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.

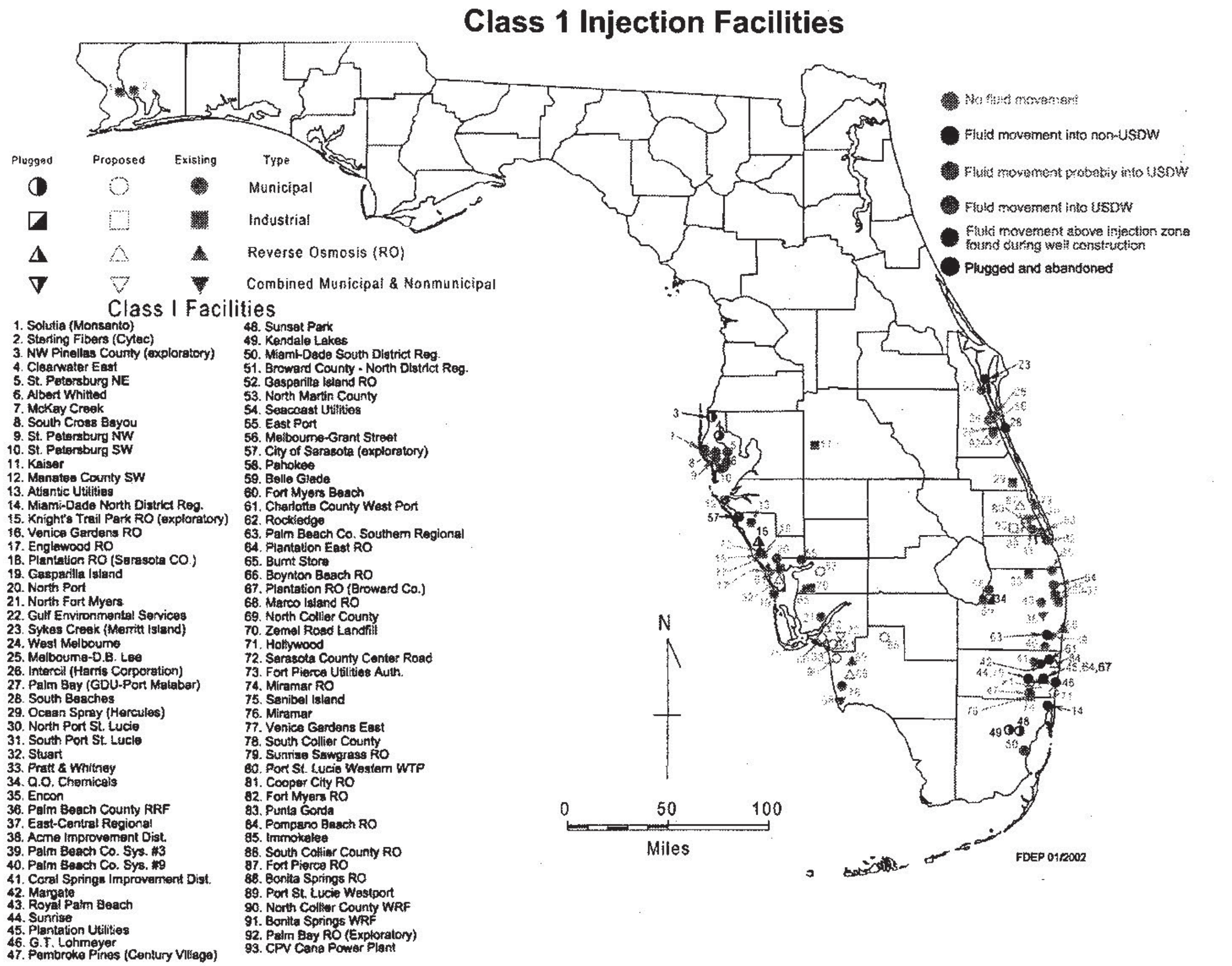

Figure 4-4. Fluid Movement Associated with Class I Deep Well Injection Facilities in South Florida

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.

