# Chapter 3: Establishing treatment system performance requirements


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## 3.1 Introduction

This chapter outlines essential steps for characterizing wastewater flow and composition and provides a framework for establishing and measuring performance requirements. Chapter 4 provides information on conventional and alternative systems, including technology types, pollutant removal effectiveness, basic design parameters, operation and maintenance, and estimated costs. Chapter 5 describes treatment system design and selection processes, failure analysis, and corrective measures.

This chapter also describes methods for establishing and ensuring compliance with wastewater treatment performance requirements that protect human health, surface waters, and ground water resources. The chapter describes the characteristics of typical domestic and commercial wastewaters and discusses approaches for estimating wastewater quantity and quality for residential dwellings and commercial establishments. Pollutants of concern in wastewaters are identified, and the fate and transport of these pollutants in the receiving environment are discussed. Technical approaches for establishing performance requirements for onsite systems, based on risk and environmental sensitivity assessments, are then presented. Finally, the chapter discusses performance monitoring to ensure sustained protection of public health and water resources.

## 3.2 Estimating wastewater characteristics

Accurate characterization of raw wastewater, including daily volumes, rates of flow, and associated pollutant load, is critical for effective treatment system design. Determinating treatment system performance requirements, selecting appropriate treatment processes, designing the treatment system, and operating the system depends on an accurate assessment of the wastewater to be treated.

There are basically two types of onsite system wastewaters—residential and nonresidential. Single-family households, condominiums, apartment houses, multifamily households, cottages, and resort residences all fall under the category of residential dwellings. Discharges from these dwellings consist of a number of individual waste streams generated by water-using activities from a variety of plumbing fixtures and appliances. Wastewater flow and quality are influenced by the type of plumbing fixtures and appliances, their extent and frequency of use, and other factors such as the characteristics of the residing family, geographic location, and water supply (Anderson and Siegrist, 1989; Crites and Tchobanoglous, 1998; Siegrist, 1983).

A wide variety of institutional (e.g., schools), commercial (e.g., restaurants), and industrial establishments and facilities fall into the nonresidential wastewater category. Wastewater-generating activities in some nonresidential establishments are similar to those of residential dwellings. Often, however, the wastewater from nonresidential establishments is quite different from that from of residential dwellings and should be characterized carefully before Onsite Wastewater Treatment System (OWTS) design. The characteristics of wastewater generated in some types of nonresidential establishments might prohibit the use of conventional systems without changing wastewater loadings through advanced pretreatment or accommodating elevated organic loads by increasing the size of the subsurface wastewater infiltration system (SWIS). Permitting agencies should note that some commercial and large-capacity septic systems (systems serving 20 or more people, systems serving commercial facilities such as automotive repair shops) might be regulated under USEPA's Class V Underground Injection Control Program (see **http://www.epa.gov/safewater/uic/classv.html**).

In addition, a large number of seemingly similar nonresidential establishments are affected by subtle and often intangible influences that can cause significant variation in wastewater characteristics. For example, popularity, price, cuisine, and location can produce substantial variations in wastewater flow and quality among different restaurants (University of Wisconsin, 1978). Nonresidential wastewater characterization criteria that are easily applied and accurately predict flows and pollutant loadings are available for only a few types of establishments and are difficult to develop on a national basis with any degree of confidence. Therefore, for existing facilities the wastewater to be treated should be characterized by metering and sampling the current wastewater stream. For many existing developments and for almost any new development, however, characteristics of nonresidential wastewaters should be estimated based on available data. Characterization data from similar facilities already in use can provide this information.

## 3.3 Estimating wastewater flow

The required hydraulic capacity for an OWTS is determined initially from the estimated wastewater flow. Reliable data on existing and projected flows should be used if onsite systems are to be designed properly and cost-effectively. In situations where onsite wastewater flow data are limited or unavailable, estimates should be developed from water consumption records or other information. When using water meter readings or other water use records, outdoor water use should be subtracted to develop wastewater flow estimates. Estimates of outdoor water use can be derived from discussions with residents on car washing, irrigation, and other outdoor uses during the metered period under review, and studies conducted by local water utilities, which will likely take into account climatic and other factors that affect local outdoor use.

Accurate wastewater characterization data and appropriate factors of safety to minimize the possibility of system failure are required elements of a successful design. System design varies considerably and is based largely on the type of establishment under consideration. For example, daily flows and pollutant contributions are usually expressed on a per person basis for residential dwellings. Applying these data to characterize residential wastewater therefore requires that a second parameter, the number of persons living in the residence, be considered. Residential occupancy is typically 1.0 to 1.5 persons per bedroom; recent census data indicate that the average household size is 2.7 people (U.S. Census Bureau, 1998). Local census data can be used to improve the accuracy of design assumptions. The current onsite code practice is to assume that maximum occupancy is 2 persons per bedroom, which provides an estimate that might be too conservative if additional factors of safety are incorporated into the design.

For nonresidential establishments, wastewater flows are expressed in a variety of ways. Although per person units may also be used for nonresidential wastewaters, a unit that reflects a physical characteristic of the establishment (e.g., per seat, per meat served, per car stall, or per square foot) is often used. The characteristic that best fits the wastewater characterization data should be employed (University of Wisconsin, 1978).

When considering wastewater flow it is important to address sources of water uncontaminated by wastewater that could be introduced into the treatment system. Uncontaminated water sources (e.g., storm water from rain gutters, discharges from basement sump pumps) should be identified and eliminated from the OWTS. Leaking joints, cracked treatment tanks, and system damage caused by tree roots also can be significant sources of clear water that can adversely affect treatment performance. These flows might cause periodic hydraulic overloads to the system, reducing treatment effectiveness and potentially causing hydraulic failure.

## 3.3.1 Residential wastewater flows

### Average daily flow

The average daily wastewater flow from typical residential dwellings can be estimated from indoor water use in the home. Several studies have evaluated residential indoor water use in detail (Anderson and Siegrist, 1989; Anderson et al., 1993; Brown and Caldwell, 1984; Mayer et al., 1999). A summary of recent studies is provided in table 3-1. These studies were conducted primarily on homes in suburban areas with public water supplies. Previous studies of rural homes on private wells generally indicated slightly lower indoor water use values. However, over the past three decades there has been a significant increase in the number of suburban housing units with onsite systems, and it has recently been estimated that the majority of OWTSs in the United States are located in suburban metropolitan areas (Knowles, 1999). Based on the data in table 3-1, estimated average daily wastewater flows of approximately 50 to 70 gallons per person per day (189 to 265 liters per person per day) would be typical for residential dwellings built before 1994.

In 1994 the U.S. Energy Policy Act (EPACT) standards went into effect to improve water use efficiency nationwide. EPACT established national flow rates for showerheads, faucets, urinals, and water closets. In 2004 and again in 2007 energy use standards for clothes washers will go into effect, and they are expected to further reduce water use by those appliances. Homes built after 1994 or retrofitted with EPACT-efficient fixtures would have typical average daily wastewater flows in the 40 to 60 gallons/person/day range. Energy- and water-efficient clothes washers may reduce the per capita flow rate by up to 5 gallons/person/day (Mayer et al., 2000).

Of particular interest are the results of the Residential End Uses of Water Study (REUWS), which was funded by the American Water Works Association Research Foundation (AWWARF) and 12 water supply utilities (Mayer et al., 1999). This study involved the largest number of residential water users ever characterized and provided an evaluation of annual water use at 1,188 homes in 12 metropolitan areas in North America. In addition, detailed indoor water use characteristics of approximately 100 homes in each of the 12 study areas were evaluated by continuous data loggers and computer software that identified fixture-specific end uses of water. Table 3-2 provides the

**Table 3-1. Summary of average daily residential wastewater flows[a]**

| Study | Number of residences | Study duration (months) | Study average (gal/pers/day)[b] | Study range (gal/pers/day) |
|---|---|---|---|---|
| Brown & Caldwell (1984) | 210 | | 66.2 (250.6)[b] | 57.3–73.0 (216.9–276.3)[b] |
| Anderson & Siegrist (1989) | 90 | 3 | 70.8 (268.0) | 65.9–76.6 (249.4–289.9) |
| Anderson et al. (1993) | 25 | 3 | 50.7 (191.9) | 26.1–85.2 (98.9–322.5) |
| Mayer et al. (1999) | 1188 | 1[c] | 69.3 (262.3) | 57.1–83.5 (216.1–316.1) |
| Weighted Average | 153 | | 68.6 (259.7) | |

[a] Based on indoor water use monitoring and not wastewater flow monitoring.
[b] Liters/person/day in parentheses.
[c] Based on 2 weeks of continuous flow monitoring in each of two seasons at each home.

**Table 3-2. Comparison of daily per capita indoor water use for 12 study sites**

| Study Site | Sample size (number of houses) | Mean daily per capita indoor use (gal/pers/day)[a] | Median daily per capita indoor use (gal/pers/day)[a] | Standard deviation of per capita indoor use (gal/pers/day)[a] |
|---|---|---|---|---|
| Seattle, WA | 99 | 57.1 | 54.0 | 28.6 |
| San Diego, CA | 100 | 58.3 | 54.1 | 23.4 |
| Boulder, CO | 100 | 64.7 | 60.3 | 25.8 |
| Lompoc, CA | 100 | 65.8 | 56.1 | 33.4 |
| Tampa, FL | 99 | 65.8 | 59.0 | 33.5 |
| Walnut Valley Water District, CA | 99 | 67.8 | 63.3 | 30.8 |
| Denver, CO | 99 | 69.3 | 64.9 | 35.0 |
| Las Virgenes Metropolitan Water District, CA | 100 | 69.6 | 61.0 | 38.6 |
| Waterloo & Cambridge, ON | 95 | 70.6 | 59.5 | 44.6 |
| Phoenix, AZ | 100 | 77.6 | 66.9 | 44.8 |
| Tempe & Scottsdale, AZ | 99 | 81.4 | 63.4 | 67.6 |
| Eugene, OR | 98 | 83.5 | 63.8 | 68.9 |
| 12 study sites | 1188 | 69.3 (316.5)[b] | 60.5 (289.0)[b] | 39.6 (149.9)[b] |

[a] Multiply gallons/person/day by 3.875 to obtain liters/person/day.
[b] Liters/person/day in parentheses.

Source: Mayer et al., 1999.

average daily per capita indoor water use by study site for the 1,188 homes. The standard deviation data provided in this table illustrate the significant variation of average daily flow among residences. The median daily per capita flow ranged from 54 to 67 gallons/person/day (204 to 253 liters/person/day) and probably provides a better estimate of average daily flow for most homes given the distribution of mean per capita flows in figure 3-1 (Mayer et al., 2000). This range might be reduced further in homes with EPACT-efficient fixtures and appliances.

