4.0 DEEP-WELL INJECTION

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

4.1 Definition of the Deep-Well Injection Option

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

4.2 Deep-Well Capacity and Use in South Florida

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

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

4.3 Environment into Which Treated Wastewater is Discharged

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

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

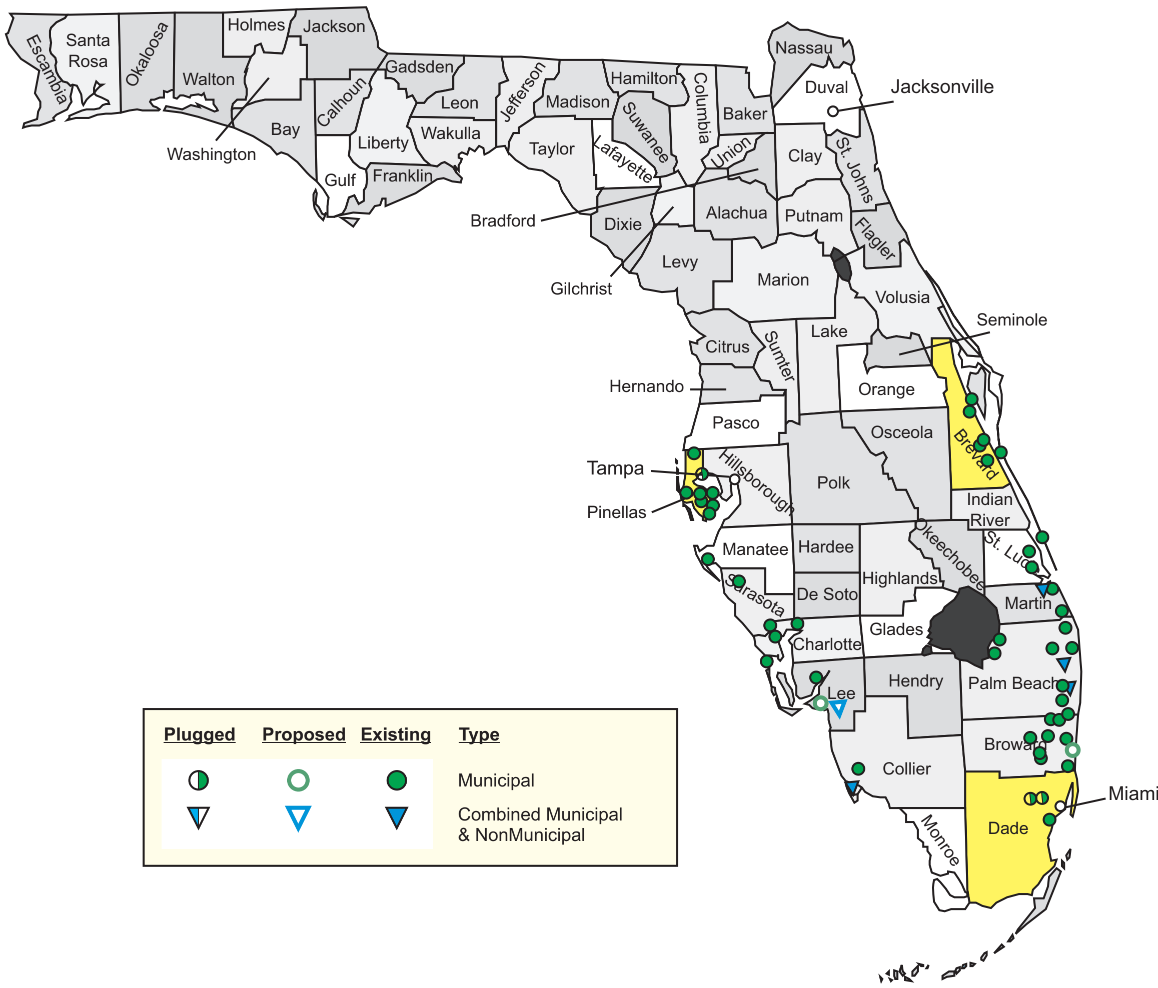

Figure 4-1. Locations of Class I Injection Wells in South Florida

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

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

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

4.3.1 Aquifers in South Florida

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

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

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

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

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

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

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

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

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

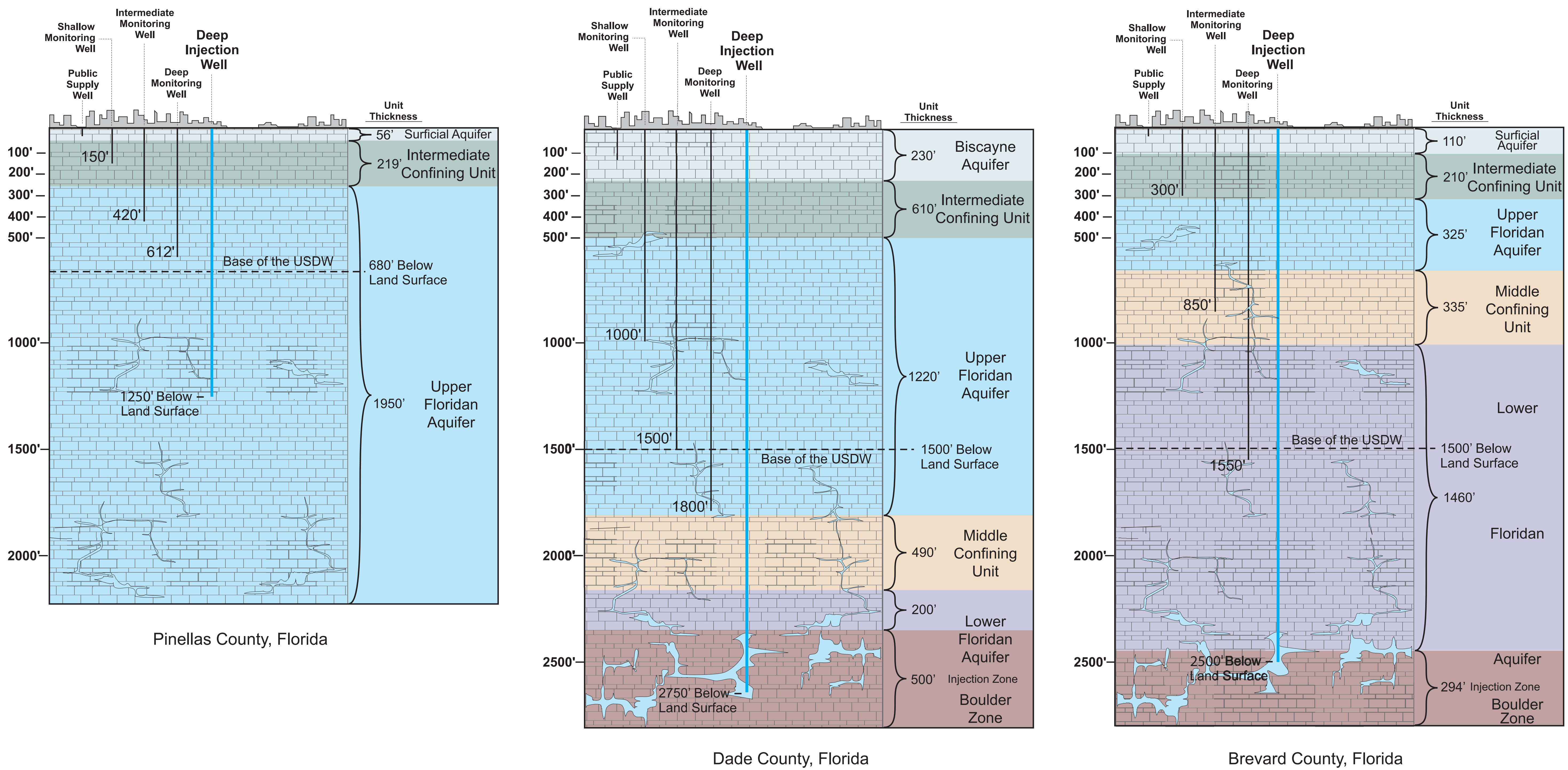

Figure 4-2. Representative Hydrogeologic Cross Sections

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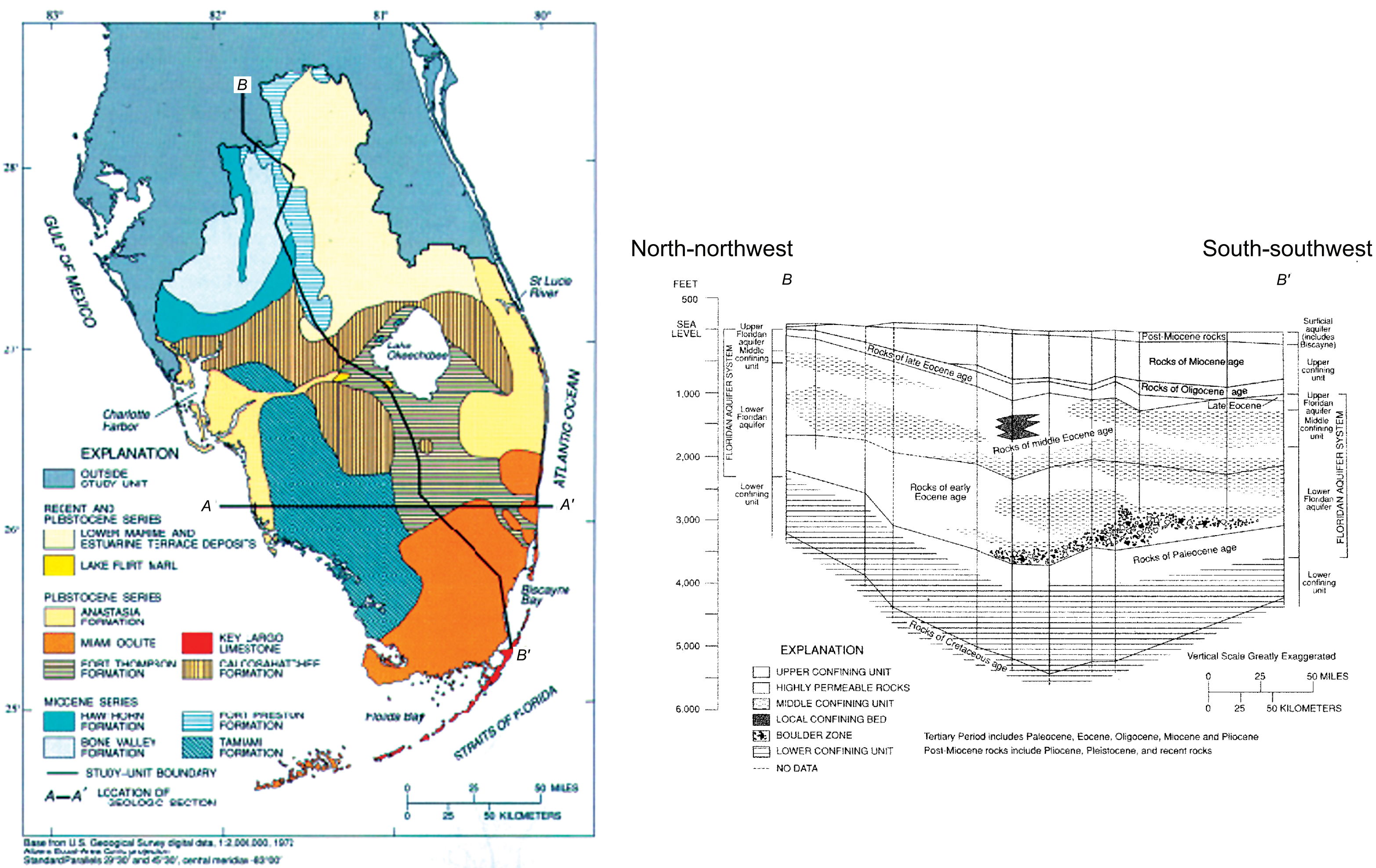

Source: McPherson et al (2000)

Figure 4-3. Geologic Profile of South Florida

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

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

4.3.2 Regional Conditions in Dade County

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

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

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

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

4.3.3 Regional Conditions in Pinellas County

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

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

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

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

4.3.4 Regional Conditions in Brevard County

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

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

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

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

4.4 Groundwater Quality and Fluid Movement in South Florida

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

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

Data are available for characterizing the quality of groundwater resources since injection-well construction and operation began. Englehardt et al. (2001) compiled a limited data set that includes information about the levels of inorganic contaminants present in lower and upper native (or ambient) groundwater monitoring zones (Appendix Table 1-1). Though it cannot be said conclusively that these data characterize preoperation conditions, the data are sufficient for illustrating two points. First, deep native groundwater in southeast Florida does appear to exceed several primary or secondary drinking-water standards (maximum contaminant levels, or MCLs). Second, for some contaminants (for example, cadmium, lead, antimony, aluminum, iron), there is reason to conclude that these levels are of natural origin (resulting, for example, from the dissolution of the native aquifer matrix) and not attributable to any aspect of well construction or operation. For some other contaminants (for example, thallium, beryllium), it is less clear why there are slightly elevated levels present in upper and lower groundwater monitoring zones.