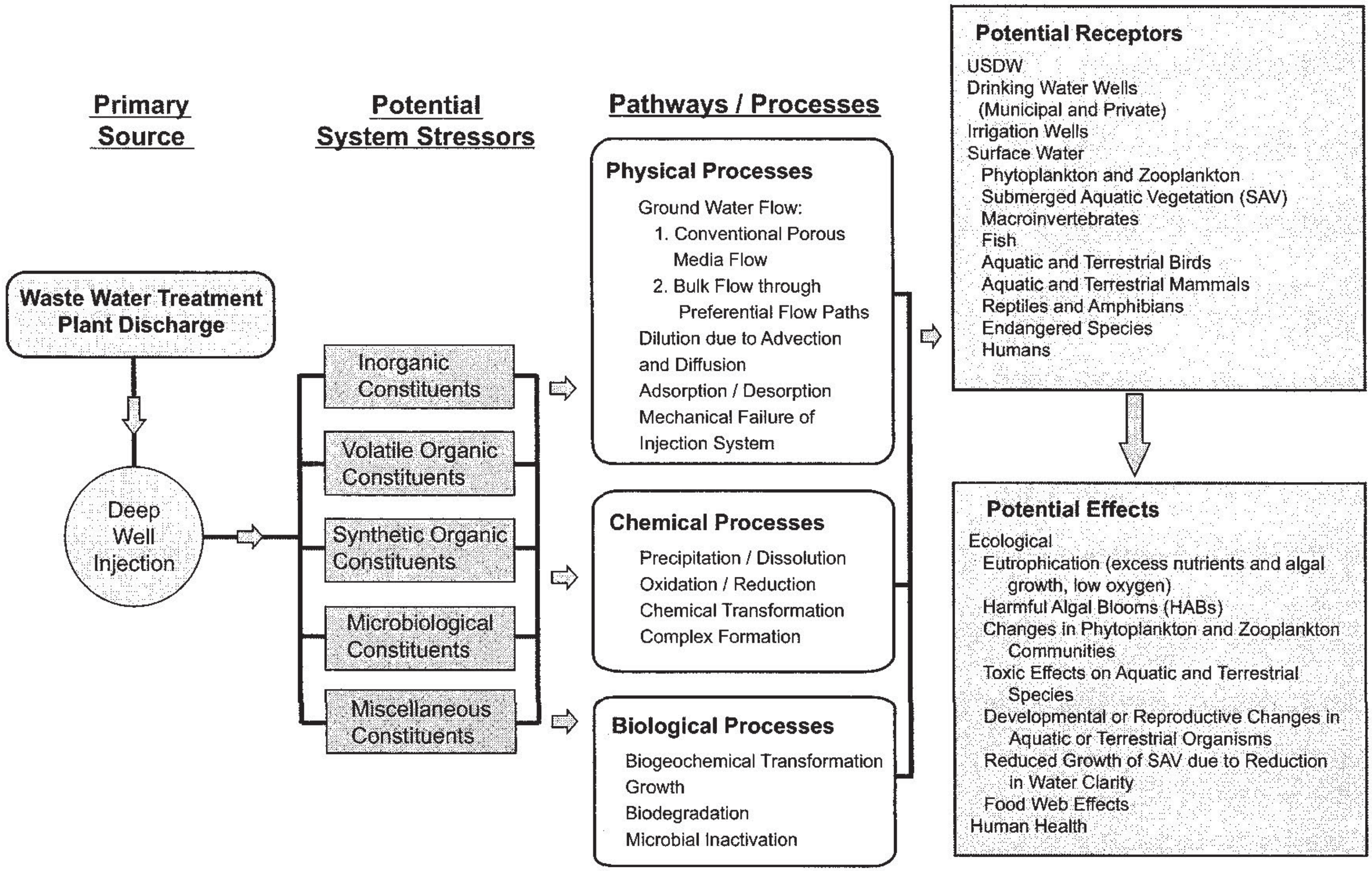

Figure 4-5. Conceptual Model of Potential Risks for the Deep Well Injection Option

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.