### Individual activity flows

Average daily flow is the average total flow generated on a daily basis from individual wastewater-generating activities in a building. These activities typically include toilet flushing, showering and bathing, clothes washing and dishwashing, use of faucets, and other miscellaneous uses. The average flow characteristics of several major residential water-using activities are presented in table 3-3. These data were derived from some 1 million measured indoor water use events in 1,188 homes in 12 suburban areas as part of the REUWS (Mayer et al., 1999). Figure 3-2 illustrates these same data graphically.

One of the more important wastewater-generating flows identified in this study was water leakage from plumbing fixtures. The average per capita leakage measured in the REUWS was 9.5 gallons/person/day (35.0 liters/person/day). However, this value was the result of high leakage rates at a relatively small percentage of homes. For example, the average daily leakage per household was 21.9 gallons (82.9 liters) with a standard deviation of 54.1 gallons (204.8 liters), while the median leakage rate was only 4.2 gallons/house/day (15.9 liters/house/day). Nearly 67 percent of the homes in the study had average leakage rates of less than 10 gallons/day (37.8 liters/day), but 5.5 percent of the study homes had leakage rates that averaged more than 100 gallons (378.5 liters) per day. Faulty toilet flapper valves and leaking faucets were the primary sources of leaks in these high-leakage-rate homes. Ten percent of the homes monitored accounted for 58 percent of the leakage measured. This result agrees with a previous end use study where average leakage rates of 4 to 8 gallons/person/day (15.1 to 30.3 liters/person/day) were measured (Brown and Caldwell, 1984). These data point out the importance of leak detection and repair during maintenance or repair of onsite

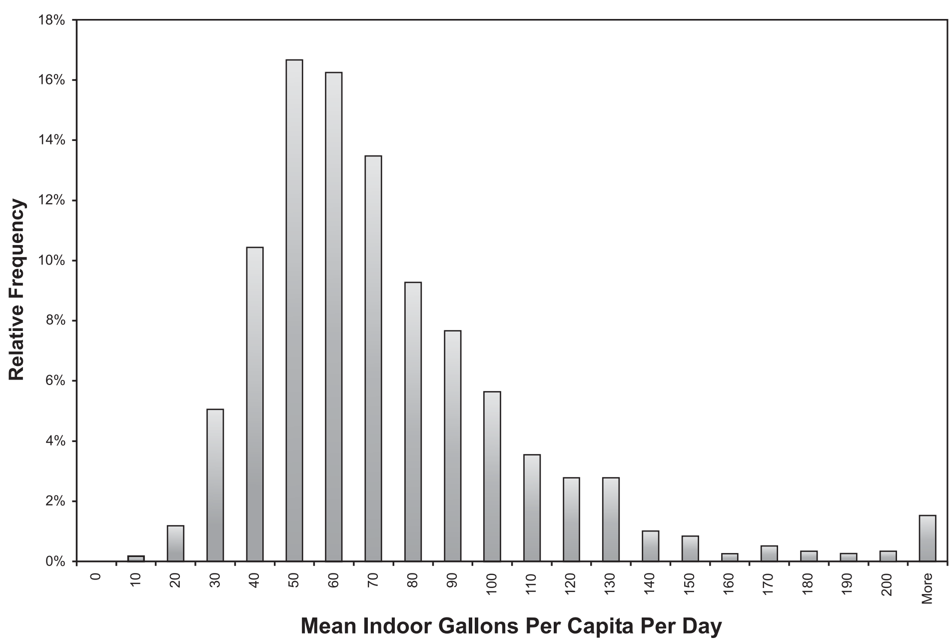
**Table 3-3. Residential water use by fixture or appliance[a,b]**

| Fixture/use | Gal/use: Average range | Uses/person/day: Average range | Gal/person/ day: Average range[c] | % Total: Average range |
|---|---|---|---|---|
| Toilet | 3.5<br>2.9–3.9 | 5.05<br>4.5–5.6 | 18.5<br>15.7–22.9 | 26.7<br>22.6–30.6 |
| Shower | 17.2[d]<br>14.9–18.6 | 0.75[d]<br>0.6–0.9 | 11.6<br>8.3–15.1 | 16.8<br>11.8–20.2 |
| Bath | See shower | See shower | 1.2<br>0.5–1.9 | 1.7<br>0.9–2.7 |
| Clothes washer | 40.5<br>— | 0.37<br>0.30–0.42 | 15.0<br>12.0–17.1 | 21.7<br>17.8–28.0 |
| Dishwasher | 10.0<br>9.3–10.6 | 0.10<br>0.06–0.13 | 1.0<br>0.6–1.4 | 1.4<br>0.9–2.2 |
| Faucets | 1.4[e]<br>— | 8.1[f]<br>6.7–9.4 | 10.9<br>8.7–12.3 | 15.7<br>12.4–18.5 |
| Leaks | NA | NA | 9.5<br>3.4–17.6 | 13.7<br>5.3–21.6 |
| Other Domestic | NA | NA | 1.6<br>0.0–6.0 | 2.3<br>0.0–8.5 |
| Total | NA | NA | 69.3<br>57.1–83.5 | 100 |

[a] Results from AWWARF REUWS at 1,188 homes in 12 metropolitan areas. Homes surveyed were serve by public water supplies, which operate at higher pressures than private water sources. Leakage rates might be lower for homes on private water supplies.
[b] Results are averages over range. Range is the lowest to highest average for 12 metropolitan areas.
[c] Gal/person/day might not equal gal/use multiplied by uses/person/day because of differences in the number of data points used to calculate means.
[d] Includes shower and bath.
[e] Gallons per minute.
[f] Minutes of use per person per day.

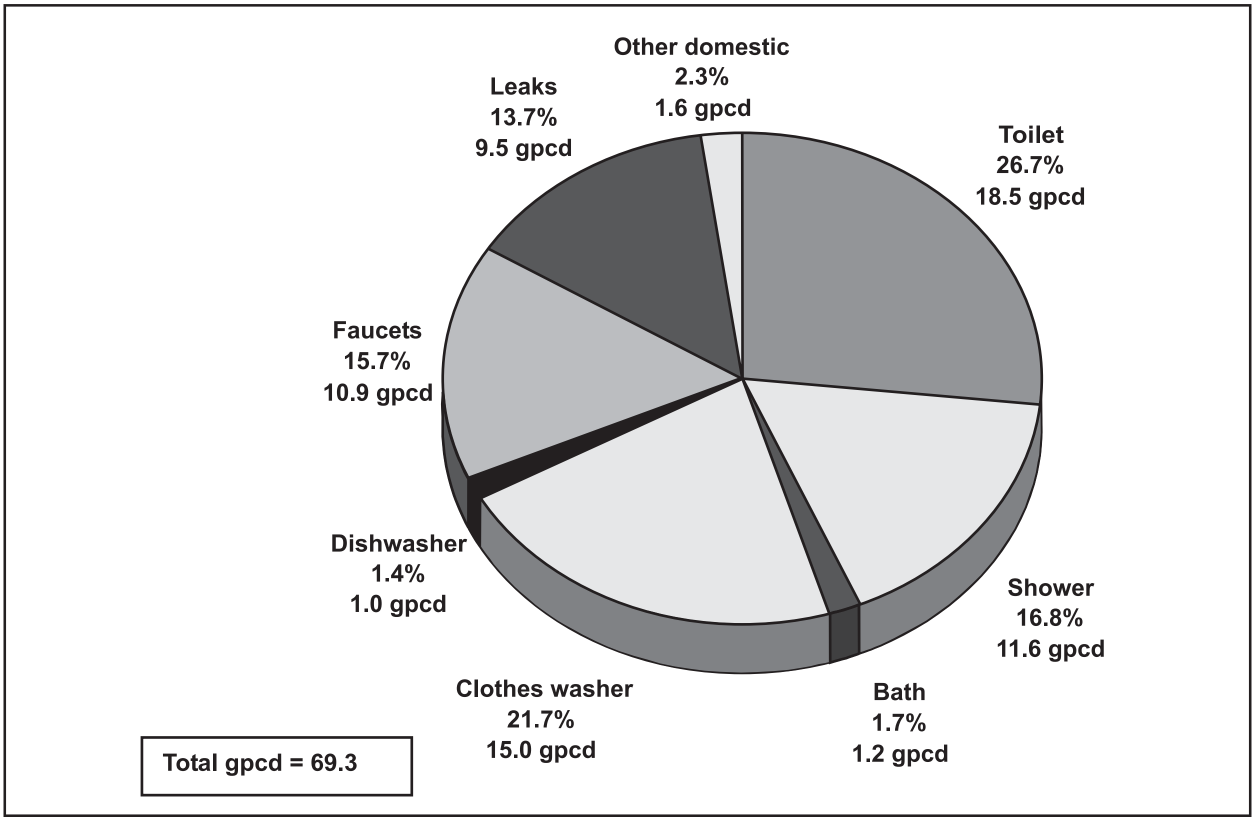
Source: Mayer et al., 1999.

**Figure 3-1. Distribution of mean household daily per capita indoor water use for 1,188 data-logged homes**




Source: Mayer et al., 1999.

**Figure 3-2. Indoor water use percentage, including leakage, for 1,188 data logged homes[a]**




[a] gpcd = gallons per capita (person) per day
Source: Mayer et al. 1999.

systems. Leakage rates like those measured in the REUWS could significantly increase the hydraulic load to an onsite wastewater system and might reduce performance.

### Maximum daily and peak flows

Maximum and minimum flows and instantaneous peak flow variations are necessary factors in properly sizing and designing system components. For example, most of the hydraulic load from a home occurs over several relatively short periods of time (Bennett and Lindstedt, 1975; Mayer et al., 1999; University of Wisconsin, 1978). The system should be capable of accepting and treating normal peak events without compromising performance. For further discussion of flow variations, see section 3.3.3.

## 3.3.2 Nonresidential wastewater flows

For nonresidential establishments typical daily flows from a variety of commercial, institutional, and recreational establishments are shown in tables 3-4 to 3-6 (Crites and Tchobanoglous, 1998; Tchobanoglous and Burton, 1991). The typical values presented are not necessarily an average of the range of values but rather are weighted values based on the type of establishment and expected use. Actual monitoring of specific wastewater flow and characteristics for nonresidential establishments is strongly recommended. Alternatively, a similar establishment located in the area might provide good information. If this approach is not feasible, state and local regulatory agencies should be consulted for approved design flow guidelines for nonresidential establishments. Most design flows provided by regulatory agencies are very conservative estimates based on peak rather than average daily flows. These agencies might accept only their established flow values and therefore should be contacted before design work begins.

