Comparative Assessment of Municipal Wastewater Disposal Methods in Southeast Florida

Frederick Bloetscher, James D. Englehardt, David A. Chin, Joan B. Rose, George Tchobanoglous, Vincent P. Amy, Sinem Gokgoz

ABSTRACT: A comparative assessment of the risks of three effluent disposal alternatives currently available to wastewater utilities in Southeast Florida is presented in this paper. The alternatives are: deep well injection and ocean outfalls following secondary treatment, and surface water (canal) discharges following secondary wastewater treatment, filtration and nutrient removal. Water quality data, relative to disposal of wastewater treatment plant effluent were gathered, along with water quality data on the receiving waters, from utilities. Comparisons and conclusions regarding potential health concerns associated with the three disposal alternatives are presented. The results indicated that health risks associated with deep wells were generally lower than those of the other two alternatives. The proximity of injection wells to aquifer storage and recovery wells was a determining factor relative to injection well risk. Urban ecological risks were also indicated to be lower, though impacts of urban water use/reuse to the Everglades were not studied. Additional data collection and analysis were recommended to understand the effects of wastewater management on the cycling of water, nutrients and other constituents on southeast Florida. In particular, it was recommended that monitoring of effluents for nitrosamines and pharmaceutically active substances be implemented on a broad scale. *Water Environ. Res.*, **77,** 480 (2005).

doi:10.2175/106143005X67395

Introduction

Regulatory, political and economic constraints have shaped wastewater management strategies in Southeast Florida. Three options are currently available: Class I injection wells, ocean outfalls and reclaimed water. Ocean outfalls were constructed in the early 1970s, with deep well construction starting after 1977, after canal discharges of secondary wastewater were found to impact the surface water environment. Both alternatives are less expensive due to the less stringent treatment requirements relative to canal discharges. Reuse has not generally been implemented in the region for several reasons. First, coastal Southeast Florida has access to the Florida Current, allowing for ocean outfall disposal, and an underlying geologically isolated formation, allowing for injection well disposal. Second, present regulations (Rule 62-610, Florida Administrative Code) require that reuse systems have a back-up disposal system capable of disposing of 100 percent of the reuse water in the event of wet weather, water quality concerns or adverse weather conditions. The back-up disposal systems would include injection wells and/or ocean outfalls, making the capital investment redundant. Third, lot sizes in Southeast Florida are generally small, limiting the amount of reuse water that can be disposed of relative to other, less densely developed areas of the state. For these reasons, the costs of retrofitting large residential areas with a reuse distribution system, as was done by the City of St. Petersburg in the 1980s (with 80 percent Federal matching grants) have been considered prohibitive. Of note, the Tampa Bay/St. Petersburg area does not have deep aquifers with the same degree of geologic isolation as in Southeast Florida, nor access to large ocean currents that would permit outfall disposal.

The use of class I injection wells is currently the principal alternative to ocean outfalls for disposing municipal wastewater effluent in Southeast Florida. In the past several years, elevated concentrations of ammonia and total Kjeldahl nitrogen, and depressed salinity relative to native water in the Floridan Aquifer, have been reported in monitoring wells in zones overlying the injection zone in Miami-Dade County (Englehardt, et al, 2001). This finding has raised concerns of the U.S. EPA and others regarding the extent of migration of injected water.

Risk analysis applications to water management planning have included assessment of risks associated with water distribution system and reclaimed water programs, particularly with regard to viral pathogens (Tanaka, et al., 1998, Gerba and Rose, 1983; Hutzler and Boyle, 1982; Hutzler and Boyle, 1980; Regli, et al., 1988; and Asano and Sakiji, 1990). The Southeast Florida Ocean Outfall Experiments I and II, conducted in a utility collaboration with the National Oceanic and Atmospheric Administration from 1987–1994, characterized ocean currents and plume movement associated with four of the six open ocean outfalls (Hazen and Sawyer, 1994 and Fergen, et al., 1994). Viral risks to beach bathers were studied by Cabelli, et al. (1979). Several authors have discussed health effects of sewage tainted surface waters, however virtually no data exists for this practice in Southeast Florida because no known wastewater effluents are disposed in this manner. Limited injection data exists, including limited information on the direction of flow in the injection zone (Kahout, et al., 1988). Reynolds (2000) found that the fate and transport of viruses and metals may create health concerns in groundwater recharge systems, so uncertainty in the flow of groundwater is a primary concern. Kent and Bentley (1985) reviewed perceived injection well failures, but no case was akin to those of South Florida. Garcia-Bengochea (1983) noted that the safety of injection wells in Florida was related to the ability of the aquifer to accept wastewater, and to the monitoring of adjacent groundwater. CH_2MHill (1993) modeled the transport of wastewater from injection wells in Pinellas County. However, buoyancy of the injected fresh wastewater in the brackish injection zone was not accounted for.

Generally, the use of risk assessment for comparison of water and wastewater management plans is hindered by variability among sites and the lack of general data and other information needed to compare technologies. Bayesian methods are well-developed for

analysis of limited information, and allow the explicit and rigorous integration of expert opinion and numeric data, in contrast with resampling methods (e.g., the bootstrap method, which have no subjective capability), and fuzzy logic methods (which have limited numeric data capability). However, Bayesian methods have been used principally for estimating parameters and associated uncertainty in models, and have not been used previously in comparative (or relative) evaluations of water and wastewater management plans.

A predictive Bayesian approach was proposed previously for the comparative evaluation of engineering and management alternatives (Englehardt 1997). The predictive method involves the use of unconditional, believed probability distributions for potential human health and ecological risk. Such distributions represent both uncertainty (due to information limitations) and variability (due to natural random variation). Resulting unconditional distributions become broader with decreasing levels of available information, giving estimates of risk directly, and avoiding the arbitrary assumption of confidence limits. Because the distributions are broader in general, their means are typically larger than the corresponding mean frequencies of occurrence. Therefore, these probabilities are termed *believed probabilities*, and are not interpreted as frequencies. Predictive Bayesian methods require no arbitrary assumption of confidence limits in interpretation. Results are more sensitive to the form of the distribution for the quantity of interest, and typically no account is taken of uncertainty in this form. However, this is an important advantage when there is theoretical or empirical basis for the form of the distribution, as it allows risk to be assessed from any level of available information.

In this research, indicators of the human health and urban ecological risks of available alternatives for disposal of treated municipal wastewater effluent in southeast Florida were evaluated on a relative basis (Englehardt, et al., 2001). Injection well disposal, ocean outfall disposal, and indirect reuse via surficial aquifer recharge (canal discharge) were considered. Because of the complexity of potential exposure paths, time scales, and population characteristics, and the need to avoid site-specificity in the assessment, a predictive Bayesian assessment of relative risks of the various alternatives was undertaken. Assessment of risks of reuse of treated effluent for irrigation of golf courses, residential lawns, and parks was outside of the scope of this study.

Methods

The project comprised two phases: data collection, and a comparative (relative) risk assessment based on the predictive Bayesian compound Poisson model proposed previously (Englehardt, 1997). The data collection step included review of data on the quality of wastewater effluents from different levels of treatment, and the quality of ground water at depths to approximately 3000 feet. Next, a conceptual model of the operating environment was developed for each disposal option, including regulatory constraints, hydrogeological and hydrological considerations, and potential pathways of health and ecological exposure. From these conceptual models, a probabilistic computer model was developed using the programming software package Matlab® Version 5.2, for computation of believed mean days of regulatory water quality violations in 30 years. Expert opinion was elicited for input the model developed using a modified Delphi Method. Once completed, the relative risks of the three wastewater treatment plant disposal alternatives were compared.

