, in many cases, it is not feasible for the risk manager to manipulate the factors affecting time and distance to reduce risk. For example, increasing the distance to potential receptors may be difficult or impossible for existing treatment facilities.

The time of travel needed for effluent water and effluent constituents to reach possible drinking-water, human, or ecological receptors is related to the distance, the nature of the environment through which the effluent must travel, and the nature of the stressors remaining in the effluent. In general, the longer the time of travel and the greater the distance the effluent must migrate, the lower the risk. However, if problems are identified in a given situation, long times of travel may mean that the benefits of corrective actions will not be realized for some time.

In general, higher relative risks are related to fast times of travel because of the potentially rapid exposure of receptors and the limited attenuation that may achieved by filtering or straining. However, in the case of ocean disposal, where the time of travel may be almost instantaneous, attenuation by dilution can greatly reduce potential risk.

Direct comparisons between the distances and times of travel for the four wastewater treatments options provide no useful assessment of risk because the four options involve very different processes. As an example, there is virtually immediate transport between the discharge point and potential receptors for discharge to the ocean or to a surface-water body, whereas contact between a stressor and a receptor for some deep-well injection can be on the order of hundreds of years (Table 8–1).

8.1.4 Attenuation Processes

Attenuation results in a decrease in concentration of wastewater constituents. Depending on the disposal option being used, attenuation can have a significant role in reducing the concentrations of effluent constituents, including potential stressors. The attenuation processes and the degree to which they are effective in reducing concentrations depends on the media through which the effluent moves and how the constituents interact with those media.

In the ocean and in surface-water bodies, attenuation processes may include dilution, microbial and biological processes, photo-oxidation (by natural sunlight) of organic compounds, inactivation of viruses and bacteria by ultraviolet rays (in sunlight), adsorption onto sediment or organic particles, and settling of particles containing adsorbed wastewater constituents (Table 8-1).

In the subsurface, attenuation processes include dilution, adsorption to geologic material, entrapment or filtration of microorganisms and other constituents, oxidation, reduction, or other chemical processes that affect the mobility of constituents, and biological degradation of organic compounds (Table 8-1). There may be some microbial transformation (denitrification) of nutrients nearer to the surface, but overall microbial decomposition or other microbial activities is not expected to be significant.

The highest potential risks are associated with the least attenuation of stressors. Although all four management options provide attenuation, the least attenuation is probably associated with deep-well injection. In the absence of information to the contrary, the subsurface environment may have low rates of biological and chemical degradation, compared to surface-water bodies and soils. However, for deep injection wells, all constituents except nitrate and metals typically decrease to lower levels by the time the effluent water reaches the USDWs. This is because of the long travel times associated with deep-well injection.

For deep injection wells in Dade and Brevard counties, the concentrations of all constituents except nitrate and metals decrease to lower levels by the time the effluent water reaches the drinking-water receptors. Nitrate and metals may remain at the same concentration as the discharge point unless local geochemical conditions facilitate attenuation. In Pinellas County, effluent water may reach drinking-water receptors because of the short overall vertical travel time. However Pinellas County uses a higher level of treatment, and so the initial effluent may have low concentrations of stressors, which are further reduced by the time the effluent water reaches receptors.

Microbial survival in the deep subsurface and in groundwater is also an important issue, because wastewater injected into deep-injection wells is not disinfected or filtered. The processes involved in microbial survival are not well understood and constitute an information gap. Inactivation rates for fecal coliforms range up to tens of days for 90% inactivation (Bitton et al., 1983; Medema et al., 1997). As a result, the microorganisms likely cannot survive the months, decades, or years of transport before reaching drinking-

water receptors. However, there are no studies that examine long-term survival and transport of microorganisms in the context of deep-well injection. Inactivation times for pathogenic protozoans, such as *Cryptosporidium*, may be in a range that would pose a human health risk if significant numbers of *Cryptosporidium* were present initially in the discharged effluent (Table 8-1).

For aquifer recharge, travel times are shorter than for deep-well injection, but the effluent must travel through soils and, in some cases, surface vegetation. Uptake of potential stressors by soils and vegetation may constitute an important attenuation process for disposal by aquifer recharge. Also, reclaimed water for aquifer recharge does not pose the same degree of microbial risk as deep-well injection or ocean outfalls because the level of treatment and disinfection is higher.

Ocean outfalls have designated mixing zones associated with each outfall. Water-quality standards are usually met for ocean disposal because of the rapid attenuation within the mixing zone from dilution. Within the mixing zone, the level of stressors may temporarily exceed standards; however, by the time the effluent reaches the boundary of the mixing zone, dilution has reduced the levels of stressors.

Treated wastewater discharged to surface waters generally meets surface-water quality standards (for Class III waters). In some cases, monitoring data indicate that the discharged water is of higher quality than the receiving water. Treated wastewater still contains small amounts of nutrients and other constituents. This is especially significant for phosphorus, which can stimulate algal blooms in nearshore or brackish environments. Since high-level disinfection and filtration are provided, risks from pathogenic microorganisms are very low.

8.1.5 Factors That Contribute To or Diminish Risk

In general, factors that when present contribute to risk are the same factors that when eliminated diminish risk. For example, proximity to human or ecological or drinking water receptors will increase risk, whereas increasing the distances to or travel times for these receptors will diminish the risk (Table 8-1). Also, the factors that may contribute to risk for one particular disposal option may have no effect on other disposal methods. For example, a lack of natural straining and filtering by geologic media will increase risk for deep-well injection when flow is preferentially through cracks, fissures, and cavernous openings. However, for ocean disposal, this lack of attenuation by natural straining or filtering may be insignificant as far as human and ecological health effects because of the dilution of effluent by the ocean.

The major factors that decrease risk are use of a higher degree of treatment, a high degree of dilution in receiving water, long travel times to receptors, and the ability of the system to recover quickly if input of wastewater constituents were to decrease or cease. Aquifer recharge and surface-water discharge are characterized by higher degrees of treatment and by rapid potential recovery rates. Ocean outfalls and surface-water discharge are characterized by rapid dilution, more so for ocean outfalls than for surface water discharges. Class I injection wells are characterized by very long travel times for effluent

to reach drinking-water receptors in Dade and Brevard Counties (but short travel times in Pinellas County).

8.1.6 Data and Knowledge Gaps

For all four wastewater disposal options, there is limited site-specific information concerning potential ecological effects, bioaccumulation of wastewater constituents, survival and transport of pathogenic microorganisms, and of specific evidence (such as tracers) that link stressors from disposal options to ecological or biological or human health effects. The potential effects of local geochemical conditions on fate and transport of nitrate and metals cannot be assessed with available information.

Table 8–1 lists the major areas where information and data are lacking. Key general areas where information is needed to better design, manage, and control wastewater treatment and disposal include the following:

- Microbial survival, inactivation, and transportation rates in groundwater
- Rates for microbial straining or filtration by geologic media under different flow scenarios
- Extent of hydrologic connection between groundwater, surface water, and the ocean
- Definitive tracer studies to conclusively prove that monitored stressors are derived from discharged treated wastewater and to conclusively demonstrate the most likely transportation pathways
- Monitoring ecological or human health effects
- Monitoring effects of climate change on large-scale transportation processes.

8.2 Risk Issues Relevant to Human Health

The potential human health risks associated with each wastewater disposal option differ, but overall they can be considered low (Table 8–2). Just as for the general risk-related issues discussed above, quantitative comparisons between the four disposal options are not feasible. However, the information in Table 8–2 identifies key issues for human health and allows the reader to relate these issues between the four wastewater treatment options. Of the various human health stressors identified, pathogenic protozoans (*Cryptosporidium, Giardia*) are the most important for all but the surface water option where high level disinfection is provided. The deep-well injection process is dominated by porous media flow, long travel times and fine pore spaces may attenuate and retain microorganisms including protozoans. When wastewater treatment includes filtration, the risk posed by pathogenic protozoans decreases significantly but does not disappear, partly because filtration must be maintained at a high level in order to remove protozoans. When wastewater treatment does not include high-level disinfection or basic disinfection, the risks posed by viruses and bacteria are significantly higher.

Table 8-2. Relevant Issues for Human Health

Issues	Deep-Well Injection	Aquifer Recharge (using RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Human Health Stressors Remaining After Treatment	Infective bacteria or viruses and pathogenic protozoans. Nitrates and ammonia.	Bacteria and viruses are effectively inactivated; there may remain low mean numbers of pathogenic protozoans with occasional instances of higher numbers. Disinfection byproducts, such as trihalomethanes, may remain.	Small numbers of infective bacteria or viruses may remain, as well as pathogenic protozoans (those that can survive basic disinfection). Remaining nitrogen and phosphorus, in excess, can cause harmful algal blooms, which are secondary stressors. Metals or organic compounds; these may bioaccumulate in fish or shellfish consumed by humans.	Infective bacteria and viruses are effectively inactivated; low numbers of pathogenic protozoans may remain. Remaining nitrogen and phosphorus, in excess, can cause harmful algal blooms, which are secondary stressors. Metals or organic compounds; these may bioaccumulate in fish or shellfish consumed by humans.

Table 8-2. Relevant Issues for Human Health

Issues	Deep-Well Injection	Aquifer Recharge (using RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Treatment Adequacy	Disinfection is not conducted and so pathogenic bacteria and viruses are not inactivated. Levels of *Cryptosporidium* and *Giardia* are uncertain. This wastewater is usually not filtered and likely exceeds the State health-based limits (reuse limits) for pathogenic protozoans. Levels of disinfection byproducts may rarely exceed health standards. Levels of nitrate occasionally exceed the drinking-water standard (MCL). Levels of ammonia meet the EPA lifetime health-advisory limit; exceed stringent risk-based criteria that account for indoor air exposure. Levels of regulated metals and organic compounds typically meet drinking-water MCLs.	High-level disinfection inactivates pathogenic bacteria and viruses. Filtration is generally adequate to remove *Cryptosporidium* and *Giardia*; levels occasionally exceed health-based limits. Levels of disinfection by-products (for example, total trihalomethanes) or ammonia may rarely exceed health-based standards. Levels of nitrate, regulated metals, and organic compounds typically meet drinking-water MCLs.	Pathogenic bacteria and viruses are inactivated by basic disinfection. However, the levels of bacteria may occasionally exceed the fecal coliform limit for recreational waters (14 per 100 milliliters). Levels of the pathogenic protozoans *Cryptosporidium* and *Giardia* are uncertain. This wastewater is usually not filtered, and so it may exceed the State health-based limits (reuse limits). Levels of nitrate, regulated metals, and organic compounds typically meet drinking-water MCLs. Nutrient levels (nitrogen, phosphorus) typically exceed ambient concentrations. These nutrients can cause localized harmful algal blooms.	AWT inactivates pathogenic bacteria and viruses. Filtration is generally adequate to remove *Cryptosporidium* and *Giardia*; levels are typically below ambient concentrations in surface waters. Levels of disinfection byproducts (for example, total trihalomethanes) may rarely exceed health-based standards. Levels of nitrate, regulated metals, and organic compounds typically meet drinking-water MCLs.

Table 8-2. Relevant Issues for Human Health

Issues	Deep-Well Injection	Aquifer Recharge (using RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Known Significant Exposure Pathways	Transport of stressors to deep USDWs.	Transport of stressors to shallow USDWs.	No known significant exposure pathways.	Dermal contact or accidental ingestion associated with recreational use of water bodies.
Potential Exposure Pathways	Transport of stressors to shallow USDWs and to public or private water-supply wells. Exposure to significant stressor concentrations is unlikely, but depends upon proximity and site-specific vertical times of travel.	Transport of stressors to public or private water-supply wells. Exposure to significant stressor concentrations is unlikely, but depends upon drinking water well proximity and highly variable horizontal times of travel. Additional pathways are associated with other forms of reuse not discussed here (such as inhalation exposure to aerosols created by spray irrigation).	Dermal contact or accidental ingestion associated with recreational use. Ingestion of contaminated fish or shellfish. Possible stimulation of harmful algal blooms (that is, “red tide”); these can increase algal toxins in marine water and air.	Recharge to shallow USDWs and subsequent transport; exposure to significant stressor concentrations is unlikely (pathogens are a possible exception). Ingestion of contaminated fish or shellfish.

Table 8-2. Relevant Issues for Human Health

Issues	Deep-Well Injection	Aquifer Recharge (using RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Data and Knowledge Gaps	Survival and transport of pathogenic microorganisms in the deep subsurface. Exact means of transport at specific locations (for example, porous media flow versus bulk flow through conduits). Rates of biogeochemical transformation for conservative compounds (such as nitrate and ammonia). This is of particular relevance in relatively shallow aquifers.	There is incomplete information regarding the presence and numbers of pathogens in reclaimed water. Little is known about the survival and transport of pathogenic microorganisms in the shallow subsurface and surficial aquifers. Extent of surface-water recharge to surficial USDWs.	Downstream monitoring information from outside of the mixing zones is not available. The potential for changes in the circulation of ocean currents is unknown, as is the subsequent effect changes may have on transport within the effluent plume. Potential for bioaccumulation or bioconcentration of metals and persistent organic compounds is not known or understood.	Survival and transport of pathogenic microorganisms in surface-water bodies and coastal embayments. Extent of surface-water recharge to surficial USDWs. Potential for bioaccumulation or bioconcentration of metals and persistent organic compounds.

Table 8-2. Relevant Issues for Human Health

Issues	Deep-Well Injection	Aquifer Recharge (using RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Overall Estimate of Human Health Risk	Low where proper siting, construction, and operation result in physical isolation of stressors, with no fluid movement. Low where there have been impacts to deep USDWs; however, exposure of current water supplies is unlikely. Increased risk where short times of travel prevail and where exposure of current water supplies is more likely. In all cases, the risk would be further reduced when injected wastewater is treated to reclaimed water standards.	Low when treated with high-level disinfection, filtration, and treatment to reclaimed-water standards. Increased risk where filtration is not adequate to meet health-based recommendations for *Giardia* or *Cryptosporidium*. Increased risk where chlorination results in high levels of disinfection byproducts (that is, failure to dechlorinate).	Low because of rapid dilution and an absence of drinking-water receptors. The low probability (less than 4%) that current flow is towards the coast means that human exposure along coastal beaches is reduced. Increased risk where recreational use is near the discharge. Increased risk where discharges contribute to stimulation of harmful algal blooms.	Low when treated with high-level disinfection and treatment to AWT standards. Increased risk where filtration is not provided or is inadequate to meet health-based recommendations for *Giardia* or *Cryptosporidium*. Increased risk where surface-water discharges are near recreational use of water bodies. Increased risk where discharges contribute to stimulation of harmful algal blooms.

Other lower-priority human health stressors included nitrate and ammonia associated with deep-well injection and nitrogen and phosphorus associated with ocean outfalls (because of the potential for causing harmful algal blooms). Persistent organic compounds may pose some risks in the deep-well injection option when shorter travel times occur and when treatment is not adequate to reduce concentrations below the MCL (Table 8-2).

For aquifer recharge, disinfection byproducts, such as trihalomethanes, also may be of concern in reclaimed water that is not dechlorinated (Table 8-2).

Other human health stressors, including metals and organic compounds, are associated with all options. For aquifer recharge and surface-water discharge, nutrients are lower-priority human health stressors, because treatment of wastewater for these options removes significant amounts of nutrients.

Wastewater treatment is adequate for metals and most organic compounds to meet existing regulatory standards and drinking-water MCLs (Table 8-2). However, there are no quantitative standards for unregulated substances, such as endocrine disruptors and detergents, or for *Cryptosporidium* and other pathogenic protozoans.

8.3 Risk Issues Relevant to Ecological Health

Just as for human health risks, the potential ecological health risks differ, depending on the option. However, there is somewhat more of a gradation between the different disposal options (Table 8-3). The overall risk is likely very low (but probably not zero) for aquifer recharge, discharge to surface waters, and deep injection wells in Dade and Brevard counties; low for discharges to the ocean; and moderate for deep injection wells in Pinellas County.

Nutrients are the major ecological stressors for all four disposal options. Nutrients can potentially stimulate primary production, and this can lead to eutrophication or other adverse changes in community structure. Because of its mobility in groundwater, nitrogen is the primary nutrient of concern for deep injection wells and aquifer recharge. Phosphorus is not a concern for these disposal options because phosphorus tends to adsorb quickly to sediment or soil. Nitrogen is also the primary nutrient of concern for ocean outfalls because it is generally the limiting nutrient for primary production in the ocean. For discharges to fresh-to-brackish surface water, phosphorus poses the greatest concern because it is generally limiting in such systems and is not as quickly immobilized as it is in soil.

Table 8-3. Relevant Issues for Ecological Health

Issue	Deep-Well Injection	Aquifer Recharge (RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Ecological Health Stressors Remaining After Treatment	Nitrogen. Metals, organic compounds, phosphorus, pathogenic micro-organisms.	Nitrogen. Disinfection by-products. Metals, organic compounds, phosphorus, pathogenic micro-organisms.	Nitrogen and phosphorus. Metals, organic compounds, pathogenic micro-organisms.	Nitrogen and phosphorus. Metals, organic compounds, pathogenic micro-organisms.
Treatment Adequacy	Post-treatment nutrient levels are high enough to pose potential ecological risks for surface-water bodies. However, no off-site ground water monitoring is conducted and so actual subsurface levels are not known. The presence of subsurface receptors is also not known.	Post-treatment nutrient levels may exceed recommended levels for unimpacted water bodies, but are lower than concentrations in secondary-treated wastewater and also lower than some ambient levels in surface-water bodies.	Nutrient levels are high enough to pose potential ecological risks if dilution does not occur or if there are cumulative effects over time. However, no ecological monitoring is conducted, and so individual or cumulative effects are not understood or identified.	Nutrient levels may exceed recommended levels for unimpacted water bodies. Discharges to sensitive water bodies (such as Tampa Bay) must meet a 3-milligram-per-liter limit on total nitrogen (a 70% reduction).

Table 8-3. Relevant Issues for Ecological Health

Issue	Deep-Well Injection	Aquifer Recharge (RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Known and Potential Exposure Pathways	Transport of stressors to surface-water bodies is feasible, but would occur over long time frames. Pinellas County has shorter times of travel but a higher level of treatment is used and so risk is reduced.	Pathways include contact, ingestion and inhalation. Exposure of ecological receptors may occur in areas where there is significant surface water or groundwater interaction or exchange. Exposure is most likely where surface-water bodies are near RIBs or under direct influence from groundwater.	Pathways include contact, ingestion and inhalation. Fish and other marine organisms within the mixing zone are exposed to potential stressors. The effects of cumulative or chronic exposure to elevated concentrations of some potential stressors (such as metals) are not known.	Pathways include contact, ingestion and inhalation. Ecological receptors near the discharge points are exposed to potential stressors. Discharges may contribute to cumulative effects such as nutrient loading and bioaccumulation. Discharges may aggravate conditions in some surface-water ecosystems already under stress.
Recommended Water Quality for Ecological Protection	If injectate reaches surface waters, the nitrate level may exceed recommended surface-water levels in the absence of denitrification in the subsurface.	Reclaimed water standards do not meet ecological protection recommended standards, but does meet Florida standards for receiving waters.	Exceeds the recommended levels within the allowed mixing zone (502,655 square meters). Effluent plume may occasionally exceed Class III marine water-quality standards outside the mixing zone.	AWT may not be sufficient for ecological health water-quality standards, but otherwise meets Florida Class III standards for receiving waters.

Table 8-3. Relevant Issues for Ecological Health

Issue	Deep-Well Injection	Aquifer Recharge (RIBs)	Discharge to the Ocean	Discharge to Surface Waters
Data and Knowledge Gaps	Survival and transport of microorganisms in the deep subsurface; microbial transformation processes in deep subsurface; cumulative impacts of long-term disposal.	Impact of aquifer recharge on groundwater movement and the transport of existing groundwater contaminants; ecological impacts on nearby wetlands; cumulative and long-term impacts.	Cumulative impacts of long-term disposal of nutrients; ecological impacts or bioaccumulation of metals or other compounds in the biota at or near discharge points; impact of global climate change on ocean currents and effluent dispersal.	Ecological impacts of nutrient phosphorus or bioaccumulation of metals or other compounds in the biota; cumulative effects.
Overall Ecological Health Risk	The risks from chemical constituents are low, but not zero, because of possible hydrologic connectivity. Risks related to pathogenic microorganisms are low to moderate for Dade and Brevard counties because of lack of disinfection and filtration. Microbial risk is very low in Pinellas County because of use of disinfection and filtration.	Low because of possibility of hydrologic connectivity between wetlands and surficial aquifer. Cumulative and long-term effects are not known.	Low because of the concentrations of nutrients in the discharged effluent. No ecological monitoring is currently conducted. Cumulative and long-term effects are not known.	Low because of the concentration of nutrients in the discharged effluent.

Metals and organic compounds are also ecological stressors for all options. However, they are considered a lower stressor than nutrients because the information reviewed did not identify toxic effects over the short-term at either acute or chronic exposure levels. Pathogenic microorganisms are also considered a lower-priority ecological stressor, although there is evidence to suggest that aquatic organisms suffer from high concentrations of enteric microorganisms, just as humans do. The low concentrations of microorganisms associated with aquifer recharge and discharge to surface water implies that there probably are few, if any, ecological effects.

8.4 Conclusion

This relative risk assessment analyzed and characterized potential human health and ecological risks associated with four wastewater management options currently in use in South Florida. The relative risk assessment emphasized analysis and characterization of the processes involved in each option and, in particular, of the processes that affect fate and transport of disposed wastewater effluent. There are many physical, chemical, and biological factors that affect risk. Their degree of influence varies widely, depending on the particular disposal option. Some factors can be readily manipulated and managed to control or reduce risk.

Each of the four wastewater management options is associated with existing State programs that have been operating over a period of years and that have levels of control focused on the risks posed by that management option. As demonstrated by the range of information and data presented in the four chapters dealing with the individual options, each management option for treatment and disposal is extremely complex and can vary, depending on site-specific conditions and constraints. This makes the task of interpreting the data and presenting the relative risk assessment very difficult. In spite of this, for all options, there is either low or no risk.

There is a decrease in the level of confidence concerning deep-well injection. In some cases, a lack of confinement of the injected effluent has been confirmed, and the areal extent of the fluid is unknown. This migration of effluent seems to be associated with very few site-specific cases but warrants attention. Also, although risks to ecological health are also considered low, there are considerable data gaps concerning the biota and natural systems. Additional or new information and data could provide additional insight into the actual risks.

For all four wastewater disposal options, the type of wastewater treatment used may be the most simple factor for comparing the concentrations of stressors that may come in contact with a receptor. Treatment type and the resulting concentration of stressors is a risk factor that can be managed. However, the feasibility of using a particular type of treatment is not equal across the four disposal options.

Another significant issue for both human and ecological health is the distance that must be traveled by discharged effluent in order to reach a receptor. The longer the distance

traveled to a receptor (and the greater the time of travel), the lower the risk. Distance and the associated travel time is also a risk factor that can be manipulated by risk managers.

Natural attenuation processes can significantly reduce risks in all of the options. The type and opportunity for attenuation is very specific to the particular disposal option and the local conditions under which it is used. Natural attenuation processes include filtration by geologic media, dispersion by groundwater or ocean currents, biological degradation, adsorption, and photo-oxidation. The distance between the receptors and stressors and the resulting travel times are important factors that can further enhance attenuation.

Depending on the geographic location, there are significant differences in hydrogeology, coastal hydrology, and water quality in South Florida. These site-specific and regional characteristics can determine whether there is a very low risk or a significant risk. For example, deep-well disposal in Dade and Brevard counties have long travel times in comparison to Pinellas County. However, this potential increased risk for Pinellas County is ameliorated by providing a higher level of wastewater treatment in Pinellas County. As another example, the coastal conditions off southeast Florida are favorable for ocean disposal because the local currents result in rapid dispersion and dilution, whereas the circulation and water-quality conditions along Florida’s Gulf Coast would probably preclude placement of outfalls.

The relative risk assessment identified major data and knowledge gaps for all of the disposal options. This is particularly the case for how natural processes may influence attenuation in deep-well injection and in the extent and nature of ecological impacts. The relative risk assessment relied on existing information and data and some modeling of that data. It is clear that for deep-well injection, many issues have never been addressed because of the belief that there would be no movement of the effluent into USDWs once the fluid was injected. The confirmation of fluid movement, even in the few cases reported, reveals that there is much about the pathways, flow, attenuation, and so forth that is little understood, given the fact that injected fluid can reach USDWs in some cases.

For all options, there is very limited information concerning ecological health effects. Water-quality standards do not exist for this area, and in many cases, the numbers and types of receptors may not be known. Also, compared to human health effects, there is little information on the impacts of specific stressors on specific populations (such as zooplankton, fisheries, marine mammals, birds).

Definitive studies are needed to track stressors back to their origins or sources because there are many potential sources other than wastewater disposal for the same stressors. It is important to identify and recognize the contributions of various sources of stressors. Cumulative effects are not well understood for either human or ecological receptors and may go unrecognized. As more demand develops for additional wastewater treatment capacity in South Florida, these data and information gaps will likely need to be addressed so that new facilities can be designed, constructed, operated and maintained with full confidence that public health and the environment are protected.

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DESCRIPTION OF APPENDIX TABLES 1-1 AND 1-2.

1.0 General

Appendix Table 1-1 includes data collected from various sources. These sources include information compiled in reports by the Florida Water Environment Association Utility Council and by SEFLOE, as well as sampling data sent directly from the Miami-Dade North District Wastewater Treatment Plant and the Brevard County South Beaches Wastewater Treatment Facility. KEMRON Environmental Services, Inc. provided sampling data from the Albert Whitted Water Reclamation Facility in St. Petersburg. The Florida Department of Environmental Protection provided sampling results for the wastewater treatment facility in the City of Cape Canaveral and the Howard Curren Wastewater Treatment Plant in Tampa Bay.

1.1 Florida Water Environment Association Utility Council

The Florida Water Environment Association Utility Council (FWEAUC) report (Englehardt et al., 2001) provided analysis of sampling and monitoring results of effluent that had been treated to different standards (advanced wastewater treatment, secondary treatment, and advanced secondary treatment) as well as "native" ambient water in injection zones and monitoring zones in target aquifers. In all, eight (8) categories of sampling data were summarized in the Florida Utility Council report. The data that we present in Appendix Table 1-1 represents "digested" data that has already been processed by the FWEAUC authors. Those authors include raw concentration data for each of the sampling stations in appendices B and C of their report. For each of the sampling dates for each of the stations, the authors provide two lists of monitoring data; the first list includes the concentrations of all detected constituents and sets each of the "non-detect" values to zero (0), and the second list duplicates the first, but sets "non-detect" values at their detection limit. For Each of these two lists, the average concentration of each parameter was calculated from all of the sampling results at all of the stations within a category (e.g., advanced wastewater treatment), resulting in average values for each constituent with non-detects as zero and non-detects at the detection limit, respectively. Processing the data in this manner has the same effect as assigning values one-half of the detection limit to all non-detects, a standard approach not inconsistent with risk assessment methodologies (US EPA. 1998).