**Table 3-4. Typical wastewater flow rates from commercial sources[a,b]**

| Facility | Unit | Flow, gallons/unit/day Range | Typical | Flow, liters/unit/day Range | Typical |
|---|---|---|---|---|---|
| Airport | Passenger | 2–4 | 3 | 8–15 | 11 |
| Apartment house | Person | 40–80 | 50 | 150–300 | 190 |
| Automobile service station[c] | Vehicle served | 8–15 | 12 | 30–57 | 45 |
| | Employee | 9–15 | 13 | 34–57 | 49 |
| Bar | Customer | 1–5 | 3 | 4–19 | 11 |
| | Employee | 10–16 | 13 | 38–61 | 49 |
| Boarding house | Person | 25–60 | 40 | 95–230 | 150 |
| Department store | Toilet room | 400–600 | 500 | 1,500–2,300 | 1,900 |
| | Employee | 8–15 | 10 | 30–57 | 38 |
| Hotel | Guest | 40–60 | 50 | 150–230 | 190 |
| | Employee | 8–13 | 10 | 30–49 | 38 |
| Industrial building (sanitary waste only) | Employee | 7–16 | 13 | 26–61 | 49 |
| Laundry (self-service) | Machine | 450–650 | 550 | 1,700–2,500 | 2,100 |
| | Wash | 45–55 | 50 | 170–210 | 190 |
| Office | Employee | 7–16 | 13 | 26–61 | 49 |
| Public lavatory | User | 3–6 | 5 | 11–23 | 19 |
| Restaurant (with toilet) | Meal | 2–4 | 3 | 8–15 | 11 |
| Conventional | Customer | 8–10 | 9 | 30–38 | 34 |
| Short order | Customer | 3–8 | 6 | 11–30 | 23 |
| Bar/cocktail lounge | Customer | 2–4 | 3 | 8–15 | 11 |
| Shopping center | Employee | 7–13 | 10 | 26–49 | 38 |
| | Parking space | 1–3 | 2 | 4–11 | 8 |
| Theater | Seat | 2–4 | 3 | 8–15 | 11 |

[a] Some systems serving more than 20 people might be regulated under USEPA's Class V Underground Injection Control (UIC) Program. See http://www.epa.gov/safewater/uic.html for more information.
[b] These data incorporate the effect of fixtures complying with the U.S. Energy Policy Act (EPACT) of 1994.
[c] Disposal of automotive wastes via subsurface wastewater infiltration systems is banned by Class V UIC regulations to protect ground water. See http://www.epa.gov/safewater/uic.html for more information.

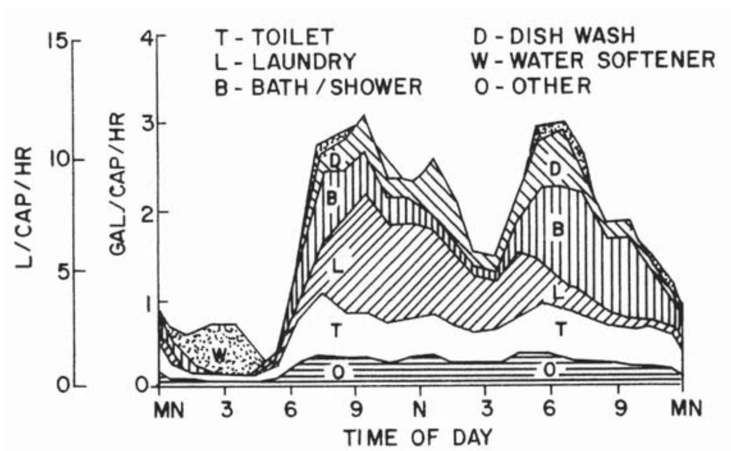
Source: Crites and Tchobanoglous, 1998.

## 3.3.3 Variability of wastewater flow

Variability of wastewater flow is usually characterized by daily and hourly minimum and maximum flows and instantaneous peak flows that occur during the day. The intermittent occurrence of individual wastewater-generating activities can create large variations in wastewater flows from residential or nonresidential establishments. This variability can affect gravity-fed onsite systems by potentially causing hydraulic overloads of the system during peak flow conditions. Figure 3-3 illustrates the routine fluctuations in wastewater flows for a typical residential dwelling.

Wastewater flow can vary significantly from day to day. Minimum hourly flows of zero are typical for

**Figure 3-3. Daily indoor water use pattern for single-family residence**




Source: University of Wisconsin, 1978.

**Table 3-5. Typical wastewater flow rates from institutional sources** [a]

| Facility | Unit | Flow, gallons/unit/day Range | Flow, gallons/unit/day Typical | Flow, liters/unit/day Range | Flow, liters/unit/day Typical |
|---|---|---|---|---|---|
| Assembly hall | Seat | 2–4 | 3 | 8–15 | 11 |
| Hospital, medical | Bed | 125–240 | 165 | 470–910 | 630 |
| | Employee | 5–15 | 10 | 19–57 | 38 |
| Hospital, mental | Bed | 75–140 | 100 | 280–530 | 380 |
| | Employee | 5–15 | 10 | 19–57 | 38 |
| Prison | Inmate | 80–150 | 120 | 300–570 | 450 |
| | Employee | 5–15 | 10 | 19–57 | 38 |
| Rest home | Resident | 50–120 | 90 | 190–450 | 340 |
| | Employee | 5–15 | 10 | 19–57 | 38 |
| School, day-only: | | | | | |
| With cafeteria, gym, showers | Student | 15–30 | 25 | 57–110 | 95 |
| With cafeteria only | Student | 10–20 | 15 | 38–76 | 57 |
| Without cafeteria, gym, or showers | Student | 5–17 | 11 | 19–64 | 42 |
| School, boarding | Student | 50–100 | 75 | 190–380 | 280 |

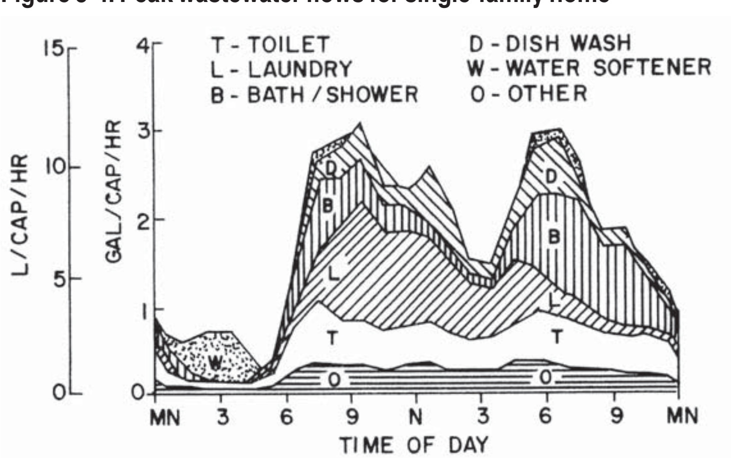
[a] Systems serving more than 20 people might be regulated under USEPA's Class V UIC Program. See http://www.epa.gov/safewater/uic.html for more information.

Source: Crites and Tchobanoglous, 1998.

residential dwellings. Maximum hourly flows as high as 100 gallons (380 L/hr) (Jones, 1976; Watson et al., 1967) are not unusual given the variability of typical fixture and appliance usage characteristics and residential water use demands. Hourly flows exceeding this rate can occur in cases of plumbing fixture failure and appliance misuse (e.g., broken pipe or fixture, faucets left running).

Wastewater flows from nonresidential establishments are also subject to wide fluctuations over time and are dependent on the characteristics of water-using fixtures and appliances and the business characteristics of the establishment (e.g., hours of operation, fluctuations in customer traffic).

**Figure 3-4. Peak wastewater flows for single-family home**




Source: University of Wisconsin, 1978.

The peak flow rate from a residential dwelling is a function of the fixtures and appliances present and their position in the plumbing system configuration. The peak discharge rate from a given fixture or appliance is typically around 5 gallons/minute (19 liters/minute), with the exception of the tank-type toilet and possibly hot tubs and bathtubs. The use of several fixtures or appliances simultaneously can increase the total flow rate above the rate for isolated fixtures or appliances. However, attenuation occurring in the residential drainage system tends to decrease peak flow rates observed in the sewer pipe leaving the residence. Although field data are limited, peak discharge rates from a single-family dwelling of 5 to 10 gallons/minute (19 to 38 liters/minute) can be expected. Figure 3-4 illustrates the variability in peak flow from a single home.

## 3.4 Wastewater quality

The qualitative characteristics of wastewaters generated by residential dwellings and nonresidential establishments can be distinguished by their physical, chemical, and biological composition. Because individual water-using events occur intermittently and contribute varying quantities of

**Table 3-6. Typical wastewater flow rates from recreational facilities[a]**

| Facility | Unit | Flow, gallons/unit/day Range | Flow, gallons/unit/day Typical | Flow, liters/unit/day Range | Flow, liters/unit/day Typical |
|---|---|---|---|---|---|
| Apartment, resort | Person | 50–70 | 60 | 190–260 | 230 |
| Bowling alley | Alley | 150–250 | 200 | 570–950 | 760 |
| Cabin, resort | Person | 8–50 | 40 | 30–190 | 150 |
| Cafeteria | Customer | 1–3 | 2 | 4–11 | 8 |
| | Employee | 8–12 | 10 | 30–45 | 38 |
| Camps: | | | | | |
| Pioneer type | Person | 15–30 | 25 | 57–110 | 95 |
| Children's, with central toilet/bath | Person | 35–50 | 45 | 130–190 | 170 |
| Day, with meals | Person | 10–20 | 15 | 38–76 | 57 |
| Day, without meals | Person | 10–15 | 13 | 38–57 | 49 |
| Luxury, private bath | Person | 75–100 | 90 | 280–380 | 340 |
| Trailer camp | Trailer | 75–150 | 125 | 280–570 | 470 |
| Campground-developed | Person | 20–40 | 30 | 76–150 | 110 |
| Cocktail lounge | Seat | 12–25 | 20 | 45–95 | 76 |
| Coffee Shop | Customer | 4–8 | 6 | 15–30 | 23 |
| | Employee | 8–12 | 10 | 30–45 | 38 |
| Country club | Guests onsite | 60–130 | 100 | 230–490 | 380 |
| | Employee | 10–15 | 13 | 38–57 | 49 |
| Dining hall | Meal served | 4–10 | 7 | 15–38 | 26 |
| Dormitory/bunkhouse | Person | 20–50 | 40 | 76–190 | 150 |
| Fairground | Visitor | 1–2 | 2 | 4–8 | 8 |
| Hotel, resort | Person | 40–60 | 50 | 150–230 | 190 |
| Picnic park, flush toilets | Visitor | 5–10 | 8 | 19–38 | 30 |
| Store, resort | Customer | 1–4 | 3 | 4–15 | 11 |
| | Employee | 8–12 | 10 | 30–45 | 38 |
| Swimming pool | Customer | 5–12 | 10 | 19–45 | 38 |
| | Employee | 8–12 | 10 | 30–45 | 38 |
| Theater | Seat | 2–4 | 3 | 8–15 | 11 |
| Visitor center | Visitor | 4–8 | 5 | 15–30 | 19 |