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

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

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

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

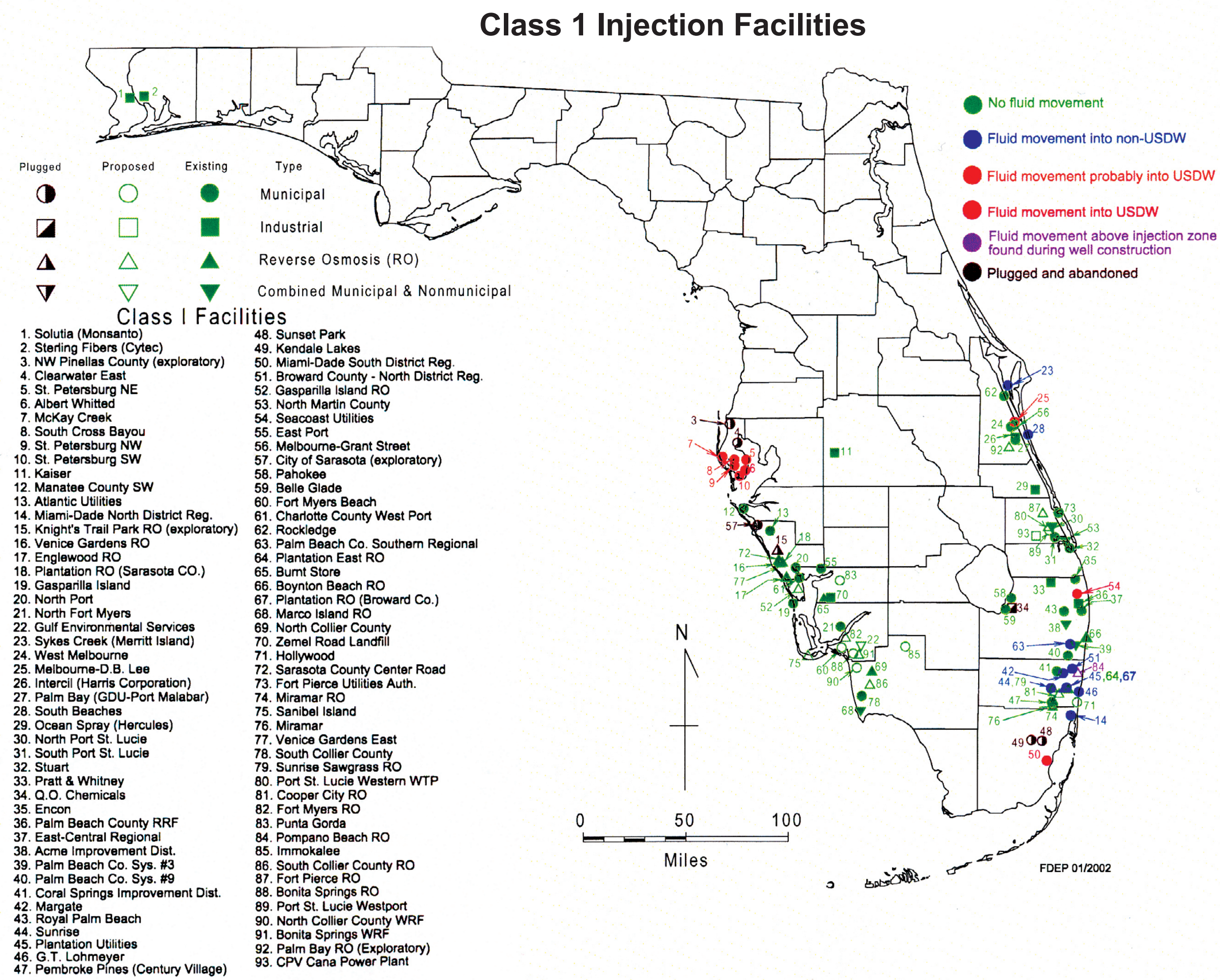

Figure 4-4. Fluid Movement Associated with Class I Deep Well Injection Facilities in South Florida

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

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

4.4.1 Dade County Groundwater Monitoring Information

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

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

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

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

A purging report from December 1998 (SDWWTP, 1998) indicates that there was a slight decrease in the concentrations of ammonia detected by monitoring well FA-16 in response to purging. In another well, FA-15, there was a larger drop in ammonia concentrations after purging but subsequently these concentrations stabilized at a lower, but still elevated, level. Detected levels of ammonia were higher than background levels for these depths, and as such, were interpreted as an indicator of potential contamination resulting from movement of injected fluids.

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

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

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

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

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

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

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

4.4.2 Pinellas County Groundwater Monitoring Information

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

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

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

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

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

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

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

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

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

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

4.4.3 Brevard County Groundwater Monitoring Information

4.4.3.1 South Beaches

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

A 2001 report prepared for the South Beaches facility (CDM, 2001) includes ground-water monitoring data for three monitoring wells at the site. A shallow well, MW-1, monitors the Ocala formation from 300 to 350 feet. Well MW-3, placed at an intermediate depth, monitors the middle of the Upper Floridan Aquifer from 1,200 feet to 1,320 feet. A deep well, MW-2, monitors the lower part of the Upper Floridan Aquifer from 1,550 feet to 1,700 feet.

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

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

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

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

4.4.3.2 Palm Bay

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

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

4.5 Regulations and Requirements for the Deep-Well Injection Option

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

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

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

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

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

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

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

In Florida, Class I wells injecting treated wastewater into Class G-IV waters must provide secondary treatment, at a minimum, and must meet pH limitations. Class G-IV waters are defined as groundwater for nonpotable use or groundwater in confined aquifers, that has a TDS content of 10,000 mg/L or greater (FAC 62-520.410). Disinfection is not required, but all Class I well permittees must maintain the capability to disinfect (FAC 62-600.540).

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

4.6 Problem Formulation

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

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

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

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

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

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

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

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

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

The subsurface heterogeneity that is characteristic of South Florida introduces complexity. Unit properties (for example, hydraulic conductivity, porosity, effective porosity) vary from one unit to the next, within a given unit from one site to another, and even within a given unit at a given site. Accounting for this heterogeneity presents a significant challenge in evaluating risk. In an effort to explore possibilities where available data are limited or inconclusive, this analysis relies on an exhaustive review of available data concerning unit properties and considers two different scenarios as it examines uncertainty.

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

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

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

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

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

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

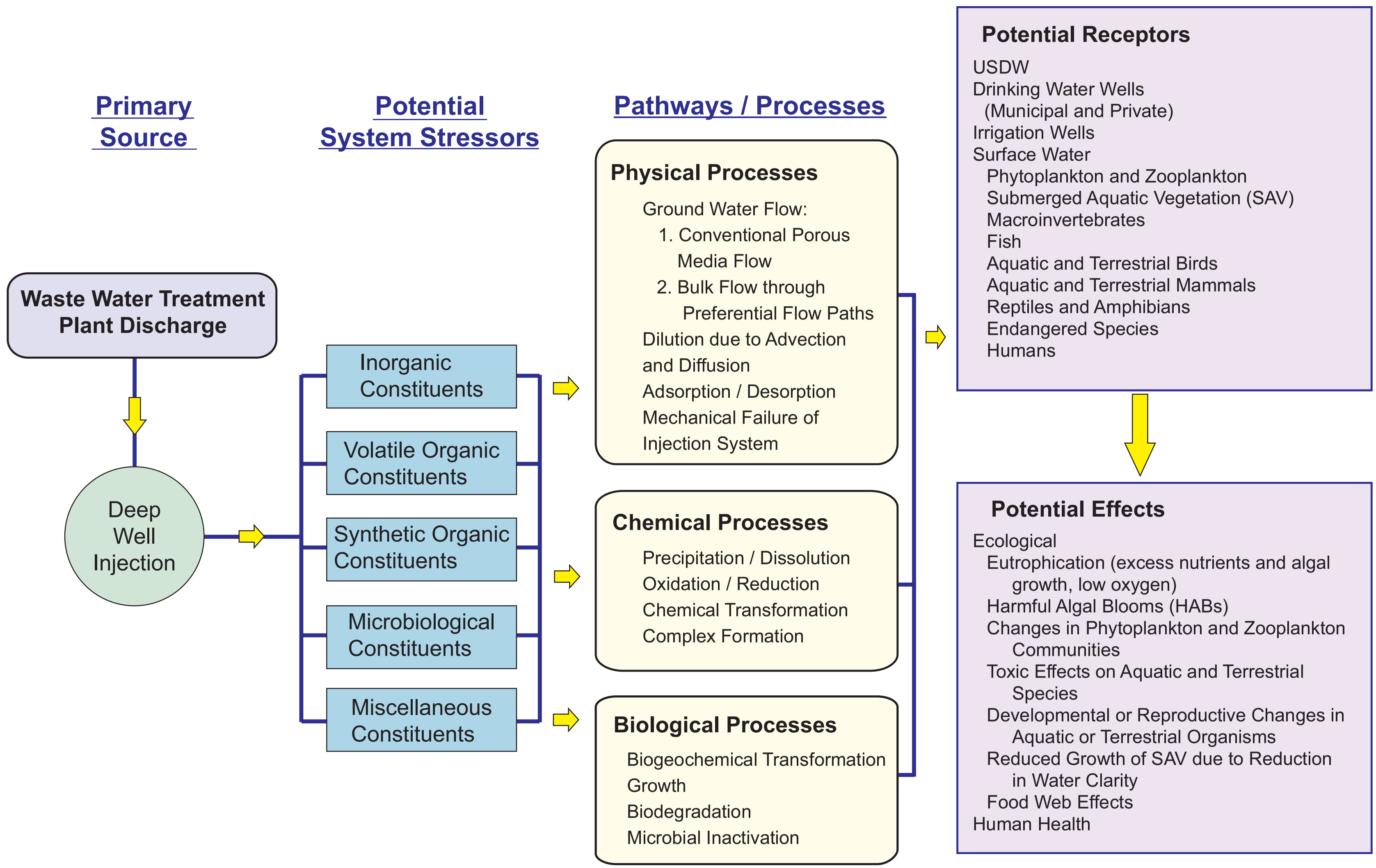

Figure 4-5. Conceptual Model of Potential Risks for the Deep Well Injection Option