The Florida Utility Council study processed all of the raw data in this manner. The utilities that supplied monitoring data to the authors of the report include:

- City of Hollywood
- City of Boca Raton
- City of Fort Lauderdale
- City of Sunrise
- City of Boynton Beach
- City of West Palm Beach
- Broward County North Regional Wastewater Treatment Plant
- Miami-Dade County North and South District Wastewater Treatment Plants
- Seacoast Utilities

- South Central Regional Wastewater Treatment Plant
- Florida Governmental Utility Authority (FGUA) Sarasota plants (Southgate and Gulf Gate Wastewater Treatment Plants)
- The FGUA Golden Gate Plant

1.2 Miami-Dade North District Wastewater Treatment Facility, Dade County

Sampling results from one round of tests characterizing a full suite of waste contaminants in screen effluent were obtained from the Miami-Dade Water and Sewer (North District) utility directly (Miami-Dade Water/Sewer Submission # 9903001041). This facility provides secondary treatment for wastewater effluent before discharging through an ocean outfall to the Atlantic Ocean. The sampling date for these results is March 19, 1999; this is the same sampling date as the results used in the Florida Utility Council report; a comparison of the raw data sent by the facility to the data in the Florida Utility Council report confirms that this is the same data set. Data from this set were entered into Appendix Table 1-1 directly; no processing of the data was performed except for the conversion of values from mg/L to μg/L (or vice versa). Constituents that were below the detection limit are indicated in Table 11 with a less than (<) sign preceding the reported detection limit.

1.3 South Beaches Wastewater Treatment Facility, Brevard County

Sampling results from one round of tests characterizing a full suite of waste contaminants were obtained from the Brevard County Water Resources Department (South Beaches Wastewater Treatment Facility, 2001) for effluent analyses conducted on December 7 and 28, 2000. This facility discharges effluent via a Class I deep injection well, reuse, or surface water discharge. Wastewater that is discharged through deep well injection receives secondary treatment. Water that is reused receives secondary treatment and high level disinfection with chlorine. Finished water destined for reuse has a concentration of 1 ppm chlorine and is filtered to reduce the concentration of total suspended solids to less than 5 ppm. Effluent is occasionally discharged directly to the Indian River during heavy rain and hurricanes. This effluent receives secondary treatment, plus chlorination and dechlorination as well as nutrient removal to lower the concentration of nitrogen, phosphorus and chlorine (Chuck Caron, personal communication).

These data represent single (not averaged) results. Data from this set were entered into Appendix Table 1-1 directly; no processing of the data was performed except for the conversion of values from mg/L to μg/L (or vice versa). Constituents that were below the detection limit are indicated in Appendix Table 1-1 with a less than (<) sign preceding the reported detection limit.

1.4 City of St. Petersburg, Albert Whitted Water Reclamation Facility, Pinellas County

Sampling results from the Albert Whitted Water Reclamation Facility were obtained by KEMRON Environmental Services, Inc. The records supplied by Kemron include effluent monitoring data from a range of dates, as well as minimum, maximum, and

average concentrations for each constituent; not all constituents were tested for on all the dates. Sampling and analysis occurred on September 16, 1998, January 4, 1999, April 6, 1999, June 29, 1999, July 1, 1999, September 26, 2000, and January 24, 2001.

Volatile organic constituents, synthetic organic constituents, secondary drinking water standard regulated constituents, and inorganic constituents were all sampled in September 1998, January and April 1999, September 2000, and January 2001. Radionuclides were sampled in September 1998, April and June 1999, and September 2000. Trihalomethanes were sampled in September 2000, and microbes were sampled in January 1999 and January 2001. Kemron provided constituent concentration data in two sets: one set of data included data qualifiers to indicate concentrations that were below detection limits, and the other set of data had the qualifiers removed in order to calculate the average concentration of each constituent. The average concentration of each constituent was entered directly into Appendix Table 1-1 from the Kemron table lacking qualifiers. Then, if any of the values used in the calculation had actually been below the detection limit, a "less than" (<) sign was added to the value entered into Appendix Table 1-1. For this reason, a "less than" sign preceding a concentration value **does not** indicate that the numeric value is the detection limit. The "less than" sign simply means that the average concentration of the constituent in question is less than the value reported in the table.

Ammonia, total nitrogen, total Kjeldahl nitrogen, orthophosphate, and water temperature were sampled in November 2000; those results were obtained from a Reclamation Facility Monitor Well and Effluent Study Report dated December 26, 2000 that was also provided by Kemron. These single sampling values were added to Appendix Table 1-1. No processing of the data was performed except for the conversion of values from mg/L to μg/L (or vice versa).

1.5 City of Cape Canaveral

The Cape Canaveral treatment plant serves the City of Cape Canaveral. In the mid 1990s, the plant was upgraded to an advanced wastewater treatment facility. The plant is part of a reclaimed water system that supplements the City of Cocoa Beach's reclaimed water supply. Discharge to the Banana River, a segment of the Indian River Lagoon, occurs during wet weather or other periods when reclaimed water demands are low.

The Florida Department of Environmental Protection, Central District, supplied comprehensive sampling results from a round of sampling at the Cape Canaveral Wastewater Treatment Plant conducted on October 1, 1999. The City of Cape Canaveral provided comprehensive sampling results from analyses conducted on April 3, 2001. These sampling results were entered into Appendix Table 1-1 directly without processing other than conversion of concentration units to be compatible with the other records in the table (i.e., conversion of values from mg/L to μg/L or vice versa).

In addition, the Florida Department of Environmental Protection (DEP) provided weekly, monthly, and annual sampling results for constituents that were monitored as part of Cape Canaveral's compliance with its National Pollution Discharge Elimination System (NPDES) permit. These constituents include total nitrogen, total phosphorus, and total suspended solids. These data were provided for calendar years 1999 through 2001. To

supplement the other sampling results for Cape Canaveral (dated October 1999 and April 2001), the annual average of each of the three constituents were calculated from monthly averages provided in the Florida DEP spreadsheet. Twelve monthly average records were used to calculate the annual average for each constituent in 1999; however, May and June 2001 data were unavailable. For this reason, only ten monthly averages were used to calculate the annual average of each constituent in 2001. Annual averages for each of these three constituents were included in Appendix Table 1-1 in the Cape Canaveral 1999 and 2001 columns; superscripted footnote numbers distinguish these average values from the comprehensive raw data.

1.6 SEFLOE II Data

Concentrations of several parameters in effluent from four wastewater treatment plants (Broward County North Regional Wastewater Treatment Plant, City of Hollywood, Miami-Dade North District WWTP, and Miami-Dade Central District WWTP) were provided in the SEFLOE II report (Appendix Table 1-2) (Hazen and Sawyer, 1994). Data for ammonia, total Kjeldahl nitrogen, total phosphorus, nitrate, nitrite, and oil and grease were supplied in that report as arithmetic averages for each utility. These average values were collected from separate tables and entered into Appendix Table 1-2 in columns labeled with the utility names (SEFLOE data for Miami-Dade North District are in a different column than the monitoring data provided by the utility directly). For the remaining parameters that were analyzed, raw sampling data were provided for each facility. For example, results from one round of sampling were reported for the City of Hollywood, results from two sampling dates were reported for each of the Miami-Dade facilities (February 27, 1991 and February 18, 1992 for North District and February 22, 1991 and September 20, 1991 for Central District), and results from four sampling dates were reported for Broward County (February 13, 1991, September 20, 1991, February 11, 1992, and March 24, 1992). The average concentration for each parameter at each utility was calculated using these concentration data and excluding data points that were identified by the SEFLOE authors as “questionable” (i.e., single values for arsenic, copper, and zinc in Broward County; total silver in the City of Hollywood and Miami-Dade Central; and heptachlor in Miami-Dade North). In instances where the reported concentration was “BDL” (Below Detection Limit), no detection limit was reported; for this reason, a value of zero (0) was used in the calculation of average concentrations.

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Appendix Table 1-1. Drinking Water Standards and Sampling Results for Treated Wastewater and Native Water in South Florida

		Advanced Wastewater Treatment				**Reclaimed Water Treatment**		**Secondary Effluent**		**Native Water Monitoring Zones**				
		Various Counties	**Brevard County**			**Various Counties**	**Pinellas County**	**Various Counties**	**Dade County**	**Florida Utility Council[2] - Various Counties**				
Parameter Name	Drinking Water MCL[1]	Florida Utility Council[2]	South Beaches WWTF[3]	Cape Canaveral WWTP Sample Date 10/1/99[4]	Cape Canaveral WWTP Sample Date 4/3/01[5]	Florida Utility Council[2]	Albert Whitted WRF, St. Petersburg[6]	Florida Utility Council[2]	Miami-Dade North District[7]	Effluent Injection Zone	Lower Monitoring Zone	Upper Monitoring Zone	ASR Injection Zone	Biscayne Monitoring Zone
Inorganic Analysis														
Arsenic (mg/L)	0.050	0.001	< 0.0050	< 0.005	< 0.0045	0.003	<0.00314	0.003	< 0.01	0.010	0.007	0.005	0.002	0.015
Barium (mg/L)	2.000		0.031		0.0070	0.094	<0.0103	0.023	< 0.05	0.184	0.363	0.089	0.404	0.244
Cadmium (mg/L)	0.005	0.000	< 0.0010	< 0.0001	< 0.00005	0.001	<0.0022	0.001	< 0.005	0.004	**0.012**	**0.065**	0.003	0.001
Chromium (mg/L)	0.100	0.001	< 0.0050	< 0.005	< 0.0009	0.003	<0.00625	0.005	< 0.005	0.014	0.023	0.006	0.010	0.004
Cyanide (mg/L)	0.200				< 0.006	0.002	<0.025800	0.015	< 0.004	0.006	0.009	0.004	0.002	0.004
Fluoride (mg/L)	4.000	0.940	0.080		0.505	0.420	0.73700	0.790	0.75	0.700	0.860	1.470	1.580	0.190
Lead (mg/L)	A.L.=0.015	0.000	< 0.0010	< 0.003	0.0014	0.001	<0.00255	0.004	< 0.005	**0.069**	**0.108**	**0.022**	0.002	0.009
Mercury (mg/L)	0.002	0.000	< 0.00020	0.0004	0.00021	0.000	<0.0002	0.000	< 0.001	0.000	0.001	0.001	0.000	0.000
Nickel (mg/L)	0.100	0.002		< 0.03	0.0031	0.005	<0.008	0.011	< 0.005	0.023	0.036	0.025	0.004	0.003
Nitrate (mg/L)	10.00		9.6		0.0620	3.690	0.28000	3.820	0.64	0.420	0.070	0.040	0.030	0.190
Nitrite (mg/L)	1.000				< 0.0035	0.013	0.18000	0.575	< 0.05	0.009	0.025	0.012	0.006	0.005
Selenium (mg/L)	0.050	0.001	< 0.0020	< 0.005	< 0.0026	0.004	<0.00388	0.004	< 0.01	**0.637**	0.007	0.004	0.005	0.001
Sodium (mg/L)	160.0	64.00	**230**		121.0	75.00	28.03500	114.0	181	**8062**	**5514**	**1357**	**1215**	80.0
Antimony (mg/L)	0.006			0.011	0.0034	**0.142**	<0.002175	**0.013**	< 0.005	0.003	**0.019**	**0.010**	0.004	0.001
Beryllium (mg/L)	0.004			< 0.004	< 0.0001	0.004	<0.000525	0.001	< 0.002	0.008	**0.010**	**0.005**	0.001	0.000
Thallium (mg/L)	0.002			< 0.002	< 0.0010	0.001	<0.0012	0.002	< 0.002	**0.305**	**0.013**	**0.007**	0.001	0.001
Secondary Analysis														
Aluminum (mg/L)	0.200				0.0992	0.050	<0.1135	0.074	< 0.1	0.20	**0.917**	**0.744**	0.163	**0.823**
Chloride (mg/L)	250.0	82.20	160		165	116.9	189	151.85	218	**15302.5**	**9897.0**	**2203.3**	**2448.4**	176.2
Copper (mg/L)	A.L.=1.3	0.003	< 0.01	< 0.01	< 0.0005	0.021	0.0086	0.004	< 0.01	0.21	0.032	0.132	0.010	0.005
Iron (mg/L)	0.300	0.000	< 0.040		0.0225	0.177	0.0963333	0.183	0.209	**3.151**	**4.450**	**19.294**	**1.079**	**0.420**
Manganese (mg/L)	0.050		0.0058		0.0185	0.024	0.012567	0.018	< 0.05	0.038	0.046	0.027	0.043	0.013
Silver (mg/L)	0.100		< 0.010	< 0.01	0.0040	0.001	<0.00392	0.002	< 0.001	0.037	0.008	0.005	0.004	0.003
Sulfate (mg/L)	250.0	179.5	110		111	76.20	41.7	56.623	71.9	**2379.2**	**1117.9**	**401.0**	**521.8**	38.80
Zinc (mg/L)	5.000	0.000	0.037	< 0.03	0.0676	0.023	0.036500	0.014	0.02	0.008	0.015	0.059	0.082	0.025
Color (PtCo units)	15.00				5	**33.00**	**30**	**43.91**		7.400	6.300	12.60	12.00	**21.90**
Color (APHA units)									50					
Odor (TON)	3.000				2	2.500	**10**	**10.95**	**75**	1.200	**3.300**	2.100	**13.50**	0.700
pH	6.5-8.5		7.42		6.99	7.000	7.0775	6.863	6.93	7.700	7.900	7.700	7.500	8.100
TDS (mg/L)	500.0		**1200**		648	**528.0**	**557**	**550.71**	**610**	**28682**	**18328**	**4128**	**5240**	**533.0**
TSS (mg/L)				1.066[9]	0.868[9]									
Foaming Agents (mg/L)	1.500		0.13			0.143	0.305	**2.518**		0.080	0.253	0.118	0.074	0.193
Trihalomethane Analysis														
Bromodichloromethane (μg/L)				6.4	10.200				< 0.5					
Dibromochloromethane (μg/L)				40.9	10.700				< 0.5					
Tribromomethane (Bromoform; μg/L)				122	< 0.31000				< 0.5					
Trichloromethane (Chloroform; μg/L)				1.2	3.6900				7.18					
Total THMs (μg/L)	80.00		**230**		24.600	26.850	6.7	61.584	7.18	0.167	0.650	0.500	2.607	0.026

Appendix Table 1-1. Drinking Water Standards and Sampling Results for Treated Wastewater and Native Water in South Florida

		Advanced Wastewater Treatment				Reclaimed Water Treatment		Secondary Effluent		Native Water Monitoring Zones				
		Various Counties	Brevard County			Various Counties	Pinellas County	Various Counties	Dade County	Florida Utility Council[2] - Various Counties				
				Cape Canaveral WWTP										
Parameter Name	Drinking Water MCL[1]	Florida Utility Council[2]	South Beaches WWTF[3]	Sample Date 10/1/99[4]	Sample Date 4/3/01[5]	Florida Utility Council[2]	Albert Whitted WRF, St. Petersburg[6]	Florida Utility Council[2]	Miami-Dade North District[7]	Effluent Injection Zone	Lower Monitoring Zone	Upper Monitoring Zone	ASR Injection Zone	Biscayne Monitoring Zone
Radiological Analysis														
Gross Alpha (pCi/L)	15				< 1.60	3.167	<6.775 +/-1.4	0.400	< 1+/-0.5	9.675	7.300	4.100	**24.660**	5.550
Gross Alpha excl. radon & uranium (pCi/L)			< 4.0+/-2.3											
Radium-226 (pCi/L)					< 0.30		0.4 +/-0.15							
Radium-228 (pCi/L)					< 0.90		<0.75 +/-0.45							
Radium-226 and Radium-228			0.5+/-0.1											
Microbiological Analysis														
Total Coliform (col/100ml)	1, or 5%[8]							**394.071**		**33.50**	**7.000**	**0.500**	**6.000**	
Fecal Coliform (cfu/100ml)	0						<1							
Miscellaneous Analysis														
Ammonia-N (mg/L)							18.0	8.753		3.766	0.561	0.644	0.575	
Nitrogen, total (mg/L)				0.752 [9]	0.97 [9]	13.30	18.3	17.000		9.350	0.881	1.330		
Nitrogen, organic (mg/L)								1.584		0.998	0.374	0.432	0.307	
Nitrogen, total Kjeldahl (mg/L)						4.075	17.9	9.783		5.528	0.474	0.678	0.830	
Nitrate/Nitrite (as N; mg/L)					0.062		0.426667		0.64					
Ortho-phosphate (mg/L)							2.18	1.431		0.234	0.045	0.023	0.133	
Phosphorus, total (mg/L)				0.152 [9]	0.119 [9]	1.375		1.327		0.271	0.261	0.129	0.255	
BOD (mg/L)								8.300		4.300	5.400	7.000	1.400	
CBOD 5 (mg/L)														
Hormonally Active Agents														
Oil and Grease (mg/L)														
Hazardous Algal Bloom including aerosol dist.														
Water Temperature (°C)							26.6	25.333		22.80	23.50	24.30		24.40
Turbidity (NTU)							2.515		11.59					
MBAS Surfactants (mg/L)					< 0.020				0.063					
Synthetic Organic Constituent and Volatile Organic Constituent Analysis														
1,2-Dibromo-3-Chloropropane (DBCP; µg/L)	0.2				< 0.02000		<0.14		< 0.02					
Ethylene Dibromide (EDB; µg/L)	0.05		< 0.01		< 0.01000		<0.02		< 0.02					
Hexachlorocyclopentadiene (µg/L)	50			< 10	< 0.0200		<1.94		< 0.01					
Hexachlorobenzene (µg/L)	1			< 10	< 0.0100		<0.233		< 0.01					
v-BHC (Lindane; µg/L)	0.2		< 0.023	< 0.05	< 0.0240		<0.01		< 0.01					
Alachlor (µg/L)	2				< 0.0625		<1.17		< 0.01					
Heptachlor (µg/L)	0.4			< 0.05	< 0.0540		<0.045		< 0.01					
Heptachlor epoxide (µg/L)	0.2			< 0.05	< 0.0245		<0.035		< 0.01					
Endrin (µg/L)	2		< 0.023	< 0.1	< 0.0100		<0.13		< 0.01					
Methoxychlor (µg/L)	40		< 0.023	< 0.1	< 0.250		<0.21		< 0.01					
Arochlor 1016 (µg/L)				< 1.0					< 0.01					
Arochlor 1221 (µg/L)				< 1.0					< 0.01					
Arochlor 1232 (µg/L)				< 1.0					< 0.01					
Arochlor 1242 (µg/L)				< 1.0					< 0.01					
Arochlor 1248 (µg/L)				< 1.0					< 0.01					
Arochlor 1254 (µg/L)				< 1.0					< 0.01					

Appendix Table 1-1. Drinking Water Standards and Sampling Results for Treated Wastewater and Native Water in South Florida

		Advanced Wastewater Treatment				Reclaimed Water Treatment		Secondary Effluent		Native Water Monitoring Zones				
		Various Counties	Brevard County			Various Counties	Pinellas County	Various Counties	Dade County	Florida Utility Council[2] - Various Counties				
				Cape Canaveral WWTP										
Parameter Name	Drinking Water MCL[1]	Florida Utility Council[2]	South Beaches WWTF[3]	Sample Date 10/1/99[4]	Sample Date 4/3/01[5]	Florida Utility Council[2]	Albert Whitted WRF, St. Petersburg[6]	Florida Utility Council[2]	Miami-Dade North District[7]	Effluent Injection Zone	Lower Monitoring Zone	Upper Monitoring Zone	ASR Injection Zone	Biscayne Monitoring Zone
Arochlor 1260 (µg/L)				< 1.0					< 0.01					
Toxaphene (µg/L)	3		< 0.57	< 1.0	< 0.500		<1.77		< 0.01					
Chlordane (µg/L)	2			< 0.05	< 0.500		<0.64		< 0.01					
Simazine (µg/L)	4				< 0.176		<2.67		< 0.5					
Atrazine (µg/L)	3				< 0.625		<1.4		< 0.2					
Dalapon (µg/L)	200				< 0.802		<1.0		< 1.3					
2,4-D (µg/L)	70		< 0.1		< 0.362		<0.1		< 0.2					
Pentachlorophenol (µg/L)	1			< 50	< 0.0545		<0.04		< 0.2					
Phenols (total; µg/L)														
2,4,5-TP (silvex; µg/L)	50		< 0.2		< 0.0250		<0.2		< 0.2					
Dinoseb (µg/L)	7				< 0.125		<0.2		< 0.2					
Picloram (µg/L)	500				< 0.250		<0.1		< 0.2					
Vinyl Chloride (µg/L)	2		< 1	< 10	< 0.29000		<0.625		< 0.5					
1,1-Dichloroethene (µg/L)	7			< 5	< 0.02000		<0.625		< 0.5					
Methylene Chloride (µg/L)				< 5	< 0.31000				< 0.5					
Trans-1,2-Dichloroethene (µg/L)	100			< 5	< 0.12000		<2.5		< 0.5					
Cis-1,2-Dichloroethene (µg/L)	70				< 0.03000		<0.625		< 0.5					
1,1,1-Trichloroethane (µg/L)	200		< 1	< 5	< 0.21000		<2.5		< 0.5					
Carbon Tetrachloride (µg/L)	5		< 1	< 5	< 0.29000		<0.625		< 0.5					
Benzene (µg/L)	5		< 1	< 5	< 0.05000		<0.625		< 0.5					
1,2-Dichloroethane (µg/L)	5		< 1	< 5	< 0.02000		<0.625		< 0.5					
Trichloroethene (µg/L)	5		< 1	< 5	< 0.02000		<0.875		< 0.5					
1,2-Dichloropropane (µg/L)	5			< 5	< 0.33000		<0.625		< 0.5					
Toluene (µg/L)	1000			< 5	< 0.41000		<2.5		< 0.5					
1,1,2-Trichloroethane (µg/L)	5			< 5	< 0.23000		<0.625		< 0.5					
Tetrachloroethene (µg/L)	5		< 1	< 5	< 0.21000		<0.625		4.66					
Chlorobenzene (µg/L)	100			< 5	< 0.23000		<0.625		< 0.5					
Ethylbenzene (µg/L)	700			< 5	< 0.47000		<2.5		< 0.5					
m & p-Xylene (µg/L)									< 0.5					
o-Xylene (µg/L)									< 0.5					
Xylenes (total; µg/L)	10000				< 0.24000		<0.5							
Styrene (µg/L)	100				< 0.47000		<0.5		< 0.5					
1,4-Dichlorobenzene (para) (µg/L)	75		< 1	< 5	< 0.02000		<0.517		< 0.5					
1,2-Dichlorobenzene (ortho) (µg/L)	600			< 5	< 0.05000		<0.5		< 0.5					
1,2,4-Trichlorobenzene (µg/L)	70			< 10	< 0.22000		<10.9		< 0.5					
Di(2-Ethylhexyl)phthalate (µg/L)	6			< 10	< 1.32		<1.25		< 5					
Di(2-Ethylhexyl)adipate (µg/L)	400				< 0.600		<1.12		< 5					
Benzo(a)pyrene (µg/L)	0.2			< 10	< 0.0400		<0.1		< 0.2					
Carbofuran (µg/L)	40				< 0.900		<2		< 10					
Oxamyl (vydate; µg/L)	200				< 1.13		<2		< 50					
Glyphosate (µg/L)	700				< 2.4		<6		< 10					
Endothall (µg/L)	100				< 3.00		<9.0		< 10					

Appendix Table 1-1. Drinking Water Standards and Sampling Results for Treated Wastewater and Native Water in South Florida

		Advanced Wastewater Treatment				Reclaimed Water Treatment		Secondary Effluent		Native Water Monitoring Zones				
		Various Counties	Brevard County			Various Counties	Pinellas County	Various Counties	Dade County	Florida Utility Council[2] - Various Counties				
				Cape Canaveral WWTP										
Parameter Name	Drinking Water MCL[1]	Florida Utility Council[2]	South Beaches WWTF[3]	Sample Date 10/1/99[4]	Sample Date 4/3/01[5]	Florida Utility Council[2]	Albert Whitted WRF, St. Petersburg[6]	Florida Utility Council[2]	Miami-Dade North District[7]	Effluent Injection Zone	Lower Monitoring Zone	Upper Monitoring Zone	ASR Injection Zone	Biscayne Monitoring Zone
Diquat (µg/L)	20				< 4.00		<0.4		< 0.5					
Paraquat (µg/L)									< 1					
1,1-dichloroethane (µg/L)			< 1	< 5	< 0.10000									
PCB-1242 (mg/L)							<0.00075							
PCB-1254 (mg/L)							<0.00075							
PCB-1221 (mg/L)							<0.00125							
PCB-1232 (mg/L)							<0.00075							
PCB-1248 (mg/L)							<0.00075							
PCB-1260 (mg/L)							<0.00075							
PCB-1016 (mg/L)							<0.00075							
Polychlorinated byphenyls (PCBs; mg/L)	0.0005				< 0.250 mg/L		<0.00023							
2,3,7,8-TCDD (Dioxin; mg/L)	$3x10^{-8}$													
Dichloromethane (mg/L)	0.005						<0.000625							

1 National Primary Drinking Water Regulations, 40 CFR 141 et seq.
2 Englehardt et. al. 2001. Comparative Assessment of Human and Ecological Impacts from Municipal Wastewater Disposal Methods in Southeast Florida, Table 2. Numbers are the average of the means of the measurements calculated with non-detects as zero and non-detects at their detection limit values.
3 South Beaches Wastewater Treatment Facility, Melbourne Beach, FL. 2001. Reclaimed Water or Effluent Analysis Report, Report Period 1/1/2000 - 12/31/2000. If monitoring result below the detection limit, this was indicated by showing a less than (<) sign preceding the detection limit.
4 Florida Department of Environmental Protection. 1999. Annual Reclaimed Water/Effluent Analysis for Primary and Secondary Drinking Water Standards, Cape Canaveral Wastewater Treatment Plant. Samples collected October 1, 1999. Laboratory Order Number B9-10-019.
5 City of Cape Canaveral. 2001. Laboratory Order Number 11926.
6 City of St. Petersburg, Public Utilities Department, Albert Whitted Water Reclamation Facility. Values are the average of sampling results from 9/98, 1/99, 4/99, 9/00, and 1/01, except values for ammonia, total nitrogen, total Kjeldahl nitrogen, orthophosphate, and water temperature, which are actual values measured 11/9/00. Values that were non-detects with a detection limit greater than the MCL were excluded from the calculation of the averages. A "less than" sign preceding a value indicates that at least one of the annual sampling results was below the detection limit. It does not necessarily indicate that all annual sampling results were below the detection limit for any given constituent.
7 Miami Dade Water/Sewer, North District. 1999. Submission #9903001041, pp. 47-52. Screen effluent collected 3/19/99. Monitoring results below detection limits are indicated by showing a less than (<) sign preceding the reported detection limit.
8 For systems that collect >40 samples per month, MCL is 5% monthly samples are positive; for systems that collect <40 samples per month, MCL is 1 positive sample.

9 Annual Average calculated from monthly averages in 1999 and 2001 supplied by Florida Department of Environmental Protection (Cape Canaveral NPDES constituent data). Data from May and June 2001 are unavailable; therefore, annual averages for 2001 are calculated from 10 monthly averages.