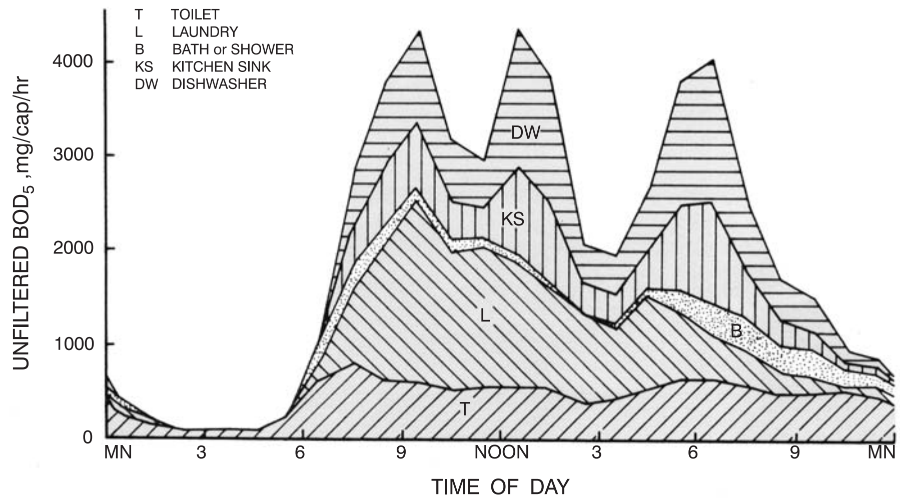
[a] Some systems serving more than 20 people might be regulated under USEPA's Class V UIC Program.

Source: Crites and Tchobanoglous, 1998.

pollutants, the strength of residential wastewater fluctuates throughout the day (University of Wisconsin, 1978). For nonresidential establishments, wastewater quality can vary significantly among different types of establishments because of differences in waste-generating sources present, water usage rates, and other factors. There is currently a dearth of useful data on nonresidential wastewater organic strength, which can create a large degree of uncertainty in design if facility-specific data are not available. Some older data (Goldstein and Moberg, 1973; Vogulis, 1978) and some new information exists, but modern organic strengths need to be verified before design given the importance of this aspect of capacity determination.

Wastewater flow and the type of waste generated affect wastewater quality. For typical residential sources peak flows and peak pollutant loading rates do not occur at the same time (Tchobanoglous and Burton, 1991). Though the fluctuation in wastewater quality (see figure 3-5) is similar to the water use patterns illustrated in figure 3-3, the fluctuations in wastewater quality for an individual home are likely to be considerably greater than the multiple-home averages shown in figure 3-5.

**Figure 3-5. Average hourly distribution of total unfiltered $BOD_5$**




Source: University of Wisconsin, 1978.

OWTSs should be designed to accept and process hydraulic flows from a residence (or establishment) while providing the necessary pollutant removal efficiency to achieve performance goals. The concentrations of typical pollutants in raw residential wastewaters and average daily mass loadings are summarized in table 3-7. Residential water-using activities contribute varying amounts of pollutants to the total wastewater flow. Table 3-8 contains a summary of the average mass loading of several key pollutants from the sources identified in table 3-7.

If the waste-generating sources present at a particular nonresidential establishment are similar to those of a typical residential dwelling, an approximation of the pollutant mass loadings and concentrations in the wastewater can be derived using the residential wastewater quality data for those categories presented in tables 3-7 and 3-8. However, the results of previous studies have demonstrated that in many cases nonresidential wastewater is considerably different from residential wastewater. Restaurant wastewater, for example, contains substantially higher levels of organic matter, solids, and grease compared to typical residential wastewater (Siegrist et al., 1984; University of Wisconsin, 1978). Restaurant wastewater $BOD_5$ concentrations reported in the literature range from values similar to those for domestic waste to well over 1,000 milligrams/liter, or 3.5 to 6.5 times higher than residential $BOD_5$. Total suspended solids and grease concentrations in restaurant wastewaters were reported to be 2 to 5 times higher than the concentrations in domestic wastewaters (Kulesza, 1975; Shaw, 1970). For shopping centers, the average characteristics determined by one study found $BOD_5$ average concentrations of 270 milligrams/liter, with suspended solids concentrations of 337 milligrams/liter and grease concentrations of 67 milligrams/liter (Hayashida, 1975).

More recent characterizations of nonresidential establishments have sampled septic tank effluent, rather than the raw wastewater, to more accurately identify and quantify the mass pollutant loads delivered to the components of the final treatment train (Ayres Associates, 1991; Siegrist et al., 1984). Because of the variability of the data, for establishments where the waste-generating sources are significantly different from those in a residential dwelling or where more refined characterization data might be appropriate, a detailed review of the pertinent literature, as well as wastewater sampling at the particular establishment or a similar establishment, should be conducted.

## 3.5 Minimizing wastewater flows and pollutants

Minimizing wastewater flows and pollutants involves techniques and devices to (1) reduce water use and resulting wastewater flows and (2) decrease the quantity of pollutants discharged to the waste stream. Minimizing wastewater volumes and pollutant concentrations can improve the efficiency of onsite treatment and lessen the risk of hydraulic or treatment failure (USEPA, 1995). These meth-

**Table 3-7. Constituent mass loadings and concentrations in typical residential wastewater [a]**

| Constituent | Mass loading (grams/person/day) | Concentration[b] (mg/L) |
|---|---|---|
| Total solids (TS) | 115–200 | 500–880 |
| Volatile solids | 65–85 | 280–375 |
| Total suspended solids (TSS) | 35–75 | 155–330 |
| Volatile suspended solids | 25–60 | 110–265 |
| 5-day biochemical oxygen demand ($BOD_5$) | 35–65 | 155–286 |
| Chemical oxygen demand (COD) | 115–150 | 500–660 |
| Total nitrogen (TN) | 6–17 | 26–75 |
| Ammonia ($NH_4$) | 1–3 | 4–13 |
| Nitrites and nitrates ($NO_2$-N; $NO_3$-N) | <1 | <1 |
| Total phosphorus (TP)[c] | 1–2 | 6–12 |
| Fats, oils, and grease | 12–18 | 70–105 |
| Volatile organic compounds (VOC) | 0.02–0.07 | 0.1–0.3 |
| Surfactants | 2–4 | 9–18 |
| Total coliforms (TC)[d] | – | $10^8$–$10^{10}$ |
| Fecal coliforms (FC)[d] | – | $10^6$–$10^8$ |

[a] For typical residential dwellings equipped with standard water-using fixtures and appliances.
[b] Milligrams per liter; assumed water use of 60 gallons/person/day (227 liters/person/day).
[c] The detergent industry has lowered the TP concentrations since early literature studies; therefore, Sedlak (1991) was used for TP data.
[d] Concentrations presented in Most Probable Number of organisms per 100 milliliters.

Source: Adapted from Bauer et al., 1979; Bennett and Linstedt, 1975; Laak, 1975, 1986; Sedlak, 1991; Tchobanoglous and Burton, 1991.

**Table 3-8. Residential wastewater pollutant contributions by source [a,b]**

| Parameter | | Garbage disposal (gpcd)[c] | Toilet (gpcd)[c] | Bathing, sinks, appliances (gpcd)[c] | Approximate total (gpcd)[c] |
|---|---|---|---|---|---|
| $BOD_5$ | mean | 18.0 | 16.7 | 28.5 | 63.2 |
| | range | 10.9–30.9 | 6.9–23.6 | 24.5–38.8 | |
| | % of total | (28%) | (26%) | (45%) | (100%) |
| Total suspended solids | mean | 26.5 | 27.0 | 17.2 | 70.7 |
| | range | 15.8–43.6 | 12.5–36.5 | 10.8–22.6 | |
| | % of total | (37%) | (38%) | (24%) | (100%) |
| Total nitrogen | mean | 0.6 | 8.7 | 1.9 | 11.2 |
| | range | 0.2–0.9 | 4.1–16.8 | 1.1–2.0 | |
| | % of total | (5%) | (78%) | (17%) | (100%) |
| Total phosphorus[d] | mean | 0.1 | 1.6 | 1.0 | 2.7 |
| | range | — | — | — | — |
| | % of total | (4%) | (59%) | (37%) | (100%) |

[a] Adapted from USEPA, 1992.
[b] Means and ranges for BOD, TSS, and TN are results reported in Bennett and Linstedt, 1975; Laak, 1975;Ligman et al., 1974; Olsson et al., 1968; and Siegrist et al., 1976.
[c] Grams per capita (person) per day.
[d] The use of low-phosphate detergents in recent years has lowered the TP concentrations since early literature studies; therefore, Sedlak (1991) was used for TP data.

ods have been developed around two main strategies—wastewater flow reduction and pollutant mass reduction. Although this section emphasizes residential flows, many of the concepts are applicable to nonresidential establishments. (For more information on both residential and nonresidential water use reduction, see **http://www.epa.gov/OW/you/intro.html**.)

### 3.5.1 Minimizing residential wastewater volumes

The most commonly reported failure of residential OWTS infiltration systems is hydraulic overloading. Hydraulic overloads can be caused by wastewater flow or pollutant loads that exceed system design capacity. When more water is processed than an OWTS is designed to handle, detention time within the treatment train is reduced, which can decrease pollutant removal in the tank and overload the infiltration field. Reducing water use in a residence can decrease hydraulic loading to the treatment system and generally improve system performance. If failure is caused by elevated pollutant loads, however, other options should be considered (see chapter 5).