Interactions of the research team were conducted electronically and in three principal meetings. The team members and advisors (Appendix A) represented experience in risk analysis, reuse and disposal technology, geology, surface and groundwater fate and transport and epidemiology. At the first meeting, a conceptual model of the technological and environmental setting for wastewater disposal in Southeast Florida was constructed, including available wastewater treatment technologies, water quality regulations, hydrologic characteristics, and conventional and emerging wastewater constituents of concern. At the second meeting, the analysis of collected water quality data was presented, along with tree diagrams describing potential exposure pathways, and the conceptual model for the risk analysis was developed. At this meeting, risk was defined for the study in terms of the number and duration of water quality violations projected for each alternative. Professional judgment was elicited from team members in their areas of self-appraised expertise, based on assembled data, literature information, previous reports and regulations. A probabilistic assessment of risks of injection well disposal was developed, based on (a) published literature, (b) available data, and (c) expert opinion, relative to those of ocean outfall and canal aquifer recharge methods of treated-effluent disposal. Initial results were presented at a third meeting of the research team, and discussed for revision of both input and model form. Final input was then collected, and results computed for review by team members. All team members were interviewed regarding the final results, which were included in the final report and circulated all members for final review.

Water Quality Data Analysis. Data were collected from participating utilities and regulatory agencies, summarized, and analyzed. In addition, local and regional geologic data were collected. Data for effluents and ground waters were compared to drinking water standards. The majority of constituents were found in low concentration in the effluent in comparison to the receiving waters. Constituents found in higher concentrations in effluent included cyanide, nitrogen, phosphorous, color, odor, foaming agents, total trihalomethanes (THMs), biochemical oxygen demand (BOD), and total coliform count. In addition, treated effluents were somewhat higher in temperature and lower in pH, on average. Based on wastewater effluent analyses obtained, on average, treated effluents met both primary and secondary standards for drinking water, with the exceptions of primary standards for antimony and total coliform, and secondary standards for color, odor, TDS, and foaming agents.

System Conceptual Models. Conceptual models for each alternative, including potential exposure routes associated with each disposal method, are shown in Figures 1 through 3. For ocean outfalls, the assumption was that secondary treated and disinfected wastewater is discharged through existing outfalls at a depth of 90 feet, within the western edge of the Florida Current portion of the Gulfstream (Figure 1). Potential exposures occur to swimmers and fish that enter plumes from pipe leaks or the actual discharge. For canal discharges, exposure routes include direct canal affects on swimmers and fish, and impacts to consumers of water where the canal water is used for potable water supplies or may migrate toward water supply wells (Figure 2).

Injection wells have more complex potential exposure routes, including such remote possibilities as migration of secondary treated wastewater via leaks in the casing, and migration from the injection zone into surficial waters and the ocean. Migration can impact potable water supply wells, the ocean, canals and treated and untreated aquifer storage and recovery wells. The simplified tree diagrams in Figures 4 to 6 were developed from Figures 1 through 3 to show specific exposure routes and nodes for which water quality standards apply, based on the definition of risk adapted for the project.

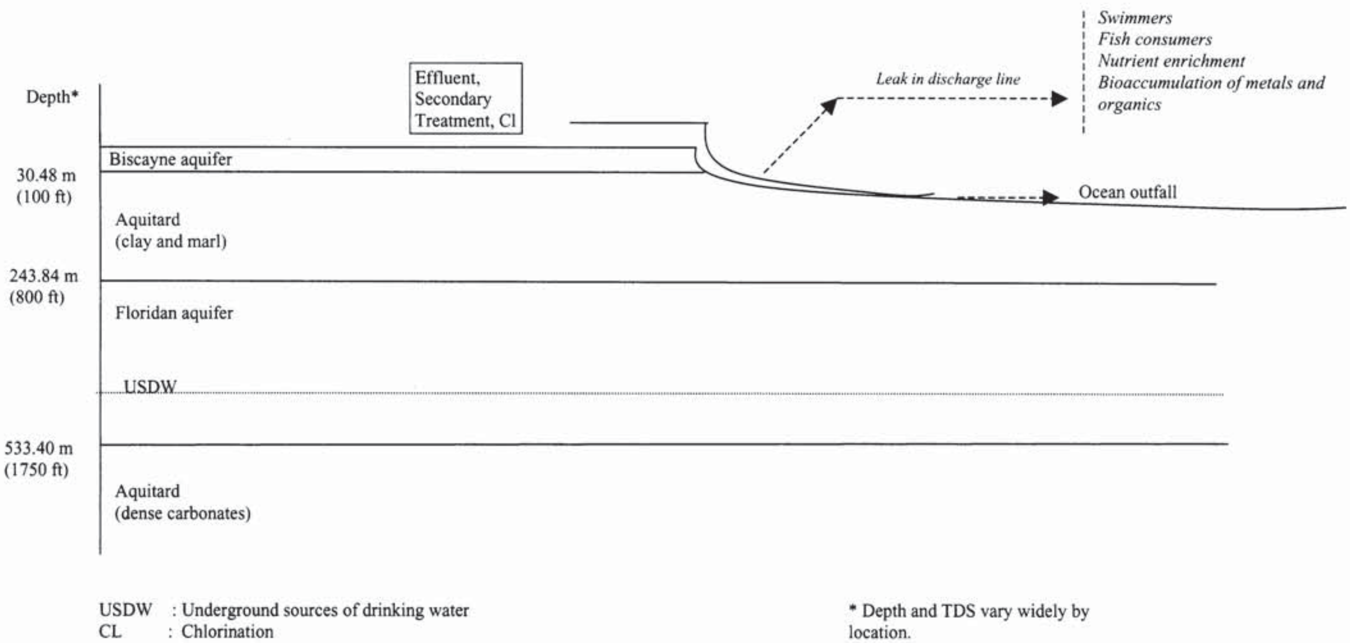

Figure 1—Ocean outfall disposal method.

The relative risk assessment was developed using several tracer constituents and comparing the relative likelihood of those indicators showing up in quantities that exceed the receiving water or drinking water standards, as appropriate. To select appropriate constituents that could be used to measure health risks, the following were evaluated: presence in wastewater; concentration higher in wastewater than ambient receiving waters; and potential for health impacts. Arsenic, NDMA, and *Cryptosporidium parvum* (for surface water) or rotavirus (for ground water) were selected by the team as indicators of human health risk based on concentrations in effluent and native waters, and on health effects; TKN was used as an indicator of potential eutrophication and ecological risk, considering current concern regarding nutrient loading impacts on the coastal ecosystem. It was not possible to conduct a quantitative assessment of risks associated with pharmaceutically active substances (PASs), because such compounds were still being identified chemically, concentrations in treated and natural waters were largely unknown, and environmental fates were uncertain.

Expert Opinion Elicitation. A modified Delphi elicitation of expert judgment was conducted electronically, to assess risks associated with the three disposal alternatives based on data collected for the region, applicable published literature information, and the experience of the research team. Because a predictive Bayesian approach was adopted for development of the model, the formal Delphi method was modified to reflect the variation in responses, rather than the consensus, of the experts. In a true Delphi process, responses outside the range of the central 50 percent of responses are eliminated prior to each iteration, with the ultimate goal of reaching a consensus likelihood of the group response. However, group decisions have been found to polarize opinions in the direction of lower or higher risk, and can tend to coalesce around the opinions of dominant members or otherwise change without a corresponding change in the underlying facts. While polarization can be a positive force for agreement when value or preference-based choices are made, it was minimized in this study so as not to artificially reduce the actual assessed uncertainty.

A questionnaire was prepared as a Microsoft Excel® Workbook, consisting of three sheets. Each sheet was dedicated to a method of disposal, corresponding to the three modified tree diagrams (Figures 4 to 6). The questionnaire posed a series of questions that followed exposure pathways from the point of discharge through to each point in the environment or water management system where a water quality standard existed. Health concerns, if any, would be expected to be associated with migration of effluent out of the injection zone, onto the shore from the ocean outfalls or into drinking water supplies from surface waters.