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 80° Fahrenheit, whereas native groundwater has a temperature far closer to 60° 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**
	Components	**To Receptor Well**	**To USDW**
Pinellas County Injection rate = 7 mgd	Buoyancy	18 ft	16 ft
	Injection	533 ft	533 ft
	Total Head [2]	**551 ft**	**549 ft**
	Components	**To Receptor Well**	**To USDW**
Brevard County Injection rate = 5 million mgd	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 30° and 35° Celsius; poor growth occurs at temperatures below 5 °Celsius (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 (1×10^{-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 °C*	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:

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 °C	Bitton et al., 1983
Fecal streptococci	0.0204	49.02	Florida groundwater sample at 22 °C	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 °C, 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 °C, 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 °C (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 °C (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.

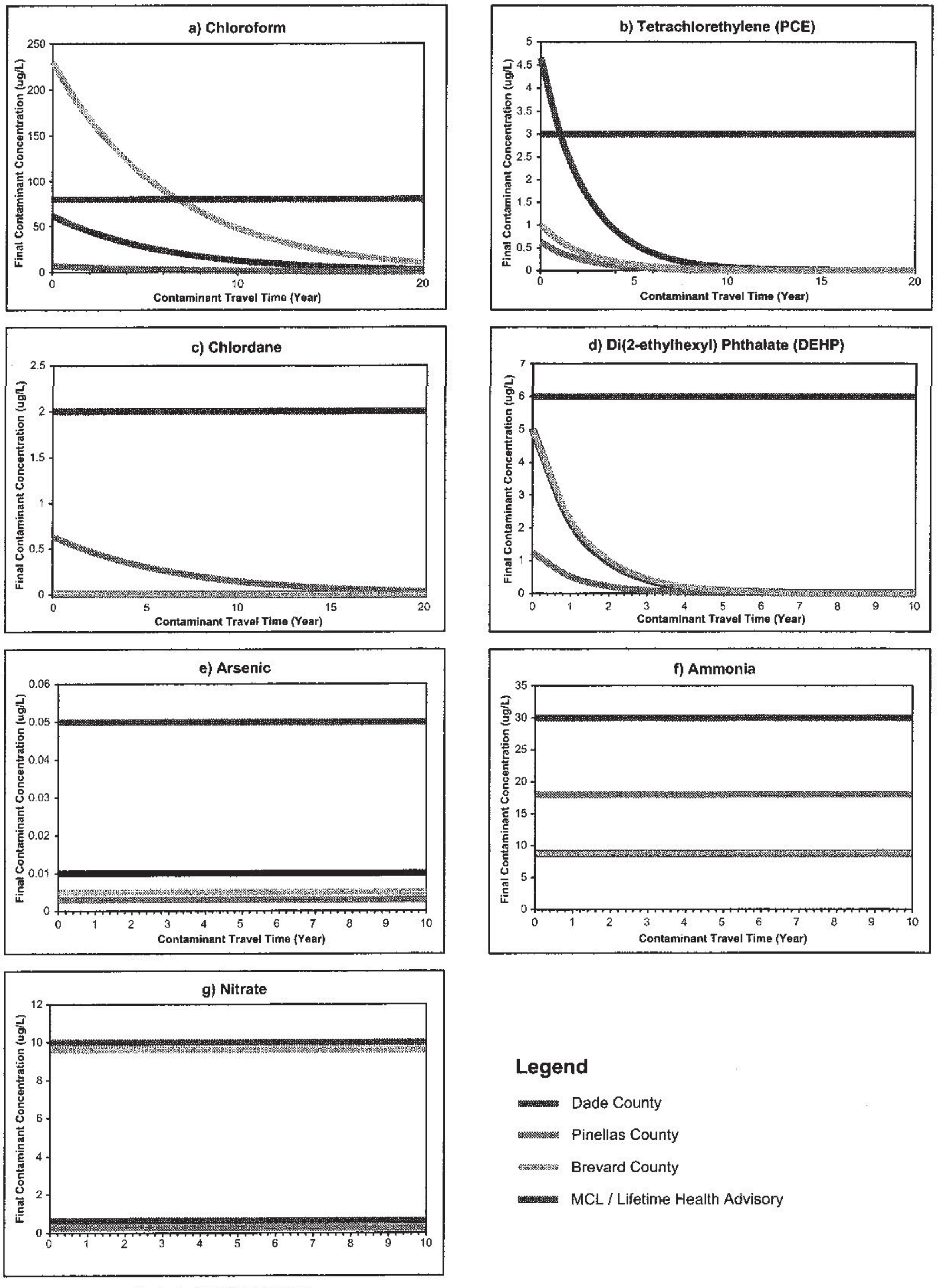

Figure 4-8. Final Concentrations of Representative Stressors Versus Time

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 1×10^{-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.