Indoor residential water use and resulting wastewater flows are attributed mainly to toilet flushing, bathing, and clothes washing (figure 3-2). Toilet use usually accounts for 25 to 30 percent of indoor water use in residences; toilets, showers, and faucets in combination can represent more than 70 percent of all indoor use. Residential wastewater flow reduction can therefore be achieved most dramatically by addressing these primary indoor uses and by minimizing wastewater flows from extraneous sources. Table 3-9 presents many of the methods that have been applied to achieve wastewater flow reduction.

#### Eliminating extraneous flows

Excessive water use can be reduced or eliminated by several methods, including modifying water use habits and maintaining the plumbing system appropriately. Examples of methods to reduce water use include

- Using toilets to dispose of sanitary waste only (not kitty litter, diapers, ash tray contents, and other materials.)
- Reducing time in the shower
- Turning off faucets while brushing teeth or shaving
- Operating dishwashers only when they are full
- Adjusting water levels in clothes washers to match loads; using machine only when full
- Making sure that all faucets are completely turned off when not in use
- Maintaining plumbing system to eliminate leaks

These practices generally involve changes in water use behavior and do not require modifying of plumbing or fixtures. Homeowner education programs can be an effective approach for modifying water use behavior (USEPA, 1995). Wastewater flow reduction resulting from eliminating wasteful water use habits will vary greatly depending on past water use habits. In many residences, significant water use results from leaking plumbing fixtures. The easiest ways to reduce wastewater flows from indoor water use are to properly maintain plumbing fixtures and repair leaks when they occur. Leaks that appear to be insignificant, such as leaking toilets or dripping faucets, can generate large volumes of wastewater. For example, a 1/32-inch (0.8 millimeters) opening at 40 pounds per square inch (207 mm of mercury) of pressure can waste from 3,000 to 6,000 gallons (11, 550 to 22,700 liters) of water per month. Even apparently very slow leaks, such as a slowly dripping faucet, can generate 15 to 20 gallons (57 to 76 liters) of wastewater per day.

#### Reducing wastewater flow

Installing indoor plumbing fixtures that reduce water use and replacing existing plumbing fixtures or appliances with units that use less water are successful practices that reduce wastewater flows (USEPA, 1995). Recent interest in water conservation has been driven in some areas by the absence of adequate source water supplies and in other areas by a desire to minimize the need for expensive wastewater treatment. In 1992 Congress passed the U.S. Energy Policy Act (EPACT) to establish national standards governing the flow capacity of showerheads, faucets, urinals, and water closets for the purpose of national energy and water conservation (table 3-10). Several states have also implemented specific water conservation practices

**Table 3-9. Wastewater flow reduction methods**

**Elimination of extraneous flows**

- Improved water-use habits
- Improved plumbing and appliance maintenance and monitoring
- Elimination of excessive water supply pressure

**Reduction of existing wastewater flows**

- Toilets

| Water-carriage toilets | Non-water-carriage toilets |
|---|---|
| - Toilet-tank inserts | - Biological (compost) toilets |
| - Ultra-low flush (ULF) toilets (1.6 gal or 6 L per flush or less) Wash-down flush Pressurized tank | - Incinerator toilets |

- Bathing devices, fixtures, and appliances
  - Shower flow controls
  - Reduced-flow showerheads
  - On/off showerhead valves
  - Mixing valves
  - Air-assisted, low-flow shower system
- Clothes-washing devices, fixtures, and appliances
  - High-efficiency washer
  - Adjustable cycle settings
  - Washwater recycling feature
- Miscellaneous
  - Faucet inserts
  - Faucet aerators
  - Reduced-flow faucet fixtures
  - Mixing valves
  - Hot water pipe insulation
  - Pressure-reducing valves
  - Hot water recirculation

**Wastewater recycle/reuse systems**

- Sink/bath/laundry wastewater recycling for toilet flushing
- Recycling toilets
- Combined wastewater recycling for toilet flushing
- Combined wastewater recycling for outdoor irrigation

Sources: Adapted from USEPA, 1992, 1995.

(USEPA, 1995; for case studies and other information, see **http://www.epa.gov/OW/you/intro.html**.

Several toilet designs that use reduced volumes of water for proper operation have been developed. Conventional toilets manufactured before 1994 typically use 3.5 gallons (13.2 liters) of water per flush. Reduced-flow toilets manufactured after 1994 use 1.6 gallons (6.1 liters) or less per flush. Though studies have shown an increased number of flushes with reduced-flow toilets, potential savings of up to 10 gallons/person/day (37.8 liters/person/day) can be achieved (Aher et al., 1991; Anderson et al., 1993; Mayer et al., 1999, 2000). Table 3-11 contains information on water carriage toilets and systems; table 3-12 contains information on non-water-carriage toilets. The reader is cautioned that not all fixtures perform well in every application and that certain alternatives might not be acceptable to the public.

The volume of water used for bathing varies considerably based on individual habits. Averages indicate that showering with common showerheads using 3.0 to 5.0 gallons/minute (0.19 to 0.32 liters/second) amounts to a water use of 10 to 12.5

**Table 3-10. Comparison of flow rates and flush volumes before and after U.S. Energy Policy Act**

| Fixture | Fixtures installed prior to 1994 in gallons/minute (liters/second) | EPACT requirements (effective January, 1994) | Potential reduction in water used (%) |
|---|---|---|---|
| Kitchen faucet | 3.0 gpm (0.19 L/s) | 2.5 gpm (0.16 L/s) | 16 |
| Lavatory faucets | 3.0 gpm (0.19 L/s) | 2.5 gpm (0.16 L/s) | 16 |
| Showerheads | 3.5 gpm (0.22 L/s) | 2.5 gpm (0.16 L/s) | 28 |
| Toilet (tank type) | 3.5 gal (13.2 L) | 1.6 gal (6.1 L) | 54 |
| Toilet (valve type) | 3.5 gal (13.2 L) | 1.6 gal[a] (6.1 L) | 54 |
| Urinal | 3.0 gal (11.4 L) | 1.0 gal (3.8 L) | 50 |

Source: Konen, 1995.

**Table 3-11. Wastewater flow reduction: water-carriage toilets and systems [a]**

| Generic type | Description | Application considerations | Operation & maintenance | Water use per event gal (L) | Total flow reduction in gpcd (Lpcd); % of use [b] |
|---|---|---|---|---|---|
| Toilets with tank inserts | Displacement devices placed into storage tank of conventional toilet to reduce volume but not height of stored water.<br><br>Varieties: Plastic bottles, flexible panels, drums, or plastic bags | Device must be compatible with existing toilet and not interfere with flush mechanism<br><br>Installation by owner<br><br>Reliability low; failure can result in large flow increase | Frequent post-installation inspections to ensure proper positioning | 3.3–3.8 (12.5–14.4) | 1.8–3.5 (6.8–13.2)<br><br>4%–8% |
| Water-saving toilets | Variation of conventional flush toilet fixture; similar in appearance and operation. Redesigned flushing rim and priming jet to initiate siphon flush in smaller trapway with less water. | Interchangeable with conventional fixture | Essentially the same as for a conventional unit | 1.0–1.6 (3.8–13.2) | 5.3–13 (12.1–49.2)<br><br>6%–20% |
| Washdown flush toilets | Flushing uses only water, but substantially less due to washdown flush<br><br>Varieties: Few<br><br>Note: Water usage may increase due to multiple flushings | Rough-in for unit may be nonstandard<br><br>Drain-line slope and lateral-run restrictions<br><br>Plumber installation advisable | Similar to conventional toilet<br><br>Cleaning possible | 0.8–1.6 (3.0–6.1)<br><br>(but more frequent flushings possible) | 9.4–12.2 (35.6–46.2)<br><br>21%–27% |
| Pressurized-tank toilets | Specially designed toilet tank to pressurize air contained in toilet tank. Upon flushing, compressed air propels water into bowl at increased velocity<br><br>Varieties: Few | Compatible with most conventional toilet units<br><br>Increased noise level<br><br>Water supply pressure of 35–120 psi (180–620 cm Hg) required | Periodic maintenance of compressed air source | 2.0–2.5 (7.6–9.5) | 6.3–8.0 (23.8–30.3)<br><br>14%–18% |

[a] Adapted from USEPA, 1992. Compared to conventional toilet usage (4.3 gallons/flush [16.3 liters/flush], 3.5 uses per person per day, and a total daily flow of 45 gallons/person/day [170 liters/person/day]).

[b] gpcd = gallons per capita (person) per day; Lpcd = liters per capita (person) per day.

**Table 3-12. Wastewater flow reduction: non-water-carriage toilets [a]**

| Generic type | Description | Application considerations | Operation and maintenance |
|---|---|---|---|
| Biological toilets | Large units with a separated decomposition chamber. Accept toilet wastes and other organic matter, and over a long time period partially stabilize excreta through biological activity and evaporation. | Installation requires 6- to 12-in (150-mm to 300-mm)-diameter roof vent, space beneath floor for decomposition chamber, ventilation system, and heating<br>Handles toilet waste and some kitchen waste<br>Restricted usage capacity cannot be exceeded<br>Difficult to retrofit and expensive | Periodic addition of organic matter<br>Removal of product material at 6- to 24-month intervals should be performed by management authority due to risk of exposure to pathogens in wastes<br>Heat loss through vent |
| Incinerator toilets | Small self-contained units that volatilize the organic components of human waste and evaporate the liquids. | Installation requires 4-in-diameter roof vent<br>Handles only toilet waste<br>Power or fuel required<br>Increased noise level<br>Residuals disposal<br>Limited usage rate (frequency) | Weekly removal of ash<br>Semiannual cleaning and adjustment of burning assembly or heating elements<br>Fuel units could pose safety concerns |

[a] Adapted from USEPA, 1992. None of these devices uses any water; therefore, the amount of flow and pollutant reduction equal to those of conventional toilet use (see table 3-3). Significant quantities of pollutants (including N, $BOD_5$, SS, P, and pathogens) are therefore removed from the wastewater stream (table 3-8).