Appendix Table 1-2. Summary of Treated Wastewater Effluent Characteristics - Southeast Florida Outfall Experiment (SEFLOE)

	Parameter Name	Broward	Hollywood	Dade-North	Dade-Central
Arithmetic Average	Ammonia (mg/L)	12.48	5.96	10.46	
	Nitrogen, total Kjeldahl (mg/L)	14.31	9.38	13.4	
	Phosphorus, Total (mg/L)	1.66	0.97	1.6	
	Nitrates (mg/L)	0.42	1.70		
	Nitrites (mg/L)	2.01			
	Nitrates + Nitrites (mg/L)	2.07			
	Oil & Grease (mg/L)	2.17	19.24	3.27	2.54

	Broward		Hollywood	Dade-North		Dade-Central	
	Arithmetic Average of data from four sampling dates		Data from one sampling date	Arithmetic Average of data from two sampling dates		Arithmetic Average of data from two sampling dates	
Parameter Name	Average	Average, excluding "questionable" data points	Sampling Data	Average	Average, excluding "questionable" data points	Average	Average, excluding "questionable" data points
1,1,1 Trichloroethane (µg/L)	1.1	1.1					
Antimony Total (mg/L)				0.013	0.013	0.022	0.022
Arsenic Total (mg/L)	0.032	0.001		0.0004	0.0004		
Cadmium Total (mg/L)	0.002	0.002		0.002	0.002	0.005	0.005
Chloroform (µg/L)	1.65	1.65	10	9.01	9.01		
Chromium Total (mg/L)	0.047	0.047					
Copper Total (mg/L)	0.037	0.012		0.018	0.018	0.023	0.023
Cyanide Total (mg/L)				0.008	0.008	0.005	0.005
Dichlorobromomethane (µg/L)	0.615	0.615					
Ethylbenzene (µg/L)				0.25	0.25		
Heptachlor (µg/L)				0.092	0		
Lead Total (mg/L)	0.004	0.004		0.010	0.010	0.02	0.02
Nickel total (mg/L)	0.015	0.015		0.003	0.003	0.003	0.003
Phenols, Total (µg/L)			70			11	11
Selenium Total (mg/L)	0.007	0.007		0.0005	0.0005		
Silver Total (mg/L)	0.0005	0.0005	0.010 (ques)			0.007	0
Tetrachloroethylene (µg/L)						3	3
Thallium Total (mg/L)				0.019	0.019	0.007	0.007
Toluene (µg/L)				1.07	1.07		
Zinc Total (mg/L)	0.054	0.036	0.015	0.015	0.015	0.041	0.041

1 Hazen and Sawyer, Inc. 1994. SEFLOE II Final Report. pp. III-182-185; III-202-205; and III-210-213. Average values represent 7-43 sampling records from Broward County North Regional Wastewater Treatment Plant, the Miami-Dade North District and Central District Wastewater Treatment Plants, and the City of Hollywood Wastewater Treatment Plant.

Appendix Table 1-2. Summary of Treated Wastewater Effluent Characteristics - SEFLOE data[1] Continued

Broward

Parameter	2/13/1991	9/20/1991	2/11/1992	3/24/1992	average	average -ques points
Dichlorobromomethane (µg/L)	**n/a**	**n/a**	0	1.23	0.615	0.615
Chloroform (µg/L)	1.7	0.0	2.72	2.18	1.65	1.65
1,1,1 Trichloroethane (µg/L)	1.5	0.0	2.68	0	1.0525	1.0525
Arsenic Total (µg/L)	0.0	124.0	1.7	2.3	32	1.333333333
Cadmium Total (µg/L)	0.0	8.3	0	0.3	2.15	2.15
Chromium Total (µg/L)	2.8	3.3	3.2	179	47.075	47.075
Copper Total (µg/L)	2.1	20.0	111.3	14.4	36.95	12.16666667
Lead Total (µg/L)	0.0	5.0	4.8	6.7	4.125	4.125
Nickel total (µg/L)	4.2	44.0	6.8	6.7	15.425	15.425
Selenium Total (µg/L)	0.0	23.3	1	2	6.575	6.575
Silver Total (µg/L)	0.0	0.5	0.9	0.5	0.475	0.475
Zinc Total (µg/L)	20.0	52.5	111	34	54.375	35.5

Hollywood

Parameter	sampling point (11/25/91)
Chloroform (µg/L)	10
Silver, Total (µg/L)	10 (ques)
Zinc, Total (µg/L)	15
Phenols, Total (µg/L)	70

Dade-North

Parameter	2/27/1991	2/18/1992	average	ave - ques points
Chloroform (µg/L)	10.01	8.0	9.005	9.005
Ethylbenzene (µg/L)	0.5	0	0.25	0.25
Toluene (µg/L)	2.14	0	1.07	1.07
Heptachlor (µg/L)	0.183	0	0.0915	0
Antimony Total (µg/L)	26.3	0	13.15	13.15
Arsenic Total (µg/L)	0.83	0	0.415	0.415
Cadmium Total (µg/L)	3.0	0	1.5	1.5
Copper Total (µg/L)	19.0	16.0	17.5	17.5
Lead Total (µg/L)	20.2	0	10.1	10.1
Nickel total (µg/L)	5	0	2.5	2.5
Selenium Total (µg/L)	0.91	0	0.455	0.455
Thallium Total (µg/L)	38.9	0	19.45	19.45

Dade-Central

Parameter	2/22/1991	9/20/1991	average	ave - ques points
Tetrachloroethylene (µg/L)	0	6	3	3
Antimony Total (µg/L)	44.8	0	22.4	22.4
Cadmium Total (µg/L)	9	0	4.5	4.5
Copper Total (µg/L)	35	10	22.5	22.5
Lead Total (µg/L)	40	0	20	20
Nickel total (µg/L)	5	0	2.5	2.5
Silver, Total (µg/L)	14	0	7	0
Thallium Total (µg/L)	13	0	6.5	6.5
Zinc Total (µg/L)	82	0	41	41
Cyanide Total (µg/L)	9.6	0	4.8	4.8
Phenols, Total (µg/L)		11	11	11

Appendix Table 1-3. Microbial Standards and Concentrations in Treated Wastewater.

Microbial Standards

Microbial Pathogens and Sewage Indicators	**Drinking Water Maximum Containment Level[1]**	**Florida Department of Environmental Protection Recommended Limits[2]**		**Summary of Requirements For Disinfection Used in South Florida**		
		Average	Maximum	Basic disinfection	Intermediate disinfection	High-Level disinfection
Total Coliform (col/100ml)	1, or 5%					
Fecal Coliform (cfu/100ml)	0			$\leq$ 200	$\leq$ 14	BDL
Enterovirus (mpn/100 L)						
Enterovirus (PFU/100 L)[2a]		0.044	0.165			
Enterovirus (PFU/100 L)[2b]		14	50			
Cryptosporidium (oocysts/100 L)		5.8	22	5	5	5
Giardia lamblia (cysts/100 L)		1.4	5			
Enterococci (cfu/100 mL)						
Clostridium perfringens (cfu/100 mL)						
Coliphages (pfu/100 mL)						
Enterovirus (PFU/100L)						
Coliphages Host E. coli (pFamp) (PFU/100 mL)						
Coliphages Host E. coli C (PFU/100 mL)						

BLD = Below Detection Limit

Microbial Surface Water Concentrations (1)

Microbial Pathogens and Sewage Indicators	**Surface Water**							
	Sarasota County						**Hillsborough County**	
	5 Streams in the vicinity of Sarasota[1]		Sarasota Bay [1]		Phillippi Creek[1]		Tampa Bypass Canal[1]	
	Average	Range	Average	Range	Average	Range	Average	Range
Total Coliform (col/100ml)								
Fecal Coliform (cfu/100ml)								
Enterovirus (mpn/100 L)								
Cryptosporidium (oocysts/100 L)	6.6	ND-157	ND	ND	3.1	ND-11	3.1	ND-11
Giardia lamblia (cysts/100 L)	ND	ND	ND	ND	0.42	ND-2.9	0.42	ND-2.9
Enterococci (cfu/100 mL)								
Clostridium perfringens (cfu/100 mL)								
Coliphages (pfu/100 mL)								
Enterovirus (PFU/100L)								
Coliphages Host E. coli (pFamp) (PFU/100 mL)								
Coliphages Host E. coli C (PFU/100 mL)								

ND= Nondetect

Microbial Surface Waters Concentrations (2)

Microbial Pathogens and Sewage Indicators	Surface Waters
	Brevard County
	Duda Ditches[2]
	single sampling date
Total Coliform (col/100ml)	
Fecal Coliform (cfu/100ml)	100.9
Enterovirus (mpn/100 L)	
Cryptosporidium (oocysts/100 L)	
Giardia lamblia (cysts/100 L)	
Enterococci (cfu/100 mL)	
Clostridium perfringens (cfu/100 mL)	
Coliphages (pfu/100 mL)	
Enterovirus (PFU/100L)	
Coliphages Host E. coli (pFamp) (PFU/100 mL)	
Coliphages Host E. coli C (pfu/100 mL)	

Microbial Concentrations in Untreated and Treated Wastewater (1)

Microbial Pathogens and Sewage Indicators	**Raw Wastewater**	**Secondary Treated Wastewater**	**Secondary Treated Wastewater**					
	United States	**Dade County**	**Broward County**					
	Urban Communities within the United States[1]	MDWSD North District IW3[2]	City of Fort Lauderdale[3]	City of Hollywood[3]	Sunrise (IW1 and IW2)[3]	Sunrise (IW3)[3] Sawgrass	Hollywood WTP (reuse filter)[3]	Broward County Regional WWTP (Reuse Composite Sampler)[4]
	sampling dates unknown	sampling date 3/19/99	sampling date 4/25/96	sampling date 4/25/96	sampling date 2/11/00	sampling date unknown	sampling date 7/9/99	average of 30 values taken in September 2001
Total Coliform (col/100ml)	22 x (10^6)	0.0005	2100	0.5	280	180	0.5	0.033
Fecal Coliform (cfu/100ml)	8 x (10^6)							0
Enterovirus (mpn/100 L)								
Cryptosporidium (oocysts/100 L)								
Giardia lamblia (cysts/100 L)								
Enterococci (cfu/100 mL)								
Clostridium perfringens (cfu/100 mL)								
Coliphages (pfu/100 mL)								
Enterovirus (PFU/100L)								
Coliphages Host E. coli (pFamp) (PFU/100 mL)								
Coliphages Host E. coli C (pfu/100 mL)								

Microbial Concentrations in Treated Wastewater (2)

Microbial Pathogens and Sewage Indicators	**Secondary Effluent**				
	Brevard County[5]				
	BCUD/ South Central Regional WWTF[6]	South Beaches WWTF[7]	BCUD/Port St. John WWTF	Barefoot Bay Advanced Wastewater Treatment Facility	BCUD/ Sykes Creek Regional WWTF
	daily sample results	daily sample results	daily sample results	daily sample results-R001 reuse irrigation	daily sample results-reuse
	(mon site EFA-1; avg of monthly)	(mon site EFA-1; avg of monthly)	(mon site EFA-1; avg of monthly)	(mon site EFA-1; avg)	(mon site EFA-2; avg)
Total Coliform (col/100ml)					
Fecal Coliform (cfu/100ml)	0.03	0.04	0.18	0	0
Enterovirus (mpn/100 L)					
Cryptosporidium (oocysts/100 L)					
Giardia lamblia (cysts/100 L)					
Enterococci (cfu/100 mL)					
Clostridium perfringens (cfu/100 mL)					
Coliphages (pfu/100 mL)					
Enterovirus (PFU/100L)					
Coliphages Host E. coli (pFamp) (PFU/100 mL)					
Coliphages Host E. coli C (pfu/100 mL)					

Microbial Concentrations in Treated Wastewater (3)

Microbial Pathogens and Sewage Indicators	**Reclaimed Water**			**Advanced Wastewater Treatment**			
	Pinellas County			**Brevard County**		**Hillsborough County**	
	Albert Whitted WRF, St. Petersburg[8]	St. Petersburg[9]		Cape Canaveral WWTP NPDES database		Howard Curren WWTP[12]	
	Sampling date 11/28/00	Average	Maximum	Annual average 1999[10]	Annual average 2001[11]	Sample date 5/5/00	Sample date 5/16/01
Total Coliform (col/100ml)							
Fecal Coliform (cfu/100ml)				0.125	1.15		
Enterovirus (mpn/100 L)							
Cryptosporidium (oocysts/100 L)	1.85	0.75	5.35			<0.7	2.33
Giardia lamblia (cysts/100 L)	0.26	0.49	3.3			<0.7	<0.29
Enterococci (cfu/100 mL)							
Clostridium perfringens (cfu/100 mL)							
Coliphages (pfu/100 mL)							
Enterovirus (PFU/100L)		0.01	0.133				
Coliphages Host E. coli (pFamp) (PFU/100 mL)							
Coliphages Host E. coli C (pfu/100 mL)							

Microbial Concentrations in Monitoring Wells (1)

Microbial Pathogens and Sewage Indicators	**Deep Injection Monitoring Wells**			
	Pinellas County			
	Well ID AWWRF 757[1]	Well ID AWWRF 758[1]	Well ID AWWRF 779[1]	St Petersburg[1]
	Average of the results from 20 sampling events between 4/98 and 12/00	Average of the results from 20 sampling events between 4/98 and 12/00	Average of the results from 20 sampling events between 4/98 and 12/00	Sampling date 10/13/00[2]
Total Coliform (col/100ml)				<0.058
Fecal Coliform (cfu/100ml)				<0.058
Enterovirus (mpn/100 L)	<1.0	<1.0	<1.0	<0.1
Cryptosporidium (oocysts/100 L)				<0.17
Giardia lamblia (cysts/100 L)				<0.17
Enterococci (cfu/100 mL)				<0.058
Clostridium perfringens (cfu/100 mL)				<0.058
Coliphages (pfu/100 mL)				
Enterovirus (PFU/100L)				
Coliphages Host E. coli (pFamp) (PFU/100 mL)				<5
Coliphages Host E. coli C (pfu/100 mL)				<5

Microbial Concentrations in Ground Water Monitoring Wells (2)

Microbial Pathogens and Sewage Indicators	**Deep Injection Monitoring Wells**				
	Various Counties				
	Native Water Monitoring Zones				
	Effluent Injection Zone	Lower Monitoring Zone	Upper Monitoring Zone	ASR Injection Zone	Biscayne Monitoring Zone
Total Coliform (col/100ml)	33.50	7.00	0.500	6.00	
Fecal Coliform (cfu/100ml)					
Enterovirus (mpn/100 L)					
Cryptosporidium (oocysts/100 L)					
Giardia lamblia (cysts/100 L)					
Enterococci (cfu/100 mL)					
Clostridium perfringens (cfu/100 mL)					
Coliphages (pfu/100 mL)					
Enterovirus (PFU/100L)					
Coliphages Host E. coli (pFamp) (PFU/100 mL)					
Coliphages Host E. coli C (pfu/100 mL)					

Microbial Concentrations in Ground Water Monitoring Wells (3)

Microbial Pathogens and Sewage Indicators	Deep Injection Monitoring Wells					
	Pinellas County					
	AWWRF well 779[2]	SWWRF well 765[2]	SWWRF well 768[2]	NWWRF well 798[2]	AWWRF well 758[2]	NEWRF well 784[2]
	Sample date 12/1/99	Sample date 12/1/99	Sample date 12/1/99	Sample date 12/1/99	Sample date 12/1/99	Sample date 12/1/99
Total Coliform (col/100ml)	<0.058	<0.058	<0.058	<0.058	<0.058	<0.058
Fecal Coliform (cfu/100ml)	<0.058	<0.058	<0.058	<0.058	<0.058	<0.058
Enterovirus (mpn/100 L)	<0.071	<0.060	<0.075	<0.05	0.074	<0.080
Cryptosporidium (oocysts/100 L)	1.18	<0.30	0.74	0.36	0.14	<0.11
Giardia lamblia (cysts/100 L)	<0.29	<0.30	<0.15	<0.30	<0.14	<0.11
Enterococci (cfu/100 mL)	<0.058	<0.058	<0.058	<0.058	<0.058	<0.058
Clostridium perfringens (cfu/100 mL)	<0.058	<0.058	<0.058	<0.058	<0.058	<0.058
Coliphages (pfu/100 mL)	<5	<5	<5	<5	<5	<5
Enterovirus (PFU/100L)						
Coliphages Host E. coli (pFamp) (PFU/100 mL)						
Coliphages Host E. coli C (pfu/100 mL)						

FOOTNOTES TO APPENDIX 1-3 (MICROBIAL PATHOGEN TABLE).

Footnotes for Table – Microbial Standards

1 Maximum Contaminant Level (MCL). National Primary Drinking Water Regulations, 40 CFR 141 et seq.

2 York, D.W., P. Menendez, and L. Walker-Coleman. 2002. Pathogens in Reclaimed Water: the Florida Experience. 2002 Water Sources Conference.
- 2a. Assumes all Enterovirus is highly infective Rotavirus.
- 2b. Assumes all Enterovirus is moderately infective Echovirus.

Footnotes for Table - Microbial Concentrations in Treated Effluent

1 Geldriech, E.E. 1978 in Wood, I.R. et al. 1993. Ocean Disposal of Wastewater. Advanced Series on Ocean Engineering. Volume 8. World Scientific Publishing Co. Pte. Ltd. Samples taken from several urban communities in the United States.

2 Englehardt al. 2001. Comparative Assessment of Human and Ecological Impacts from Municipal Wastewater Disposal Methods in Southeast Florida. Florida Water and Environment Utility Council.

3 Englehardt al. 2001. Comparative Assessment of Human and Ecological Impacts from Municipal Wastewater Disposal Methods in Southeast Florida. Florida Water and Environment Utility Council.

4 Broward County Office of Environmental Services, Environmental Operations Division, Compliance and Monitoring Section. Facsimile. Contact: Richard Walker.

5 Florida Department of Environmental Protection Discharge Monitoring Reports for Brevard County.

6 Florida Department of Environmental Protection Discharge Monitoring Reports for Brevard County. No detection limit was given; zero was used in calculations where non-detect (ND) was entered on data form. Values are averages from monthly reported values for March, April, and May 2001, except for “created wetlands” value, which is the average of March and April reported values (no discharge to wetlands in May 2001) and for “surface water” value.

7 Florida Department of Environmental Protection Discharge Monitoring Reports for Brevard County. Values are averages from monthly reported values for March, April, and May 2001, calculated by Horsley & Witten, Inc. No detection limit was given; zero was used in calculations where non-detect (ND) was entered on data form.

8 Sampling results provided by Mr. Alfredo Crafa, Environmental Compliance Division, Albert Whitted Wastewater Reclamation Facility, March 18, 2002, City of St. Petersburg, Florida.

9 York, D.W., P. Menendez, and L. Walker-Coleman. 2002. Pathogens in Reclaimed Water: the Florida Experience. 2002 Water Sources Conference.

10 Annual average calculated from monthly averages in 1999 supplied by Florida Department of Environmental Protection (Cape Canaveral National Pollution Discharge Elimination System constituent data). Data from May, June, November, and December are unavailable; therefore, coliforms per 100 mL annual averages for 1999 are calculated from 8 monthly averages. Seven of eight months reported 0 fecal coliforms; one month detected <1 fecal rk. Personal Communication (February 22, 2002). Results are for pathogens in reclaimed wastewater intended for reuse.

11 Annual average calculated from twelve monthly averages in 2001 supplied by Florida Department of Environmental Protection (Cape Canaveral NPDES constituent data). Eleven months reported <1 cfu/100 mL; one month (January) reported 2.8 cfu/100 mL.

12 David York, Ph.D., P.E., Reuse Coordinator, Florida Department of Environmental Protection, personal communication (February 22, 2002). Results are the pathogens in reuse effluent from the Howard Curran Wastewater Treatment Plant.

Footnotes for Table - Microbial Data from Monitoring Wells

1 The Albert Whitted Wastewater Reclamation Facility provided sampling data for microbes from effluent treated to Advanced treatment standards. Values are the average of 20 sampling events for microbial concentrations in three (3) monitoring wells between the period of March 1998 and December 2000.

2 Sampling results provided by Mr. Alfredo Crafa, Environmental Compliance Division, Albert Whitted Wastewater Reclamation Facility, March 18, 2002, City of St. Petersburg, Florida.

3 Rose, J.B., and W. Quintero-Betancourt, J. Jarrel, E. Lipp, S. Farrah, G. Lukasik, and T. Scott. 2001. Deep Injection Monitoring Well: Water Quality Monitoring Report. Report to the Florida Department of Environmental Protection.

Footnotes for Table - Microbial Data from Surface Waters

1 Florida Department of Environmental Protection, Risk Impact Statement, Phase II Revisions to Chapter 62-610, F.A.C., Docket No. 95-08R.

2 Average of samples taken on 3/13/01 at 10 surface water-sampling stations. Florida Department of Environmental Protection Discharge Monitoring Reports.

Footnotes for Table - Microbial Data from Ohio River in the Cincinnati, Ohio Area

1 York, D.W., P. Menendez, and L. Walker-Coleman. 2002. Pathogens in Reclaimed Water: the Florida Experience. 2002 Water Sources Conference. Values are the average of five separate sampling events.

2 York, D.W., P. Menendez, and L. Walker-Coleman. 2002. Pathogens in Reclaimed Water: the Florida Experience. 2002 Water Sources Conference. Values are the average of four separate sampling events.

3 York, D.W., P. Menendez, and L. Walker-Coleman. 2002. Pathogens in Reclaimed Water: the Florida Experience. 2002 Water Sources Conference. Values are the average of two separate sampling events.

Footnotes for Table - SDWTP Monitoring Well Data, Dade County

1 South District Wastewater Treatment Plant, Miami-Dade County, Florida. Monitoring Well Purging Report.

Appendix 1-3. Microbial Pathogens and Description of Data Sources

1.0 General

Appendix Table 1-3 provides data on microbial concentrations in treated effluent and monitoring samples, collected from various sources. These sources include information compiled from:

- The National Pollutant Discharge Elimination System (NPDES) effluent quality database for Cape Canaveral WWTP including years 1999-2001;
- David York (personal communication), regarding microbial concentrations in treated effluent at the Howard Curren WWTP in Hillsborough County;
- Several sets of microbial data from the Alfred Whitted AWT facility, including results for treated effluent and deep injection monitoring wells;
- A Florida Water Environment Association Utility Council (FWEAUC) report, including monitoring data from several groundwater monitoring zones and data from secondary-treated effluent from facilities in Broward and Dade counties;
- A report to the Florida Department of Environmental Protection authored by JB Rose, W. Quintero-Betancourt, J. Jarrel, E. Lipp, S. Farah, G. Lukasic, and T. Scott in 2001, which includes data for six deep injection monitoring well wells in St. Petersburg, Pinellas County;
- Florida Department of Environmental Protection Discharge Monitoring Reports for several wastewater treatment facilities in Brevard County, including:
    - BCUD/South Central Regional WWTF
    - Barefoot Bay WWTF
    - BCUD/Sykes Creek Regional WWTP
    - BCUD/Port St. John WWTF
    - South Beaches WWTF
- The Broward County Office of Environmental Services, which provided water quality data for reclaimed water (including total coliform and fecal coliform values) from the Broward County Regional Wastewater Treatment Plant for the month of September 2001;
- A report by D.W. York, P. Menendez, and L. Walker-Coleman entitled *Pathogens in Reclaimed Water: The Florida Experience* 2002, which includes a review of reclaimed water quality in St. Petersburg as reported by J.B. Rose and R. P. Carnahan; and
- A Risk Impact Statement prepared in 1998 by the Florida Department of Environmental Protection, which includes surface water monitoring data for microbes for Sarasota and Hillsborough Counties.

2.0 The National Pollutant Discharge Elimination System (NPDES)

National Pollutant Discharge Elimination System (NPDES) data for Cape Canaveral for the years 1999-2001were obtained in spreadsheet format from the Florida Department of Environmental Protection. The average annual concentrations of fecal coliform bacteria in treated effluent from this facility were calculated from monthly averages in 1999 and

2001. Data from May, June, November, and December 1999 were unavailable; therefore, the annual average number of fecal coliform colony forming units (cfu) per 100 mL was calculated from 8 monthly averages. Seven of eight months reported zero (0) fecal coliforms; one month detected fewer than 1 cfu per 100 mL of treated effluent. During 2001, eleven of twelve monthly results were less than 1 cfu/100 mL; one month (January) reported 2.8 cfu/100 mL.

3.0 Howard Curren WWTP

David York, Ph.D., Reuse Coordinator for the Florida Department of Environmental Protection, provided results from two sampling events at the Howard Current Wastewater Treatment Plant. These sampling events measured *Giardia* and *Cryptosporidium* in effluent treated to reuse standards. These sampling events occurred on May 5, 2000, and on May 16, 2001.

4.0 Albert Whitted Water Reclamation Facility

The Albert Whitted Wastewater Reclamation Facility provided sampling data for microbes from effluent treated to Advanced Treatment standards (sampling date November 28, 2000) as well as from three deep monitoring wells (sampling date October 13, 2000). The microbial results for the treated effluent sample were obtained from a single sample of 378.5 liters. The deep monitoring well results in the table reflect the data from each of the three wells as well as duplicate samples for each monitoring well; all microbial parameters were below detection limits (indicated by the "less than" (<) sign) in all the monitoring well samples.

5.0 Florida Water Environment Association Utility Council

The Florida Water Environment Association Utility Council (FWEAUC) report (Englehardt et al., 2001) provided analysis of sampling and monitoring results of effluent that had been treated to different standards (advanced wastewater treatment, secondary treatment, and advanced secondary treatment) as well as "native" ambient water in injection zones and monitoring zones in target aquifers. The data that presented in the "Native Water Monitoring Zones" columns in Table X.X represents "digested" data that have already been processed by the FWEAUC authors, who calculated the average concentrations of each parameter from several sampling locations and events. The "digested" data effectively assign a value of one-half the detection limit to non-detects, a standard approach not inconsistent with risk assessment methodologies (US EPA. 1998).

The authors of the FWEAUC report also include raw concentration data for each of the sampling stations in appendices B and C of their report. The concentration of total coliform bacteria for several wastewater treatment facilities in south Florida were obtained from these appendices and entered into Table X.X. Microbial data were only available for facilities treating to secondary treatment standards. Non-detects (for the City of Hollywood treated effluent and reuse filter) were assigned a value of one-half the

detection limit of 1.0 cfu/100 mL (0.5 cfu/100 mL) in the table. Facilities for which total coliform bacteria concentrations were available include:

- City of Hollywood (Broward County);
- City of Sunrise Sawgrass Facility (IW3; Broward County);
- City of Ft. Lauderdale (Broward County);
- Miami-Dade Water and Sewer Department North District (IW3; Dade County);
- City of Hollywood Reuse Filter (Broward County); and
- City of Sunrise (IW1 + IW2; Broward County).

St. Petersburg, Pinellas County

A report to the Florida Department of Environmental Protection authored by JB Rose, W. Quintero-Betancourt, J. Jarrel, E. Lipp, S. Farah, G. Lukasic, and T. Scott in 2001 includes monitoring data for six deep monitoring wells associated with Class I municipal injection wells at four wastewater facilities in St. Petersburg: the Southwest Wastewater Reclamation Facility (SWWRF), the Northwest Wastewater Reclamation Facility (NWWRF), the Northeast Wastewater Reclamation Facility (NEWRF) and the Albert Whitted Wastewater Reclamation Facility (AWWRF). Values entered into Table X.X (monitoring wells) represent results from single sampling events at each monitoring well.

6.0 Florida Department of Environmental Protection Discharge Monitoring Reports

The Florida Department of Environmental Protection provided monthly Discharge Monitoring Reports covering March, April, and May 2001, for several wastewater treatment facilities in Brevard County. Those facilities include:

- BCUD/South Central Regional WWTF
- Barefoot Bay WWTF
- BCUD/Sykes Creek Regional WWTP
- BCUD/Port St. John WWTF
- South Beaches WWTF

For each of the facilities daily monitoring data were provided for Fecal Coliform levels. The values entered in Table XX (treated effluent) are the averages for March, April and May that are then averaged together.

7.0 Broward County Office of Environmental Services

Richard Walker provided monitoring data, via facsimile, from the Broward County Office of Envirnomental Services Analytical Laboratory, for the Broward County Regional WWTP. Daily monitoring data was supplied for the month of September 2001 for advanced secondary treated effluent. The sampling location was the Reuse Composite Sampler. Total and Fecal Coliform levels were reported in counts/100 mL.

The values in Table XX (treated effluent) are the average of the 30 values reported for September 2001.