4.7.1 Potential Stressors

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

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

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

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

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

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

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

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

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

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

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

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

Ammonia is an important potential human health stressor for which there is no MCL. The EPA has published a Lifetime Health Advisory for ammonia of 30 mg/L (US EPA, 2000). This Lifetime Health Advisory is an estimate of the acceptable level of ammonia in drinking water, based on health effects information. According to the advisory, at this concentration, a lifetime of exposure to ammonia is not expected to cause adverse health effects. Ammonia is not considered a suspected or human carcinogen. Ammonia and other nitrogenous materials (as measured by the parameters total nitrogen and TKN) may also be of further significance to human health as sources of combined nitrogen that may be converted to nitrate.

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

Table 4-5. Representative Pathogenic Stressors

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

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

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

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

Data to describe concentrations of some representative pathogenic stressors (for example, rotaviruses, *Cryptosporidium parvum, Giardia lamblia*) are incomplete and not widely available. Rose et al. (1991) reported that secondary-treated wastewater contains concentrations of *Giardia* ranging from 0.2 to 130 cysts/L (average 0.88 cysts/L). Levels of *Cryptosporidium* and *Giardia* in advanced treated and reclaimed wastewater compare favorably with risk-based criteria recommended by York and Walker-Coleman (1999) and York et al. (2002).

4.7.2 Potential Exposure Pathways

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

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

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

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

4.7.3 Potential Receptors and Assessment Endpoints

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

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

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

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

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

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

4.8 Risk Analysis of the Deep-Well Injection Option

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

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

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

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

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

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

4.8.1 Application of the Analytical Transport Model

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

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

Bulk flow through preferential flow paths (channel or fracture flow) is a scenario where fluid flows through naturally occurring or man-made conduits. Naturally occurring conduits include fractures, solution channels, and fissures. Man-made conduits might include injection wells with faulty construction, monitoring wells with faulty construction, abandoned wells, or fractures created because of well drilling or injection. Figure 4-7 illustrates the flow of injectate where bulk flow is the primary mechanism of plume migration. It is important to note that preferential flow pathways may result from the presence of naturally occurring solution channels or fractures in geologic strata or from mechanical problems associated with wells.

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

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

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

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

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

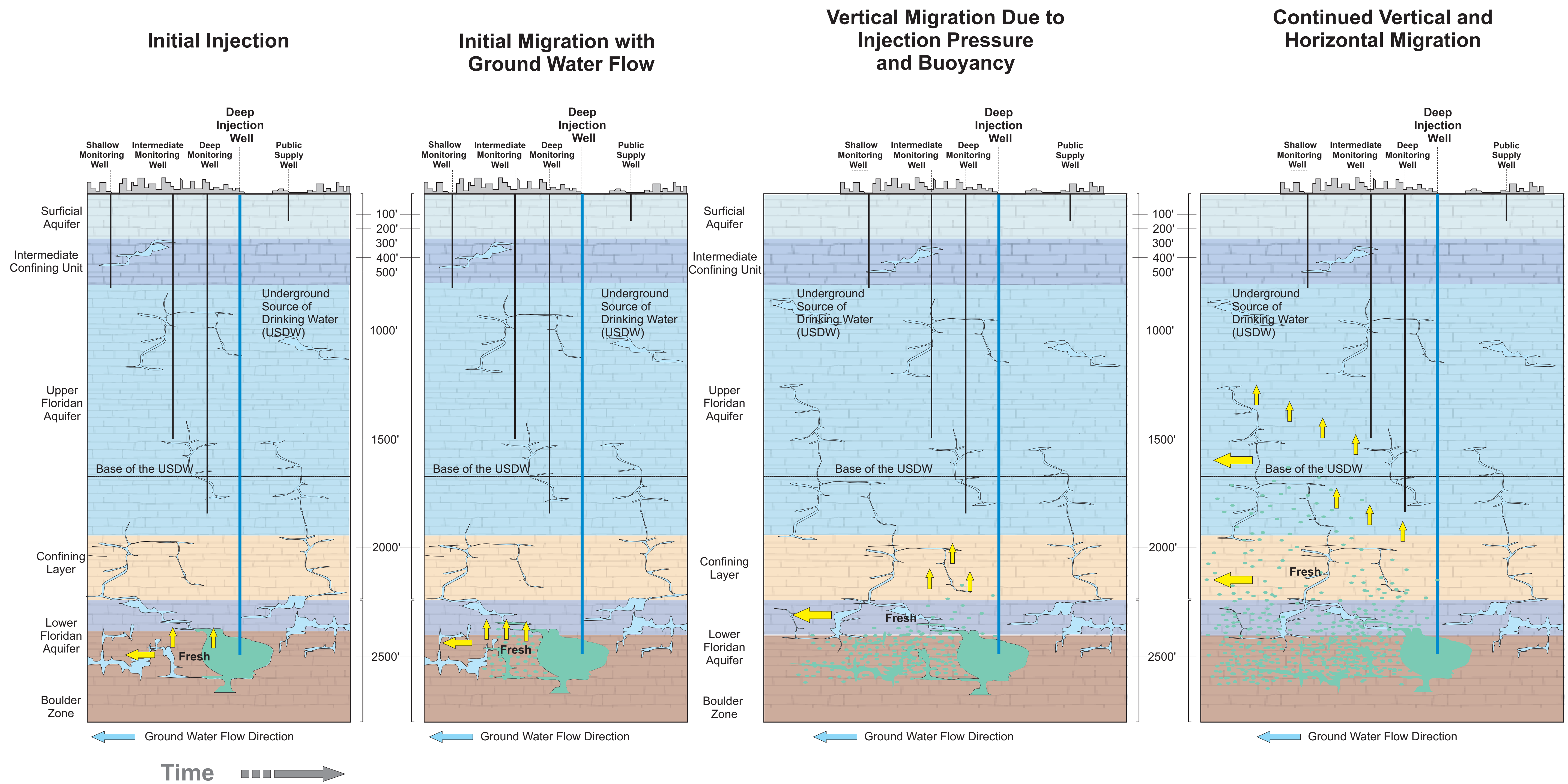

Figure 4-6. Migration Following Deep Well Injection; Fluid Flow Through Porous Media (Scenario 1)