**Table 3-13. Wastewater flow reduction: showering devices and systems [a]**

| Generic type | Description | Application considerations | Water use rate |
|---|---|---|---|
| Shower flow-control inserts and restrictors | Reduce flow rate by reducing diameter of supply line ahead of showerhead | Compatible with most existing showerheads. User habits may negate potential savings by extended shower duration | 1.5–3.0 gal/min<br>(0.09–0.19 L/s) |
| Reduced-flow showerheads | Fixtures similar to conventional, except restrict flow rate<br>Varieties: Many manufacturers, but units similar | Compatible with most conventional plumbing<br>Installed by user | 1.5–2.5 gal/min<br>(0.09–0.19 L/s) |
| On/off showerhead valve | Small valve device placed in supply line ahead of showerhead allows shower flow to be turned on and off without readjustment of volume or temperature | Compatible with most conventional plumbing and fixtures<br>Usually installed by plumber | Unchanged, but total duration and use are reduced |
| Mixing valves | Specifically designed valves maintain constant temperature of total flow. Faucets may be operated (on and off) without temperature adjustment | Compatible with most conventional plumbing and fixtures<br>Usually installed by plumber | Unchanged, but daily duration and use are reduced |
| Air-assisted, low-flow shower system | Specifically designed system uses compressed air to atomize water flow and provide shower sensation | May be difficult and expensive to retrofit<br>Requires shower location less than 50 ft (15.3 m) away from water heater<br>Requires compressed air and power source<br>Requires maintenance of air compressor | 0.5 gal/min<br>(0.3 L/s) |

Note: gal/min = gallons per minute; L/s = liters per second.
[a] Adapted from USEPA, 1992.

gallons/person/day (37.9 to 47.3 liters/person/day). Table 3-13 provides an overview of showering devices available to reduce wastewater flows associated with shower use. A low-flow showerhead can reduce water flow through the shower by 2 or 3 gallons/minute (0.13 to 0.19 liters/second), but if the user stays in the shower twice as long because the new showerhead does not provide enough pressure or flow to satisfy showering preferences, projected savings can be negated.

Indoor water use can also be reduced by installing flow reduction devices or faucet aerators at sinks and basins. More efficient faucets can reduce water use from 3 to 5 gallons/minute (0.19 to 0.32 liters/second) to 2 gallons/minute (0.13 liters/second), and aerators can reduce water use at faucets by as much as 60 percent while still maintaining a strong flow. Table 3-14 provides a summary of wastewater flow reduction devices that can be applied to water use at faucets.

### Reducing water pressure

Reducing water pressure is another method for reducing wastewater flows. The flow rate at faucets and showers is directly related to the water pressure in the water supply line. The maximum water flow from a fixture operating on a fixed setting can be reduced by reducing water pressure. For example, a reduction in pressure from 80 pounds per square inch (psi) (414 cm Hg) to 40 psi (207 cm Hg) can reduce the flow rate through a fully opened faucet by about 40 percent. Reduced pressure has little effect on the volume of water used by fixtures that operate on a fixed volume of water, such as toilets and washing machines, but it can reduce wastewater flows from sources controlled by the user (e.g., faucets, showerheads).

## 3.5.2 Reducing mass pollutant loads in wastewater

Pollutant mass loading modifications reduce the amount of pollutants requiring removal or treatment in the OWTS. Methods that may be applied for reducing pollutant mass loads include modifying product selection, improving user habits, and eliminating or modifying certain fixtures. Household products containing toxic compounds, commonly referred to as "household hazardous waste," should be disposed of properly to minimize threats to human health and the environment. For more information on disposal options and related issues, visit the USEPA Office of Solid Waste's *Household Hazardous Waste* web site at **http://www.epa.gov/epaoswer/non-hw/muncpl/hhw.htm**.

**Table 3-14. Wastewater flow reduction: miscellaneous devices and systems**

| Generic type | Description | Application considerations |
|---|---|---|
| Faucet insert | Device that inserts into faucet valve or supply line and restricts flow rate with a fixed or pressure-compensating orifice | Compatible with most plumbing<br>Installation simple |
| Faucet aerator | Devices attached to faucet outlet that entrain air into water flow | Compatible with most plumbing<br>Installation simple<br>Periodic cleaning of aerator screens |
| Reduced-flow faucet | Similar to conventional unit, but restricts flow rate with a fixed or pressure-compensating orifice | Compatible with most plumbing<br>Installation identical to conventional faucet |
| Mixing valves | Specifically designed valve units that allow flow and temperature to be set with a single control | Compatible with most plumbing<br>Installation identical to conventional valve units |
| Hot-water system insulation | Hot-water heater and piping are wrapped with insulation to reduce heat loss and water use (faucet delivers hot water quicker) | May be difficult to wrap entire hot-water piping system after house is built. |

[a] Adapted from USEPA, 1992.

Source: Adapted from USEPA, 1992.

### Selecting cleaning agents and household chemicals

Toilet flushing, bathing, laundering, washing dishes, operating garbage disposals, and general cleaning are all activities that can include the use of chemicals that are present in products like disinfectants and soaps. Some of these products contribute significant quantities of pollutants to wastewater flows. For example, bathing, clothes washing, and dish washing contribute large amounts of sodium to wastewater. Before manufacturers reformulated detergents, these activities accounted for more than 70 percent of the phosphorus in residential flows. Efforts to protect water quality in the Chesapeake Bay, Great Lakes, and major rivers across the nation led to the first statewide bans on phosphorus in detergents in the 1970s, and other states issued phosphorus bans throughout the 1980s. The new low-phosphorus detergents have reduced phosphorus loadings to wastewater by 40 to 50 percent since the 1970s.

The impacts associated with the daily use of household products can be reduced by providing public education regarding the environmental impacts of common household products. Through careful selection of cleaning agents and chemicals, pollution impacts on public health and the environment associated with their use can be reduced.

**Improving onsite system performance by improving user habits**

The University of Minnesota Extension Service's *Septic System Owner's Guide* recommends the following practices to improve onsite system performance:

- Do not use "every flush" toilet bowl cleaners.
- Reduce the use of drain cleaners by minimizing the amount of hair, grease, and food particles that go down the drain.
- Reduce the use of cleaners by doing more scrubbing with less cleanser.
- Use the minimum amount of soap, detergent, and bleach necessary to do the job.
- Use minimal amounts of mild cleaners and only as needed.
- Do not drain chlorine-treated water from swimming pools and hot tubs into septic systems.
- Dispose of all solvents, paints, antifreeze, and chemicals through local recycling and hazardous waste collection programs.
- Do not flush unwanted prescription or over-the-counter medications down the toilet.

*Adapted from University of Minnesota, 1998.*

### Improving user habits

Everyday household activities generate numerous pollutants. Almost every commonly used domestic product—cleaners, cosmetics, deodorizers, disinfectants, pesticides, laundry products, photographic products, paints, preservatives, soaps, and medicines—contains pollutants that can contaminate ground water and surface waters and upset biological treatment processes in OWTSs (Terrene Institute, 1995). Some household hazardous waste (HHW) can be eliminated from the wastewater stream by taking hazardous products to HHW recycling/reuse centers, dropping them off at HHW collection sites, or disposing of them in a solid waste form (i.e., pouring liquid products like paint, cleaners, or polishes on newspapers, allowing them to dry in a well-ventilated area, and enclosing them in several plastic bags for landfilling) rather than dumping them down the sink or flushing them down the toilet. Improper disposal of HHW can best be reduced by implementing public education and HHW collection programs. A collection program is usually a 1-day event at a specific site. Permanent programs include retail store drop-off programs, curbside collection, and mobile facilities. Establishing HHW collection programs can significantly reduce the amount of hazardous chemicals in the wastewater stream, thereby reducing impacts on the treatment system and on ground water and surface waters.

Stopping the practice of flushing household wastes (e.g., facial tissue, cigarette butts, vegetable peelings, oil, grease, other cooking wastes) down the toilet can also reduce mass pollutant loads and decrease plumbing and OWTS failure risks. Homeowner education is necessary to bring about these changes in behavior. Specific homeowner information is available from the National Small Flows Clearinghouse at **http://www.estd.wvu.edu/nsfc/NSFC_septic_news.html.**

Table 3-15. Reduction in pollutant loading achieved by eliminating garbage disposals

| Parameter | Reduction in pollutant loading (%) |
|---|---|
| Total suspended solids | 25–40 |
| Biochemical oxygen demand | 20–28 |
| Total nitrogen | 3.6 |
| Total phosphorus | 1.7 |
| Fats, oils, and grease | 60–70 |

Source: University of Wisconsin, 1978.

### Eliminating use of garbage disposals

Eliminating the use of garbage disposals can significantly reduce the amount of grease, suspended solids, and BOD in wastewater (table 3-15). Reducing the amount of vegetable and other food-related material entering wastewater from garbage disposals can also result in a slight reduction in nitrogen and phosphorus loads. Eliminating garbage disposal use also reduces the rate of sludge and scum accumulation in the septic tank, thus reducing the frequency of required pumping. OWTSs, however, can accommodate garbage disposals by using larger tanks, SWISs, or alternative system designs. (For more information, see Special Issue Fact Sheets 2 and 3 in the Chapter 4 Fact Sheets section.)

### Using graywater separation approaches

Another method for reducing pollutant mass loading to a single SWIS is segregating toilet waste flows (blackwater) from sink, shower, washing machine, and other waste flows (graywater). Some types of toilet systems provide separate handling of human excreta (such as the non-water-carriage units in table 3-14). Significant quantities of suspended solids, BOD, nitrogen, and pathogenic organisms are eliminated from wastewater flows by segregating body wastes from the OWTS wastewater stream through the use of composting or incinerator toilets. This approach is more cost-effective for new homes, homes with adequate crawl spaces, or mobile or modular homes. Retrofitting existing homes, especially those with concrete floors, can be expensive. (For more information on graywater reuse, see Special Issue Fact Sheet 4 in the Chapter 4 Fact Sheets section and **http://www.epa.gov/OW/you/chap3.html**.)

Graywaters contain appreciable quantities of organic matter, suspended solids, phosphorus, grease, and bacteria (USEPA, 1980a). Because of the presence of significant concentrations of bacteria and possibly pathogens in graywaters from bathing, hand washing, and clothes washing, caution should be exercised to ensure that segregated graywater treatment and discharge processes occur below the ground surface to prevent human contact. In addition, siting of graywater infiltration fields should not compromise the hydraulic capacity of treatment soils in the vicinity of the blackwater infiltration field.