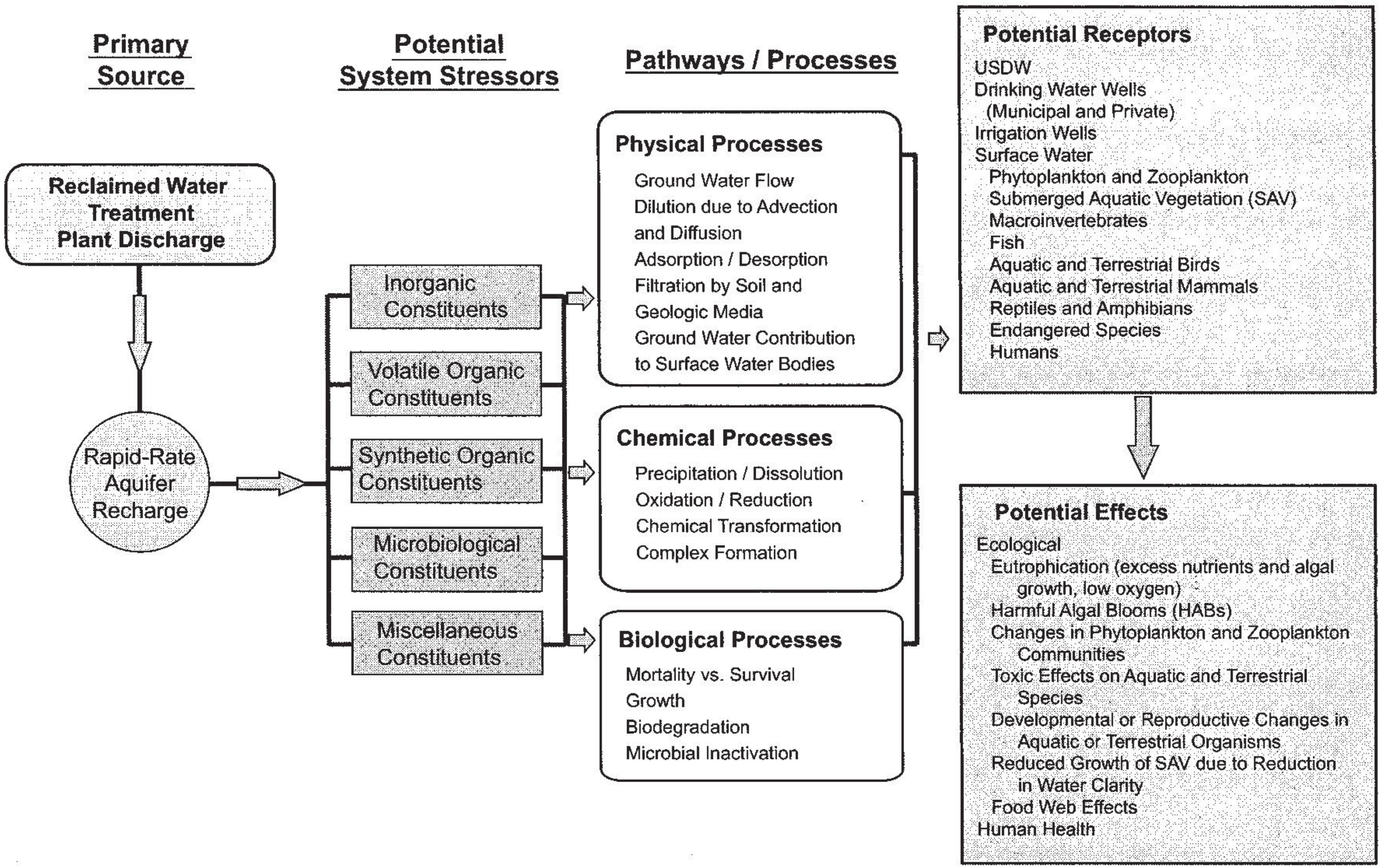

Figure 5-1. Conceptual Model of Potential Risks for the Aquifer Recharge Option

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 *Cryptosporidum 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.