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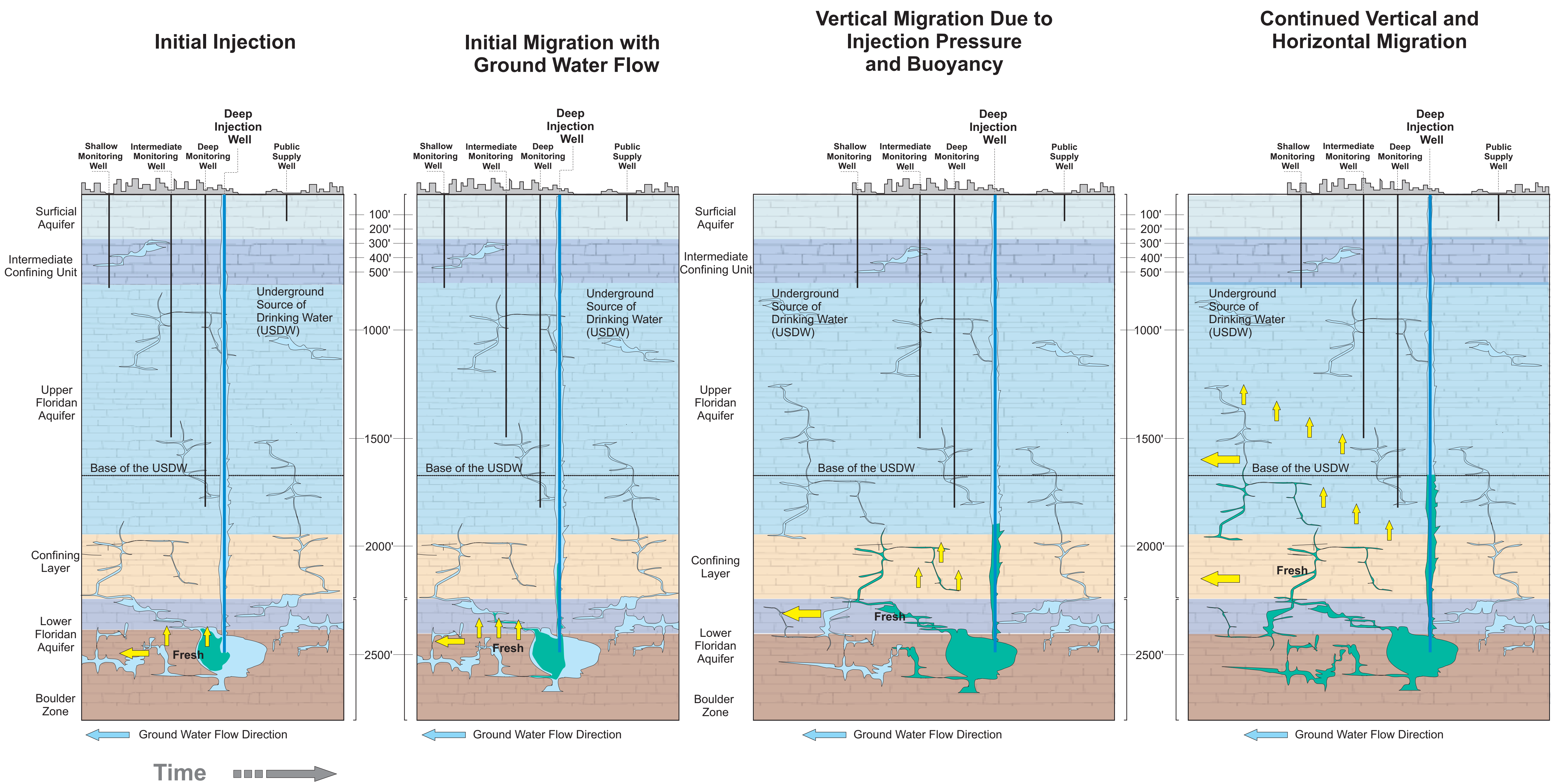

Figure 4-7. Migration Following Deep Well Injection; Bulk Flow Through Preferential Flow Paths (Scenario 2)

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

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

4.8.2 Vertical Times of Travel and Horizontal Migration

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

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

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

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

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

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

4.8.2.1 Governing Assumptions for the Transport Model

The following are the governing assumptions for the transport model:

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

4.8.2.2 Vertical Time-of-Travel Results and Discussion

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

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

	Components	To Receptor Well	To USDW
Dade County Injection rate = 112.5 mgd [1]	Buoyancy	73 ft	68 ft
	Injection	0 ft	0 ft
	Total Head [2]	**73 ft**	**68 ft**
	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 30º and 35º Celsius; poor growth occurs at temperatures below 5 ºCelsius (Buswell et al., 1954; Deppe and Engel, 1960, summarized in Fenchel, 1983). Nitrifying bacteria can survive under anoxic conditions but experience high rates of mortality wherever hydrogen sulfide is produced by anaerobic sulfate-reducing bacteria (reviewed in Blackburn, 1983).

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

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

4.8.3.3 Fate and Transport of Pathogenic Microorganisms

Assessing the potential human health risks from microbial pathogens in injected treated wastewater depends to a large extent on evaluating the fate and transport of pathogenic microorganisms. A crucial step in risk assessment is determining whether pathogens can be transported in an infective form to drinking water receptors and to human receptors. Thus, there are four risk questions to address:

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

Assessment endpoints used in this microbial risk assessment include a 1 in 10,000 (1×10^{-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.

<u>Microbial Concentrations Needed to Cause Risk</u>

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.

<u>Microbial Survival in Groundwater</u>

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

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

Table 4-16. Survival of Microorganisms in Water

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

4.9 Final Conceptual Model of Risk for Deep-Well Injection

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

4.9.1 Injection Pressure Head and Buoyancy Pressure

Vertical migration of effluent constituents depends on two major components: pressure head from injection and pressure head from buoyancy. Pressure head from injection is a result of injected effluent displacing native groundwater in the injection zone. Pressure head from buoyancy is a result of salinity and temperature differences between the injectate and native groundwater. Fluids that are more saline tend to be denser than fluids that are less saline. Warmer fluids tend to be less dense relative to cooler fluids.

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

4.9.2 Vertical Time of Travel

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

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

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

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

4.9.3 Horizontal Distance Traveled in a Given Travel Time

Based on horizontal hydrogeologic conditions and estimated vertical travel times, the extent of horizontal migration was estimated for each county. For scenario 1, the expected horizontal migration in Dade County is approximately 16 miles. Dade County has the furthest horizontal migration relative to Brevard and Pinellas counties, which have an expected horizontal migration of 1.5 and 1.2 miles, respectively. For scenario 2, as expected, Dade County has the furthest horizontal migration distance of 1.6 miles, while Brevard and Pinellas counties have horizontal travel distances of 0.1 and 0.6 miles, respectively.