Each of these points represented a point of potential water quality violation due to the discharge alternative, and was represented by a node in the corresponding decision tree. For each node and each discharge alternative, the research team was asked four principal questions, worded as follows:

1. How many times in 30 years will the regulatory standard be exceeded at the receiving node? (One such exceedance event may last any number of days.)
2. What is your confidence in the numbers of exceedance events you entered? Please select low (L), medium (M) or high (H).
3. How many days will exceedance events last (minimum, mean, maximum)?
4. What is your confidence in the event sizes you entered? Please select from low (L), medium (M) or high (H).

For each node, the following three types of information were provided to the team members, for use in answering the questions posed: applicable effluent concentrations found in Southeast Florida effluent, as collected during the project; applicable actual or assumed regulatory standard; and relevant assumptions and supporting information, including studies such as the Southeast Florida Ocean Outfall Experiments I and II and the references noted at the end of this paper.

The questionnaire set was circulated electronically three times to develop input for the initial assessment presented at Meeting 3.

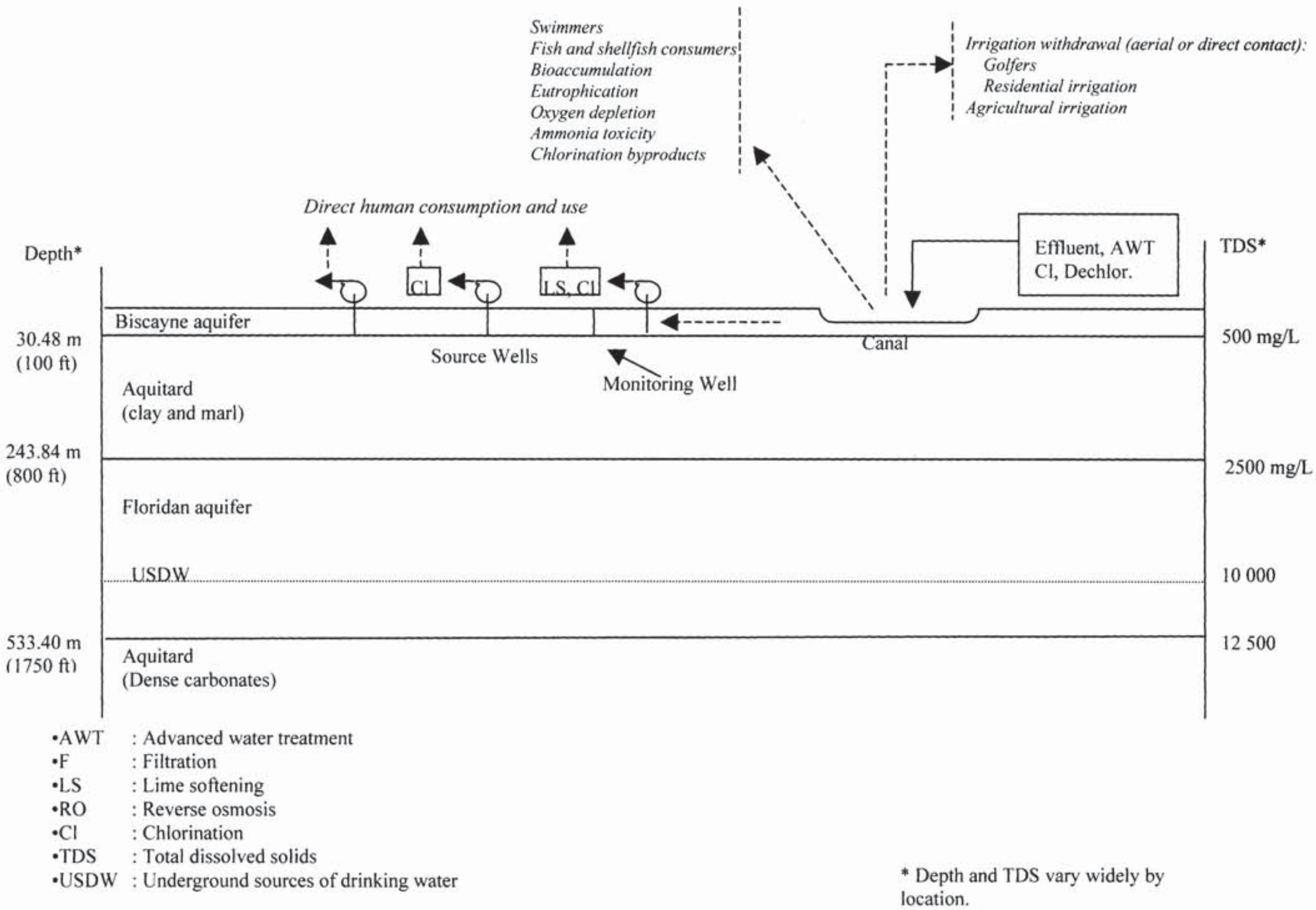

Figure 2—Canal discharge (aquifer recharge) method.

Team members were free to answer only those questions for which they felt they had expertise. A final iteration was conducted to refine input based on group discussion at Meeting 3, to develop input for the final assessment. After each iterative circulation of the questionnaires, geometric mean, weighted geometric mean, maximum, and minimum answers were tabulated and reported to the group for each question anonymously, along with reasons given for individual answers, if any. Reasons were particularly requested for answers outside the range of the group in general. Team members were asked to review and refine their individual opinions, based on the opinions and reasons of other members. Extreme answers were not removed, and differences of opinion were allowed.

Predictive Bayesian Assessment Model. A predictive Bayesian model of health risks associated with injection well, ocean outfall, and canal aquifer recharge disposal alternatives was constructed, based on available data and other information, including expert opinion. The approach involved the assignment of probability distributions, termed sampling distributions, to (a) the number of water quality standard violations expected in 30 years due to a discharge alternative, and (b) the expected magnitude (number of days) of individual violations. The Poisson distribution can be theoretically and empirically shown to predict the number of incidents, or infrequent events, over a period, and is widely used for this purpose. Therefore, the Poisson distribution was chosen as the sampling distribution for the number of violations in 30 years. The Pareto distribution generally represents the size of individual incidents (Englehardt, 1995).

Sampling distributions describe the natural variability in, for example, the number of violations. That is, they describe variability in the number of violations experienced among several imagined, repeated 30-year periods. The parameters of these distributions determine the location, scale, and shape of the distributions. When limited data are available to specify the parameters of these distributions, probability distributions, termed prior distributions, can then be assigned to the parameters themselves. Priors can be determined based on any combination of subjective or numeric information using Bayes Theorem. Because data were not available, model input consisted of professional judgment based on available data (e.g., effluent and water quality, hydraulic conductivity, and dispersion data collected during SEFLOE) and other local knowledge.