8.0 Pathogens in Reclaimed Water: The Florida Experience

The paper authored by David York, Ph.D., and Lauren Walker-Coleman of the Florida Department of Environmental Protection and Pepe Menendez, P.E., of the Florida Department of Health outlines Florida's addition of required monitoring for the protozoan pathogens *Cryptosporidium* and *Giardia* in domestic wastewater, to the state's regulations regarding water reuse. The paper contains summarized monitoring data through September 2001 taken in Monterey County, California and St. Petersburg Florida. The data provided in Table XX (treated effluent) represents the average of all data collected through September 2001 and the maximum concentration of pathogens found reclaimed water. The paper also contains fecal and total coliform data from the Ohio River in the Cincinnati, Ohio area. These data were taken over a four month period from September 8, 1975 through December 1, 1975. Three separate sampling stations were monitored and the values in table XY (surface water) are the averages of the combined sampling events at each separate station.

9.0 Risk Impact Statement, Florida Department of Environmental Protection

Surface water monitoring data for microbes for Sarasota and Hillsborough Counties were taken from the Risk Impact Statement, Phase II Revisions prepared by the Florida Department of Environmental Protection. The data provided in Table XX represents the average concentrations and the range of oocysts/100L of water of Giardia and Cryptosporidium in reclaimed water in St. Petersburg and surface waters in Sarasota and Hillsborough Counties. The sampling dates for this study are unknown. The surface waters samples collected in Sarasota County include 24 samples taken in five streams, four samples taken from a high quality estuary within Sarasota Bay, and 16 samples taken from Phillippi Creek, an urban stream within Sarasota. The samples collected in Hillsborough County include seven samples taken from the Tampa Bypass Canal

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BCUD/South Central Regional Wastewater Treatment Facility, Brevard County Water Resources Department, Brevard County. Discharge Monitoring Reports April 01, 2001- April 30, 2001.

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BCUD/Sykes Creek Regional Wastewater Treatment Plant, Brevard County Water Resources Department, Brevard County. Discharge Monitoring Reports March 01, 2001- March 31, 2001.

BCUD/Sykes Creek Regional Wastewater Treatment Plant, Brevard County Water Resources Department, Brevard County. Discharge Monitoring Reports April 01, 2001- April 30, 2001.

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BCUD/Port St. John Wastewater Treatment Facility, Brevard County Water Resources Department, Brevard County. Discharge Monitoring Reports March 01, 2001- March 31, 2001.

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Cape Canaveral National Pollutant Discharge Elimination System (NPDES) Monitoring Database 1999-2001. Permit Number FL0020541.

City of St. Petersburg, Environmental Compliance Division. Sampling data for October 13, 2000 at the Albert Whitted Wastewater Treatment Facility. Contact Alfredo J. Crafa, Manager.

David York, Ph.D., P.E., Reuse Coordinator, Florida Department of Environmental Protection, 2600 Blair Stone Rd.- MS 3540 Tallahassee, Florida 32399-2400, phone: (850) 922-2034, fax: (850) 921-6385, email: david.york@dep.state.fl.us

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South Beaches Wastewater Treatment Facility, Brevard County Water Resources Department, Brevard County. Discharge Monitoring Reports May 01, 2001 – May 31, 2001.

South District Wastewater Treatment Plant, Miami Dade Water and Sewer Department, Miami Dade County, Florida. Monitoring Well Purging Report (Report A.) December 26, 2002.

Appendix Table 1-4. Fecal Coliform Concentrations in Secondary Treated Wastewater Effluent, South Dade Wastewater Treatment Plant, Dade County.

Effluent[1]		
Number of times sampled and range of dates	**Fecal coliform colonies/100 mL**	**Date(s) detected**
44 5/7/91-5/4/93	0	5/7/91
	40,000	5/14/91
	>400	5/21/91
	9,200	5/28/91
	800	6/4/91
	72,000	6/18/91
	152,000	6/25/91
	180,000	8/6/91
	4,000	8/13/91
	80,000	8/20/91
	190,000	8/27/91
	150,000	9/17/91
	260,000	9/24/91
	490,000	10/1/91
	430,000	10/16/90
	300	10/22/91
	300,000	10/29/91
	160,000	11/5/91
	80,000	11/13/91
	50,000	11/19/91
	280,000	11/26/91
	170,000	12/3/91
	18,000	12/10/91
	24,000	12/17/91
	580,000	12/26/91
	17,000,000	1/2/92
	150,000	1/14/92
	0	1/21/91
	20,000	1/28/92
	80,000	2/4/92
	50,000	2/13/92
	30,000	2/18/92
	30,000	2/25/92
	510,000	3/3/92
	50,000	3/10/92
	50,000	3/17/92
	21	3/2/93
	10,000	3/9/93
	50,000	3/16/93
	160,000	3/30/93

Effluent[1]		
Number of times sampled and range of dates	**Fecal coliform colonies/100 mL**	**Date(s) detected**
44 5/7/91 - 5/4/93 (cont.)	10,000	4/6/93
	32,000	4/13/93
	160,000	4/20/93
	124	4/27/93
52 5/11/93-4/4/95	18,000	5/4/93
	29,000	5/11/93
	12,000	5/18/93
	7,000	5/25/93
	4,800	6/3/93
	55,000	6/8/93
	52,000	6/15/93
	16,000	6/21/93
	152,000	6/29/93
	19,400	7/8/93
	13,800	7/13/93
	58,000	7/20/93
	61,000	7/27/93
	39,000	8/5/93
	21,000	8/9/93
	31,000	8/18/93
	6,000	8/25/93
	6,600	9/1/93
	26,000	9/8/93
	22,000	9/14/93
	2,650	9/21/93
	17,400	9/28/93
	32,500	10/5/93
	5,000	10/13/93
	235	10/19/93
	12,000	10/28/93
	900	11/2/93
	40	11/9/93
	48,000	11/16/93
	140,000	11/30/93
	3,000	12/7/93
	210,000	12/21/93
	36,000	12/28/93
	310,000	1/6/94
	760,000	1/11/94
	74,000	1/18/94
	4	1/10/95
	2	1/31/95

Effluent[1]		
Number of times sampled and range of dates	**Fecal coliform colonies/100 mL**	**Date(s) detected**
52 5/11/93-4/4/95	4	2/6/95
	2	2/13/95
	19,800	2/21/95
	9,800	2/28/95
	156	3/8/95
	230,000	3/28/95
	120,000	4/4/95
49 4/18/95-2/4/97	62	4/18/95
	6	5/2/95
	12,000	5/16/95
	46,000	5/23/95
	100,000	5/31/95
	180,000	6/7/95
	8	6/13/95
	154,000	6/20/95
	70,000	6/27/95
	80,300	7/5/95
	120,000	7/18/95
	64,000	7/21/95
	24,000	8/3/95
	58,000	8/8/95
	13,000	8/15/95
	58,000	8/23/95
	14,000	8/29/95
	42,000	9/5/95
	18,500	9/12/95
	56,000	9/19/95
	70,000	9/28/95
	32,000	10/3/95
	29,000	10/11/95
	70,000	10/17/95
	29,000	10/31/95
	42,000	11/7/95
	67,000	10/8/96
	480,000	10/16/96
	140,000	10/22/96
	33,000	10/29/96
	45,500	11/5/96
	45,000	11/12/96
	99,000	11/19/96
	62,500	11/26/96
	93,000	12/3/96
	26,000	12/10/96

Effluent[1]		
Number of times sampled and range of dates	**Fecal coliform colonies/100 mL**	**Date(s) detected**
4/18/95-2/4/97 (cont.)	128,000	12/17/96
	680,000	12/24/96
	780,000	12/31/96
	130,000	1/7/97
	61,000	1/14/97
	780,000	1/21/97
	705,000	1/28/97
	50,000	2/4/97
40 2/11/97- 10/2/97	38,000	2/11/97
	22,500	2/18/97
	38,000	2/25/97
	38,000	3/4/97
	670,000	3/11/97
	92,300	3/18/97
	53,000	3/25/97
	25,500	4/2/97
	23,500	4/8/97
	21,500	4/15/97
	30,000	4/22/97
	150,000	4/29/97
	140,000	5/13/97
	90,000	5/20/97
	42,000	5/27/97
	142,000	6/3/97
	72,500	6/10/97
	25,600	6/17/97
	37,000	6/24/97
	42	7/1/97
	21,500	7/8/97
	176,000	7/15/97
	24,000	7/22/97
	36,000	8/5/97
	64,000	8/12/97
	16	8/19/97
	12,200	8/26/97
	415,000	7/28/98
	380,000	8/4/98
	715,000	8/11/98
	665,000	8/25/98
	2,300,000	9/1/98
	2,400,000	9/8/98
	9,000,000	9/15/98
	400,000	9/29/98

Effluent[1]		
Number of times sampled and range of dates	**Fecal coliform colonies/100 mL**	**Date(s) detected**
2/11/97- 10/2/97 (cont.)	2,400,000	9/8/98
	9,000,000	9/15/98
	400,000	9/29/98
	3,100,000	10/6/98
	7,050,000	10/14/98
	7,050,000	10/20/98
	1,090,000	10/28/98

[1] Reference: South District Wastewater Treatment Plant, Miami Dade Water and Sewer Department, Miami Dade County, Florida. Monitoring Well Purging Report (Report A) December 26, 2002.

Appendix Table 1-5. Fecal Coliform Concentrations From Monitoring Wells, South District Wastewater Treatment Plant, Dade County

Monitoring well[1]	Depth (feet)	Number of times sampled	Number of fecal coliforms detections (>0)	Fecal coliforms, colonies/100 mL	Date(s) detected
FA 1-U	980-1,090	230	7	2	3/1/87
				4	3/1/90
				14	1/1/91
				94	10/1/92
				400	12/8/92
				150	4/13/93
				6	1/10/95
FA 1-L	1,840-1,927	227	3	4	3/1/90
				4	10/1/92
				58	12/8/92
FA 2-U	980-1,020	208	6	2	12/1/90
				>2000	9/24/92
				520	10/1/92
				340	10/8/92
				110	10/14/92
				4	10/20/92
FA 2-L	1,645-1,672	213	1	10	10/1/92
FA 3-U	981-1,050	183	5	6	3/9/93
				2	3/16/93
				8	4/13/93
				200	11/28/94
				4	6/13/95
FA 3-L	1,771-1,892	184	2	2	5/21/91
				4	4/13/93
Zone 4	1,702-1,840	151	1	100	9/24/92
FA 5-U	1,490-1,588	95	1	2	1/10/95
FA 5-L	1,790-1,890	121	1	6	4/4/95
FA 6-U	1,490-1,584	99	1	4	1/10/95

Monitoring well[1]	Depth (feet)	Number of times sampled	Number of fecal coliforms detections (>0)	Fecal coliforms colonies/100 mL	Date(s) detected
FA 6-L	1,790-1,890	101	0	N/A	N/A
FA 7-U	1,488-1,580	105	1	18	4/4/95
FA 7-L	1,805-1,872	116	0	N/A	N/A
FA 8-U	1,490-1,575	103	1	2	6/13/95
FA 8-L	1,790-1,890	103	0	N/A	N/A
FA 9-U	1,490-1,587	94	1	14	6/13/95
FA 9-L	1,790-1,880	84	1	2	6/13/95
FA 10-U	1,490-1,592	67	0	N/A	N/A
FA 10-L	1,790-1,890	84	0	N/A	N/A
FA 11-U	1,490-1,588	42	0	N/A	N/A
FA 11-L	1,790-1,890	75	0	N/A	N/A
FA 12-U	1,495-1,597	87	0	N/A	N/A
FA 12-L	1,790-1,890	78	0	N/A	N/A
FA 13-U	1480-1,585	89	0	N/A	N/A
FA 13-L	1,740-1,845	81	0	N/A	N/A
FA 14-U	1,490-1,575	87	0	N/A	N/A
FA 15-U	1,490-1,575	83	0	N/A	N/A
FA 15-L	1,790-1,890	79	0	N/A	N/A
FA 16-U	1,490-1,590	89	0	N/A	N/A
FA 16-L	1,790-1,890	80	0	N/A	N/A
BZ 1	1,005-1,037	190	33	40	11/1/90
				16	12/1/90
				13	1/1/91
				50	3/31/92
				14	4/7/92
				116	4/21/92
				362	5/5/92
				62	5/12/92

Monitoring well[1]	Depth (feet)	Number of times sampled	Number of fecal coliforms detections (>0)	Fecal coliforms colonies/100 mL	Date(s) detected
				86	5/20/92
				22	5/26/92
				106	6/9/92
				14	6/16/92
				212	6/23/92
				2	6/30/92
BZ 1(cont.)	1,005-1,037	(cont)	33	2	7/7/92
				6	7/14/92
				4	7/21/92
				2	8/11/92
				2	8/18/92
				30	9/24/92
				400	10/1/92
				130	10/8/92
				500	10/14/92
				88	10/20/92
				208	10/27/92
				54	11/3/92
				530	11/10/92
				20	11/17/92
				200	1/8/93
				24	3/9/93
				400	3/16/93
				192	3/30/93
				496	4/6/93
BZ-2	1,577-1,664	97	0	N/A	N/A

[1] Reference: South District Wastewater Treatment Plant, Miami Dade Water and Sewer Department, Miami Dade County, Florida. Monitoring Well Purging Report (Report A) December 26, 2002.

Appendix 1-6. Class 1 Facilities in South Florida.

FACILITY AND WELL DATA

Facility	Injection Wells			
	Active	Inactive	Under Construction	Proposed
Albert Whitted	2	-	-	-
MDW&S South District Regional	13	4	4	-
Seacoast Utilities	1	-	-	-
McKay Creek	2	-	-	-
South Cross Bayou	3	-	-	-
St. Petersburg NE	3	-	-	-
St. Petersburg NW	2	-	-	-
St. Petersburg SW	3	-	-	-
Broward County - North District Regional	4	-	2	2
G.T. Lohmeyer	5	-	-	-
Margate	2	-	-	-
MDW&S North District Regional	-	2	2	-
Palm Beach County - Southern Regional	2	-	-	-
Plantation Regional (Broward County)	2	-	-	-
South Beaches	1	-	-	-
Sunrise	3	-	-	-
Sykes Creek (Merritt Island)	2	-	-	-
Belle Glade	1	-	-	-
Brentwood WWTP (Atlantic Utilities)	1	-	-	-
Coral Springs Improvement District	2	-	-	-
East Port (Charlotte)	2	-	-	-
East-Central Regional	6	-	-	1
Encon	1	-	-	-
Ft. Myers Beach	1	-	-	-
Ft. Pierce Utility Authority	1	-	-	-
Gasparilla Island	1	-	-	-
Immokalee	-	-	1	-
Manatee County SW - Subregional	1	-	-	-
Melbourne - Grant St.	1	-	-	-
Miramar WWTP	2	-	-	-
North Ft. Myers Utilities	1	-	-	-
North Port (Charlotte)	1	-	-	-
Pahokee	1	-	-	-
Palm Bay (GDU-Port Malabar)	1	-	-	-
Palm Beach County System #9	1	-	-	-
Pembroke Pines	2	-	-	-
Port St. Lucie Westport	-	-	-	1
Punta Gorda	-	-	2	-
Rockledge	1	-	-	-
Royal Palm Beach	1	-	-	-
South Collier County	1	-	-	-
South Port St. Lucie	1	-	-	-
Stuart	2	-	-	-
West Melbourne	1	-	-	-
West Port (Charlotte)	1	-	-	-

Appendix Table 1-6. Class I Facility Treatment and Flow Data

Facility	Permitted Treatment Capacity (MGD)	Permitted Injection Rate (Well Capacity) (MGD)		Total Well Capacity (MGD)	Injectate Characteristics/ Current Treatement in Place	Emergency Disposal Practice
Albert Whitted	12.40	24.00	Per 2 wells	48.00	Activated sludge process with chlorinated effluent to a reclaimed water spray irrigation system and back-up/wet weather disposal to wells.	Injection wells used for backup/wet-weather disposal.
MDW&S South District Regional		15.9 17.5 16.9 17.8 10.18 15.00 16.7 15.0 16.9 17.5 17.2 17.2 16.1 14.9 14.9 14.9 14.9 (208)	IW-1 IW-2 IW-3 IW-4 IW-5 IW-6 IW-7 IW-8 IW-9 IW-10 IW-11 IW-12 IW-13 IW-14 IW-15 IW-16 IW-17 (Total 1-13)	269.48	Secondary treated domestic wastewater effluent	
Seacoast Utilities		15.00	IW-1	15.00	Secondary treated domestic wastewater	
McKay Creek	6.00	6.35	Per 2 wells	12.70	Secondary treated municipal effluent from a Type 1 contact stabilization municipal sewage treatment plant with filtered chlorinated effluent.	
South Cross Bayou	24.50	10.20	Per 3 wells	20.40	Filtered and chlorinated effluent from a conventional activated sludge municipal treatment plant.	
St. Petersburg NE	16.00	27.00	Total 3 wells	27.00	Activated sludge process with chlorinated effluent to a reclaimed water spray irrigation system and back-up/wet weather disposal to wells.	
St. Petersburg NW	20.00	16.00	Per 2 wells	32.00	Activated sludge process with chlorinated effluent to a reclaimed water spray irrigation system and back-up/wet weather disposal to wells.	
St. Petersburg SW	20.00	9.00	Per 3 wells	27.00	Municipal effluent from a Type 1 activated sludge plant with chlorinated effluent to a reclaimed water spray irrigation system and backup disposal to wells.	
Broward County - North District Regional		15 18.7	Per 4 wells IW-5, 6	97.40	Secondary treated domestic wastewater (effluent)	
G.T. Lohmeyer		18.3 18.7 (67)	IW-1-4 IW-5 (Pump Cap.)	91.90	Secondary treatment	
Margate		8.15 15	IW- 1 IW- 2	23.15	Secondary treated domestic wastewater.	Excess wastewater from East WWTP discharged to Margate canal and excess from West WWTP discharged to One Mile canal.
MDW&S North District Regional		18.70	Per 2 wells	33.40	Secondary treated domestic wastewater (effluent)	
Palm Beach County - Southern Regional		15.00	Per 2 wells	30.00	Secondary treated domestic wastewater	3.5 MGD and 4.5 MGD diverted to Palm Beach County System #3 and #9, respectively. Remaining effluent will be disinfected and allowed to overflow to on-site stormwater detention ponds.
Plantation Regional (Broward County)		15 (24.00)	IW-1,2 (Pump cap.)	30.00	Secondary treated domestic wastewater.	
South Beaches		9.00		9.00	Secondary treated domestic wastewater (effluent)	Existing percolation ponds for overflow (15 million gallons storage). Ability to store water at Indian River and South Patrick treatment plants. A last option is discharging to the Indian River.
Sunrise		18.70	IW-1,2,3	56.10	Secondary treated domestic wastewater, may include membrane softening concentrate during planned	
Sykes Creek (Merritt Island)		8.2 8.1	Well 1 Well 2	16.30	Minimum to secondary treatment levels - no chlorination is necessary.	

Yellow highlighted denotes facilities reviewed for risk assessment

Source: Florida status reports -January 2002; Florida Discharge Monitoring Reports-February 2002

Appendix 1-6 - Class I Facility Treatment and Flow Data

Facility	Permitted Treatment Capacity (MGD)	Permitted Injection Rate (Well Capacity) (MGD)		Total Well Capacity (MGD)	Injectate Characteristics/ Current Treatement in Place	Emergency Disposal Practice
Belle Glade		10.20		10.20	Secondary treated domestic wastewater	
Brentwood WWTP (Atlantic Utilities)		3.41		3.41	Secondary treated domestic wastewater	
Coral Springs Improvement District		4.87 15	IW-1 IW-2	19.87	Secondary treated domestic wastewater	IW-1 serves as the backup well when IW-2 is out of service.
East Port (Charlotte)		2.04 7.56	IW-1 IW-2	9.60	Domestic wastewater	To existing onsite spray irrigation system, onsite storage pond and discharge to surface waters.
East-Central Regional		15.3 15 17.3 20.7 18.7 18.7 (98.00)	IW-1 IW-2 IW-3 IW-4 IW-5 IW-6 (Pump Cap.)	105.70	Secondary treated domestic wastewater	Treated, chlorinated effluent to remaining injection wells and an equalization basin with capacity of 8 million gallons, and then to the Atlantic Ocean via drainage system.
Encon		18.00		18.00	Secondary treated domestic wastewater	Chlorinated effluent to stabilization pond, overflow to recharge lake to a tributary of the Loxahatchee River.
Ft. Myers Beach		7.92		7.92	Domestic wastewater	Sent to reclaimed water system, then percolation ponds with injection used for excess effluent disposal.
Ft. Pierce Utility Authority	10.00	14.92		14.92	Conventional activated sludge secondary domestic wastewater plant with influent screening, grit removal, aeration, secondary clarification, chlorination, and dechlorination.	Surface water discharge to Indian River Lagoon
Gasparilla Island		0.81		0.81	Back up disposal of secondary treated domestic wastewater following filtration and disinfection	Well is a backup discharge mechanism to golf course irrigation. There is 2.13 million gallon onsite holding pond.
Immokalee		2.50		2.50	Backup disp of secondary treated domestic effluent	If well is out of service, flow directed to existing effluent holding ponds.
Manatee County SW - Subregional		15.00		15.00	Treated municipal effluent receiving min. of secondary treatment	
Melbourne - Grant St.		14.92		14.92	Pretreated domestic wastewaters	Suface water discharge directed to Crane Creek and on to the Indian River.
Miramar WWTP		18.50	per 2 wells	37.00	Secondary treated municipal effluent	Directed to plant's stormwater collection system, which flows into a drainage canal.
North Ft. Myers Utilities		4.00		4.00	Secondary treated domestic wastewater; following filtration and disinfection	Well is back up for plant. Additional disposal is to onsite storage pond.
North Port (Charlotte)		4.75		4.75	Secondary treated domestic wastewater	
Pahokee		4.00		4.00	Secondary treated domestic wastewater	Directed to onsite polishing ponds.
Palm Bay (GDU-Port Malabar)		10.00		10.00	Secondary treated domestic wastewater	Directed to South Harris Ditch to Turkey Creek and on to the Indian River.
Palm Beach County System #9		12.70		12.70	Concentrate rejected waters from low-pressure membrane softening process generated from the water treatment facility	
Pembroke Pines		7.69 15.27 (7.69)	IW-1 IW-2 (Pump Cap.)	22.96		IW-1 is used for emergency disposal. If flows exceed permitted amount, part of flow will be diverted to existing percolation pond.
Port St. Lucie Westport						
Punta Gorda		12.00	For 1 well	12.00	Secondary treated domestic effluent	Directed to existing effluent disposal ponds.
Rockledge		4.50		4.50	Secondary treated effluent	Directed to Indian River via 2934 feet of effluent pipeline.
Royal Palm Beach		6.34		6.34	Secondary treated domestic wastewater	onsite percolation ponds.
South Collier County		18.00		18.00	Secondary treated domestic effluent	
South Port St. Lucie		3.41		3.41	Secondary treated domestic wastewater	
Stuart		3.5 10.00	IW-1 IW-2	13.50	Secondary treated domestic wastewater; IW-1 (emergency back-up well) will inject potable water once a month	Additional emergency flow is diverted to outfall system into the St. Lucie River.
West Melbourne	2.50	4.80		4.80	Secondary treated effluent	Emergency ponds store 3.2 million gallons, additional flow diverted to Crane Creek drainage canal.
West Port (Charlotte)		4.75		4.75	Secondary treated domestic wastewater	Flow directed to 3 existing percolation ponds (capacity 6.3 MGD) and to onsite spray irrigation system.

Source: Florida status reports -January 2002; Florida Discharge Monitoring Reports-February 2002

Appendix 1-6. Injectate Characteristics for Class I Injection Wells.

Facility	Injectate Characteristics/ Current Treatement in Place	Emergency Disposal Practice
Albert Whitted	Activated sludge process with chlorinated effluent to a reclaimed water spray irrigation system and back-up/wet weather disposal to wells.	Injection wells used for backup/wet-weather disposal.
MDW&S South District Regional	Secondary treated domestic wastewater effluent	
Seacoast Utilities	Secondarily treated domestic wastewater	
McKay Creek	Secondary treated municipal effluent from a Type 1 contact stabilization municipal sewage treatment plant with filtered chlorinated effluent.	
South Cross Bayou	Filtered and chlorinated effluent from a conventional activated sludge municipal treatment plant.	
St. Petersburg NE	Activated sludge process with chlorinated effluent to a reclaimed water spray irrigation system and back-up/wet weather disposal to wells.	
St. Petersburg NW	Activated sludge process with chlorinated effluent to a reclaimed water spray irrigation system and back-up/wet weather disposal to wells.	
St. Petersburg SW	Municipal effluent from a Type 1 activated sludge plant with chlorinated effluent to a reclaimed water spray irrigation system and backup disposal to wells.	
Broward County - North District Regional	Secondarily treated domestic wastewater	
G.T. Lohmeyer	Secondary treatment	
Margate	Secondary treated domestic wastewater.	Excess wastewater from East WWTP discharged to Margate canal and excess from West WWTP discharged to One Mile canal.
MDW&S North District Regional	Secondary treated domestic wastewater (effluent)	
Palm Beach County - Southern Regional	Secondarily treated domestic wastewater	3.5 MGD and 4.5 MGD diverted to Palm Beach County System #3 and #9, respectively. Remaining effluent will be disinfected and allowed to overflow to on-site stormwater detention ponds.
Plantation Regional (Broward County)	Secondary treated domestic wastewater.	
South Beaches	Secondarily treated domestic wastewater (effluent)	Existing percolation ponds for overflow (15 million gallons storage). Ability to store water at Indian River and South Patrick treatment plants. A last option is discharging to the Indian River.
Sunrise	Secondary treated domestic wastewater, may include membrane softening concentrate during planned outages of Injection Well CW-1.	
Sykes Creek (Merritt Island)	Minimum to secondary treatment levels - no chlorination is necessary.	
Belle Glade	Secondary treated domestic wastewater	
Brentwood WWTP (Atlantic Utilities)	Secondary treated domestic wastewater	
Coral Springs Improvement District	Secondary treated domestic wastewater	IW-1 serves as the backup well when IW-2 is out of service.
East Port (Charlotte)	Domestic wastewater	To existing onsite spray irrigation system, onsite storage pond and discharge to surface waters.

Appendix 1-6. Injectate Characteristics for Class I Injection Wells.