4.9.4 Fate of Chemical Constituents

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

4.9.5 Comparison with Monitoring-Well Data

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

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

4.9.6 Mechanical Integrity as a Risk Factor

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

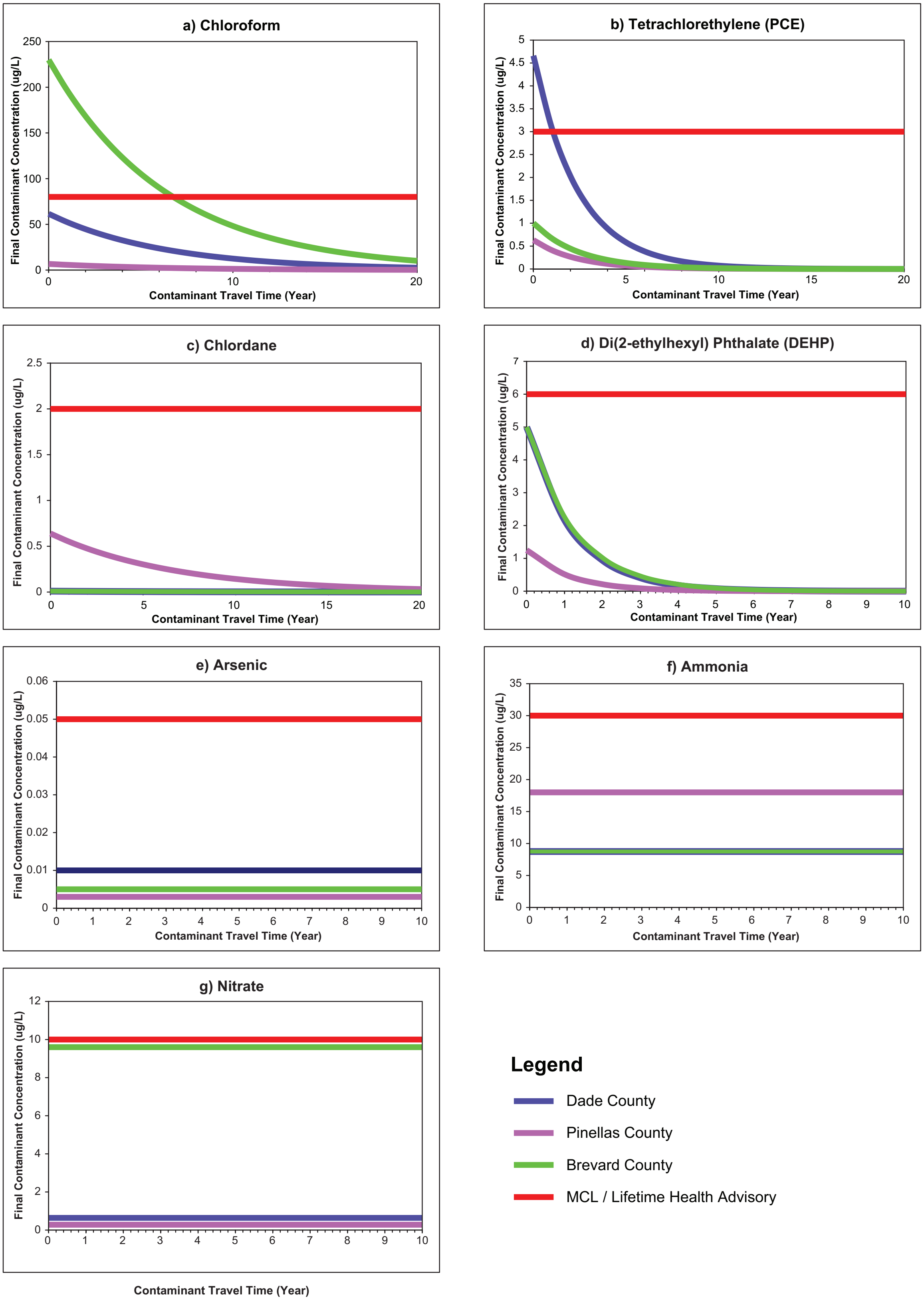

Figure 4-8. Final Concentrations of Representative Stressors Versus Time

4.9.7 Fate and Transport of Pathogenic Microorganisms

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

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

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

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

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

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

4.9.8 Effects of Data Gaps

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

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

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

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

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

REFERENCES

Adams K. 1992. A three dimensional finite difference flow model of the Surficial Aquifer in Martin County, Florida. SFWMD Technical Publication 92-02.

Anan'ev NI, and Demin ND. 1971. On the spread of pollutants in subsurface waters, *Hyg. Sanit.,* 36:292-294.

Aulenbach DB. 1979. *Long-Term Recharge of Trickling Filter Effluent into Sand*. Rep. EPA-600/2-79-068. Ada (OK): US EPA.

Barr GL. 1996. *Hydrogeology of the Surficial and Intermediate Aquifer Systems in Sarasota and Adjacent Counties, Florida*. USGS WRI 96-4063. Reston (VA): USGS.

BC&E and CH2M Hill. 1977. *Drilling and Testing of the Test-Injection Well. Prepared for the Miami-Dade Water and Sewer Authority*. EPA Contract No. C120377020.

Beard PJ. 1940. Longevity of *Eberthella typhosus* in various soils. *American Journal of Public Health* 30: 1077.

Bitton G, Farrah SR, Ruskin RH, Butner J, and Chou YJ. 1983. Survival of pathogenic and indicator organisms in groundwater. *Ground Water*. 21:405-410.

Blackburn TH. 1980. Seasonal variations in the rate of organic-N mineralization in anoxic marine sediments. In: *Biogeochimie de la Matiere Organique a l'Interface Eau-Sediment Marin. Edition du Center National se la Recherche Scientifique.* Paris: Colloque International C.N.R.S.

Bowman DD and Jenkins MB. 1996. The effects of ammonia on the viability of *Cryptosporidium parvum* as determined by a dye-permeability assay. Paper: *45th Annual Meeting, American Society of Tropical Medicine and Hygiene*. 1996; Baltimore, MD.

Broska JC and Barnette HL. 1999. *Hydrogeology And Analysis Of Aquifer Characteristics In West-Central Pinellas County, Florida*. USGS OFR 99-185.

Bush PW. and Johnston RH. 1988. *Ground-Water Hydraulics, Regional Flow, and Ground-Water Development of the Floridan Aquifer System in Florida, and in Parts of Georgia, South Carolina, and Alabama*. USGS Professional Paper 1403-B. Reston (VA): USGS.

Camp, Dresser, McKee, Inc. 1992. *Point-/Nonpoint Source Pollution Loading Assessment. Phase I. Final Report*. Sarasota (FL): Sarasota Bay National Estuary Program.

CH2M Hill. 1977. *Drilling and Testing of the Test-Injection Well for the Miami-Dade Water and Sewer Authority*. EPA Contract No. C120377020. Gainesville (FL): CH2M Hill.

CH2M Hill. 1981. *Drilling and Testing of the 8 Injection Wells and the 3 Monitoring Wells for the South District Regional Wastewater Treatment Plant of the Miami-Dade Water and Sewer Authority, Dade County, Florida*. EPA Contract No. C120377020. Gainesville (FL): CH2M Hill.

CH2M Hill. 2001. *Annual summary report of the City of St. Petersburg's Four Water Reclamation Facilities' Deep Injection Well System (January 1999 through December 1999)*. Tampa (FL): CH2M HILL.

Duerr AD. 1995. *Types of secondary porosity of carbonate rocks in injection and test wells in Southern Peninsular Florida*. USGS WRI 94-4013. Reston (VA): USGS.

Duerr AD and Enos GM. 1991. *Hydrogeology of the Intermediate Aquifer System and Upper Floridan Aquifer, Hardee and De Soto Counties, Florida*. USGS WRI 90-4104. Reston (VA): USGS.

Duncan JG, Evans WL, and Taylor KL. 1994. *Geologic framework of the lower Floridan Aquifer System, Brevard County, Florida*. Bulletin No. 64. Florida Geological Survey, Florida Department of Environmental Protection.

Earle JE, and Meyer FW. 1973. *Reconnaissance of the water resources in the vicinity of proposed deep-well injection sites in southeast Dade County, Florida*. USGS WRI 78-73. Reston (VA): USGS.

Elliot EL and Colwell RR. 1985. Indicator organisms for estuarine and marine waters. *FEMS Microbiology Reviews*. 32: 61-79.

Englehardt JD, Amy VP, Bloettscher F, Chin DA, Fleming LE, Gokgoz S, Rose JB, Solo-Gabriele H, and Tchobanoglons G. 2001. *Comparative Assessment of Human and Ecological Impacts Form Municipal Wastewater Disposal Methods in Southeast Florida*. University of Miami. Submitted to the Florida Water Environmental Association Utility Council.

Fenchel TH. 1983. The Microbial Nitrogen Cycle. In: *Microbial Geochemistry*. Krumbein WE, ed. Boston: Blackwell Scientific Publications.

Fetter CW. 1994. *Applied Hydrogeology.* 3rd ed. New York: Macmillan College Publishing Company, Inc.

Fish JE and Stewart M. 1991. *Hydrogeology of the Surficial Aquifer System, Dade County, Florida*: USGS WRI 90-4108. Reston(VA): USGS.

Fletcher MW and Myers RL. 1974. Groundwater tracing in karst terrain using phase T-4. Paper: *Annual Meeting, American Society of Microbiology*; Chicago, Ill.