Predictive Bayesian methods involve finding the unconditional probability of the numbers and sizes of violations, given the forms of the sampling distributions, and the specified distributions for sampling distribution parameters. That is, uncertainty in the parameters was integrated into the sampling distributions, resulting in predictive distributions for the numbers and sizes of violations. Predictive Bayesian distributions evolve in shape systematically in response to information content. As predicted by information theory, predictive distributions are broader when less is known, becoming narrower as information increases, converging on the underlying sampling distribution of variability. The increased breadth of a predictive distribution given limited information accounts for uncertainty due to information limitations and/or variability. For

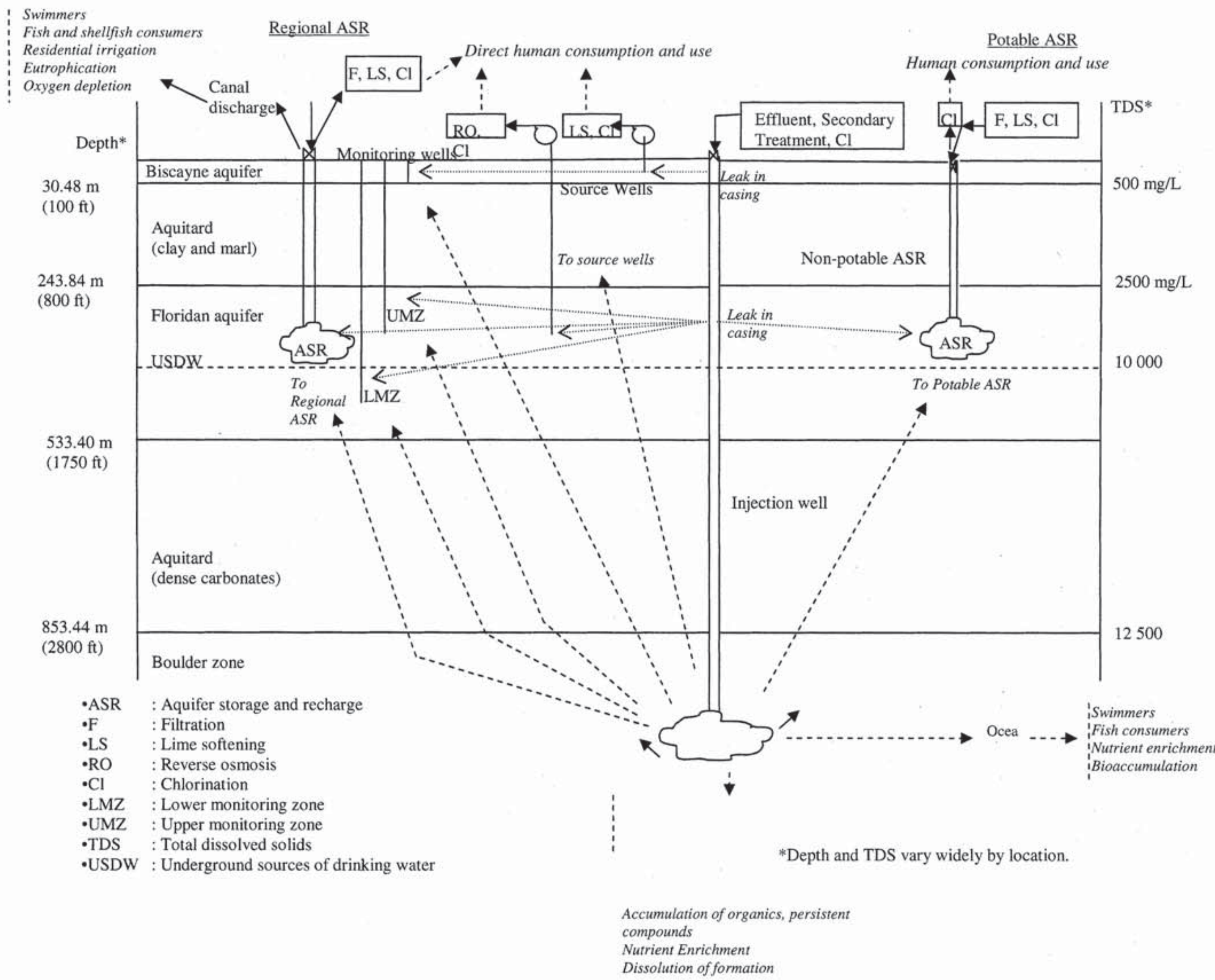

Figure 3—Deep well injection disposal method.

example, because an ocean outfall in one location may have one expected number of violations and the same outfall in another setting may have another, the distribution of the number of outfall violations is broader to account for this variability among locations.

Predictive distributions were used to compute distributions for total violations over the period, and for the number of days per violation. The model used to compute cumulative risks over a planning period is the compound Poisson model, in which the number of incidents over the period are described by a predictive Bayesian version of the Poisson distribution, and the severity of individual incidents is described by a predictive Bayesian version of the Pareto distribution. Distributions for total believed violation days were found by Monte Carlo simulation as the product of the number of violations times the length of individual violations, summed for all typed of violations to occur for a particular water quality constituent and discharge alternative. For example, surface water and drinking water violation days were summed for NDMA for ocean discharge. The resulting probability distributions give mean impacts, as well as variability around the mean. For example, the levels of total loss having a 5% exceedance probability for the planning period, are used for comparative purposes in this study.

Incident Number Assessment. For incident number, the distribution was based on the Poisson distribution of the number of incidents over a period, which can be written:

$$f_n(n) = \frac{\lambda^n}{n!} e^{-\lambda}, \quad n = 0, 1, 2, \ldots \tag{1}$$

in which λ is the mean number of incidents over the period. Team members were asked not to assess the expected number of occurrences in 30 years at zero for events considered highly unlikely, but rather to estimate such a mean in terms of small fractions. In a few cases where team members felt this to be impossible, a default value of 10^{-12} was used. Because responses regarding mean numbers of occurrences in 30 years varied over several orders of magnitude, geometric means (equal to the exponential of the average of the natural logs) were used as the measure of central tendency, to best represent the centroid of the probability mass for skewed data. The reported confidence of each team member was accounted for by counting high-confidence answers three times, moderate-confidence answers two times, and low-confidence answers once, in calculating a weighted geometric mean. This was

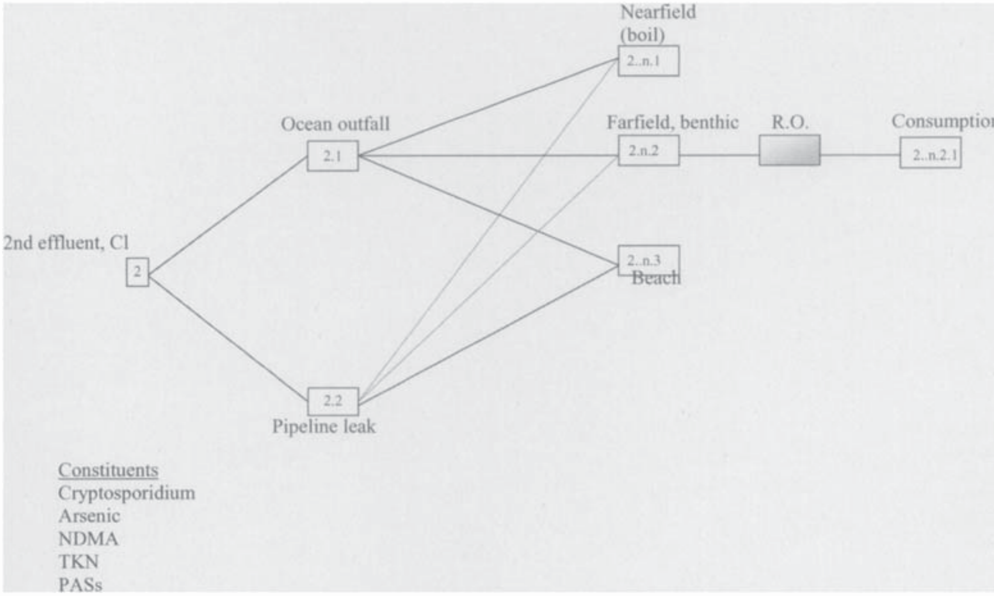

Figure 4—Ocean outfall trees used in Delphi survey.

seen as a straightforward way to account for differences in specific expertise areas of team members. Analysis of sensitivity to these weights was considered beyond the scope of this study. Because of the high level of uncertainly for the assessment, mean incident number was assumed log-normally distributed, and a predictive Bayesian version of the Poisson based on professional judgment was computed numerically. The standard deviation of this normal distribution was fixed at unity. That is, it was assumed that the weighted mean incident number could vary over plus or minus two orders of magnitude, with 95% confidence. Thus, team members fixed the mean of the prior, while associated variability was standardized according to the range of answers obtained. Predictive Bayesian distributions are as a rule robust to second-order decisions (e.g., magnitude of variability around prior means) regarding parameter estimation, relative to first order decisions (the form of the sampling distribution, the number of uncertain parameters, and the mean of the uncertain parameter vector) (Englehardt 1995). Thus, the prior for λ was assumed as:

$$f_\lambda(\lambda) = \frac{1}{\lambda\sigma\sqrt{2\pi}}\exp\left[-\frac{(\ln\lambda - \mu)^2}{2\sigma^2}\right] \quad 0 < \lambda \qquad (2)$$

in which $\sigma = 1$ and μ is the weighted geometric mean the mean of the natural logarithms of team members estimates of the average number of incidents over a 30 year planning period, weighted as described above to account for their confidence.