Facility	Injectate Characteristics/ Current Treatement in Place	Emergency Disposal Practice
East-Central Regional	Secondary treated domestic wastewater	Treated, chlorinated effluent to remaining injection wells and an equalization basin with capacity of 8 million gallons, and then to the Atlantic Ocean via drainage system.
Encon	Secondary treated domestic wastewater	Chlorinated effluent to stabilization pond, overflow to recharge lake to a tributary of the Loxahatchee River.
Ft. Myers Beach	Domestic wastewater	Sent to reclaimed water system, then percolation ponds with injection used for excess effluent disposal.
Ft. Pierce Utility Authority	Conventional activated sludge secondary domestic wastewater plant with influent screening, grit removal, aeration, secondary clarification, chlorination, and dechlorination.	Surface water discharge to Indian River Lagoon
Gasparilla Island	Back up disposal of secondarily treated domestic wastewater following filtration and disinfection	Well is a backup discharge mechanism to golf course irrigation. There is 2.13 million gallon onsite holding pond.
Immokalee	Backup disp of secondarily treated domestic effluent	If well is out of service, flow directed to existing effluent holding ponds.
Manatee County SW - Subregional	Treated municipal effluent receiving min. of secondary treatment	
Melbourne - Grant St.	Pretreated domestic wastewaters	Suface water discharge directed to Crane Creek and on to the Indian River.
Miramar WWTP	Secondary treated municipal effluent	Directed to plant's stormwater collection system, which flows into a drainage canal.
North Ft. Myers Utilities	Secondary treated domestic wastewater; following filtration and disinfection	Well is back up for plant. Additional disposal is to onsite storage pond.
North Port (Charlotte)	Secondary treated domestic wastewater	
Pahokee	Secondary treated domestic wastewater	Directed to onsite polishing ponds.
Palm Bay (GDU-Port Malabar)	Secondary treated domestic wastewater	Directed to South Harris Ditch to Turkey Creek and on to the Indian River.
Palm Beach County System #9	Concentrate rejected waters from low-pressure membrane softening process generated from the water treatment facility	
Pembroke Pines		IW-1 is used for emergency disposal. If flows exceed permitted amount, part of flow will be diverted to existing percolation pond.
Port St. Lucie Westport		
Punta Gorda	Secondary treated domestic effluent	Directed to existing effluent disposal ponds.
Rockledge	Secondary treated effluent	Directed to Indian River via 2934 feet of effluent pipeline.
Royal Palm Beach	Secondarily treated domestic wastewater	Surficial aquifer recharge through rapid rate infiltration in onsite percolation ponds.
South Collier County	Secondarily treated domestic effluent	
South Port St. Lucie	Secondary treated domestic wastewater	
Stuart	Secondary treated domestic wastewater; IW-1 (emergency back-up well) will inject potable water once a month	Additional emergency flow is diverted to outfall system into the St. Lucie River.
West Melbourne	Secondary treated effluent	Emergency ponds store 3.2 million gallons, additional flow diverted to Crane Creek drainage canal.
West Port (Charlotte)	Secondary treated domestic wastewater	Flow directed to 3 existing percolation ponds (capacity 6.3 MGD) and to onsite spray irrigation system.

Appendix Table 2-1 Dade County

	Authors	Title	Date	Source	Hydrogeologic Unit	Depth Below Land Surface (ft)	Thickness (ft)	Hydraulic Conductivity K (ft/day)	Transmissivity T (ft^2/day)	Effective Porosity	Notes
1	Reese, R.S.	Hydrogeology and the Distribution and the Origin of Salinity in the Floridan Aquifer System, **Southeastern** Florida	1994	USGS WRI 94-4010	**Surficial Aquifer**						**Salinity (mg/L)**
					Surficial Aquifer (Biscayne)	0 - 270	175 - 270				
					Intermediate Confining Unit	270 - 990	600 - 1050				
					Floridan Aquifer System	990 - ???	2500 - 3000				1840
					Upper Floridan Aquifer	990 - 1530	500 - 600		10000 - 60000 2700, 10000, 31000		
					Suwannee		120 - 300				
					Ocala		150 - 200			0.2 - 0.45	
					Avon Park	1100 - 1199	100 - 270			0.5, 0.3 - 0.4	3800
					Middle Confining Unit	1530 - 2450		0.003 - 3 (V) $1.3x10^{-4}$ - 0.24 (n=8)		0.336 - 0.464, Avg 0.402 (n=6)	11800 - 35600
					Lower Floridan Aquifer	2450 - ???	1000 - 1200 lower 300 - 600 (Boulder Zone)		$3.2x10^6$ - $24.6x10^6$ (boulder zone)		
2	Reese, R.S. and Cunningham, K.J.	Hydrogeology of the Gray Limestone Aquifer in **Southern Florida**	1-Jan-00	USGS WRI 99-4213	**Surficial Aquifer**	0 - 270					
					Biscayne Aquifer		0 - 120				
					Upper Semiconfining to Confining Unit		0 - 130				
					Gray Limestone Aquifer		0 - 130	148 - 2900	5800 - 160000		
					Lower Semiconfining Unit		0 - 20				
					Sand Aquifers		0 - 100				
3	Maliva, R.G. and Walker, C.W.	Hydrogeology of Deep-Well Disposal of Liquid Wastes in Southwestern Florida, USA	Aug-98	Hydrogeology Journal	**Surficial Aquifer**	0 - 50	50				
					Intermediate Confining Unit	50 - 200	150				
					Floridan Aquifer System						
					Upper Floridan Aquifer	200 - 450	250				
					Middle Confining Unit	450 - 700	250				
					Lower Floridan Aquifer	700 - 1000	300				Temp (°F)
					Bolder Zone, 750 - 950 bls						74, 61.3, 60.5, 60.6
4	Meyer, F.W.	Hydrogeology, Ground-Water Movement, and Subsurface Storage in the Floridan Aquifer System in Southern Florida	1989	USGS Prof. Paper 1403-G	**Surficial Aquifer**	0 - 275	275				Temp (°F) 98-73
					Intermediate Confining Unit	275 - 850	575				73 - 67
					Floridan Aquifer System						
					Upper Floridan Aquifer	850 - 1550	700		10000 - 60000	0.30	67 - 63
					Middle Confining Unit	1550 - 1990	440				63 - 58
					Lower Floridan Aquifer	1990 - 3900	1910				58 - 60
					Bolder Zone	2900 - 3500	600		$3.2x10^6$ - $24x10^6$	0.30	53, 74, 61.3, 60.5, 60.6
5	Reese, R.S. and Memburg, S.J.	Hydrogeology and the Distribution of Salinity in the Floridan Aquifer System, Palm Beach County, Florida	1999	USGS WRI 99-4061	**Surficial Aquifer**		150 - 830				
					Intermediate Confining Unit		600 - 700				
					Floridan Aquifer System						
					Upper Floridan Aquifer		500 - 700		10000 - 100000 32000 - 132000, 24000, 64800, 49100		
					Middle Confining Unit		0 - 900	V: $2x10^{-2}$ - $2x10^{-5}$ (n=9), $1.3x10^{-4}$ (n=8)			
					Lower Floridan Aquifer		1800				
					Bolder Zone	2500 - 3100	300 - 650		$3.2x10^6$ - $24x10^6$		
6	Duerr, A.D.	Types of Secondary Porosity of Carbonate Rocks in Injection and Test Wells in Southern Peninsular Florida	1995	USGS WRI 94-4013	**Surficial Aquifer**	0 - 105	105, 180				
					Intermediate Confining Unit	105 - 910	805, 590				
					Floridan Aquifer System	910 - 3190	2280				
					Upper Floridan Aquifer						
					Middle Confining Unit						
					Lower Floridan Aquifer						
					Bolder Zone						
					USDW	1750, 1415, 2170, 1640					
7	Englehardt, J.D., et al.	Comparative Assessment of Human and Ecological Impacts from Municipal Wastewater Disposal Methods in Southeast Florida	23-Apr-01	University of Miami	**Intermediate Confining Unit**						
					Hawthorn Formation (Upper Confining Unit)		300	V: 0.089 - 8.8e-5			Geraghty & Miller 1975
					Floridan Aquifer System						
					Upper Floridan Aquifer				10000 - 60000		Meyer 1989
					Middle Confining Unit			V: 0.80, H: 0.73		0.32	
					Lower Floridan Aquifer						

21 - 26°C

Appendix Table 2-2 Pinellas County

	Authors	Title	Date	Source	Hydrogeologic Unit	Depth Below Land Surface (ft)	Thickness (ft)	Hydraulic Conductivity K (ft/day)	Transmissivity T (ft^2/day)	Effective Porosity	Notes		
1	Hutchinson, C.B. and Trommer J.T.	Model Analysis of Hydraulic Properties of a Leaky Aquifer System, **Sarasota County**, Florida	December, 1992	USGS WSP 2340	**Surficial Aquifer**								
					Surficial Aquifer	0-50	50	10					
					Intermediate Confining Unit								
					Semiconfining Unit	50-60	10	0.005					
					Tamiami-upper Hawthorn Aquifer	60-100	40	125	5100				
					Semiconfining Unit	100-240	140	0.1					
					Lower Hawthorn-upper Tampa Aquifer	240-410	170	59	9600				
					Lower Tampa Semiconfining Unit	410-500	90	10					
					Floridan Aquifer System								
					Upper Floridan Aquifer								
					Suwannee Permeable Zone	500-750	250	70	13000				
					Lower Suwannee-Ocala Semiconfining Unit	750-1100	350	0.1					
					Avon Park Upper Permeable Zone	1100-1400	300	25					
					Avon Park Highly Permeable Dolomite	1400-2075	675	100					
					Middle Confining Unit	2075-2400					**Unused**		
2	Knochenmus, L.A. and Robinson, J.L.	Description of Anisotropy and Heterogeneity and Their Effect on Ground Water Flow and Areas of Contribution to Public Supply Wells in a Karst Carbonate Aquifer System	January, 1996	USGS WSP 2475	**Intermediate Confining Unit**								
					Tampa		99			0.41			
							137			0.21			
							203			0.30			
					Upper Floridan Aquifer		1125		29400 - 130000 mean 57000				
					Suwannee		232			0.3			
							273			0.42			
							397			0.39			
					Ocala		419			0.38			
							436			0.40	as cited in CH2M Hill (1990a)		
							452			0.27	as cited in CH2M Hill (1990a)		
							490			0.34	as cited in CH2M Hill (1990a)		
							500			0.49			
							540			0.25			
							616			0.48	as cited in Hickey (1977)		
					Avon Park		892			0.20	as cited in Hickey (1977)		
							957			0.07	as cited in Hickey (1977)		
							1028			0.02	as cited in Hickey (1977)		
							1155			0.11	as cited in CH2M Hill (1990a)		
							1174			0.24	as cited in CH2M Hill (1990a)		

Appendix Table 2-2 Pinellas County

	Authors	Title	Date	Source	Hydrogeologic Unit	Depth Below Land Surface (ft)	Thickness (ft)	Hydraulic Conductivity K (ft/day)	Transmissivity T (ft^2/day)	Effective Porosity	Notes		
3	Hickey, J.J.	Hydrogeology and Results of Injection Tests at Waste Injection Test Sites in **Pinellas County**, Florida	1982	USGS WSP 2183							Salinity (mg/L)	Density	
					Surficial Aquifer	0 - 85	20 - 85	V: 0.36 - 13, Avg 2.6 H: 13 - 33		0.292, 0.322	25, 32, 259, 404, 3040, 210, 120,		
					Intermediate Confining Unit								
					Hawthorn Formation (Upper Confining Unit)	85 - 200	115	0.00011 - 0.021 (n=12), Avg 0.0083 (Clay)6.6×10^{-5} - 2.8×10^{-3} (n=16), Avg 7.6×10^{-4}			52		
					Floridan Aquifer								
					Upper Floridan Aquifer								
					Tampa (Zone A)	200 - 350	112 - 245, Avg 180		21000 - 43200, 29000, 29000, 25000 - 30000	0.26, 0.31	450, 6540, 508, 969, 32900, 1530, 5530	1.003, 1.0, 1.0, 1.018, 0.998, 1.002	
					Suwannee (Semiconfining Zone)		90	0.0013		0.22 - 0.36, Avg 0.3	19000		
					Suwannee (Zone B)	350 - 500	50 - 75, Avg 60			0.19, 0.24, 0.30	19000 - 20000, 35500, 38400, 30400, 26700	1.025, 1.024, 1.024, 1.020, 1.018	
					Ocala (Semiconfining Zone)	500 - 750	250	0.1 - 1		0.22 - 0.36, Avg 0.3			
					Avon Park (Zone C)		300 - 386, Avg 330		900000 - 1200000 Avg 1000000	0.15, 0.14, 0.21	20001 - 21000, 39200, 38200, 38100, 41800, 37900, 37800	1.025, 1.025, 1.025, 1.026	Injection Zone
					Avon Park (Semiconfining Zone)	750 - 1250	100			0.26, 0.33	21000 - 25000		
					Avon Park (Zone D)		22 - 121, Avg 70			0.22, 0.39	39800, 43500, 38600, 42500	1.026, 1.026, 1.026, 1.026	
					Middle Confining Unit						25000 - 31000		
					Lake City	1250 - 2000	750	6.6×10^{-7}, 4.0×10^{-5}, 3.0×10^{-3}, 3.3×10^{-3}, 5.2×10^{-2}, 1.1, 2.0					
					Lower Floridan Aquifer Oldsmar	2000 - 3260	1260		2000 - 3000				
4	Knochenmus, L.A. and Bowman, G	Transmissivity and Water Quality of Water-Producing Zones in the Intermediate Aquifer System, **Sarasota County**, Florida	1-Jan-98	USGS WRI 98-4091	**Intermediate Confining Unit**		400						
					Confining Unit Producing Zone 1 Confining Unit		80		1100 - 8000, 8000		Barr, 1996, Duerr and others, 1988		
					Tamiami-upper Hawthorn Aquifer (Producing Zone 2)		200		100 - 26,000 (n=18), 200 - 5000, 740 - 2400, 500 - 3500		Barr, 1996, Duerr and others, 1988, Wolansky, 1983		
					Confining Unit								
					Lower Hawthorn-upper Tampa Aquifer (Producing Zone3)		150		1300 - 6200 (n=4), 5600 - 15400, 500 - 10000		Barr, 1996, Wolansky, 1983		
					Lower Tampa Semiconfining Unit (Confining Unit)								

Appendix Table 2-2 Pinellas County

	Authors	Title	Date	Source	Hydrogeologic Unit	Depth Below Land Surface (ft)	Thickness (ft)	Hydraulic Conductivity K (ft/day)	Transmissivity T (ft^2/day)	Effective Porosity	Notes	
5	Hutchinson, C.B.	Assessment of Hydrogeologic Conditions with Emphasis on Water Quality and Wastewater Injection, S**outhwest Sarasota and West Charlotte Counties**, Florida	1991	USGS OFR 90-709							**Salinity (mg/L)**	
					Surficial Aquifer	0-50	50		1340 - 1850, 1100		<500	
					Intermediate Confining Unit							
					Semiconfining Unit	50-60	10		7800, 5500, 1260, 3320, 3800, 1525, 1608, 2970			
					Tamiami-upper Hawthorn Aquifer	60-100	40		200, 650, 300, 400, 650, 550, 800, 5000		660, 1200, 1300, 390, 1900, 660, 1240, 228, 326, 3000, 21000, 500, 546, 458, 650, 756, 791, 590, 484, 423, 406, 1630, 3240, 980, 330, 4700, 2040, 5500	
					Semiconfining Unit	100-240	140					
					Lower Hawthorn-upper Tampa Aquifer	240-410	170		8200, 5600, 10000		2170, 1700, 420, 1400, 2200, 2300, 1900, 2750, 2930, 2058, 890, 3000, 21000, 2900, 2750, 2930, 2058, 1600, 1200, 250, 1700, 2170, 2600, 2200, 1910, 1700, 4000, 1200	
					Lower Tampa Semiconfining Unit	410-500	90		17900, 15400			
					Floridan Aquifer System							
					Upper Floridan Aquifer							
					Suwannee Permeable Zone	500-750	250		13000, 8900, 72000, 13000		3210, 2900, 2500, 1600, 2300, 3780, 3520, 21000, 3000, 1500, 10900, 4490, 1500, 15000, 14000	
					Lower Suwannee-Ocala Semiconfining Unit	750-1100	350	H: 65 V: 0.01, 0.01, 0.09, 0.57, 2.27, 0.28, 0.09, 0.06, 0.06, 0.03, 0.02, 0.01, 0.1, 0.01, 0.005 H: 0.023, 0.03, 0.11, 0.25, 0.57, 0.14, 0.09, 0.06, 0.06, 0.02, 0.01, 0.19, 0.52, 0.23, 0.1, 0.08, 0.007,		0.37, 0.4, 0.45, 0.37, 0.37, 0.37, 0.31, 0.24, 0.22, 0.22, 0.22, 0.27, 0.09,0.43, 0.03, 0.28, 0.24, 0.28, 0.22, 0.25	CH2M Hill, Inc., 1986, Geraghty and Miller, Inc. 1985, Hutchinson and Trommer (in press), Law Environmental, Inc. 1989, Post, Buckley, Schuh and Jernigan, Inc. 1982	
					Avon Park Upper Permeable Zone	1100-1400	300	H: 100	64000, 48000, 80000, 67000, 24000, 150000, 140000, 300000,		35000, 1800, 37500, 32100, 34090, 2100, 3000, 25200, 32800, 35000, 33300, 35200,	Injection Zone
					Avon Park Highly Permeable Dolomite	1400-2075	675					
					Middle Confining Unit	2075-2400						
					Lower Floridan Aquifer	2400-?			370000			
6	Broska, J.C. and Barnette, H.L.	Hydrogeology and Analysis of Aquifer Charactersitics in **West-Central Pinellas County**, Florida	January 1, 1999	USGS OFR 99-185	**Surficial Aquifer**		50	V: 0.36 - 13 H: 13 - 32			**Salinity (mg/L)**	
					Intermediate Confining Unit		50 - 140 Avg 90	V: $1.3x10^{-4}$ - $6.9x10^{-3}$				
					Floridan Aquifer System							
					Upper Floridan Aquifer						1000 - 10000 @ 100 - 400 ft bls	
					Tampa (Zone A)		115 - 250 Avg 180	H: 18	10000 - 40000			
					Suwannee (Semiconfining Zone B)	350 - 547	125 - 170 Avg 150	V: $1.3x10^{-3}$ - 2 H: 0.1				
					Suwannee (Zone B)		50 - 75 Avg 62		5000			

Appendix Table 2-2 Pinellas County

	Authors	Title	Date	Source	Hydrogeologic Unit	Depth Below Land Surface (ft)	Thickness (ft)	Hydraulic Conductivity K (ft/day)	Transmissivity T (ft^2/day)	Effective Porosity	Notes
7	Barr, G.L.	Hydrogeology of the Surficial and Intermediate Aquifer Systems in Sarasota and Adjacent Counties, Florida	1-Jan-96	USGS WRI 96-4063	**Surficial Aquifer** **Intermediate Confining Unit** Confining Unit Permeable Zone 1 Confining Unit Permeable Zone 2 Confining Unit Permeable Zone3 Confining Unit		3 - 60 221 - 745 5 - 150 80 0 - 29 20 - 190 15 - 240 0 - 300 10 - 240	H: $2x10^{-3}$ - 159 (n=15) V: $2.4x10^{-3}$ H: 17 - 56 V: $2.4x10^{-3}$ V: 0.1 - 10 V: 0.1 - 10	150 - 1800 1100 - 8000 200 - 5000 5600 - 15400		300 mg/L, 23.6 - 32.4°C 316 - 10300 mg/L, 24.4 - 27.5°C 1120 - 7700 mg/L, 25.6 - 28.4°C
8	Knochenmus, L.A. and Thompson, T.H.	Hydrogeology and Simulated Development of the Brackish Ground-Water Resources in Pinellas County, Florida	Jan 1 1991	USGS WRI 91-4026	**Surficial Aquifer**		0 - 132	H: 13 - 33 V: 0.36 - 13		0.292, 0.322	25
					Intermediate Confining Unit		0 - 115				52
					Floridan Aquifer						
					Upper Floridan Aquifer Tampa (Zone A) Suwannee (Semiconfining Zone) Suwannee (Zone B) Ocala (Semiconfining Zone) Avon Park (Zone C) Avon Park (Semiconfining Zone) Avon Park (Zone D)		100 - 250 50 - 75	H: 0.4 V: $1.3x10^{-3}$ - 2, 0.4, 0.1 - 1	$2.2x10^{4}$ - $3.5x10^{4}$	0.26, 0.32, 0.41	450 19000 20000 21000
					Middle Confining Unit						25000
					Lower Floridan Aquifer						31000
9	Duerr, A.D. and Enos, G.M.	Hydrogeology of the Intermediate Aquifer System and Upper Floridan Aquifer, Hardee and De Soto Counties, Florida	1991	USGS WRI 90-4104	**Surficial Aquifer**		25 - 100		1100		
					Intermediate Confining Unit		200 - 500		400 - 7000		24.5, 26 deg C
					Floridan Aquifer System	200 - 600					
					Upper Floridan Aquifer				100000 - 850000		28, 31 deg C
					Middle Confining Unit						
					Lower Floridan Aquifer						
10	Hickey, J.J.	Hydrogeology, Estimated Impact, and Regional Well Monitoring of Effects of Subsurface Wastewater Injection, Tampa Bay Area, Florida	1981	USGS WRI 80-118	**Surficial Aquifer**		85	V: 0.36 - 13 Avg 2.6			
					Intermediate Confining Unit			V: 0.01 - 0.0001, Avg 0.008			
					Floridan Aquifer System						
					Upper Floridan Aquifer			V: 0.0013 - 2.5, Avg 0.6, 1	$7.5x10^{4}$, $5.1x10^{4}$, $3.0x10^{4}$, $2.9x10^{4}$		
					Middle Confining Unit			V: 6e-7, 3.3e-3, 1.1, 4e-5, 3e-3, 5.2e-2, 2			
					Lower Floridan Aquifer				$1x10^{1}$, $1.2x10^{6}$, $0.9x10^{6}$		
11	Duerr, A.D.	Types of Secondary Porosity of Carbonate Rocks in Injection and Test Wells in Southern Peninsular Florida	1995	USGS WRI 94-4013	**Surficial Aquifer**		15				
					Intermediate Confining Unit		50				
					Floridan Aquifer System						
					Upper Floridan Aquifer		555				80 - 84 deg F
					Middle Confining Unit						
					Lower Floridan Aquifer		705				
					Bolder Zone						
					USDW	100 - 140, 1013 - 1053, 1300, 540 - 590					

Appendix Table 2-3 Brevard County

	Authors	Title	Date	Source	Hydrogeologic Unit	Depth Below Land Surface (ft)	Thickness (ft)	Hydraulic Conductivity K (ft/day)	Transmissivity T (ft^2/day)	Effective Porosity	Notes
1	Duerr, A.D.	Types of Secondary Porosity of Carbonate Rocks in Injection and Test Wells in Southern Peninsular Florida	1995	USGS WRI 94-4013	**Surficial Aquifer** **Intermediate Confining Unit** **Floridan Aquifer System** ***Upper Floridan Aquifer*** ***Middle Confining Unit*** ***Lower Floridan Aquifer*** Bolder Zone USDW	0 - 95, 0 - 120 95 - 360, 120 - 250 360 - 2977, 250 - 2906 2070 - 2770 1190, 1634, 1670, 1700, 1190	95, 120 360, 130 2617, 2656 700				
2	Schiner, G.R.	Geohydrology of Osceola County, Florida	1993	USGS WRI 92-4076	**Surficial Aquifer** **Intermediate Confining Unit** **Floridan Aquifer System** ***Upper Floridan Aquifer*** ***Middle Confining Unit*** ***Lower Floridan Aquifer*** Bolder Zone USDW	2500 - 3000	30 - 270 40 - 300 2400 - 2900 300 - 350 450 - 700 1400 - 2100 500 - 800	H: 20 - 100 V: $1.5x10^{-2}$ - $7.8x 10^{-7}$ - $5x10^{-3}$	400, 2000, 1000 65000 - 250000, 100000 - 200000, 10000 - 35000, 10000 - 50000 5600, 8900, 43000 6500 - 60000, 60000 $1x10^7$		
3	Duncan, J.G., Evans, W.L., Taylor, K.L.	Geologic Framework of the Lower Floridan Aquifer System, Brevard County, Florida	1994	Florida Geological Survey Bulletin No. 64	**Surficial Aquifer** **Intermediate Confining Unit** **Floridan Aquifer System** ***Upper Floridan Aquifer*** ***Middle Confining Unit*** ***Lower Floridan Aquifer*** Boulder Zone		90 - 150 2300 - 2900 110 - 250 1500 - 2000 85 - 190	H: 0.78 V: 0.07, 0.06, V: <0.28 H: 0.28, 0.00283	50e3 - 100e3, 100e3 - 250e3	0.20 0.10 - 0.30	
4	Tibbals, C.H.	Hydrogeology of the Floridan Aquifer System in East-Central Florida	1990	USGS Prof. Paper 1403-E	**Floridan Aquifer System** ***Upper Floridan Aquifer*** ***Middle Confining Unit*** ***Lower Floridan Aquifer***		2300 - 2900 500 - 900		120000, 10000 - 400000, 74000, 210000, 510000, 30000 - 130000		
5	Adams, Karin	A Three Dimensional Finite Difference Flow Model of the Surficial Aquifer In Martin County, Florida	March-92	SFWMD Technical Publication 92-02	**Surficial Aquifer**			45, 53, 33, 58, 50, 100, 78, 78, 32, 40, 42, 52, 49, 50, 71, 33, 59, 62, 97, 33, 33, 49, 35, 36, 121, 53, 53, 48, 37, 38, 56, 33, 44, 91, 58, 57, 126, 76	3610, 3743, 1337, 3476, 6016. 12032, 4679, 2674, 4011, 4011, 6016, 7353, 2005, 11364, 4011, 5348, 4011, 8021, 3342, 3342, 8021, 2674, 2005, 26738, 1872, 2406, 3342, 3342, 2005, 3342, 3342, 4412, 10027, 5214, 5749, 22727, 10695		
6	Lukasiewiez, J. and Adams, K.S.	Hydrogeologic Data and Information Collected from the Surficial and Floridan Aquifer Systems, Upper East Coast Planning Area	March-96	SWFMD Technical Publication 96-02 (WRE #337)	**Surficial Aquifer** **Intermediate Confining Unit** **Floridan Aquifer System** ***Upper Floridan Aquifer*** ***Middle Confining Unit*** ***Lower Floridan Aquifer***		2700 - 3400 500 300		394 34		

Appendix 3

Weighted Mean Values

A primary literature review was conducted and all published values of hydrogeologic parameters characterizing the hydrologic units in each county studied were tabulated in this appendix and summarized in the following tables. The weighted means ($\overline{Z}$) of the data were calculated to determine representative values to be used in the risk assessment. The weighted mean method essentially reduces the effect of extreme data outliers (very high or very low values). The following equation was used to develop the weighted means for all hydrogeologic data (Mendenhall and Beaver, 1994).

$$\overline{Z} = \left[\frac{0.5Z_1 + 0.75Z_2 + \sum_{i=3}^{m-2} Z_i + 0.75Z_{m-1} + 0.5Z_m}{m - 1.5} \right] \qquad \text{(Eqn. 1)}$$

Where: Z = Hydrogeologic datum
m = Total number of values
i = Chronological interger

The above equation is not valid for data sets containing less than five values, therefore, the following equation was used.

$$\overline{Z} = \left[\frac{0.5Z_1 + \sum_{i=2}^{m-1} Z_i + 0.5Z_m}{m - 1.0} \right] \qquad \text{(Eqn. 2)}$$

For data sets with two values, an average was calculated.

In the Intermediate, Upper Floridan and Lower Floridan aquifers, hydrogeological data for the discretized geologic units within the aquifer are presented in the following tables. Representative hydrogeologic data for the entire aquifer were then determined by weighting the data in proportion to the thickness of the individual geologic units within each hydrogeologic unit.

Weighted means for the data sets are color coded in blue, while representative values used in the analysis are color coded in red.

Where there were insufficient data available, the following assumptions were made:

- Anisotropy ratios and porosity values were assumed to be consistent for equivalent aquifer units in each county, in the absence of site-specific data.

- Parameters for the horizontal and vertical hydraulic conductivities and porosities for each geologic layer in Brevard County were assumed to be consistent with data provided for the same equivalent depositional unit in Dade County.

- A horizontal hydraulic gradient of 0.001 were assumed for the injection zone and the overlying units in Dade and Brevard Counties.

- In Pinellas County, a horizontal hydraulic gradient of 0.05 in the injection zone was assumed. This accounted for the effects of pressure head due to injection. In the overlying units, a horizontal hydraulic gradient of 0.001 was used.