[FDEP] Florida Department of Environmental Protection. 1996. Florida Administrative Code Chapter 62-600, Domestic Wastewater Facilities

_________. 1997. Memorandum to Mimi Drew, Director of the Division of Water Facilities through Richard Drew, Chief, Bureau of Wastewater Facilities, from Elsa A. Plotts, P.E., Administrator, FL DEP, Domestic Wastewater Section. Re: Recommendations of the UIC Class I Fluid Movement Workgroup.

_________. 1998. Risk Impact Statement: Phase II Revisions to Chapter 62-610, F.A.C. Docket No. 95-08R.

_________. 1999a. Miami-Dade Water and Sewer Department, North District Wastewater Treatment Plant Application to Construct/Operate/Abandon Class I, III, or V Injection Well System, Class I Injection Well Operation Permit for Injection Well No. IW-2N.

_________. 1999b. Florida Administrative Code Chapter 62-528, Underground Injection Control.

_________. 2001. Department of Environmental Protection Discharge Monitoring Report – Part A, South Beaches Wastewater Treatment Facility.

_________. 2002. Map of Class I injection well facilities and status

Florida Department of Regulation. 1989. St. Petersburg Public Utilities Department - Albert Whitted Waste Water Treatment Plant Application to Construct/Operate/Abandon Class I, III, or V Injection Well System.

Freeze RA and Cherry JA. 1979. *Ground Water*. Englewood Cliffs (NJ): Prentice-Hall.

Gerba CP and Bitton G. 1984. Microbial pollutants: their survival and transport pattern to groundwater. In: *Groundwater Pollution Microbiology*. Bitton G. and Gerba CP. eds. New York: John Wiley & Sons.

Gilbert RG, Gerba CP, Rice RC, Bouwer H, Wallis C, and Melnick JL. 1976. Virus and bacteria removal from wastewater by land treatment. *Applied and Environmental Microbiology*. 32: 333.

Gray TRG, and Parkinson D, eds. 1968. *The Ecology of Soil Bacteria*. University of Toronto Press.

Gupta RS. 1995. *Hydrology & Hydraulic Systems*. Prospect Heights (IL): Waveland Press, Inc.

Haberfeld JL. 1991. Hydrogeology of effluent disposal zones, Floridan Aquifer, South Florida. *Ground Water*. 29(2): 186-190.

Hickey JJ. 1982. *Hydrogeology and Results of Injection Tests at Waste Injection Test Sites In Pinellas County, Florida*. USGS WSP 2183. Reston (VA): USGS.

Horsley & Witten, Inc. 2000. *On-site Sewage Disposal Systems Pollutant Loading Evaluation – Test and Validation of the Indian River Lagoon Nitrogen Model*.

Howard PH, Boethling RS, Jarvis WF, Meylan WM, Michalenko EM, 1991. *Handbook of Environmental Degradation Rates*. Boca Raton (FL): CRC Press LLC.

Hutchinson CB. 1991. *Assessment of Hydrogeologic Conditions with Emphasis on Water Quality And Wastewater Injection, Southwest Sarasota And West Charlotte Counties, Florida*. USGS OFR 90-709. Reston (VA): USGS.

Hutchinson CB, Sanders GL, and Duerr AD. 1993. Underground Injection in Southwest Florida. Video Corresponding to above abstract.

Hutchinson CB and Trommer JT. 1992. *Model analysis of hydraulic properties of a leaky aquifer system, Sarasota County, Florida*. USGS WSP 2340. Reston (VA): USGS.

Hwang NHC, Hita CE. 1987. *Fundamentals of Hydraulic Engineering Systems*.2nd ed. Englewood Cliffs (NJ): Prentice Hall, Inc.

Idelovitch E, et al. 1979. *Dan Region Project, Groundwater Recharge With Municipal Effluent*. Tel Aviv: Mekorot Water Co.

Jenkins MP, Bowman M, Walker and Ghiorse W. 1997. Use of sentinel *Cryptosporidium parvum* oocysts to measure effects of passive manure storage on oocyst inactivation. Paper: *7th International Coccidiosis Conference, European Union COST 820 Workshop*; Keble College, Oxford, England.

Jordan EO. 1926. The changes in bacterial content of stored normal and typhoid feces. *Journal of Infectious Diseases*. 38:306.

Knochenmus LA and Thompson TH. 1991. *Hydrogeology and Simulated Development of the Brackish Ground-Water Resources in Pinellas County, Florida*. USGS WRI 91-4026. Reston (VA): USGS.

Knochenmus LA and Bowman G. 1998. *Transmissivity and water quality of water-producing zones in the intermediate aquifer system, Sarasota County, Florida*. USGS WRI 98-4091. Reston (VA): USGS.

Koerner EL, and Haws DA. 1979. *Long-Term Effects of Land Application of Domestic Wastewater: Vineland, New Jersey, Rapid Infiltration Site*. Report No. EPA-600/2-79-072. Washington (DC): US EPA.

Lukasiewiez J and Adams KS. 1996. *Hydrogeologic data and information collected from the Surficial and Floridian Aquifer Systems, Upper East Coast Planning Area*. Technical Publication WRE #337. West Palm Beach (FL): South Florida Water Management District.

Maliva RG and Walker CW. 1998. Hydrogeology of deep-well disposal of liquid wastes in Southwestern Florida, USA. *Hydrogeology Journal*. 6(4):538-548.

Martin GN, and Noonan MJ. 1977. Effects of domestic wastewater disposal by land irrigation on groundwater quality of the central Canterbury Plains. Technical. Pub. 7. Christchurch, New Zealand: Water and Soil Division, Ministry Of Works and Development.

Martin R, and Thomas A. 1974. An example of the use of bacteriophage as a groundwater tracer. *Journal of Hydrology*. 23:73-78.

Mawdsley JL, Brooks AE, and Merry RJ. 1996. Movement of the protozoan pathogen Cryptosporidium parvum through three contrasting soil types. *Biology of Fertile Soils*. 21(1):30-36.

McKinley RM. 2000. *Comments On 21-Oct-1998 Temperature And Radioactive Tracer Surveys From Miami-Dade Water And Sewer Department Disposal Well #I-2 And Subsequent Long-Term Gamma-Ray Activity Data From Monitor Well # FA-11 (Lower Zone)*.

McNeill DF. 2000. *A Review of Upward Migration of Effluent Related to Subsurface Injection of Miami-Dade Water and Sewer South District Plant*.

Medema GJ, Bahar M, and Schets FM. 1997. Survival of *Cryptosporidium parvum, Escherichia coli,* faecal enterococci and *Clostridium perfringens* in river water: influence of temperature and autochthonous microorganisms. *Water Science and Technology*. 35:249-252.

Meyer FW. 1984. Disposal of liquid wastes in cavernous dolostones beneath Southeastern Florida. In: Vol. 1 *Hydrogeology of Karstic Terrains, Case Histories*. International Association of Hydrogeologists.

________. 1989. *Hydrogeology, Ground-Water Movement, and Subsurface Storage in the Floridan Aquifer System in Southern Florida*. USGS Prof. Paper 1403-G. Reston (VA): USGS.

[MDSWAD] Miami-Dade Water and Sewer Authority Department. 1977. *Drilling and Testing of the Test-Injection Well for the Miami-Dade Water and Sewer Authority*. Contract no. S-153, Dade County, Florida, EPA Contract No. C120377020, Project No. 559-7601.

_________. 1991. *Drilling and Testing of 3 Injection Wells and 2 Monitoring Wells for the South District Regional Wastewater Treatment Plant*. Contract No. S-330.

_________. 1997. Engineering report, plugging and reconstruction of M.D.W.S., South District's multi-zone Monitor Well BZ-1.