The likelihood function for data sampled from a Poisson distribution was set equal to one, because no available data were considered directly representative of the generalized scenario evaluated.

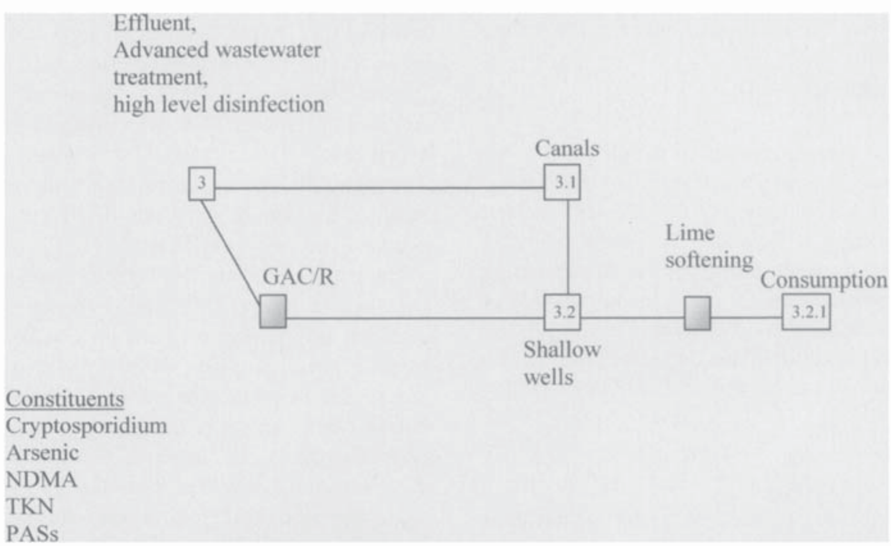

Figure 5—Surficial aquifer recharge trees used in Delphi survey.

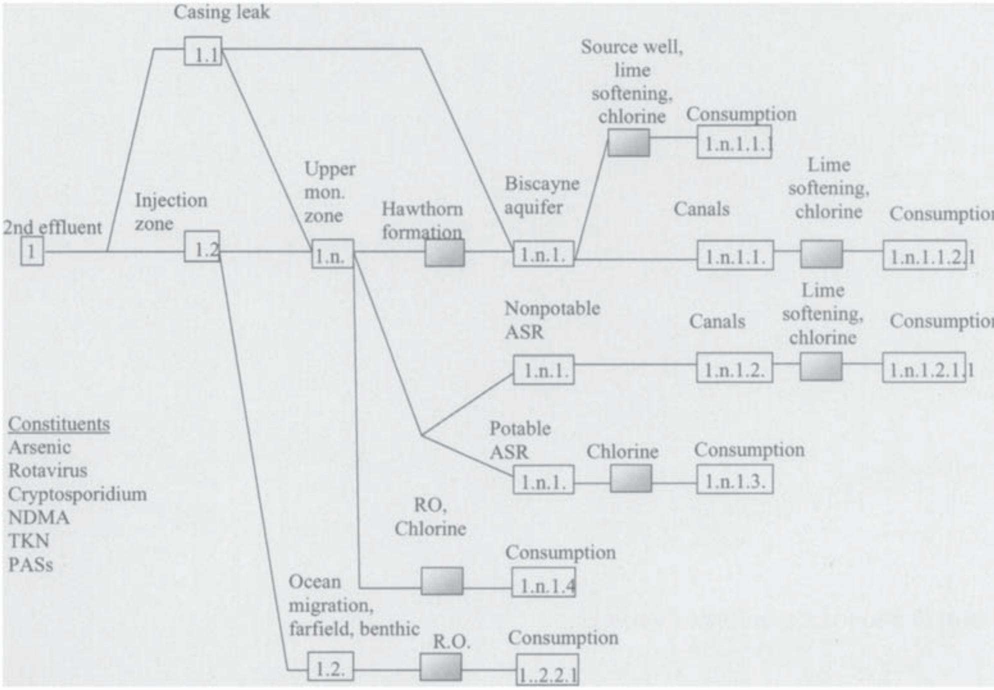

Figure 6—Deep well injection trees used in Delphi survey.

Therefore, the posterior was just equal to the prior, and the predictive distribution for n was evaluated numerically as:

$$p_n(n) = \int_0^{\infty} \frac{1}{\lambda\sigma\sqrt{2\pi}} \exp\left[-\frac{(\ln\lambda - \mu)^2}{2\sigma^2}\right] \frac{\lambda^n e^{-\lambda}}{n!} d\lambda,$$
$$n = 0,\ 1,\ \ldots\infty \tag{3}$$

The λ is integrated out of the equation leaving two variables. The parameters are σ, which was set to equal 1, and μ. The variable μ is defined above as the weighted geometric mean, which is $\hat{\lambda}$, meaning that $\mu = \ln \hat{\lambda}$.

Incident Size Assessment. For incident size, a predictive Bayesian version of the Pareto I distribution due to Englehardt (1995), that assumes a gamma prior distribution for the scale parameter, was used in the analysis. The Pareto distribution can be written:

$$f_z(z\,|\,a) = \frac{aZ_o^a}{z^{a+1}} \tag{4}$$

in which Z_0 is the location parameter equal to the minimum event size of interest, a is the scale parameter equal to the slope of the log-log plot of the CDF, and z is a realization of the size of an incident measured, in this case, in days of violation. The Pareto I distribution is very highly skewed, so much so that it is not distinguishable from the axes on linear plots. Therefore it is generally plotted on log-log scale. The parameter a largely determines the risk of large losses, and was considered uncertain. The conjugate prior for a is the gamma distribution, which can be written as follows:

$$f_a(a\,|\,\alpha,\beta) = \frac{\beta^{\alpha} a^{\alpha-1} e^{-a\beta}}{\Gamma(\alpha)}, \quad a \geq 0 \tag{5}$$

in which α and β are greater than zero. A predictive Bayesian version of the Pareto distribution can be found from the resulting posterior distribution, $f(a)$, as $\int f(z\,|\,a) f(a) da$ (Englehardt, 1995):

$$f_z(z\,|\,Z_0,\alpha,\beta,J,\overline{\ln z_j}) = \frac{\left(\beta + J\,\overline{\ln z_j} - J\ln Z_0\right)^{J+\alpha}(J+\alpha)}{z\left[\ln z + \beta + J\,\overline{\ln z_j} - (J+1)\ln Z_0\right]^{J+\alpha+1}} \tag{6}$$

The corresponding cumulative distribution function (CDF) is:

$$F_z(z) = 1 - \left(\frac{\beta + J\,\overline{\ln z_j} - J\ln Z_0}{\ln z + \beta + J\,\overline{\ln z_j} - (J+1)\ln Z_0}\right)^{J+\alpha} \tag{7}$$

For the assessment, the levels of confidence reported by individual research team members were assigned values of 1, 5, and 20, for low, medium and high confidence, respectively. These values represented increasingly narrow prior distributions for the mean incident size, as confidence increased. The value of the parameter α in Equation 4 was then specified as the average of the individual values. Weighted geometric means were then used to specify β using the relationship for the gamma prior that the (estimated) mean is equal to α/β. Because no numeric data were available for the generalized risk events assessed, the number of data points, J, was set equal to zero. Therefore, the remaining parameter representing the logarithmic mean of the data dropped out. The input values from the Delphi elicitation are shown in Table 1.