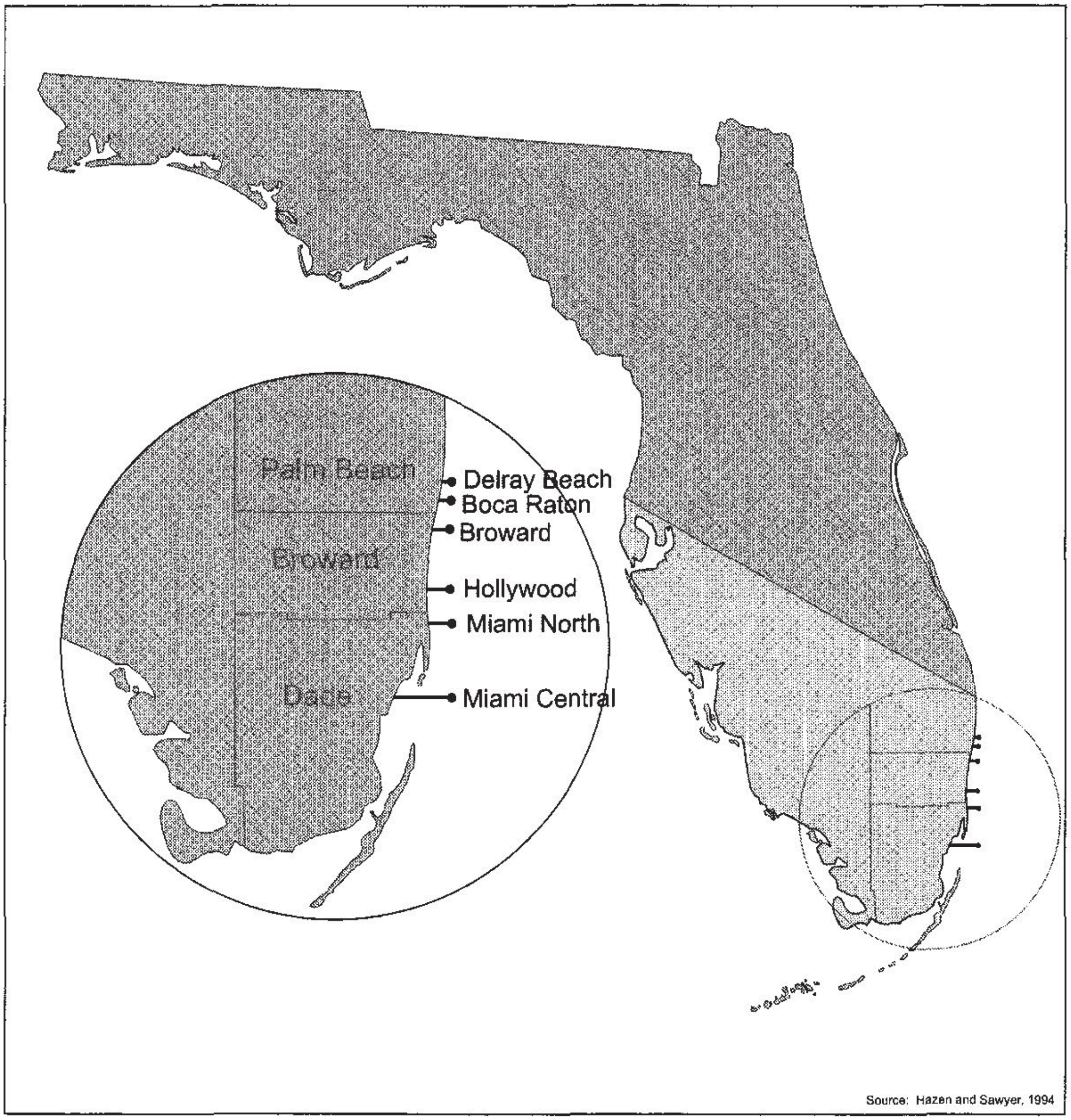


Figure 6-1. ▯ocations of Ocean Outfalls in ▯outhern Florida

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.