- A porosity of 0.5 was assumed for the Boulder Zone for horizontal ground water flow. Conduit flows occur in the Boulder Zone due to cavernous pores or large fractures in the rock (Meyer, 1984, Maliva and Walker, 1998); therefore a larger porosity is required to address this issue.

Appendix Table 3-1 Dade County

County: Dade																
Hydrogeologic Units	Surficial	Intermediate					Upper Floridan							Middle Confining Unit	Lower Floridan	
		Semiconfining Unit	Tamiami-upper Hawthorn Aquifer	Semiconfining Unit	Lower Hawthorn-upper Tampa Aquifer	Lower Tampa Semiconfining Unit	Tampa (Zone A)	Suwannee (Semiconfining Zone)	Suwannee (Zone B)	Ocala (Semiconfining Zone)	Avon Park (Zone C)	Avon Park (Semiconfining Zone)	Avon Park (Zone D)			Bolder Zone
Horizontal Conductivities	148									0.45				8.37E-07		2,307
	2900									9				0.003		2,461
	1524									55				0.03		18,923
										121.8				3		6538
										42				4.3		
														6		
														28		
														4.7		
Vertical Conductivities				8.8E-05										2.00E-05		
				0.1										1.30E-04		
														1.30E-04		
														2.00E-02		
														0.24		
														0.04		
Transmissivity	5800			10000						2,700						3.20E+06
	160000			100000						10,000						3.20E+06
	82900			55000						10,000						1.30E+07
										10,000						2.46E+07
										10,000						2.46E+07
										24,000						1.36E+07
										31,000						
										32,000						
										49,100						
										60,000						
										60,000						
										64,800						
										100,000						
										132,000						
										40,060						
Thickness	50			150			250	120		150		100		250	100	300
	105			450			500	300		200		270		440	200	300
	150			575			700	210		175		185		450	300	600
	175			590			700							900	200	600
	180			600			1100							488		650
	200			600			650									500
	270			700						1220					700	
	270			805											1300	
	275			1050											1800	
	830			614											1910	
	232														1468	
Porosity							0.3			0.2		0.3		0.336		0.3
										0.45		0.4		0.464		
										0.33		0.5		0.45		
												0.4		0.43		
										0.32						
Temperature	73.04			66.92			66.92							62.96	57.92	53
	98.06			73.04			62.96							57.92	60.08	60.5
	85.55			69.98			64.94							60.44	59.00	60.6
																61.3
																74
																61.56
Salinity								1840				3800		11800		
														35600		
USDW (below land surface)									1415							
									1640							
									1750							
									2170							
									1728							

Appendix Table 3-2 Pinellas County

County: Pinellas																
Hydrogeologic Units	Surficial	Intermediate					Upper Floridan							Middle Confining Unit	Lower Floridan	
		Semiconfining Unit	Tamiami-upper Hawthorn Aquifer	Semiconfining Unit	Lower Hawthorn-upper Tampa Aquifer	Lower Tampa Semiconfining Unit	Tampa (Zone A)	Suwannee (Semiconfining Zone)	Suwannee (Zone B)	Ocala (Semiconfining Zone)	Avon Park (Zone C)	Avon Park (Semiconfining Zone)	Avon Park (Zone D)			
Horizontal Conductivities	2.00E-03	0.005	17	0.1	18	10		0.0013	70	0.007	25		100			
	10		56		59			0.1	65	0.01		100				
	13		125		38.5			0.4	1	0.02						
	13		64					0.20	67.5	0.023						
	13			23						0.03						
	32			1.00E-04						0.06						
	33			1.1E-04						0.06						
	33			0.021						0.08						
	159			3.91						0.09						
	29									0.1						
										0.1						
										0.1						
										0.11						
										0.14						
										0.19						
										0.23						
										0.25						
										0.52						
										0.57						
										1						
										0.16						
										19.78						
										0.003						
										0.3						
										0.45						
										121.8						
										22						
Vertical Conductivities	0.36	2.40E-03		1.00E-04				1.30E-03		0.005				6.60E-07		
	0.36	2.40E-03		1.10E-04				1.30E-03		0.01				4.00E-05		
	0.36	2.40E-03		1.30E-04				0.1		0.01				3.00E-03		
	13			6.90E-03				0.4		0.01				3.30E-03		
	13			1.00E-02				1		0.01				5.20E-02		
	13			0.021				2		0.02				1.1		
	6.68			0.1				2		0.03				2		
				0.1				0.73		0.06				0.34		
				10						0.09						
				10						0.09						
				1.50						0.1						
				1.16						0.28						
										0.57						
										1						
										2.27						
										2.5						
										0.29						
Transmissivity	150	1260	100		200	15400	500		5,000		900,000			1.00E+01	2,000	3.20E+06
	1100	1525	200		200	17900	5600		8,900		1,200,000			9.00E+05	3,000	1.30E+07
	1100	1608	300		500	16650	5600		13,000		1,050,000			1.20E+06	370,000	2.46E+07
	1340	2970	400		740		10000		13,000					750003	94,500	1.35.E+07
	1800	3320	500		1300		10000		13,000							
	1850	3800	550		2400		15400		72,000							
	1248	5500	650		3500		15400		17,983							
		7800	650		5000		21000									
		3307	800		5000		22000									
			1100		5600		25000									
			1100		6200		29000									
			5100		8200		29000									
			8000		9600		30000									
			8000		10000		35000									
			1708		4071		40000									
				7259			43200									
				400			20,928									
				7000						539,999						
Transmissivity				5415						29,000						
										30,000						
										51,000						
										75,000						
										100,000						
										850,000						
										198,727						

Appendix Table 3-2 Pinellas County

County: Pinellas															
Hydrogeologic Units	Surficial	Intermediate					Upper Floridan							Middle Confining Unit	Lower Floridan
		Semiconfining Unit	Tamiami-upper Hawthorn Aquifer	Semiconfining Unit	Lower Hawthorn-upper Tampa Aquifer	Lower Tampa Semiconfining Unit	Tampa (Zone A)	Suwannee (Semiconfining Zone)	Suwannee (Zone B)	Ocala (Semiconfining Zone)	Avon Park (Zone C)	Avon Park (Semiconfining Zone)	Avon Park (Zone D)		
Thickness	3	5	40	29	20	15	100	10	50	250	300	100	22	750	1260
	20	10	40	115	170	90	112	90	50	350	386		121		
	25	10	80	140	170	90	115	125	50	419	343		72		
	50	150	80	140	190	240	150	170	75	436		515			
	50	32.5	60	113	200	103	245	240	75	452		892			
	60				157		250	127	75	490		957			
	60			465			250		250	500		1028			
	85			115			300		81	540		1155			
	100			99			1100	208		616		1174			
	132			115			256	232		453		970			
	56			137				273							
				150				397							
				200				269							
				203						1948					
				500											
				219											
Porosity	0.292			0.21			0.26	0.22	0.19	0.22	0.15	0.26	0.22	0.034	
	0.292			0.3			0.26	0.36	0.24	0.03	0.14	0.33	0.39	0.45	
	0.322			0.41			0.31	0.29	0.3	0.09	0.21	0.30	0.31	0.317	
	0.322			0.31			0.31		0.49	0.22	0.16				
	0.31						0.41		0.29	0.22		0.21			
							0.31	0.29		0.22		0.02			
								0.3		0.22		0.07			
								0.42		0.24		0.11			
								0.39		0.24		0.20			
								0.35		0.24		0.24			
										0.25		0.14			
										0.25					
										0.27					
										0.27					
										0.28					
										0.28					
										0.31					
										0.34					
										0.36					
										0.37					
										0.37					
										0.37					
										0.37					
										0.38					
										0.4					
										0.40					
										0.43					
										0.45					
										0.48					
										0.49					
										0.43					
										0.26					
Temperature	74.48			75.92						78.08					
	90.32			81.5						83.12					
				76.1						82.4					
				78.8						87.8					
Salinity	25		228	52	250		450		450	19000	19000		21000	21000	27000
	25		326		316		508		1500		20000		39800	25000	31000
	32		330		420		969		1500		37800		43500	23000	29000
	120		390		890		1120		1600		37900		38600		
	210		406		1200		1530		2300		38100		42500		
	259		423		1200		5530		2500		38200		38300		
	300		458		1400		6540		2900		39200				
	404		484		1600		7700		3000		41800				
	500		500		1700		32900		3210		34892				
	3040		546		1700		5136		3520			35480			
	383		590		1700				3780			1800			
Salinity			650		1900				4490			18000			
			660		1910				10900			20000			
			660		2058				15000			20000			
			756		2058				19,000			21000			
			791		2170				21000			21000			
			980		2,170				26700			25200			
			1200		2170				30400			30000			
			1240		2200				35500			32100			
			1300		2200				38400			32800			
			1630		2300				10755			32800			
			1900		2600							33300			
			2040		2750							34090			
			693		1583							23481			

Appendix Table 3-2 Pinellas County

County: Pinellas																
Hydrogeologic Units	Surficial	Intermediate					Upper Floridan							Middle Confining Unit	Lower Floridan	
		Semiconfining Unit	Tamiami-upper Hawthorn Aquifer	Semiconfining Unit	Lower Hawthorn-upper Tampa Aquifer	Lower Tampa Semiconfining Unit	Tampa (Zone A)	Suwannee (Semiconfining Zone)	Suwannee (Zone B)	Ocala (Semiconfining Zone)	Avon Park (Zone C)	Avon Park (Semiconfining Zone)	Avon Park (Zone D)			
Salinity			3000		2750							35,000				
			3240		2900							35000				
			4700		2930							35200				
			5500		2930							37500				
			21000		3000							29741				
			1681		4000											
					10300											
					21000											
					2609											
Density	0.997						0.998		1.025		1.025		1.026			
	0.998						1		1.024		1.025		1.026			
	0.998						1		1.024		1.025		1.026			
	0.997						1.002		1.02		1.025		1.029			
	0.998						1.003		1.018		1.026		1.027			
	0.998						1.018		1.022		1.025					
							1.003									
USDW below land surface									100							
									140							
									540							
									590							
									1013							
									1053							
									1300							
									680							

Appendix Table 3-3 Brevard County

County: Brevard																
Hydrogeologic Units	Surficial	Intermediate					Upper Floridan							Middle Confining Unit	Lower Floridan	
		Semiconfining Unit	Tamiami-upper Hawthorn Aquifer	Semiconfining Unit	Lower Hawthorn-upper Tampa Aquifer	Lower Tampa Semiconfining Unit	Tampa (Zone A)	Suwannee (Semiconfining Zone)	Suwannee (Zone B)	Ocala (Semiconfining Zone)	Avon Park (Zone C)	Avon Park (Semiconfining Zone)	Avon Park (Zone D)			Bolder Zone
Horizontal Conductivities	20													0.8	1.00E-03	500
	32														0.0028	800
	33														0.028	650
	33														8.65E-03	
	33															
	33															
	33															
	35															
	36															
	37															
	38															
	40															
	42															
	44															
	45															
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	59															
	62															
	71															
	76															
	78															
	91															
	97															
	100															
	100															
	121															
	126															
	56															
Vertical Conductivities				7.80E-07										5.10E-05	2.80E-01	
				5.00E-03											0.005	
				1.50E-02											0.06	
				6.25E-03											0.07	
															0.12	
															0.1	
Transmissivity	400									3,289				5,600	6,500	1.00E+07
	394									3,637				8,900	30,000	
	488									4,932				43,000	60,000	
	562									5,416				16,600	130,000	
	564									6,551					52,750	
	575									6,685						
	630									6,912						
	669									7,346						
	715									7,436						
	781									7,594						
	853									8,172						
	868									8,201						
	869									9,084						
	925									9,422						
	980									9,615						
	1000									9,827						

Appendix Table 3-3 Brevard County

County: Brevard																
Hydrogeologic Units	Surficial	Intermediate					Upper Floridan							Middle Confining Unit	Lower Floridan	
		Semiconfining Unit	Tamiami-upper Hawthorn Aquifer	Semiconfining Unit	Lower Hawthorn-upper Tampa Aquifer	Lower Tampa Semiconfining Unit	Tampa (Zone A)	Suwannee (Semiconfining Zone)	Suwannee (Zone B)	Ocala (Semiconfining Zone)	Avon Park (Zone C)	Avon Park (Semiconfining Zone)	Avon Park (Zone D)			Bolder Zone
Transmissivity	1042									9,953						
	1123									10,000						
	1150									10,000						
	1255									10,000						
	1315									10,050						
	1337									10,243						
	1372									10,896						
	1455									10,919						
	1482									11,064						
	1497									11,115						
	1532									11,564						
	1645									11,691						
	1724									12,271						
	1736									12,560						
	1765															
	1799									12,568						
	1859									12,609						
	1859									12,693						
	1872									13,284						
	1908									13,381						
	2000									13,815						
	2005									13,945						
	2005									13,998						
	2005									14,025						
	2006									14,266						
	2006									14,307						
	2006									14,316						
	2026									14,588						
	2313									14,707						
	2366									14,887						
	2406									14,890						
	2407									15,001						
	2644									15,988						
	2674									16,567						
	2674									16,857						
	2808									18,692						
	2947									19,926						
	2991									20,140						
	3022									20,510						
	3075									20,816						
	3209									22,023						
	3342									22,073						
	3342									22,462						
	3342									23,616						
	3342									24,485						
	3342									27,876						
	3476									28,077						
	3476									29,072						
	3476									29,204						
	3610									30,626						
	3743									31,754						
	3944									31,866						
	4011									34,537						
	4011									34,774						
	4011									35,000						
	4011									37,279						
	4319									37,365						
	4412									38,045						
	4679									41,313						
	4800									43,285						
	5214									45,672						
	5348									46,079						
	5348									49,023						
	5589									49,832						
	5749									50,000						

Appendix Table 3-3 Brevard County

County: Brevard																
Hydrogeologic Units	Surficial	Intermediate					Upper Floridan							Middle Confining Unit	Lower Floridan	
		Semiconfining Unit	Tamiami-upper Hawthorn Aquifer	Semiconfining Unit	Lower Hawthorn-upper Tampa Aquifer	Lower Tampa Semiconfining Unit	Tampa (Zone A)	Suwannee (Semiconfining Zone)	Suwannee (Zone B)	Ocala (Semiconfining Zone)	Avon Park (Zone C)	Avon Park (Semiconfining Zone)	Avon Park (Zone D)			Bolder Zone
Transmissivity	5829									50,000						
	6016									61,729						
	6016									62,037						
	6017									65,000						
	6217									67,905						
	6642									71,065						
	6819									72,090						
	7086									74,000						
	7313									74,337						
	7353									84,124						
	8021									98,225						
	8021									100,000						
	9292									100,000						
	9546									100,000						
	10027									120,000						
	10424									200,000						
	10589									210,000						
	10695									250,000						
	11364									250,000						
	12032									400,000						
	12777									510,000						
	13330									42,025						
	13343															
	22727															
	26738															
	4100															
Thickness	90			40						300				110		85
	95			130						350				250		190
	120			300						325				450		700
	150			360										500		291
	112			210										700		
														900		
														335		
															1400	
															1500	
															2000	
															2100	
															1750	
												2410				
												2300				
												2400				
												2617				
												2656				
												2900				
												2900				
												2194				
Porosity																0.2
Temperature																
Salinity																
USDW below land surface									1190							
									1190							
									1634							
									1670							
									1700							
									1499							

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Appendix 4

4.1. Total Vertical Time of Travel

Total vertical time of travel is defined as the time required for secondary treated wastewater to migrate upward from the point of injection to the USDW and hypothetical receptor wells. Given the velocity and distance of travel, the time it takes to travel the distance can be determined by dividing the distance by the velocity. To estimate the vertical travel time (t) through each hydrologic unit, the thickness of the unit (b) is divided by the seepage velocity (v_s) (Eqn. 3). Seepage velocity is defined as the velocity representing the average rate at which ground water moves (Fetter, 1994) and is estimated by dividing the Darcy flow (q) by the porosity (n) of the hydrologic unit (Eqn. 4). Porosity represents the ratio between the volume of voids over the total volume of the media (Freeze and Cherry, 1979). In this analysis, published porosity values were used. Darcy flow is defined as fluid flow through porous media (e.g. sand) (Freeze and Cherry; 1979), taking into consideration that ground water flows through porous media, Darcian assumptions must be applied. Darcy flow takes into account vertical hydraulic conductivity (K) and the hydraulic gradient (I) (Eqn. 5). Hydraulic conductivity represents the ability of the media to transmit water (Fetter, 1994). Hydraulic gradient is estimated by dividing the total pressure head (H_T) by the thickness of the hydrologic unit (Eqn. 6).

$$t = \frac{b}{v_s} \qquad \text{(Eqn. 3)}$$

$$v_s = \frac{q}{n} \qquad \text{(Eqn. 4)}$$

$$q = K \times I \qquad \text{(Eqn. 5)}$$

$$I = \frac{H_T}{b} \qquad \text{(Eqn. 6)}$$

4.2. Total Pressure Head

Pressure head can be simply viewed as a driving force for vertical migration of treated wastewater. In this analysis, two driving components of pressure head were considered. Pressure head due to injection (H_I) and pressure head due to buoyancy (H_B). These components are described separately below. The total pressure head acting on the overlying hydrogeologic unit may be expressed as the sum of the buoyancy and the injection components (Eqn. 7):

$$H_T = H_I + H_B \qquad \text{(Eqn. 7)}$$

4.2.1. Pressure Head Due to Injection

Injection-derived pressure is a controlling force that drives the wastewater plume throughout the regional ground water system. As millions of gallons of water are injected into the aquifer, that volume displaces an equivalent volume of native water in the formation. This causes a pressure build-up in the aquifer, which must be dissipated throughout the aquifer unit.

The vertical migration component due to injection-derived over-pressuring was calculated using the following leaky aquifer steady-state pressure drawdown/increase equation (Gupta, 1995).

$$H_I = \frac{Q}{2\pi T} K_O\left(\frac{r}{B}\right) \qquad \text{(Eqn. 8)}$$

$$H_I = \frac{Q}{2\pi T} \ln\left(1.123\frac{B}{r}\right) \text{ for } \frac{r}{B} < 0.05 \qquad \text{(Eqn. 9)}$$

where:

Q	= Injection rate	
K	= Vertical hydraulic conductivity	
b	= Thickness of aquifer	
T	= Transmissivity of the receiving unit = $K \times b$	(Eqn.10)
r	= Distance from injection well	
$K_O\left(\frac{r}{B}\right)$	= Zero-order modified Bessel function of the second kind (Tabulated values)	
B	= Leakage factor = $\sqrt{\frac{T}{K'/b'}}$	(Eqn. 11)
K’	= Vertical hydraulic conductivity of the overlying layer	
b’	= Thickness of the overlying layer	

A distance of one hundred feet from the injection well (r) was chosen in Pinellas County, where pressure due to injection occurs. A distance of one hundred feet was chosen because at this distance away from the injection point, it is assumed that steady upward flow would be occurring. This value will also result in a conservative travel time estimation. The closer one is to the injection point, the greater the effects of pressure due to injection, resulting in a faster travel time. Representative injection rates of 112.5 million gallons per day (mgd) in Dade County, 7 mgd in Pinellas County, and 5 mgd in Brevard County were used (Starr et al., 2001, Florida Department of Environmental Protection, 2001 and Florida Department of Regulation, 1989). In Dade and Brevard Counties the pressure head due to injection is negligible due to injection into the Boulder Zone. The Boulder Zone is highly karstified with cavernous pores and wide fractures, which does not constrain the flow of injected effluent; therefore negligible pressure build up will occur (Singh et al., 1983; Haberfeld, 1991).

4.4.2. Pressure Head Due to Buoyancy

The buoyancy pressure head component, related to variations in fluid temperature and fluid density, also influences upward migration of the injectate. The wastewater injected into the aquifer is relatively fresh in comparison to the native ground water found in the injection zone (Florida Department of Environmental Protection, 1999a). As a result, the less dense injected wastewater rises above the denser, native ground water. In hydraulic terms, the fresh water is more buoyant than the salt water.

Density is also dependent on temperature: warm water is less dense than cold water. The temperature difference between the warm injected wastewater and the comparatively cold, native formation water is yet another driving force for the upward migration of the plume.

Upward pressure heads due to the buoyancy (from salinity and temperature differences) were calculated using the following derived equation (Hwang and Hita, 1987):

$$H_B = \frac{[\rho_n h - \rho_i h]}{\rho_{water}} \qquad \text{(Eqn. 12)}$$

where:

H_B	=	Pressure head due to buoyancy (salinity and temperature gradient)
ρ	=	Density of native (n) and injected (i) fluid
h	=	Height of injected fluid (through each hydrologic unit)

Steady state conditions were assumed in this analysis. Under steady state conditions, no mixing or dispersion occurs and the injectate has a continuous path to the hypothetical water supply well or USDW. Travel times were estimated through each hydrologic unit. Therefore a simplifying assumption, valid for steady state conditions, was that the height of the injected fluid is the thickness of the hydrologic unit.

There is a natural salinity and temperature gradient in the native fluid. The native fluid in the injection zone has salinity comparable to sea water and becomes comparable to fresh water at the surficial aquifer. The injected wastewater has salinity comparable to fresh water therefore the pressure head due to buoyancy (salinity gradient) will decrease as the injectate moves closer to the hypothetical water supply well. The same result will occur with respect to temperature gradient. The temperature of the native fluid in the injection zone is approximately 60 degrees Fahrenheit and can reach up to 80 degrees in the surficial aquifer. The injected wastewater has a temperature of 80 degrees. As the injected wastewater moves closer to the hypothetical water supply well, the pressure head due to buoyancy (temperature gradient) will decrease. The buoyancy calculations were based on the discretization of the density gradient due to temperature and salinity difference.

In this analysis, two scenarios were considered: 1) porous media flow and 2) bulk flow through preferential flow paths. To assess the two scenarios, primary porosities and

hydraulic conductivities and secondary porosities and hydraulic conductivities were used in the above equations, respectively. The results are presented in the following tables for Dade, Pinellas and Brevard Counties.

Appendix Table 4-1 Vertical Travel Time to Receptor Well

(Scenario 1: Porous Media Flow)

Dade

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Biscayne Aquifer	230	100	15	0.31	130	2550	1	0	0.66	0.004	0.058	0.188	1.9	Years
Intermediate Confining Unit	840	230	0.10	0.31	610	61	5	0	4.69	0.008	0.001	0.002	674	Years
Upper Floridan Aquifer	2060	840	0.42	0.32	1220	512	19	0	18.5	0.015	0.006	0.020	168	Years
Middle Confining Unit	2550	2060	0.04	0.43	490	20	23	0	22.5	0.046	0.002	0.004	314	Years
Lower Floridan	2750	2550	0.10	0.40	200	20	15	0	14.6	0.073	0.007	0.018	30	Years
Boulder Zone	3000	2750	65	0.20	250	16250	12	0	12.0	0.048	3.13	15.7	16	Days
												Travel Time	**1,188**	**Years**

Pinellas

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Surficial Aquifer	56	30	7	0.31	26	182	0.1	0	0.10	0.004	0.027	0.087	297	Days
Intermediate Confining Unit	275	56	1.2	0.31	219	263	1.8	0	1.82	0.008	0.010	0.032	18.6	Years
Upper Floridan Aquifer	1250	275	0.3	0.226	975	293	15.6	533	548	0.563	0.169	0.747	3.58	Years
												Travel Time	**23**	**Years**

Brevard

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Surficial Aquifer	130	100	13	0.31	30	390	0	0	0.125	0.004	0.054	0.175	172	Days
Intermediate Confining Unit	340	130	0.10	0.31	210	21	2	0	1.56	0.007	0.001	0.002	240	Years
Upper Floridan Aquifer	665	340	0.20	0.26	325	65	6	0	6.13	0.019	0.004	0.015	61	Years
Middle Confining Unit	1000	665	0.04	0.43	335	13	11	0	11.0	0.033	0.001	0.003	301	Years
Lower Floridan	2460	1000	0.10	0.40	1460	146	45	0	45.4	0.031	0.003	0.008	515	Years
Boulder Zone	2754	2460	65	0.20	294	19110	47	0	46.9	0.160	10.4	51.9	5.67	Days
												Travel Time	**1118**	**Years**

Appendix Table 4-2 Vertical Travel Time to USDW

(Scenario 1: Porous Media Flow)

Dade

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Upper Floridan Aquifer	2060	1500	0.42	0.32	560	512.4	18.5	0	18.5	0.015	0.006	0.020	77	Years
Middle Confining Unit	2550	2060	0.04	0.43	490	19.6	22.5	0	22.5	0.046	0.002	0.004	314	Years
Lower Floridan	2750	2550	0.1	0.4	200	20	14.6	0	14.6	0.073	0.007	0.018	30	Years
Boulder Zone	3000	2750	65	0.2	250	16250	12.0	0	12.0	0.048	3.13	15.7	16	Days
												Travel Time	**421**	**Years**

Pinellas

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Upper Floridan Aquifer	1250	680	0.30	0.226	570	293	16	533	548	0.56	0.17	0.75	2	Years
												Travel Time	**2**	**Years**

Brevard

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Lower Floridan	2470	1500	0.1	0.4	970	146	45	0	45	0.03	0.00	0.01	342	Years
Boulder Zone	2754	2470	65.00	0.20	284	19110	47	0	47	0.160	10.378	51.892	5	Days
												Travel Time	**342**	**Years**

Appendix Table 4-3 Vertical Travel Time to Receptor Well

(Scenario 2: Preferential Flow Paths)

Dade

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Biscayne Aquifer	230	100	15	0.31	130	2550	1	0	0.7	0.004	0.058	0.19	1.9	Years
Intermediate Confining Unit	840	230	2.38	0.10	610	61	5	0	4.7	0.008	0.018	0.18	9.1	Years
Upper Floridan Aquifer	2060	840	2.38	0.10	1220	512	19	0	18.5	0.015	0.036	0.36	9.3	Years
Middle Confining Unit	2550	2060	1.5	0.10	490	20	23	0	22.5	0.046	0.069	0.69	1.9	Years
Lower Floridan	2750	2550	0.1	0.10	200	20	15	0	14.6	0.073	0.007	0.07	7.5	Years
Boulder Zone	3000	2750	65	0.2	250	16250	12	0	12.0	0.048	3.131	15.66	16	Days
												Travel Time	**30**	**Years**

Pinellas

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Surficial Aquifer	56	30	7	0.31	26	182	0.1	0.0	0.1	0.003873	0.027	0.09	297	Days
Intermediate Confining Unit	275	56	1.50	0.10	219	329	1.8	0.0	1.8	0.008313	0.012	0.12	5	Years
Upper Floridan Aquifer	1250	275	2.38	0.10	975	2321	16	122	138	0.141048	0.336	3.36	290	Days
												Travel Time	**6.4**	**Years**

Brevard

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Surficial Aquifer	130	100	13	0.31	30	390	1	0	1	0.03	0.39	1.26	24	Days
Intermediate Confining Unit	340	130	2.38	0.10	210	499.8	6	0	6	0.03	0.071	0.714	294	Days
Upper Floridan Aquifer	665	340	2.38	0.10	325	773.5	11	0	11	0.03	0.080	0.799	1	Years
Middle Confining Unit	1000	665	1.50	0.10	335	502.5	16	0	16	0.05	0.069	0.695	1	Years
Lower Floridan	2460	1000	0.10	0.10	1460	146	45	0	45	0.03	0.003	0.031	129	Years
Boulder Zone	2754	2460	65	0.20	294	19110	47	0	47	0.16	10.38	51.89	6	Days
												Travel Time	**136**	**Years**

Appendix Table 4-4 Vertical Travel Time to USDW

(Scenario 2: Preferential Flow Paths)

Dade

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Upper Floridan Aquifer	2060	1500	2.38	0.10	560	512.4	19	0	19	0.015	0.036	0.36	4	Years
Middle Confining Unit	2550	2060	1.50	0.10	490	19.6	23	0	23	0.046	0.069	0.69	2	Years
Lower Floridan	2750	2550	0.1	0.1	200	20	15	0	15	0.073	0.007	0.07	8	Years
Boulder Zone	3000	2750	65	0.2	250	16250	12	0	12	0.048	3.1	15.7	16	Days
												Travel Time	**14**	**Years**

Pinellas

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Upper Floridan Aquifer	1250	680	2.38	0.1	570	2321	16	122	138	0.14	0.34	3.36	170	Days
												Travel Time	**170**	**Days**

Brevard

Hydrogeologic Units	Injection Fluid Travel (bls)		Vertical Hydraulic Conductivity (K_V)	Porosity (n)	Aquifer Thickness (effective) (b)	Transmissivity (T)	H_B	H_I	H_T	Hydraulic Gradient (I)	Darcy Velocity (q)	Seepage Velocity (v_s)	Travel Time (t)	
	From (feet)	To (feet)	(ft/day)		(feet)	(ft^2/day)	(feet)	(feet)	(feet)		(ft/day)	(ft/day)		
Lower Floridan	2470	1500	0.1	0.1	970	146	45	0	45	0.031	0.003	0.031	86	Years
Boulder Zone	2754	2470	65.00	0.20	284	19110	47	0	47	0.160	10.4	51.9	5	Days
												Travel Time	**86**	**Years**

Appendix 5

Horizontal Travel Distance

The horizontal travel distance (X) is defined in this analysis as the distance of horizontal migration corresponding to the vertical travel time. The horizontal travel distance of the injected wastewater can be estimated by multiplying the seepage velocity (v_s) in the horizontal direction by the vertical travel time (t) estimated earlier (Eqn. 13). Seepage velocity is defined as the velocity representing the average rate at which ground water moves (Fetter, 1994) and is estimated by dividing the Darcy flow (q) by the porosity (n) of the hydrologic unit (Eqn. 14). Porosity represents the ratio between the volumes of voids over the total volume of the media (Freeze and Cherry, 1979). In this analysis, published porosity values were used. Darcy flow is defined as fluid flow through porous media (e.g. sand) (Freeze and Cherry; 1979), taking into consideration that ground water flows through porous media, Darcian assumptions must be applied. Darcy flow takes into account horizontal hydraulic conductivity (K_h) and the horizontal hydraulic gradient (i) (Eqn. 15). Hydraulic conductivity represents the ability of the media to transmit water (Fetter, 1994). Simple substitution of the seepage velocity and Darcy flow equations into Equation 13, will result in Equation 16.

$$X = v_s \times t \qquad \text{(Eqn. 13)}$$

$$v_s = \frac{q}{n} \qquad \text{(Eqn. 14)}$$[1]

$$q = K_h \times i \qquad \text{(Eqn. 15)}$$

$$X = \frac{K_h i}{n} t \qquad \text{(Eqn. 16)}$$

As in the analysis of vertical travel time, two scenarios were considered: 1) porous media flow and 2) bulk flow through preferential flow paths. To assess the two scenarios, vertical travel times respective to the two scenarios were used in estimating the horizontal travel distances.