Parameter estimates in Table 1 were used to compute distributions of believed probability for the total number of days of violation in 30 years for each node indicated by team responses to have significant risk. Arsenic, NDMA, and *Cryptosporidium parvum* (surface water) or rotavirus (ground water), as indicators of human health risk, were computed at drinking water nodes. In the case of ocean outfall disposal, the drinking water node assumed reverse osmosis treatment and therefore was considered to have negligible risk. Therefore, human health risk for ocean outfall disposal was based on violation of surface water standards at the beach, assuming accidental ingestion of ocean water by bathers at

Table 1—Summary of Input Parameters.

Constituent	Event Description		Ocean Outfall		Surface Water	Injection Well		
			Wt Avg	Wt Avg	Wt Avg	0.2 Wt Avg	1 m Wt Avg	5mi Wt Avg
Arsenic	Event No.	$\mu=$	0.097	0.001	0.0022		0.00358	
		$\sigma=$	1	1	1		1	
	Event Size	$\hat{\hat{Z}}=$	0.789	0.2593	1.603		1.17	
		$\alpha=$	3	3	3		3	
		$\beta=$	3.80	11.57	1.87		2.56	
		$J=$	0	0	0		0	
		$\overline{\ln z_i}=$	arbitrary	arbitrary	arbitrary	arbitrary	arbitrary	arbitrary
		$Z_0=$	0.1	0.1	1		0.1	
Microbiological	Event No.	$\mu=$	1.1247	0.0153	0.0191		0.00128	
		$\sigma=$	1	1	1		1	
	Event Size	$\hat{\hat{Z}}=$	8.9406	2.3956	3.354		2.385	
		$\alpha=$	3	3	3		3	
		$\beta=$	0.34	1.25	0.89		1.26	
		$J=$	0	0	0		0	
		$\overline{\ln z_i}=$	arbitrary	arbitrary	arbitrary	arbitrary	arbitrary	arbitrary
		$Z_0=$	0.1	0.1	1	0.1		
NDMA	Event No.	$\mu=$	0.6178	0.1431	0.3798	1	0.00542	0.0015
		$\sigma=$	1	1	1	1	1	1
	Event Size	$\hat{\hat{Z}}=$	4.2317	0.8524	71.57	130	15.9324	1.01
		$\alpha=$	3	3	3	3	3	3
		$\beta=$	0.71	3.52	0.04	0.02	0.19	2.97
		$J=$	0	0	0	0	0	0
		$\overline{\ln z_i}=$	arbitrary	arbitrary	arbitrary	arbitrary	arbitrary	arbitrary
		$Z_0=$	0.1	0.1	1	0.1	0.1	0.1
TKN	Event No.	$\mu=$	0.6381	0.0521	0.0359		0.01299	
		$\sigma=$	1	1	1		1	
	Event Size	$\hat{\hat{Z}}=$	1.9422	0.786	1.509		9.5	
		$\alpha=$	3	3	3			
		$\beta=$	1.54	3.82	1.99		0.0	
		$J=$	0	0	0		0	
		$\overline{\ln z_i}=$	arbitrary	arbitrary	arbitrary	arbitrary	arbitrary	arbitrary
		$Z_0=$	0.1	0.1	1		0.1	

Note: $\hat{\hat{z}}$ is the estimated $\hat{\hat{z}}$ from the Delphi solicitation.

the beach. TKN, as an indicator of ecological risk, was computed at surface water nodes. Where more than one node contributed significantly to either health or ecological risk for a discharge alternative, a distribution for the total believed violation days was computed by Monte Carlo simulation. Distributions for the number of violations, and for the total violation days, were truncated at 10,950, the total number of days in 30 years. That is, a violation day was defined as a day during which one or more water quality violations occurred.

Results

Output of the probabilistic analysis included the believed number of days during which one or more violations of existing or assumed water quality standards for arsenic, microbes, n-nitrosodimethylamine (NDMA), and total Kjeldahl nitrogen (TKN) would occur, for comparative purposes. Distributions of total believed violation days over 30 years, such as the distribution for NDMA shown in Figures 7 and 8, were developed. The log-log scale, and irregular binning of Monte Carlo data were necessary due to the extreme skew of the distribution. Believed violation days computed from these results are shown in Table 2 and 3. These numbers were larger than the expected number of days on which violations would be expected to occur, because they reflected uncertainty in addition to inherent variability.

Results were based on several assumptions. In particular, the following assumptions were considered important:

1. Rapid vertical migration to the Upper Floridan Aquifer from deep injection wells in the Lower Floridan Aquifer was assumed, although evidence of effluent in the Upper Floridan near injection wells in Miami-Dade County could have been related to construction problems. This assumption is equivalent to assuming that the Floridan Aquifer will be "impacted" with water of generally much lower dissolved solids and inorganics, yet higher organics and nutrients, than native water of the aquifer;
2. As water is withdrawn from potable and non-potable ASR wells, salinity would be monitored, and withdrawals would stop if elevated levels were detected;
3. Because marine raw water for drinking was assumed to receive reverse osmosis treatment, human consumption risks were driven by accidental ocean water ingestion by bathers at the beach, as indicated by the probability of violating marine surface water

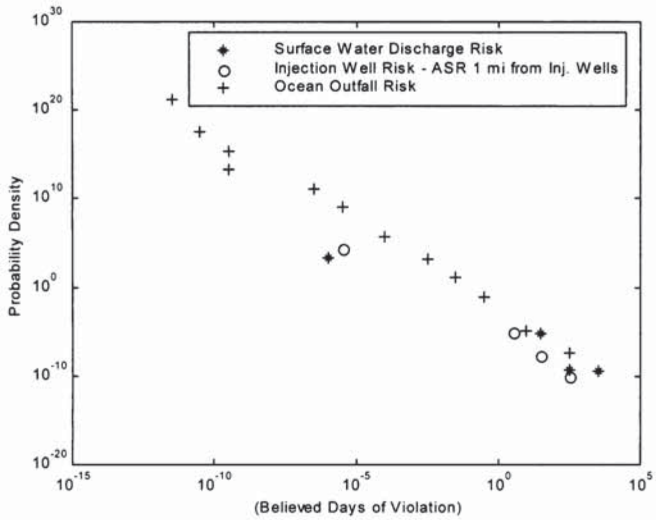

Figure 7—Comparison of predictive Bayesian risk assessment for injection wells, ocean outfalls and surface water (canal) discharges in South Florida for NDMA.

standards at the beach. In general, surface water standards are comparable to drinking water standards, though consumption of surface water in South Florida is probably three orders of magnitude less than consumption of drinking water.

4. Because specific surface water standards do not exist in Florida for NDMA and *Cryptosporidium parvum*, such standards were assumed based on California action levels (NDMA) and published dose-response data (*Cryptosporidium parvum*). Average exposure level was assumed to be 0.002 L/day, accounting for the fact that ocean and canal water are ingested only incidentally in small quantities, relative to drinking water. The existing specific surface water standard was used without modification, in keeping with the definition of risk assumed for the project.