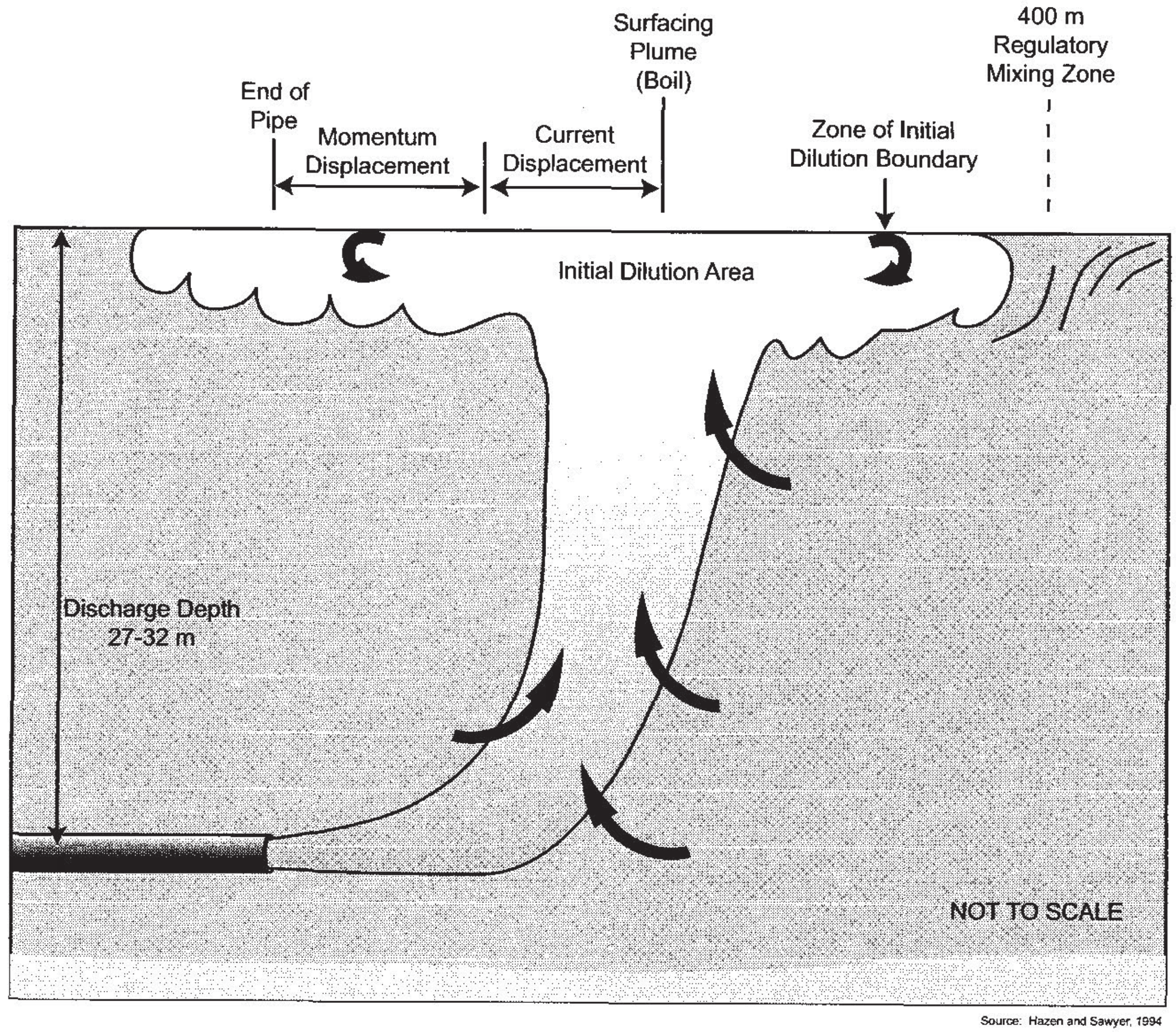


Figure 6-□. □ffluent Plume □haracteristics for Ocean Outfalls

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).