In Dade and Brevard Counties, a horizontal hydraulic gradient of 0.001 was assumed for all the hydrologic units. In Pinellas County, a horizontal hydraulic gradient of 0.05 was assumed in the injection zone and 0.001 in the overlying units. A greater horizontal hydraulic gradient in the injection zone accounts for the effects of injection pressure due to the injection of millions of gallons of wastewater a day.

Primary porosities were used in this analysis (Eqn. 16) however, in the Boulder Zone a porosity of 0.5 was assumed in Dade and Brevard Counties. A larger porosity in the Boulder Zone takes into account cavernous pores or large fractures found in the Boulder Zone (Meyer, 1984, Maliva and Walker, 1998).

The results of this analysis and a summary of the assumptions made are presented in the following tables for Dade, Pinellas and Brevard Counties (Table 5-1, 5-2 and 5-3).

[1] Same equation used in Appendix 4 (Eqn. 4)

Appendix Table 5-1 Horizontal Migration

(Scenario 1: Porous Media Flow)

Dade

Hydrogeologic Units	Horizontal Hydraulic Conductivity (K_H) (ft/day)	Hydraulic Gradient (i)	Porosity (n)	Time (t) Days	Horizontal Distance (X) ft
Biscayne Aquifer	1,524	0.001	0.31	2	9
Intermediate Confining Unit	90.0	0.001	0.31	246082	71443
Upper Floridan Aquifer	42	0.001	0.32	61270	8042
Middle Confining Unit	5	0.001	0.43	114671	1253
Lower Floridan Aquifer	0.10	0.001	0.40	10984	3
Boulder Zone	6,538	0.001	0.50	16	209
			Total Horizontal Distance		**80,959**

Pinellas

Hydrogeologic Units	Horizontal Hydraulic Conductivity (K_H) (ft/day)	Hydraulic Gradient (i)	Porosity (n)	Time (t) Days	Horizontal Distance (X) ft
Surficial Aquifer	29	0.001	0.31	297	28
Intermediate Confining Unit	4	0.001	0.31	6806	88
Upper Floridan Aquifer	22	0.05	0.226	1306	6355
			Total Horizontal Distance		**6,471**

Brevard

Hydrogeologic Units	Horizontal Hydraulic Conductivity (K_H) (ft/day)	Hydraulic Gradient (i)	Porosity (n)	Time (t) Days	Horizontal Distance (X) ft
Surficial Aquifer	56	0.001	0.31	172	31
Intermediate Confining Unit	20.00	0.001	0.31	87494	5645
Upper Floridan Aquifer	20	0.001	0.26	22406	1724
Middle Confining Unit	1	0.001	0.43	109982	205
Lower Floridan Aquifer	0.1	0.001	0.40	187918	47
Boulder Zone	650	0.001	0.50	6	7
			Total Horizontal Distance		**7,658**

Appendix Table 5-2 Horizontal Migration (Scenario 2: Preferential Flow Paths)

Dade

Hydrogeologic Units	Horizontal Hydraulic Conductivity (K_H) (ft/day)	Hydraulic Gradient (i)	Porosity (n)	Time (t) Days	Horizontal Distance (X) ft
Biscayne Aquifer	1,524	0.001	0.31	2	9
Intermediate Confining Unit	90.0	0.001	0.10	3,335	3,002
Upper Floridan Aquifer	42	0.001	0.10	3,379	1,419
Middle Confining Unit	5	0.001	0.10	711	33
Lower Floridan Aquifer	0.10	0.001	0.10	2,746	3
Boulder Zone	6,538	0.001	0.20	16	522
			Total Horizontal Distance		**4,988**

Pinellas

Hydrogeologic Units	Horizontal Hydraulic Conductivity (K_H) (ft/day)	Hydraulic Gradient (i)	Porosity (n)	Time (t) Days	Horizontal Distance (X) ft
Surficial Aquifer	29	0.001	0.31	297	28
Intermediate Confining Unit	4	0.001	0.1	1,756	70
Upper Floridan Aquifer	22	0.05	0.1	290	3,195
			Total Horizontal Distance		**3,293**

Brevard

Hydrogeologic Units	Horizontal Hydraulic Conductivity (K_H) (ft/day)	Hydraulic Gradient (i)	Porosity (n)	Time (t) Days	Horizontal Distance (X) ft
Surficial Aquifer	56	0.001	0.31	172	31
Intermediate Confining Unit	20.00	0.001	0.10	3	1
Upper Floridan Aquifer	20	0.001	0.10	724	145
Middle Confining Unit	1	0.001	0.10	682	5
Lower Floridan Aquifer	0.1	0.001	0.10	46980	47
Boulder Zone	650	0.001	0.20	6	18
			Total Horizontal Distance		**247**

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Appendix 6

Uncertainty Analysis

Upper and lower boundary travel times to use for the risk assessment were computed based on the results of the uncertainty analyses. For purposes of this risk assessment, times of travel were computed by altering one parameter in each scenario. Vertical hydraulic conductivity of the confining unit was the tested parameter for the porous media scenario (Scenario 1). Porosity was the tested parameter for the preferential flow path scenario (Scenario 2).

Vertical hydraulic conductivity was evaluated by computing travel times based on variation of the mean vertical hydraulic conductivity by up to one order of magnitude above and below the mean value calculated from review of the scientific literature. Porosity was varied from 0.01 to 0.20, a range within typical porosity values found for limestones and dolomites (Freeze and Cherry, 1979). for the travel times computed in the preferential flow path scenario. Graphical representation of the uncertainty analysis time of travel computations can be found in Appendix Figures 6-1, 6-2 and 6-3 for Dade, Brevard and Pinellas Counties.

Upper and lower bounds of times of travel were computed from the results of the uncertainty tests. The first step in developing these bounds is to determine the statistical average time of travel ($t_{average}$) (Eqn. 17).

$$t_{average} = \frac{t_{90} + t_{10}}{2} \qquad \text{(Eqn. 17)}$$

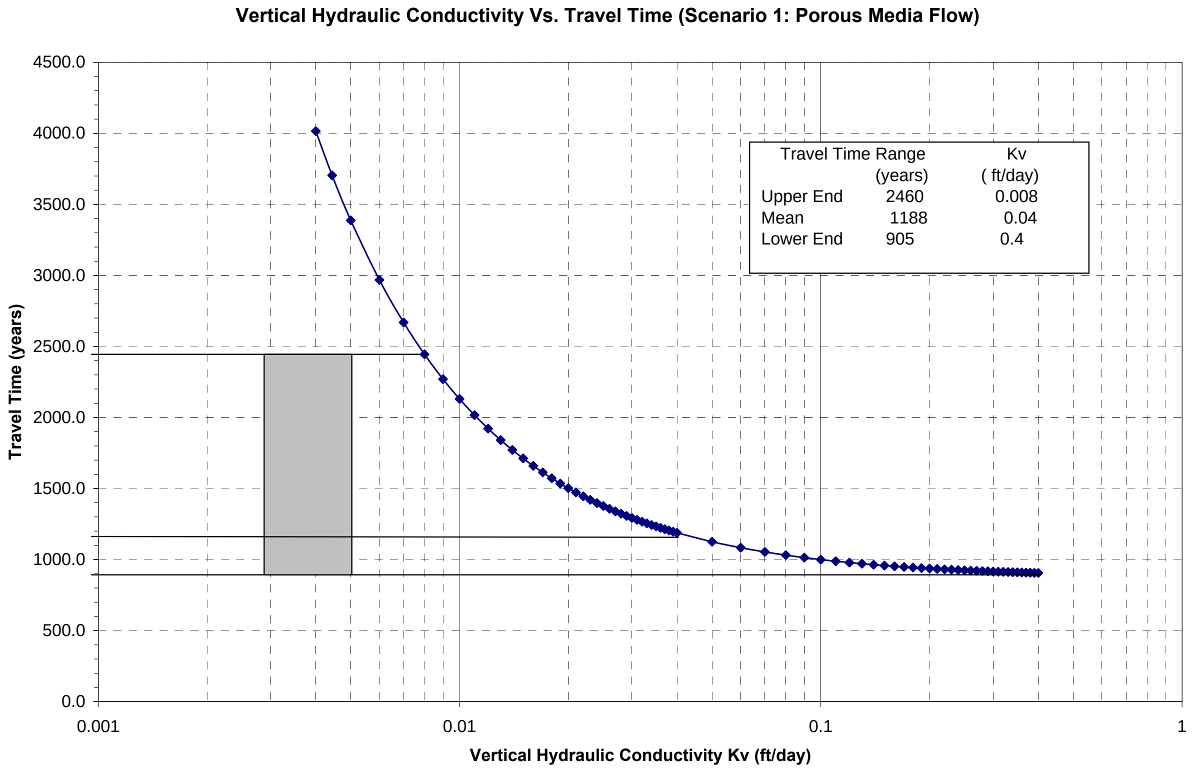
The t_{90} and t_{10} values are the vertical travel times associated with the ninetieth and the tenth percentile, respectively, within the range of the time of travel calculations for each scenario. The resulting $t_{average}$ value thus represents a statistical calculation that incorporates the weight of the travel time variations across two orders of magnitude for the lowest hydraulic conductivity unit, and across the reasonably expected range of porosity typically associated with preferential (i.e.- secondary) flow.

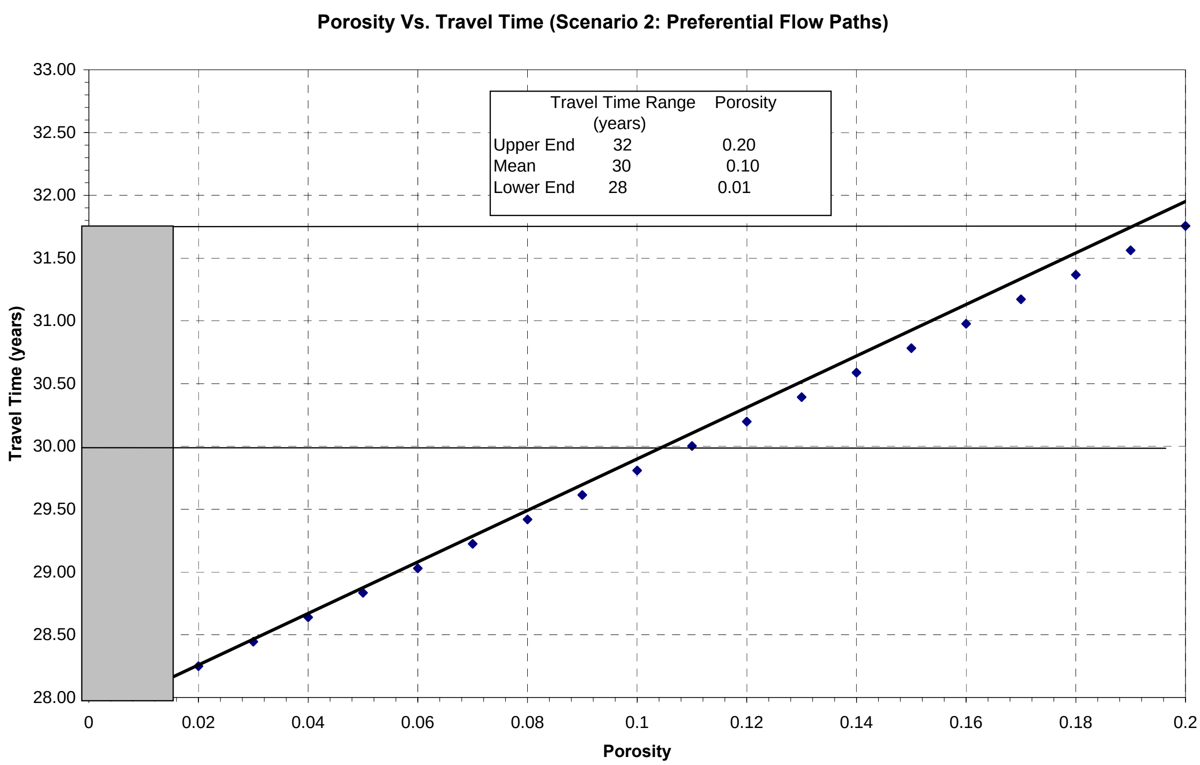
The upper and lower bounds for time of travel are then computed based on the relationship between $t_{average}$, computed in the uncertainty tests, and the vertical travel time (t) estimated earlier. Equations 18 and 19 depict the computations used to generate the upper and lower time of travel bounds, respectively:

$$t_{upper} = t + \left(t_{average} - t\right) \qquad \text{(Eqn. 18)}$$

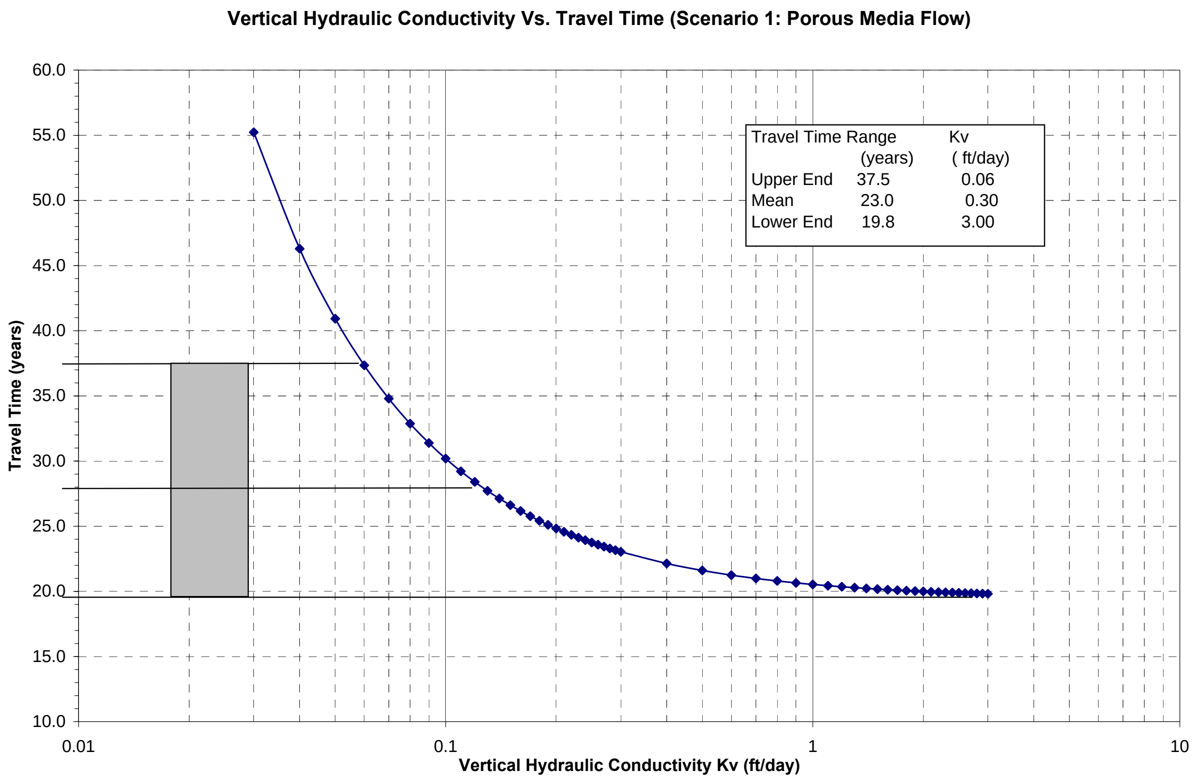
$$t_{lower} = t - \left(t_{average} - t\right) \qquad \text{(Eqn. 19)}$$

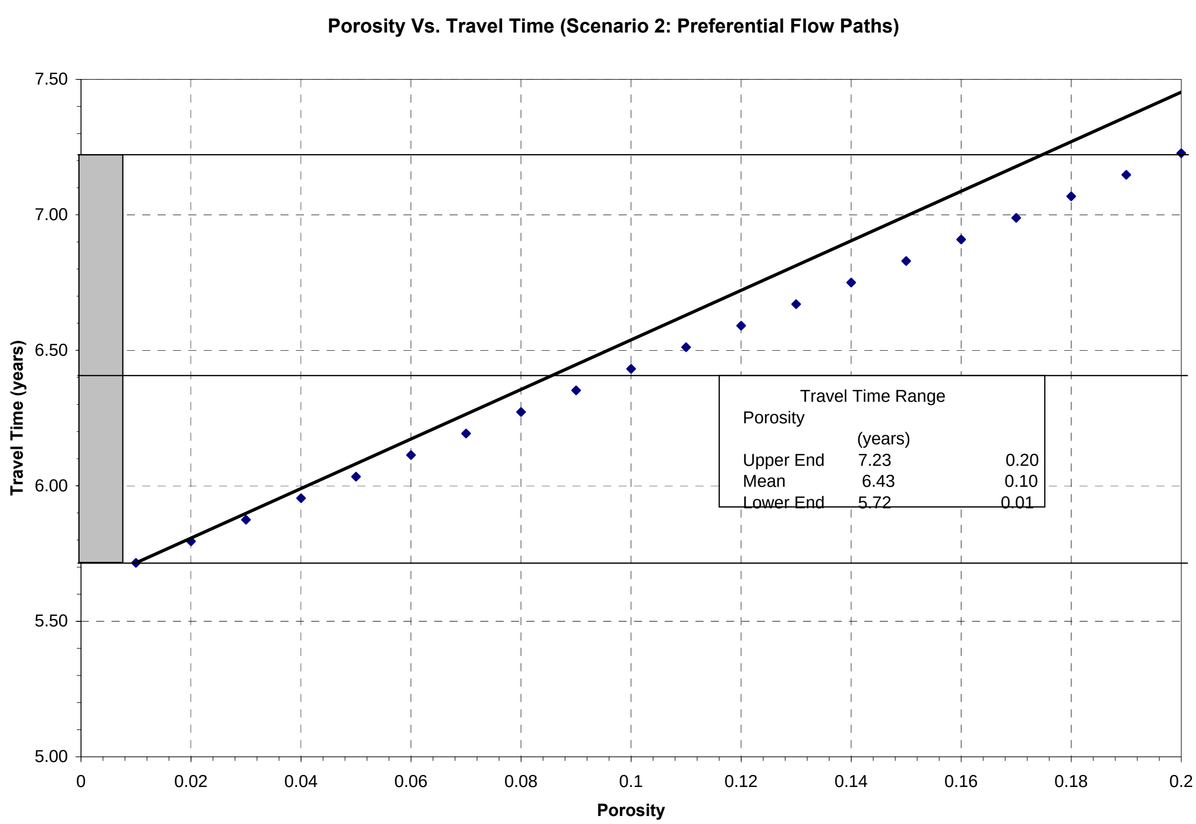
Appendix Figure 6-1
Uncertainty Analysis Results for Dade County



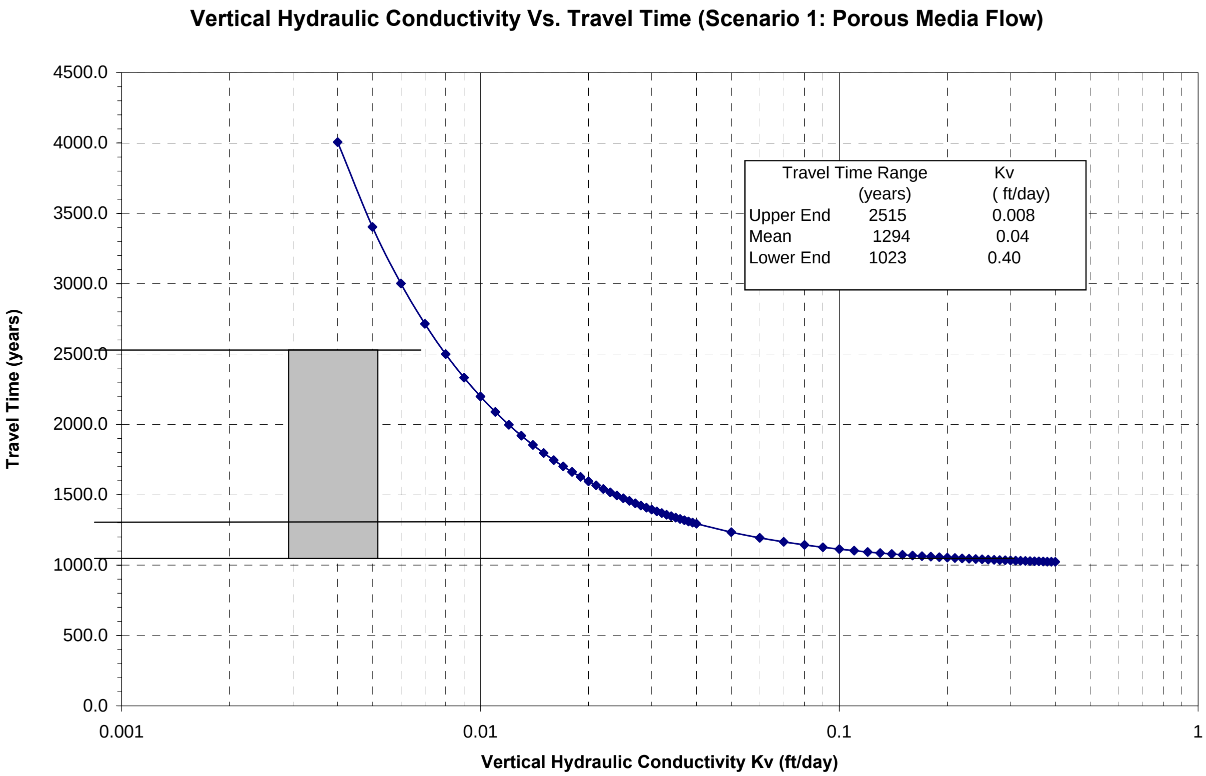


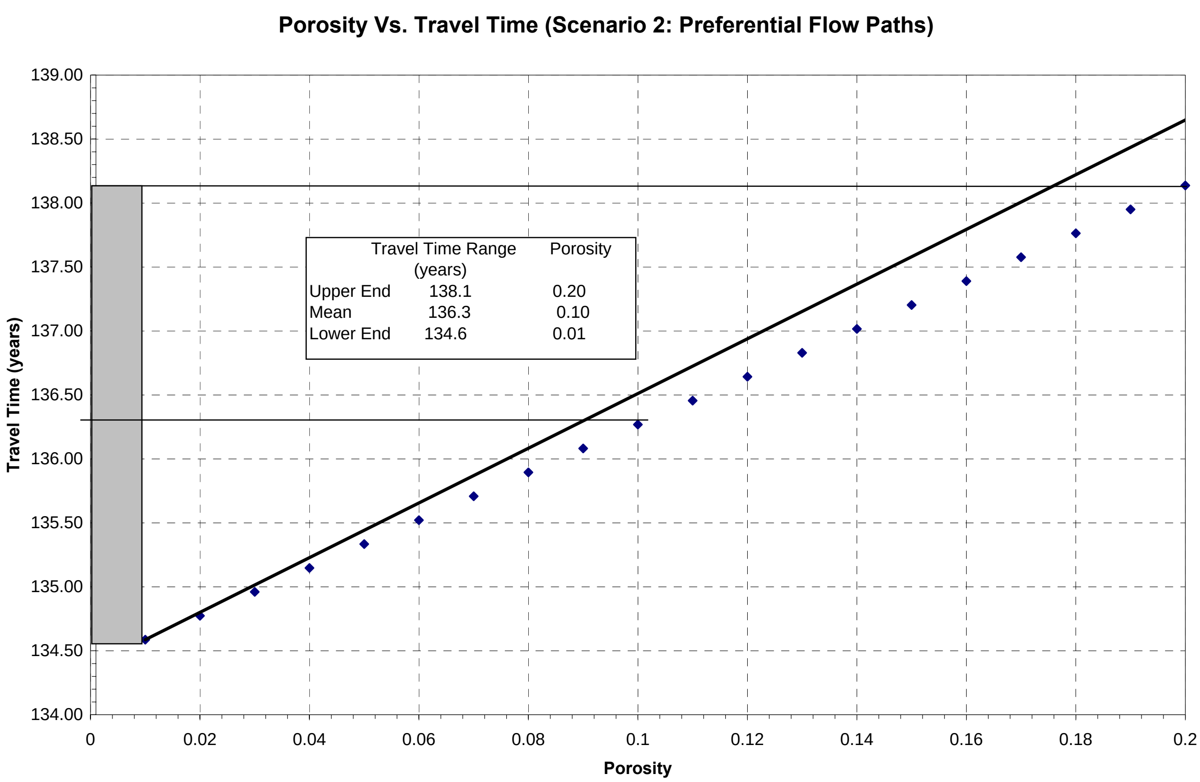
Appendix Figure 6-2
Uncertainty Analysis Results for Pinellas County





Appendix Figure 6-3
Uncertainty Analysis Results for Brevard County





Appendix 7

Fate and Transport

The fate and transport of representative stressors can be estimated by a first order decay model (Eqn. 20), which estimates the final concentration (C) of the representative stressors in correlation to vertical travel times estimated earlier. This first order decay model is appropriate for analysis of the organic constituents, because it takes into account natural attenuation processes such as biodegradation, hydrolysis and sorption (Suthersan, 2002).

$$C = C_O e^{-kt_C} \qquad \text{(Eqn. 20)}$$

where:
- C = Final concentration of stressors
- C_O = Initial concentration of stressors
- k = Decay coefficient of stressors
- t_C = Travel time of stressors

Half-life ($t_{1/2}$) is defined as the time it takes for stressors to reach half of the initial concentration. The decay coefficient (k) can be determined by rearranging Equation 20, substituting the half-life in place of the travel time of stressors (t_C) and equating the ratio of the final versus initial concentrations to 0.5 (Eqn. 21). The decay coefficient (Eqn. 22) is simplified by rearranging Equation 21. Published values for half-life are available and were identified for the selected representative stressors (Howard et al., 1991).

$$\frac{C}{C_O} = 0.5 = e^{-kt_{1/2}} \qquad \text{(Eqn. 21)}$$

$$k = \frac{0.693}{t_{1/2}} \qquad \text{(Eqn. 22)}$$

The travel time of representative stressors (t_C) are determined by multiplying the retardation coefficient (R) by the effluent travel time (t_E) (Eqn. 23). In this analysis, the effluent travel time is equivalent to the vertical travel time estimated earlier.

$$t_C = R \times t_E \qquad \text{(Eqn. 23)}$$

The retardation coefficient takes into account sorption, a natural attenuation process which increases the travel time of stressors. The greater the travel time of stressors, the more time there is for other natural attenuation process to occur, such as biodegradation and hydrolysis to a lesser extent. Biodegradation results in the degradation of organic material and may also mediate transformations in the state of inorganic material resulting in decreasing concentrations over time. Hydrolysis is the process whereby organic and inorganic solutes react with water resulting in degradation and transformation (Suthersan, 2002). Calculation for the retardation coefficient, for dissolved organic constituents, is shown below in Equation 24 (Suthersan, 2002).

$$R = 1 + \frac{\rho_b K_d}{n} \qquad \text{(Eqn. 24)}$$

where: ρ_b = Bulk density = $\rho_s(1-n)$ (Eqn. 25)

ρ_s = soil density

n = porosity

K_d = Distribution coefficient = $K_{oc} f_{oc}$ (Eqn. 26)

K_{oc} = Sorption coefficient

f_{oc} = fraction of total organic carbon

$$R = 1 + \frac{\rho_s(1-n) K_{oc} f_{oc}}{n} \qquad \text{(Eqn. 27)}$$

Sorption coefficients (K_{oc}) were obtained from published values for each representative stressor (Montgomery, 2000). For purposes of risk assessment, conservative values (indicating the least sorption) were selected to calculate the distribution coefficient and therefore the retardation coefficient. Ultimately, this produces conservative estimates of stressor concentrations at the receptors, since the data used relate to the lowest reasonably expected retardation and the shortest travel time. The calculations incorporated a typical value for sediment density of 2.63 g/cm^3 (Freeze and Cherry, 1979). Weighted mean porosity values (Appendix 3), based on unit thickness, were used in the calculations.