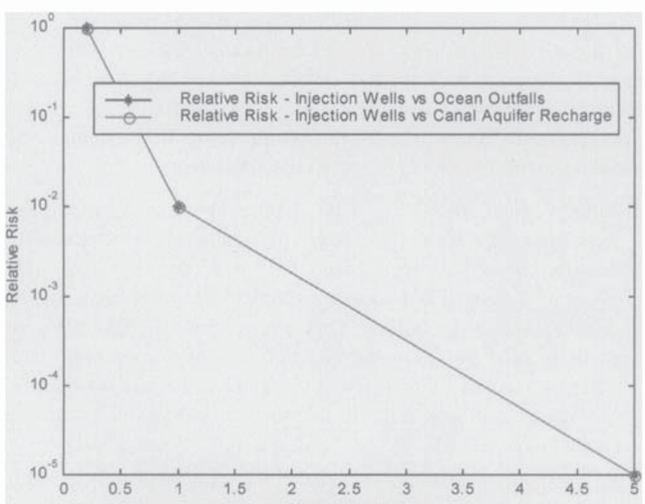

Figure 8—Comparison of predictive Bayesian risk assessment for deep wells at intervals of 1000 feet, 1 mile (used in the relative risk assessment) and 5 miles in Southeast Florida for NDMA comparing to ocean outfalls and canal aquifer recharge for sensitivity purposes.

Table 2—Comparison of Human Health Risk Indicators for Discharge Alternatives Not Considering Violation of the USDW.

Alternative Disposal Methods	Mean Believed Violation Days In 30 Years[1]		
	Arsenic	Microbial[2]	NDMA
Deep Well Injection (1 mi to ASR well)	1	0.1	1
Deep Well Injection (0.2 mi to ASR well)	n/a	n/a	50
Deep Well Injection (5 mi to ASR well)	n/a	n/a	0.002
Ocean Outfall	10	50	30
Canal Aquifer Recharge	0.3	5	40

[1] *Results reflect input developed on a relative basis, and should not be evaluated individually.*
[2] *Rotavirus for ground water nodes; Cryptosporidium parvum for surface water nodes.*

5. Levels of treatment assumed to be received by discharged effluent and treated drinking water varied according to regulatory requirements; only effluent released to surficial aquifers was assumed to receive advanced wastewater treatment (AWT). Surficial aquifer recharge by canal was assessed at lower risk than ocean outfalls for *Cryptosporidium parvum*, because the filtration step included in AWT treatment preceding canal discharge provides efficient removal of *C. parvum*, whereas present ocean outfall treatment does not include effluent filtration. Although *C. parvum* die-off in the ocean environment and dilution of the plume are normally effective exposure barriers, plumes are rapidly buoyant due to their lack of salinity and higher temperature, and may be advected with surface currents that are primarily wind-driven. Therefore, persistent on-shore wind conditions could result in inadequate dilution of ocean outfall plumes at the shore. As such, these assessed risks, and others evaluated in the study, depended upon the level of treatment assumed.
6. Consideration of cost, including those of treatment, was outside the scope of this study.

Assessed risks of injection well disposal were particularly dependent on the assumed distance to the nearest aquifer storage and recovery (ASR) well. To assess the sensitivity of results for injection wells to this assumption, team members with groundwater fate and transport expertise were asked to re-assess risks assuming

Table 3—Comparison of an Ecological Risk Indicator for three Discharge Alternatives Not Considering Violation of the USDW.

Alternative Disposal Methods	Mean Believed Violation Days In 30 Years[1]
	TKN
Deep Well Injection	10
Ocean Outfall	40
Canal Aquifer Recharge	100

[1] *Results reflect input developed on a relative basis, and should not be evaluated individually.*

Table 4—Relative Risk Indicators for The Disposal Alternatives Not Considering Violation of the USDW.

Alternative Disposal Methods	Relative Mean Believed Violation Days In 30 Years[1] (days/days)			
	Arsenic	Microbial	NDMA	TKN
Injection Well 0.2 mi/Ocean Outfall	n/a	n/a	10^{0}	n/a
Injection Well 1 mi/Ocean Outfall	10^{-1}	10^{-3}	10^{-2}	10^{-1}
Injection Well 5 mi/Ocean Outfall	n/a	n/a	10^{-5}	n/a
Injection Well (0.2 mi) /Canal Aquifer Recharge	n/a	n/a	10^{0}	n/a
Injection Well (1 mi) /Canal Aquifer Recharge	10^{0}	10^{-2}	10^{-2}	10^{-1}
Injection Well (5 mi) /Canal Aquifer Recharge	n/a	n/a	10^{-5}	n/a

[1] *Higher values represent higher potential frequency of, and/or higher uncertainty in the probability of, violating surface and drinking water standards given current treatment requirements, for a generalized scenario in Southeast Florida. Values less than one indicate a lower believed risk of violating standards for injection well disposal relative to the alternative.*

1000 foot and 5 mile injection well-ASR wells spacing in addition to the one mile spacing assumed in the first elicitation. The results of the sensitivity analysis are shown in Table 4 and Figure 8. As shown, distance has a major impact on risk–the closer the wells, the greater the risk.

Because assessing the risks of site-general scenarios involves generalized assumptions and corresponding uncertainties, the risk assessment results shown in Tables 2 and 3 are considered significant in terms of orders of magnitude only. In addition, results should be evaluated on a relative, not absolute, basis. Relative, or comparative, risks were defined as the ratio of believed violation days (that is, the mean of the believed probability distribution for violation days) for injection well disposal to those for each of the alternatives, and are shown in Table 4. Also, while comparisons between alternatives are valid, comparisons between constituents are not intended. In particular, arsenic risks may be more conservative because the existing surface water standard for arsenic in Florida mirrors the drinking water standard, whereas surface water standards assumed for *C. parvum* and NDMA were based on an assumed lower exposure rate relative to drinking water. Had surface water standards existed for *C. parvum* and NDMA, arsenic risks would presumably have been lower than other risks because average arsenic concentrations in the effluent samples analyzed were lower than either existing or proposed surface and drinking water standards, in contrast with other constituents. While assessed arsenic risks are not lower in Tables 2 and 3, the relative risks of arsenic are shown to be comparable among discharge alternatives in Table 4, as expected.

Conclusions and Recommendations

Human health risk indicators evaluated in this study were lower, in general, for injection well disposal, due to natural barriers between the injection point and population centers. Further, while there are no zero-risk options, all discharge alternatives evaluated are permitted under current regulations (though revised rules have been proposed for injection well disposal). It should be noted, however, that injection well disposal causes potential changes in Floridan Aquifer water quality in the vicinity of injection wells. As such, potential risks of injection well disposal were considered primarily associated with potential migration of effluent to ASR wells in the vicinity of injection. Of note, water from potable ASR wells is withdrawn for distribution without further treatment beyond chlorination. In addition, large-flow ASR wells are proposed to provide water for the Everglades restoration. Assessed risks were low partly due to the assumption that any contamination of ASR wells would be detected during operational withdrawals and that withdrawals would then stop.

Ecological risks within the three urban counties studied were assessed lower for injection wells; however, this finding may only apply to the urban area studied. When potential benefits and impacts to the Everglades system and to the near-shore estuarial and marine ecosystems are considered along with exponential growth in urban water demand, urban water reuse may be favored. Large-scale regional treatment facilities in Miami-Dade, Broward and Palm Beach Counties, and a number of smaller plants, routinely dispose nearly 0.5 billion gallons per day, significant in comparison with the 1.6 BGD proposed to be recovered through ASR technology to restore the Everglades. All wastewater discharges of this magnitude may impact hydrologic and chemical cycling. In particular, flows of water, carbon, oxygen, nutrients, and many life-sensitive constituents of lesser concentration through the ecosphere are affected, and should be considered in further studies. For example, indirect and direct effects on freshwater and chemical releases to estuarine environments should be considered, in the context of previous substantial alterations in such flows through large-scale drainage in South Florida, and in the context of the current Everglades restoration.

Although PASs and other emerging wastewater constituents were not able to be evaluated quantitatively, the potential risks associated with these compounds require further study. Statistical pilot monitoring programs are needed, as are studies of the fate of PASs in the surface and subsurface environment and in treatment systems. Estrogens may be useful as an indicator in future risk assessments, because the compounds and their health effects are measurable.