## 3.5.3 Wastewater reuse and recycling systems

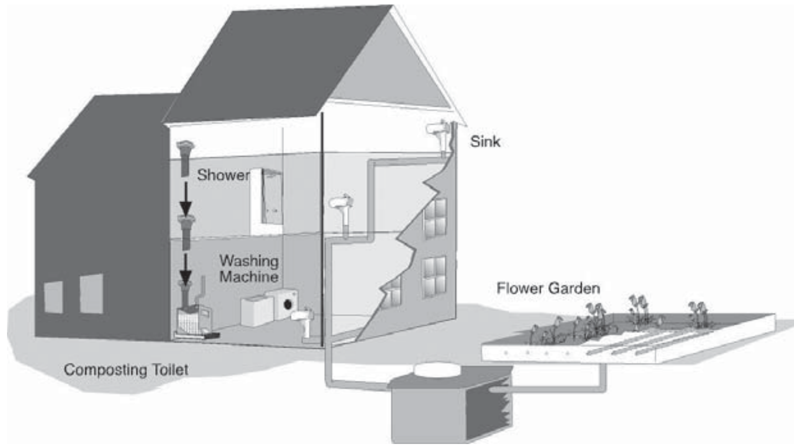
Many arid and semiarid regions in the United States have been faced with water shortages, creating the need for more efficient water use practices. Depletion of ground water and surface water resources due to increased development, irrigation, and overall water use is also becoming a growing concern in areas where past supplies have been plentiful (e.g., south Florida, central Georgia). Residential development in previously rural areas has placed additional strains on water supplies and wastewater treatment facilities. Decentralized wastewater management programs that include onsite wastewater reuse/recycling systems are a viable option for addressing water supply shortages and wastewater discharge restrictions. In municipalities where water shortages are a recurring problem, such as communities in California and Arizona, centrally treated reclaimed wastewater has been used for decades as an alternative water supply for agricultural irrigation, ground water recharge, and recreational waters.

*Wastewater reuse* is the collection and treatment of wastewater for other uses (e.g., irrigation, ornamental ponds, and cooling systems). *Wastewater recycling* is the collection and treatment of wastewater and its reuse in the same water-use scheme, such as toilet and urinal flushing (Tchobanoglous and Burton, 1991). Wastewater reuse/recycling systems can be used in individual homes, clustered communities, and larger institutional facilities such as office parks and recreational facilities. The Grand Canyon National Park has reused treated wastewater for toilet flushing, landscape irrigation, cooling water, and boiler feedstock since 1926, and other reuse systems are gaining acceptance (Tchobanoglous and Burton, 1991). Office buildings, schools, and recreational facilities using wastewater reuse/recycling systems have reported a 90 percent reduction in water use and up to a 95 percent reduction in wastewater discharges (Burks and Minnis, 1994).

Wastewater reuse/recycling systems reduce potable water use by reusing or recycling water that has already been used at the site for nonpotable purposes, thereby minimizing wastewater discharges. The intended use of wastewater dictates the degree of treatment necessary before reuse. Common concerns associated with wastewater reuse/recycling systems include piping cross-connections, which could contaminate potable water supplies with wastewater, difficulties in modifying and integrating potable and nonpotable plumbing, public and public agency acceptance, and required maintenance of the treatment processes.

A number of different onsite wastewater reuse/recycling systems and applications are available. Some systems, called combined systems, treat and reuse or recycle both blackwater and graywater (NAPHCC, 1992. Other systems treat and reuse or recycle only graywater. Figure 3-6 depicts a typical graywater reuse approach. Separating graywater and blackwater is a common practice to reduce pollutant loadings to wastewater treatment systems (Tchobanoglous and Burton, 1991).

**Figure 3-6. Typical graywater reuse approach**




## 3.5.4 Factors of safety in characterization estimates

Conservative predictions or factors of safety are typically used to account for potential variability in wastewater characteristics at a particular dwelling or establishment. These predictions attempt to ensure adequate treatment by the onsite system without requiring actual analysis of the variability in flow or wastewater quality. However, actual measurement of wastewater flow and quality from a residential dwelling or nonresidential establishment always provides the most accurate estimate for sizing and designing an OWTS. Metering daily water use and analyzing a set of grab samples to confirm wastewater strength estimates are often substituted for direct measurement of concentrations because of cost considerations.

Minimum septic tank size requirements or minimum design flows for a residential dwelling may be specified by onsite codes (NSFC, 1995). Such stipulations should incorporate methods for the conservative prediction of wastewater flow. It is important that realistic values and safety factors be used to determine wastewater characteristics in order to design the most cost-effective onsite system that meets performance requirements.

Factors of safety can be applied indirectly by the choice of design criteria for wastewater characteristics and occupancy patterns or directly through an overall factor. Most onsite code requirements for system design of residential dwellings call for estimating the flow on a per person or per bedroom basis. Codes typically specify design flows of 100 to 150 gallons/bedroom/day (378 to 568 liters/bedroom/day), or 75 to 100 gallons/person/day (284 to 378 liters/person/day), with occupancy rates of between 1.5 and 2 persons/bedroom (NSFC, 1995).

For example, if an average daily flow of 75 gallons/person/day (284 liters/person/day) and an occupancy rate of 2 persons per bedroom were the selected design units, the flow prediction for a three-bedroom home would include a factor of safety of approximately 2 when compared to typical conditions (i.e., 70 gallons/person/day and 1 person/bedroom). In lieu of using conservative design flows, a direct factor of safety (e.g., 2) may be applied to estimate the design flow from a residence or nonresidential establishment. Multiplying the typical flow estimated (140 gallons/day) by a safety factor of 2 yields a design flow of 280 gallons/day (1,058 liters/day). Factors of safety used for individual systems will usually be higher than those used for larger systems of 10 homes or more.

Great care should be exercised in predicting wastewater characteristics so as not to accumulate multiple factors of safety that would yield unreasonably high design flows and result in unduly high capital costs. Conversely, underestimating flows should be avoided because the error will quickly become apparent if the system overloads and requires costly modification.

## 3.6 Integrating wastewater characterization and other design information

Predicting wastewater characteristics for typical residential and nonresidential establishments can be a difficult task. Following a logical step-by-step procedure can help simplify the characterization process and yield more accurate wastewater characteristic estimates. Figure 3-7 is a flow chart that illustrates a procedure for predicting wastewater characteristics. This strategy takes the reader through the characterization process as it has been described in this chapter. The reader is cautioned that this flowchart is provided to illustrate one simple strategy for predicting wastewater characteristics. Additional factors to consider, such as discrepancies between literature values for wastewater flow and quality and/or the need to perform field studies, should be addressed based on local conditions and regulatory requirements.

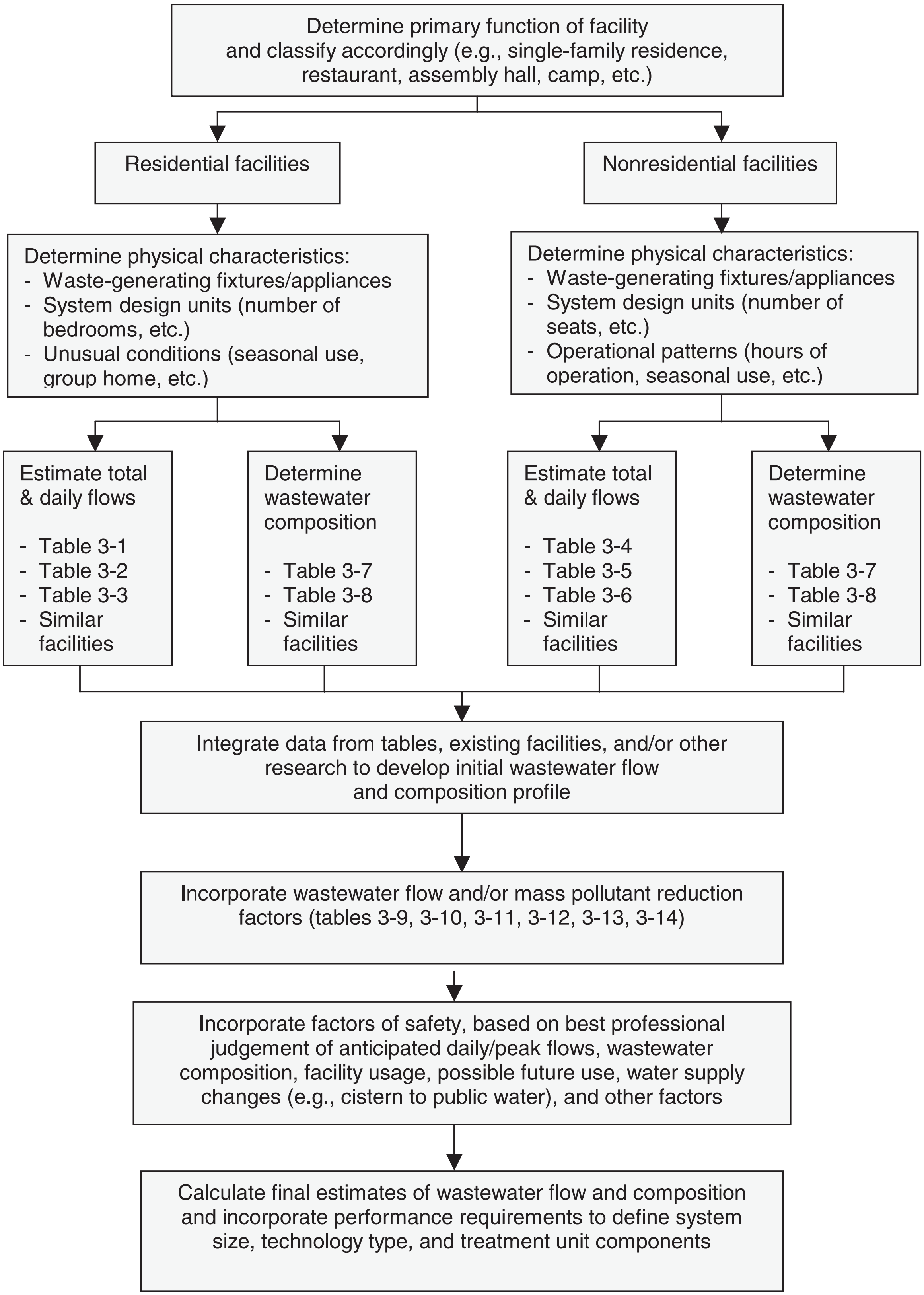
In designing wastewater treatment systems, it is recommended that designers consider the most significant or limiting parameters, including those that may be characterized as outliers, when considering hydraulic and mass pollutant treatment requirements and system components. For example, systems that will treat wastewaters with typical mass pollutant loads but hydraulic loads that exceed typical values should be designed to handle the extra hydraulic input. Systems designed for facilities with typical hydraulic loads but atypical mass pollutant loads (e.g., restaurants, grocery stores, or other facilities with high-strength wastes) should incorporate pretreatment units that address the additional pollutant loadings, such as grease traps.