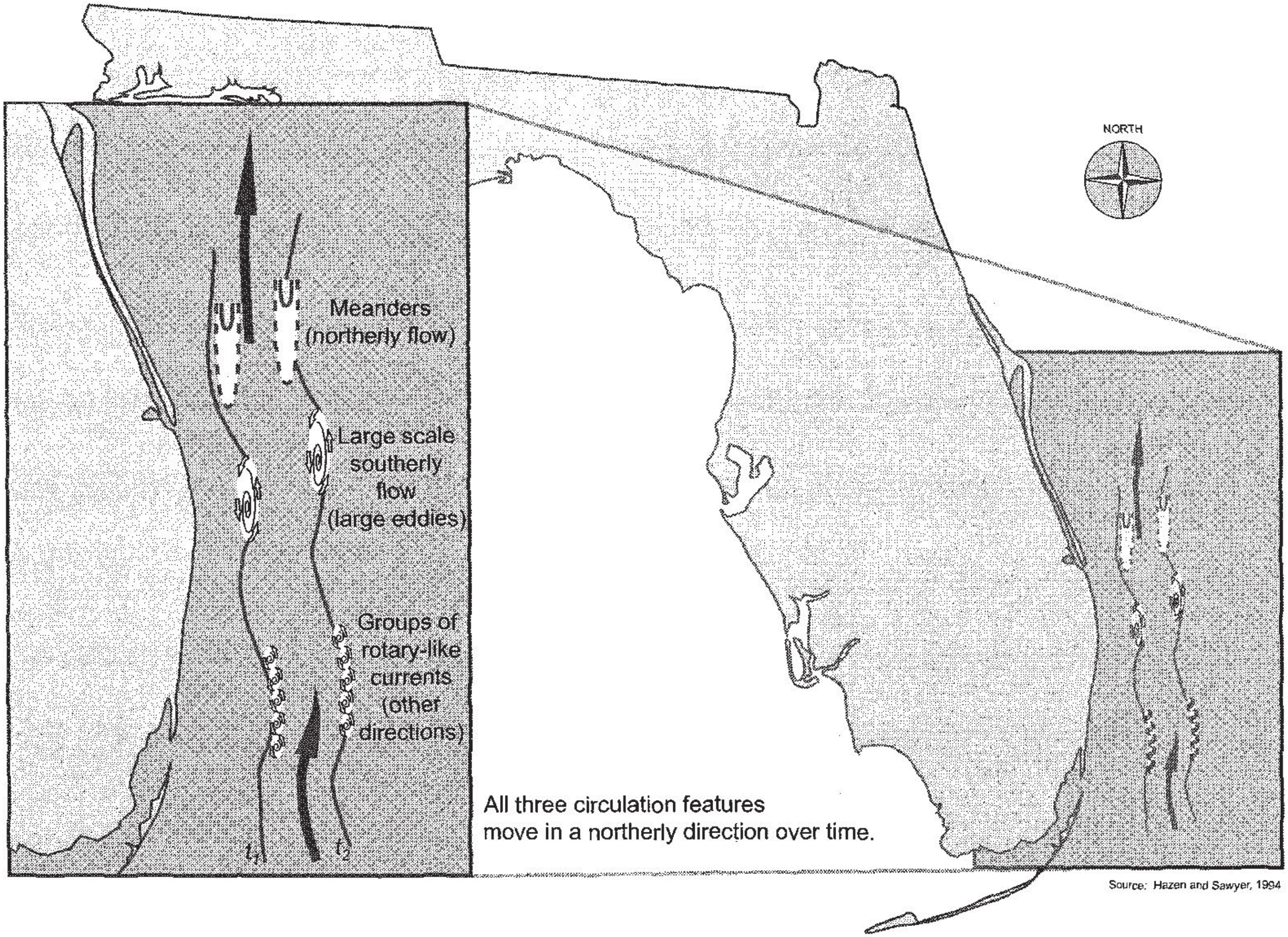
As the Florida Current travels northward off the east coast of Florida, spin-off eddies are created (Lee, 1975; Lee et al., 1995). These eddies include several components, including northerly flows associated with western meanders of the Florida Current, southerly flows, and rotary flows, composed of 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.



Source: Hazen and Sawyer, 1994

Figure 6-□. □irculation □haracteristics of the □estern Boundary □egion of the Florida □urrent.

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.