Health and ecological risks represent only one aspect of the sustainability of wastewater effluent discharge alternatives. Costs, and ecological impacts to the drainage basin at large, are also important considerations. As population pressures increase, reuse will be increasingly important part of an integrated wastewater management strategy. Therefore, treatment plant design will be driven by the need to remove non-conventional and emerging wastewater constituents, and research is needed now as a basis for future design.

Results presented reflect the uncertainty in relative risk between site-general alternatives quantitatively, convolving multiple, interrelated probabilities related to complex exposure pathways and fate effects. Limitations to the results presented are related to the scope and site-general nature of the assessment, the currently limited implementation and regulation of ASR technology, uncertainties associated with emerging wastewater constituents of concern such as NDMA and PASs, and associated assumptions for the assessment. Further, the assessment was based on professional judgment, using current data for the region, applicable published literature information, and the experience of the research team. Nevertheless, the results are considered important for water management planning and as a basis for further studies. Continued study of reclaimed water opportunities, the environmental risks of septic systems in coastal area and impact of the discharge to surface water bodies

(is) recommended for areas of different hydrologic and marine environments.

Appendix A. Team Members and External Advisors

Team Members.

James D. Englehardt, Ph.D., P.E., Professor (Principal Investigator)
University of Miami, Coral Gables, FL
Vincent P. Amy, P.G., Consulting Geologist
Tequesta, FL
Frederick Bloetscher, Ph.D., P.E., President
Public Utility Management and Planning Services, Inc., Hollywood, FL
David A. Chin, Ph.D., P.E., Professor
University of Miami, Coral Gables, FL
Sinem Gokgoz, Doctoral Student
Georgia Institute of Technology, Atlanta, GA
Joan B. Rose, Ph.D., Professor
Michigan State University, East Lansing, MI
Helena Solo-Gabriele, Ph.D., P.E., Associate Professor
University of Miami, Coral Gables, FL
George Tchobanoglous, Ph.D., P.E., Professor Emeritus
University of California–Davis, CA

External Advisors.

Robert K. Bastian, Senior Environmental Scientist
U.S. Environmental Protection Agency, Washington, DC
Pat Davis, P.E., Vice President
Hazen and Sawyer, Hollywood, FL
Jeffrey Lehnen, P.G., Group Leader, Water Resources
CH2M Hill, Gainesville, FL
John Proni, Ph.D., Director Remote Sensing Division
Atlantic Oceanographic and Meteorological Lab.
National Oceanographic and Atmospheric Administration
Miami, FL 33149

Acknowledgments

This study was made possible through support of the Florida Water Environment Association Utility Council. Bob Bastian, Pat Davis, Lora Fleming, Helena Solo-Gabriele, Tony Hui, Jeff Lehnen and John Proni are thanked for their input into the project.

Submitted for publication August 16, 2003; revised manuscript submitted June 23, 2004; accepted for publication July 12, 2004.

The deadline to submit Discussions of this manuscript is January 15, 2006.

References

Asano, T. Sakaji, R. (1990), Virus Risk analysis in Wastewater Reclamation and Reuse. In *Chemical Water and Wastewater Treatment*, H.H. Hahn and R. Klute (Eds.), Springer-Verlag, Berlin, Heidelberg, Germany.

Cabelli, V.J.; DuFour, A.P.; Levin, M.A.; McCabe, L.J.; Haberman, P.W. (1979), "Relationship of Microbial Indicators to Health Effects at Marine Bathing Beaches," *Am. J. Public Health*, Vol. **69**.

CH_2MHill (1993), *Environmental Risk and Geochemcial Analysis Related to the City of St. Petersburg's Underground Injection and Monitoring System*, CH_2MHill, Tampa, FL.

Englehardt, James D. (1995), "Predicting Incident Size from Limited Information," *Journal of Environmental Engineering*, Vol. **121**, No. 6, pp. 445–464.

Englehardt, J. (1997) "Bayesian Benefit-Risk Analysis for Sustainable Process Design," *Journal of Environmental Engineering*, vol. **123**, no.1, pp. 71–79.

Englehardt, James D.; Amy, Vincent P.; Bloetscher, Frederick; Chin, David A.; Fleming, Lora E.; Gokgoz, Sinem; Solo-Gabriele, Helena; Rose, Joan B.; Tchobanoglous, George (2001), Comparative Assessment of Human and Ecological Impacts for Municipal Wastewater Disposal Methods in Southeast Florida: Deep Wells, Ocean Outfalls, and Canal Discharges, University of Miami, Coral Gables, FL.

Fergen, Robert; Cooke, J. Phillip; Huang, Hening (1994), "An Overview of the Southeast Florida Ocean Outfall Experiment (SEFLOE) Phase II Study," MTS 94 Proceedings, Marine Technology Society, Washington D.C., pp. 748–754.

Florida Administrative Code, Chapters 62-302, 62-600, 62-610, 62-528, and 62-4, www.dep.state.fl.us/ogc/documents/rules/mainrule.htm.

Garcia-Bengochea, J.I. (1983), "Protecting Water Supply Aquifers in Areas Using Deep-Well Wastewater Disposal," *Journal–American Water Works Association*, Vol. **75**, No. 6, pp. 288–291.

Gerba, C.P. Rose, J. (1983), "Estimating Viral Disease Risk from Drinking Water," In *Comparative Environmental Risk Assessment*, C.R. Cothern (Ed.), Lewis Publishers, Boca Raton, FL.

Haile, Robert W.; Witte, John S.; Gold, Mark; Cressey, Ron; McGee, Charles; Millikan, Robert C.; Glasser, Alice; Harawa, Nina; Ervin, Carolyn; Harmon, Patricia; Harper, Janice; Dermand, John; Alamillo, James; Barrett, Kevin; Nides, Mitchell; Wang, Guang-yu (1999), "The Health Effects of Swimming in Ocean Water Contaminated by Storm Darin Run-off, Epidemiology, Vol. **10**, pp. 355–363.

Hazen and Sawyer (1994), Southeast Florida Ocean Outfall Experiment Report, Hollywood, FL.

Hutzler and Boyle (1982), "Risk Assessment in Water Reuse," In *Water Reuse*. E.J. Middlebrooks (Ed.), Ann Arbor Science Publishers, Ann Arbor, MI.

Hutzler and Boyle (1980), "Wastewater Risk Assessment," *Journal of Environmental Engineering*, Vol. **106**, EE5, p. 919.

Kahout, F.A.; Meisler, H.; Meyer, F.W.; Johnston, R.H.; Leve, G.W.; Wait, R.L. (1988), "Chapter 23 - Hydrogeology of the Atlantic Continental Margin," in *The Geology of North America, Volumes 1–2*, The Geological Society of America, p. 463–479.

Kaufman, Matthew I. (1973), "Surface Wastewater Injection, Florida," *Journal of the Irrigation and Drainage Division, ASCE*, Vol. **99**, No. IRI, pp. 53–70.

Kent, Robert T. Bentley, Michael E. (1985), "Risk Assessment of Deep Well Injection Systems," in *Proceedings: Second Canadian/American Conference on Hydrogeology* - Hitchon, Brian and Trudell, Mark R., editors. National Water Well Association, Dublin OH.

Regli, S., Pore, J.B.; Haas, C.N.: Gerba, C.P. (1991), "Modeling the Risk from *Giardia* and Viruses in Drinking Water," *Journal of the American Water Works Association*, Vol. **83**, No. 11, pp 76+.

Reynolds, Kelly A. (2000), "Human Viruses Found in Groundwater Recharge Sites," *Water Conditioning and Purification*, Vol. **42**, No. 2, pp. 148–150.

Tanaka, H.; Takashi, A.; Schroeder, E.D., Tchobanoglous, G. (1998), "Estimating the Safety of Wastewater Reclamation and Reuse using Enteric Virus Monitoring Data," *Water Environment Research*, Vol. **70**, No. 1, pp. 39–51.