Appendix Table 7-1 Representative Stressors Concentrations at Receptor Wells
(Scenario 1: Porous Media Flow)

Dade County															
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Effluent Travel Time to Receptor Wells (days)	Contaminant Travel Time (t_n) (days)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (μg/L)	1800	1.44	0.01	0.014	2.63	0.33	1.76	1.08	1188	1279	433620	466962	0.0004	61.58	0.00
Tetrachloroethylene (PCE) (μg/L)	720	2.25	0.01	0.023	2.63	0.33	1.76	1.12	1188	1331	433620	485716	0.0010	**4.66**	0.00
Chlordane (μg/L)	2772	4.72	0.01	0.047	2.63	0.33	1.76	1.25	1188	1487	433620	542907	0.0003	0.010	0.000
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.33	1.76	1.15	1188	1361	433620	496830	N/A	0.010	0.010
Di(2-ethylhexyl) Phthalate (DEHP) (μg/L)	389	4.48	0.01	0.045	2.63	0.33	1.76	1.24	1188	1472	433620	537350	0.0018	5.00	0.00
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.33	1.76	1.03	1188	1219.1	433620	444965	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	1188	N/A	433620	N/A	N/A	3.82	3.82

Pinellas County															
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Effluent Travel Time to Receptor Wells (days)	Contaminant Travel Time (t_n) (days)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (μg/L)	1800	1.44	0.01	0.014	2.63	0.24	2.00	1.12	23.00	25.8	8395	9402	0.0004	6.70	0.18
Tetrachloroethylene (PCE) (μg/L)	720	2.25	0.01	0.023	2.63	0.24	2.00	1.19	23.00	27.3	8395	9968	0.0010	0.63	0.00
Chlordane (μg/L)	2772	4.72	0.01	0.047	2.63	0.24	2.00	1.39	23.00	32.0	8395	11695	0.0003	0.64	0.03
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.24	2.00	1.23	23.00	28.23	8395	10304	N/A	0.003	0.003
Di(2-ethylhexyl) Phthalate (DEHP) (μg/L)	389	4.48	0.01	0.045	2.63	0.24	2.00	1.37	23.00	31.6	8395	11527	0.0018	1.25	0.00
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.24	2.00	1.04	23.00	23.9	8395	8738	N/A	18.00	18.00
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	23.00	N/A	8395	N/A	N/A	0.28	0.28

Brevard County															
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Effluent Travel Time to Receptor Wells (days)	Contaminant Travel Time (t_n) (days)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (μg/L)	1800	1.44	0.01	0.014	2.63	0.36	1.68	1.07	1118	1193	408070	435545	0.0004	**230**	0.00
Tetrachloroethylene (PCE) (μg/L)	720	2.25	0.01	0.023	2.63	0.36	1.68	1.11	1118	1236	408070	450999	0.0010	1.00	0.00
Chlordane (μg/L)	2772	4.72	0.01	0.047	2.63	0.36	1.68	1.22	1118	1365	408070	498125	0.0003	0.010	0.000
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.36	1.68	1.13	1118	1261	408070	460157	N/A	0.005	0.005
Di(2-ethylhexyl) Phthalate (DEHP) (μg/L)	389	4.48	0.01	0.045	2.63	0.36	1.68	1.21	1118	1352	408070	493546	0.0018	5.00	0.00
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.36	1.68	1.02	1118	1144	408070	417419	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	1118	N/A	408070	N/A	N/A	9.60	9.60

N/A = not applicable

Appendix Table 7-2 Representative Stressors Concentrations at USDW
(Scenario 1: Porous Media Flow)

Dade County													
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to USDW (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day $^{-1}$)	Concentration at Injection Pt. (C_O)	Concentration at USDW (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.33	1.76	1.08	421	453	0.0004	61.58	0.00
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.33	1.76	1.12	421	472	0.0010	**4.66**	0.00
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.33	1.76	1.25	421	527	0.0003	0.010	0.00
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.33	1.76	1.15	421	482	N/A	0.010	0.010
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.33	1.76	1.24	421	522	0.0018	5.00	0.00
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.33	1.76	1.03	421	432.0	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	421	N/A	N/A	3.82	3.82

Pinellas County													
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to USDW (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day $^{-1}$)	Concentration at Injection Pt. (C_O)	Concentration at USDW (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.24	2.00	1.12	2.00	2.2	0.0004	6.70	4.89
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.24	2.00	1.19	2.00	2.4	0.0010	0.63	0.27
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.24	2.00	1.39	2.00	2.8	0.0003	0.64	0.50
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.24	2.00	1.23	2.00	2.45	N/A	0.003	0.003
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.24	2.00	1.37	2.00	2.7	0.0018	1.25	0.21
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.24	2.00	1.04	2.00	2.1	N/A	18.00	18.00
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	2.00	N/A	N/A	0.28	0.28

Brevard County													
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to USDW (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day $^{-1}$)	Concentration at Injection Pt. (C_O)	Concentration at USDW (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.36	1.68	1.07	342	365	0.0004	**230**	0.0
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.36	1.68	1.11	342	378	0.0010	1.00	0.0
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.36	1.68	1.22	342	417	0.0003	0.010	0.0
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.36	1.68	1.13	342	386	N/A	0.005	0.005
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.36	1.68	1.21	342	414	0.0018	5.00	0.0
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.36	1.68	1.02	342	350	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	342	N/A	N/A	9.60	9.60

N/A = not applicable

Appendix Table 7-3 Representative Stressors Concentrations at Receptor Wells

(Scenario 2: Preferential Flow Paths)

Dade County

Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day $^{-1}$)	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.3	1.84	1.09	30	33	0.0004	61.58	0.63
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.3	1.84	1.14	30	34	0.0010	**4.66**	0.00
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.3	1.84	1.29	30	39	0.0003	0.010	0.000
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.3	1.84	1.17	30	35	N/A	0.010	0.010
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.3	1.84	1.27	30	38	0.0018	5.00	0.00
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.3	1.84	1.03	30	30.9	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	30	N/A	N/A	3.82	3.82

Pinellas County

Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day $^{-1}$)	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.25	1.97	1.11	6.40	7.1	0.0004	6.70	2.46
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.25	1.97	1.18	6.40	7.5	0.0010	0.63	0.04
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.25	1.97	1.37	6.40	8.8	0.0003	0.64	0.29
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.25	1.97	1.22	6.40	7.78	N/A	0.003	0.003
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.25	1.97	1.35	6.40	8.7	0.0018	1.25	0.00
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.25	1.97	1.04	6.40	6.6	N/A	18.00	18.00
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	6.40	N/A	N/A	0.28	0.28

Brevard County

Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day $^{-1}$)	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.36	1.68	1.07	136	145	0.0004	**230**	0.00
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.36	1.68	1.11	136	150	0.0010	1.00	0.00
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.36	1.68	1.22	136	166	0.0003	0.010	0.000
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.36	1.68	1.13	136	153	N/A	0.005	0.005
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.36	1.68	1.21	136	164	0.0018	5.00	0.00
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.36	1.68	1.02	136	139	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	136	N/A	N/A	9.60	9.60

N/A = not applicable

Appendix Table 7-4 Representative Stressors Concentrations at USDW
(Scenario 2: Preferential Flow Paths)

Dade County

Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to USDW (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day $^{-1}$)	Concentration at Injection Pt. (C_O)	Concentration at USDW (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.3	1.84	1.09	14	15	0.0004	61.58	7.24
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.3	1.84	1.14	14	16	0.0010	**4.66**	0.02
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.3	1.84	1.29	14	18	0.0003	0.010	0.00
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.3	1.84	1.17	14	16	N/A	0.010	0.010
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.3	1.84	1.27	14	18	0.0018	5.00	0.00
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.3	1.84	1.03	14	14.4	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	14	N/A	N/A	3.82	3.82

Pinellas County

Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to USDW (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day $^{-1}$)	Concentration at Injection Pt. (C_O)	Concentration at USDW (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.25	1.97	1.11	0.47	0.5	0.0004	6.70	6.23
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.25	1.97	1.18	0.47	0.5	0.0010	0.63	0.52
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.25	1.97	1.37	0.47	0.6	0.0003	0.64	0.60
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.25	1.97	1.22	0.47	0.57	N/A	0.003	0.003
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.25	1.97	1.35	0.47	0.6	0.0018	1.25	0.83
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.25	1.97	1.04	0.47	0.5	N/A	18.00	18.00
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.47	N/A	N/A	0.28	0.28

Brevard County

Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to USDW (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day $^{-1}$)	Concentration at Injection Pt. (C_O)	Concentration at USDW (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.36	1.68	1.07	86	92	0.0004	**230**	0.0
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.36	1.68	1.11	86	95	0.0010	1.00	0.0
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.36	1.68	1.22	86	105	0.0003	0.010	0.0
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.36	1.68	1.13	86	97	N/A	0.005	0.005
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.36	1.68	1.21	86	104	0.0018	5.00	0.0
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.36	1.68	1.02	86	88	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	86	N/A	N/A	9.60	9.60

N/A = not applicable

Appendix 8. Aquifer Recharge Calculations

To determine risk associated with aquifer recharge of treated effluent, the fate and transport of representative stressors were conducted for a range of required setbacks of 200, 500 and 2,640 feet (0.5 mile). Utilizing hydrologic data for the Surficial Aquifer, the fate and transport of the selected representative stressors can be estimated.

The time of travel to the horizontal setback distances (X) can be estimated by dividing the setback distances by the seepage velocity (v_s) (Eqn. 28). Seepage velocity is defined as the velocity representing the average rate at which ground water moves (Fetter, 1994) and is estimated by dividing the Darcy flow (q) by the porosity (n) of the hydrologic unit (Eqn. 29). Porosity represents the ratio between the volumes of voids over the total volume of the media (Freeze and Cherry, 1979). In this analysis, published porosity values were used. Darcy flow is defined as fluid flow through porous media (e.g. sand) (Freeze and Cherry; 1979), taking into consideration that ground water flows through porous media, Darcian assumptions must be applied. Darcy flow takes into account horizontal hydraulic conductivity (K_h) and the horizontal hydraulic gradient (i) (Eqn. 30). Hydraulic conductivity represents the ability of the media to transmit water (Fetter, 1994). Simple substitution of the seepage velocity and Darcy flow equations into Equation 28 will result in Equation 31.

$$t = \frac{X}{v_s} \qquad \text{(Eqn. 28)}$$

$$v_s = \frac{q}{n} \qquad \text{(Eqn. 29)}^1$$

$$q = K_h \times i \qquad \text{(Eqn. 30)}^2$$

$$t = \frac{Xn}{K_h i} \qquad \text{(Eqn. 31)}$$

Once the time of travel to the predetermined setback distances (Appendix Table 8-1) has been estimated, a fate and transport analysis can be used to determine the final concentrations of representative stressors. The fate and transport of representative stressors can be estimated by a first order decay model (Eqn. 32), which estimates the final concentration (C) of the representative stressors in correlation to vertical travel times estimated earlier. This first order decay model is appropriate for analysis of the organic constituents, because it takes into account natural attenuation processes such as biodegradation, hydrolysis and sorption (Suthersan, 2002).

$$C = C_O e^{-kt_C} \qquad \text{(Eqn. 32)}^3$$

[1] Same equation used in Appendix 4 and 5 (Eqn. 4 and Eqn. 14)
[2] Same equation used in Appendix 5 (Eqn. 15)
[3] Same equation used in Appendix 7 (Eqn. 20)

where: C = Final concentration of stressors
C_O = Initial concentration of stressors
k = Decay coefficient of stressors
t_C = Travel time of stressors

Half-life ($t_{1/2}$) is defined as the time it takes for stressors to reach half of the initial concentration. The decay coefficient (k) can be determined by rearranging Equation 32, substituting the half-life in place of the travel time of stressors (t_C) and equating the ratio of the final versus initial concentrations to 0.5 (Eqn. 33). The decay coefficient (Eqn. 34) is simplified by rearranging Equation 33. Published values for half-life are available and were identified for the selected representative stressors (Howard et al., 1991).

$$\frac{C}{C_O} = 0.5 = e^{-kt_{1/2}} \qquad \text{(Eqn. 33)}^3$$

$$k = \frac{0.693}{t_{1/2}} \qquad \text{(Eqn. 34)}^3$$

The travel time of representative stressors (t_C) are determined by multiplying the retardation coefficient (R) by the effluent travel time (t_E) (Eqn. 35). In this analysis, the effluent travel time is equivalent to the vertical travel time estimated earlier.

$$t_C = R \times t_E \qquad \text{(Eqn. 35)}^3$$

The retardation coefficient takes into account sorption, a natural attenuation process which increases the travel time of stressors. The greater the travel time of stressors, the more time there is for other natural attenuation process to occur, such as biodegradation and hydrolysis. Biodegradation results in the degradation of organic material and may also mediate transformations in the state of inorganic material, resulting in decreasing concentrations over time. Hydrolysis is the process whereby organic and inorganic solutes react with water resulting in degradation and transformation (Suthersan, 2002). Calculation for the retardation coefficient, for dissolved organic constituents, is shown below in Equation 36 (Suthersan, 2002).

$$R = 1 + \frac{\rho_b K_d}{n} \qquad \text{(Eqn. 36)}^3$$

where: ρ_b = Bulk density = $\rho_s(1-n)$ (Eqn. 37)[3]
ρ_s = soil density
n = porosity
K_d = Distribution coefficient = $K_{oc} f_{oc}$ (Eqn. 38)[3]
K_{oc} = Sorption coefficient

[3] Same equation used in Appendix 7 (Eqn. 21 to Eqn. 26)

f_{oc} = fraction of total organic carbon

$$R = 1 + \frac{\rho_s (1-n) K_{oc} f_{oc}}{n} \qquad \text{(Eqn. 39)}^3$$

Sorption coefficients (K_{oc}) were obtained from published values for each representative stressor (Montgomery, 2000). For purposes of risk assessment, conservative values (indicating the least sorption) were selected to calculate the distribution coefficient and therefore the retardation coefficient. Ultimately, this produces conservative estimates of stressor concentrations at the receptors, since the data used relate to the lowest reasonably expected retardation and the shortest travel time. The calculations incorporated a typical value for sediment density of 2.63 g/cm^3 (Freeze and Cherry, 1979). Weighted mean porosity values (Appendix 3), based on unit thickness, were used in the calculations.

Appendix Table 8-2 to 8-4 summarizes the fate and transport of the representative stressors within 200, 500 and 2640 feet (0.5 mile) from the facility in Dade, Pinellas and Brevard Counties.

[3] Same equation used in Appendix 7 (Eqn. 27)

Appendix Table 8-1. Fate Transport (200')

Dade County

Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
	Max	Max		Max				Max		Max	Max		Min
Chloroform (μg/L)	1800	1.44	0.01	0.014	2.63	0.33	1.76	1.08	0.11	0	0.0004	7.18	7.06
Tetrachloroethylene (PCE) (μg/L)	720	2.25	0.01	0.023	2.63	0.33	1.76	1.12	0.11	0	0.0010	**4.66**	**4.46**
Chlordane (μg/L)	2772	4.72	0.01	0.047	2.63	0.33	1.76	1.25	0.11	0	0.0003	0.010	0.01
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.33	1.76	1.15	0.11	0	N/A	0.010	0.010
Di(2-ethylhexyl) Phthalate (DEHP) (μg/L)	389	4.48	0.01	0.045	2.63	0.33	1.76	1.24	0.11	0	0.0018	5.00	4.57
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.33	1.76	1.03	0.11	0.1	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.11	N/A	N/A	0.64	0.64

Pinellas County

Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (μg/L)	1800	1.44	0.01	0.014	2.63	0.25	1.97	1.11	5.86	6.5	0.0004	6.70	2.68
Tetrachloroethylene (PCE) (μg/L)	720	2.25	0.01	0.023	2.63	0.25	1.97	1.18	5.86	6.9	0.0010	2.50	0.22
Hexachlorobenzene (μg/L)	4178	2.56	0.01	0.026	2.63	0.25	1.97	1.20	5.86	7.0	0.0002	**1.74**	**1.14**
Pentachlorophenol (μg/L)	1520	2.76	0.01	0.028	2.63	0.25	1.97	1.22	5.86	7.1	0.0005	**1.28**	0.39
Benzo(a)pyrene (μg/L)	1060	5.95	0.01	0.060	2.63	0.25	1.97	1.47	5.86	8.6	0.0007	**1.82**	**0.23**
Chlordane (μg/L)	2772	4.72	0.01	0.047	2.63	0.25	1.97	1.37	5.86	8.0	0.0003	0.640	0.31
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.25	1.97	1.22	5.86	7.12	N/A	0.003	0.003
Di(2-ethylhexyl) Phthalate (DEHP) (μg/L)	389	4.48	0.01	0.045	2.63	0.25	1.97	1.35	5.86	7.9	0.0018	1.25	0.01
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.25	1.97	1.04	5.86	6.1	N/A	18.00	18.00
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	5.86	N/A	N/A	0.28	0.28

Brevard County

Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (μg/L)	1800	1.44	0.01	0.014	2.63	0.36	1.68	1.07	3.03	3	0.0004	**230**	**146**
Tetrachloroethylene (PCE) (μg/L)	720	2.25	0.01	0.023	2.63	0.36	1.68	1.11	3.03	3	0.0010	1.00	0.3
Chlordane (μg/L)	2772	4.72	0.01	0.047	2.63	0.36	1.68	1.22	3.03	4	0.0003	0.010	0.0
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.36	1.68	1.13	3.03	3	N/A	0.005	0.005
Di(2-ethylhexyl) Phthalate (DEHP) (μg/L)	389	4.48	0.01	0.045	2.63	0.36	1.68	1.21	3.03	4	0.0018	5.00	0.5
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.36	1.68	1.02	3.03	3	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	3.03	N/A	N/A	9.60	9.60

N/A = not applicable

Appendix Table 8-2. Fate Transport (500')

Dade County													
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.33	1.76	1.08	0.28	0	0.0004	7.18	6.88
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.33	1.76	1.12	0.28	0	0.0010	**4.66**	**4.17**
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.33	1.76	1.25	0.28	0	0.0003	0.010	0.01
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.33	1.76	1.15	0.28	0	N/A	0.010	0.010
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.33	1.76	1.24	0.28	0	0.0018	5.00	3.99
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.33	1.76	1.03	0.28	0.3	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.28	N/A	N/A	0.64	0.64

(Appendix 8 continued)

Pinellas County													
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.25	1.97	1.11	14.64	16.3	0.0004	6.70	0.68
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.25	1.97	1.18	14.64	17.2	0.0010	2.50	0.01
Hexachlorobenzene (µg/L)	4178	2.56	0.01	0.026	2.63	0.25	1.97	1.20	14.64	17.6	0.0002	**1.74**	0.60
Pentachlorophenol (µg/L)	1520	2.76	0.01	0.028	2.63	0.25	1.97	1.22	14.64	17.8	0.0005	**1.28**	0.07
Benzo(a)pyrene (µg/L)	1060	5.95	0.01	0.060	2.63	0.25	1.97	1.47	14.64	21.5	0.0007	**1.82**	0.01
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.25	1.97	1.37	14.64	20.1	0.0003	0.640	0.10
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.25	1.97	1.22	14.64	17.80	N/A	0.003	0.003
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.25	1.97	1.35	14.64	19.8	0.0018	1.25	0.00
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.25	1.97	1.04	14.64	15.2	N/A	18.00	18.00
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	14.64	N/A	N/A	0.28	0.28

Brevard County													
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (µg/L)	1800	1.44	0.01	0.014	2.63	0.36	1.68	1.07	7.58	8	0.0004	**230**	73.7
Tetrachloroethylene (PCE) (µg/L)	720	2.25	0.01	0.023	2.63	0.36	1.68	1.11	7.58	8	0.0010	1.00	0.1
Chlordane (µg/L)	2772	4.72	0.01	0.047	2.63	0.36	1.68	1.22	7.58	9	0.0003	0.010	0.0
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.36	1.68	1.13	7.58	9	N/A	0.005	0.005
Di(2-ethylhexyl) Phthalate (DEHP) (µg/L)	389	4.48	0.01	0.045	2.63	0.36	1.68	1.21	7.58	9	0.0018	5.00	0.0
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.36	1.68	1.02	7.58	8	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	7.58	N/A	N/A	9.60	9.60

N/A = not applicable

Appendix Table 8-3. Fate Transport (0.5 mile)

Dade County													
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (μg/L)	1800	1.44	0.01	0.014	2.63	0.33	1.76	1.08	1.47	2	0.0004	7.18	5.75
Tetrachloroethylene (PCE) (μg/L)	720	2.25	0.01	0.023	2.63	0.33	1.76	1.12	1.47	2	0.0010	**4.66**	2.61
Chlordane (μg/L)	2772	4.72	0.01	0.047	2.63	0.33	1.76	1.25	1.47	2	0.0003	0.010	0.01
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.33	1.76	1.15	1.47	2	N/A	0.010	0.010
Di(2-ethylhexyl) Phthalate (DEHP) (μg/L)	389	4.48	0.01	0.045	2.63	0.33	1.76	1.24	1.47	2	0.0018	5.00	1.53
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.33	1.76	1.03	1.47	1.5	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	1.47	N/A	N/A	0.64	0.64

Pinellas County													
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (μg/L)	1800	1.44	0.01	0.014	2.63	0.25	1.97	1.11	77.32	86.1	0.0004	6.70	0.00
Tetrachloroethylene (PCE) (μg/L)	720	2.25	0.01	0.023	2.63	0.25	1.97	1.18	77.32	91.0	0.0010	2.50	0.00
Hexachlorobenzene (μg/L)	4178	2.56	0.01	0.026	2.63	0.25	1.97	1.20	77.32	92.9	0.0002	**1.74**	0.01
Pentachlorophenol (μg/L)	1520	2.76	0.01	0.028	2.63	0.25	1.97	1.22	77.32	94.2	0.0005	**1.28**	0.00
Benzo(a)pyrene (μg/L)	1060	5.95	0.01	0.060	2.63	0.25	1.97	1.47	77.32	113.6	0.0007	**1.82**	0.00
Chlordane (μg/L)	2772	4.72	0.01	0.047	2.63	0.25	1.97	1.37	77.32	106.1	0.0003	0.640	0.00
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.25	1.97	1.22	77.32	93.97	N/A	0.003	0.003
Di(2-ethylhexyl) Phthalate (DEHP) (μg/L)	389	4.48	0.01	0.045	2.63	0.25	1.97	1.35	77.32	104.6	0.0018	1.25	0.00
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.25	1.97	1.04	77.32	80.3	N/A	18.00	18.00
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	77.32	N/A	N/A	0.28	0.28

Brevard County													
Surrogate	Published Half-Life in Groundwater ($t_{1/2}$) (days)	Published Sorption Coefficient (K_{OC})	Fraction of Total Organic Carbon (f_{oc})	Distribution Coefficient (K_d)	Soil Density (ρ_s)	Porosity (n)	Bulk Density (ρ_b)	Retardation Coefficient (R)	Effluent Travel Time to Receptor Wells (t_E) (years)	Contaminant Travel Time (t_C) (years)	Decay Coefficient (k) (day^{-1})	Concentration at Injection Pt. (C_O)	Concentration at Supply Well (C)
Chloroform (μg/L)	1800	1.44	0.01	0.014	2.63	0.36	1.68	1.07	40.04	43	0.0004	**230**	0.6
Tetrachloroethylene (PCE) (μg/L)	720	2.25	0.01	0.023	2.63	0.36	1.68	1.11	40.04	44	0.0010	1.00	0.0
Chlordane (μg/L)	2772	4.72	0.01	0.047	2.63	0.36	1.68	1.22	40.04	49	0.0003	0.010	0.0
Arsenic (mg/L)	N/A	2.73	0.01	0.027	2.63	0.36	1.68	1.13	40.04	45	N/A	0.005	0.005
Di(2-ethylhexyl) Phthalate (DEHP) (μg/L)	389	4.48	0.01	0.045	2.63	0.36	1.68	1.21	40.04	48	0.0018	5.00	0.0
Ammonia (mg/L) (conservative behavior)	N/A	0.49	0.01	0.005	2.63	0.36	1.68	1.02	40.04	41	N/A	8.75	8.75
Nitrates (mg/L)(conservative behavior)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	40.04	N/A	N/A	9.60	9.60

N/A = not applicable