Miller JA. 1997. Hydrogeology of Florida. In: *The Geology of Florida*. Randazzo AF and Jones DS, eds. Gainesville: University Press of Florida.

Montgomery JH. 2000. *Ground water Chemicals Desk Reference*. 3rd ed. Boca Raton (FL): CRC Press, LLC.

Noonan MJ, and McNabb JF. 1979. Contamination of Canterbury groundwater by viruses. In *The Quality and Movement of Groundwater in Alluvial Aquifers of New Zealand, Tech., Publ. 2*. Noonan MM, ed. Canterbury, New Zealand: Department of Agriculture and Microbiology, Lincoln College.

Paul JH, Rose JB, Brown J, Shinn EA, Miller S, and Farrah SR. 1995. Viral tracer studies indicate contamination of marine waters by sewage disposal practices in Key Largo, Florida. *Applied and Environmental Microbiology*. 61: 2230-2234.

Randazzo, A.F. and D.S. Jones. 1997. *The Geology of Florida*. Gainesville: University Press of Florida.

Reese RS. 1994. *Hydrogeology and the distribution and the origin of salinity in the Floridan Aquifer System, Southeastern Florida*. USGS WRI 94-4010. Reston (VA): USGS.

Reese RS and Memburg SJ. 1999. *Hydrogeology and the distribution of salinity in the Floridan Aquifer System, Palm Beach County, Florida*. USGS WRI 99-4061. Reston (VA): USGS.

Reese RS and Cunningham KJ. 2000. *Hydrogeology of the Gray Limestone Aquifer in Southern Florida*. USGS WRI 99-4213. Reston (VA): USGS.

Rehmann LL, Campbell and Welty C. 1999. Stochastic analysis of virus transport in aquifers. *Water Resources Research*. 35:1987-2006.

Robertson LJ, Campbell AT, and Smith HV. 1992. Survival of *Cryptosporidium parvum* oocysts under various environmental pressures. *Applied and Environmental Microbiology*. 58:3494-3500.

Rose JB, Haas CN, and Regli S. 1991. Risk assessment and control of waterborne giardiasis. *American Journal of Public Health*. 81(6):709-713.

Rose JB and Carnahan RP. 1992. *Pathogen removal by full scale wastewater treatment. A report to the Florida Department of Environmental Protection*. Tampa: University of South Florida.

Rose JB, Dickson LJ, Farrah SR, and Carnahan RP. 1996. Removal of pathogenic and indicator microorganisms by a full-scale water reclamation facility. *Water Research*. 30:2785.

Rose JB, Griffin DW, and Nicosia LW. 2000. Virus Transport from Septic Tanks to Coastal Waters. *Small Flows Quarterly*. 1(3).

Schaub SA, and Sorber CA. 1977. Virus and bacteria removal from wastewater by rapid infiltration through soil. *Applied and Environmental Microbiology*. 33:609-619.

Singh UP, Eichler GE, Sproul CR, Garcia-Bengochea JI. 1983. Pump testing Boulder Zone Aquifer, South Florida. *Journal of Hydraulic Engineering*. 109(8): 1152-1160.

Skilton H and Wheeler D. 1988. Bacteriophage tracer experiments in groundwater. *Journal of Applied Bacteriology*. 65:387-395.

Smith JJ, Howington JP, and McFeters GA. 1994. Survival, physiological response, and recovery of enteric bacteria exposed to a polar marine environment. *Applied and Environmental Microbiology*. 60:2977-2984.

Sobsey MD. 1983. Transport and fate of viruses in soils. In: *Microbial Health Considerations of Soil Disposal of Domestic Wastewater*. Cincinnati (OH): US EPA.

[SDWWTP] South District Wastewater Treatment Plant. 1998. *Monitoring Well Purging Report*. Report No. A.

Starr RC, Green TS, and Hull LC. 2001. *Evaluation of Confining Layer Integrity Beneath the South District Wastewater Treatment Plant, Miami-Dade Water and Sewer Department, Dade County, Florida*. Idaho Falls (ID): Idaho National Engineering and Environmental Laboratory, Geoscience Research Department.

Suthersan SS. 2002. *Natural and Enhanced Remediation Systems*. Boca Raton (FL): CRC Press, LLC.

Tibbals CH. 1990. *Hydrogeology of the Floridan Aquifer System in East-Central Florida*. USGS Prof. Paper 1403-E. Restong (VA): USGS.

Tihansky AB and Knochenmus LA. 2001. *Karst features and hydrogeology in West-central Florida – A field perspective*. USGS WRI 01-4011. Reston (VA): USGS.

Tuffey TJ, Hunter JV, and Matulewich WA. 1974. Zones of nitrification. *Bulletin of the American Water Research Association*. 10:1-10.

[US EPA] United States Environmental Protection Agency. 1992. *Ammonia Drinking Water Health Advisory*. US EPA, NTIS#: PB92-135425, ERIC#: D-273. Washington (DC): US EPA.

_________. 2000. *Drinking Water Standards and Health Advisories*. EPA 822-B-00-001. Washington (DC): US EPA.

_________. 2002. *Technical Fact Sheet on Nitrate/Nitrite*. Internet: http://www.epa.gov/safewater/dwh/t-ioc/nitrates.html.

[USGS] United States Geological Survey. 2000. *Distribution, Movement, and Fate of Nitrate in the Surficial Aquifer Beneath Citrus Groves, Indian River, Martin and St. Lucie Counties, Florida, 2000*. WRI 00-4057. Reston (VA): USGS.

Van Donsel DJ, Geldreich EE, and Clarke NA. 1968. Seasonal variations in survival of indicator bacteria in soil and their contribution to stormwater pollution. *Applied Microbiology*. 15:1362.

Vanderborght JP, and Billen G. 1975. Vertical distribution of nitrate concentration in interstitial water of marine sediments with nitrification and denitrification. *Limnology & Oceanography*. 20:953-961.

Vaughn J, and Landry EF. 1977. *Data Report: An Assessment of the Occurrence of Human Viruses in Long Island Aquatic Systems*. Rep. BNL 50787. Upton (NY): Brookhaven National Laboratory, Department of Energy and the Environment.

Walker C. 1997. Dupont/Pinellas Membrane Research Facility. *Application to Construct Class V, Group 8 Exploratory Injection Well*. Missimer International, Inc.

Walker MJ, Montemagno CD, and Jenkins MB. 1998. Source water assessment and nonpoint sources of acutely toxic contaminants: A review of research related to survival and transport of *Cryptosporidium parvum. Water Resources Research*. 34(12):3383-3392.

Yates MV and Yates SR. 1988. Modeling microbial fate in the subsurface environment. *CRC Critical Reviews in Environmental Control*. 17(4):307-344.

Yates MV, Stetzenbach LD, Gerba CP, and Sinclair NA. 1990. The effect of indigenous bacteria on virus survival in ground water. *Journal of Environmental Science and*

Health, Part A – Environmental Science and Engineering and Toxic and Hazardous Substance Control. 25:81-100.

York DW, Menendez P, and Walker-Coleman L. 2002. *Pathogens in reclaimed water: The Florida experience*. 2002 Water Sources Conference.

York DW and Walker-Coleman L. 1999. Is it time for pathogen standards? In: *Proceedings of the 1999 Florida Water Resources Conference. AWWA, FPCA, and FW&PCOA*; 1999 April 25–28; Tallahassee, Florida.

Xu HS, Roberts N, Singleton FL, Attwell RW, Grimes DJ, and Colwell RR. 1982. Survival and viability of nonculturable *Escherichia coli* and *Vibrio cholerae* in the estuarine and marine environment. *Microbial Ecology*. 8:313-323.