## 3.7 Transport and fate of wastewater pollutants in the receiving environment

Nitrate, phosphorus, pathogens, and other contaminants are present in significant concentrations in most wastewaters treated by onsite systems. Although most can be removed to acceptable levels under optimal system operational and performance conditions, some may remain in the effluent exiting the system. After treatment and percolation of the wastewater through the infiltrative surface biomat and passage through the first few inches of soil, the wastewater plume begins to migrate downward until nearly saturated conditions exist. The worst case scenario occurs when the plume is mixing with an elevated water table. At that point, the wastewater plume will move in response to the prevailing hydraulic gradient, which might be lateral, vertical, or even a short distance upslope if ground water mounding occurs (figure 3-8). Moisture potential, soil conductivity, and other soil and geological characteristics determine the direction of flow.

Further treatment occurs as the plume passes through the soil. The degree of this additional treatment depends on a host of factors (e.g., residence time, soil mineralogy, particle sizes). Permit writers should consider not only the performance of each individual onsite system but also the density of area systems and overall hydraulic loading, the proximity of water resources, and the collective performance of onsite systems in the watershed. Failure to address these issues can lead ultimately to contamination of lakes, rivers, streams, wetlands, coastal areas, or ground water. This section examines key wastewater pollutants, their impact on human health and water resources, how they move in the environment, and how local ecological conditions affect wastewater treatment.

**Figure 3-7. Strategy for estimating wastewater flow and composition**



### 3.7.1 Wastewater pollutants of concern

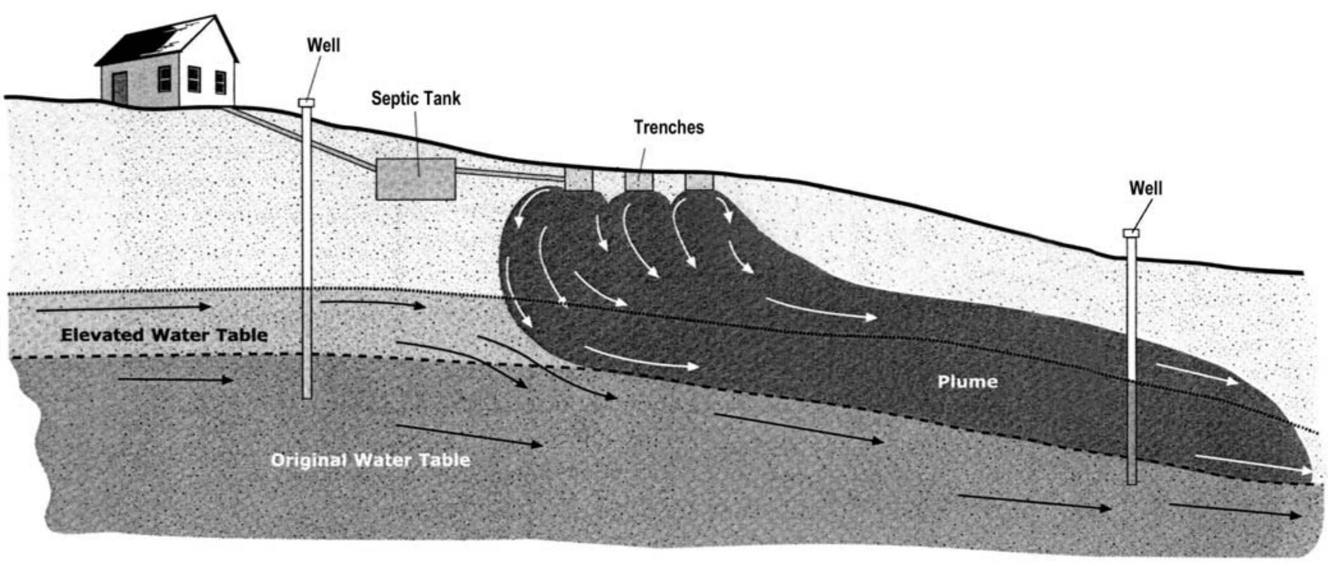
Environmental protection and public health agencies are becoming increasingly concerned about ground water and surface water contamination from wastewater pollutants. Toxic compounds, excessive nutrients, and pathogenic agents are among the potential impacts on the environment from onsite wastewater systems. Domestic wastewater contains several pollutants that could cause significant human health or environmental risks if not treated effectively before being released to the receiving environment.

A conventional OWTS (septic tank and SWIS) is capable of nearly complete removal of suspended solids, biodegradable organic compounds, and fecal coliforms if properly designed, sited, installed, operated, and maintained (USEPA, 1980a, 1997). These wastewater constituents can become pollutants in ground water or surface waters if treatment is incomplete. Research and monitoring studies have demonstrated removals of these typically found constituents to acceptable levels. More recently, however, other pollutants present in wastewater are raising concerns, including nutrients (e.g., nitrogen and phosphorus), pathogenic parasites (e.g., *Cryptosporidum parvum, Giardia lamblia*), bacteria and viruses, toxic organic compounds, and metals. Their potential impacts on ground water and surface water resources are summarized in table 3-16. Recently, concerns have been raised over the movement and fate of a variety of endocrine disrupters, usually from use of pharmaceuticals by residents. No data have been developed to confirm a risk at this time.

### 3.7.2 Fate and transport of pollutants in the environment

When properly designed, sited, constructed, and maintained, conventional onsite wastewater treatment technologies effectively reduce or eliminate most human health or environmental threats posed by pollutants in wastewater (table 3-17). Most traditional systems rely primarily on physical, biological, and chemical processes in the septic tank and in the biomat and unsaturated soil zone below the SWIS (commonly referred to as a leach field or drain field) to sequester or attenuate pollutants of concern. Where point discharges to surface waters are permitted, pollutants of concern should be removed or treated to acceptable, permit-specific levels (levels permitted under the National Pollutant Discharge Elimination System of the Clean Water Act) before discharge.

**Figure 3-8. Plume movement through the soil to the saturated zone.**




Source: Adapted from NSFC, 2000.

**Table 3-16. Typical wastewater pollutants of concern**

| Pollutant | Reason for concern |
|---|---|
| Total suspended solids (TSS) and turbidity (NTU) | In surface waters, suspended solids can result in the development of sludge deposits that smother benthic macroinvertebrates and fish eggs and can contribute to benthic enrichment, toxicity, and sediment oxygen demand. Excessive turbidity (colloidal solids that interfere with light penetration) can block sunlight, harm aquatic life (e.g., by blocking sunlight needed by plants), and lower the ability of aquatic plants to increase dissolved oxygen in the water column. In drinking water, turbidity is aesthetically displeasing and interferes with disinfection. |
| Biodegradable organics (BOD) | Biological stabilization of organics in the water column can deplete dissolved oxygen in surface waters, creating anoxic conditions harmful to aquatic life. Oxygen-reducing conditions can also result in taste and odor problems in drinking water. |
| Pathogens | Parasites, bacteria, and viruses can cause communicable diseases through direct/indirect body contact or ingestion of contaminated water or shellfish. A particular threat occurs when partially treated sewage pools on ground surfaces or migrates to recreational waters. Transport distances of some pathogens (e.g., viruses and bacteria) in ground water or surface waters can be significant. |
| Nitrogen | Nitrogen is an aquatic plant nutrient that can contribute to eutrophication and dissolved oxygen loss in surface waters, especially in lakes, estuaries, and coastal embayments. Algae and aquatic weeds can contribute trihalomethane (THM) precursors to the water column that may generate carcinogenic THMs in chlorinated drinking water. Excessive nitrate-nitrogen in drinking water can cause methemoglobinemia in infants and pregnancy complications for women. Livestock can also suffer health impacts from drinking water high in nitrogen. |
| Phosphorus | Phosphorus is an aquatic plant nutrient that can contribute to eutrophication of inland and coastal surface waters and reduction of dissolved oxygen. |
| Toxic organics | Toxic organic compounds present in household chemicals and cleaning agents can interfere with certain biological processes in alternative OWTSs. They can be persistent in ground water and contaminate downgradient sources of drinking water. They can also cause damage to surface water ecosystems and human health through ingestion of contaminated aquatic organisms (e.g., fish, shellfish). |
| Heavy metals | Heavy metals like lead and mercury in drinking water can cause human health problems. In the aquatic ecosystem, they can also be toxic to aquatic life and accumulate in fish and shellfish that might be consumed by humans. |
| Dissolved inorganics | Chloride and sulfide can cause taste and odor problems in drinking water. Boron, sodium, chlorides, sulfate, and other solutes may limit treated wastewater reuse options (e.g., irrigation). Sodium and to a lesser extent potassium can be deleterious to soil structure and SWIS performance. |

Source: Adapted in part from Tchobanoglous and Burton, 1991.

**Table3-17. Examples of soil infiltration system performance**

| Parameter | Applied concentration in milligrams per liter | Percent removal | References |
|---|---|---|---|
| $BOD_5$ | 130–150 | 90–98 | Siegrist et al., 1986<br>U. Wisconsin,1978 |
| Total nitrogen | 45–55 | 10–40 | Reneau 1977<br>Sikora et al., 1976 |
| Total phosphorus | 8–12 | 85–95 | Sikora et al., 1976 |
| Fecal coliforms | NA[a] | 99–99.99 | Gerba, 1975 |

[a] Fecal coliforms are typically measured in other units, e.g., colony-forming units per 100 milliliters.

Source: Adapted from USEPA, 1992.

Onsite systems can fail to meet human health and water quality objectives when fate and transport of potential pollutants are not properly addressed. Failing or failed systems threaten human health if pollutants migrate into ground waters used as drinking water and nearby surface waters used for recreation. Such failures can be due to improper siting, inappropriate choice of technology, faulty design, poor installation practices, poor operation, or inadequate maintenance. For example, in high-density subdivisions conventional septic tank/SWIS systems might be an inappropriate choice of technology because leaching of nitrate-nitrogen could result in nitrate concentrations in local aquifers that exceed the drinking water standard. In soils with excessive permeability or shallow water tables, inadequate treatment in the unsaturated soil zone might allow pathogenic bacteria and viruses to enter the ground water if no mitigating measures are taken. Poorly drained soils can restrict reoxygenation of the subsoil and result in clogging of the infiltrative surface.

A number of factors influence the shape and movement of contaminant plumes from OWTSs. Climate, soils, slopes, landscape position, geology, regional hydrology, and hydraulic load determine whether the plume will disperse broadly and deeply or, more commonly, migrate in a long and relatively narrow plume along the upper surface of a confining layer or on the surface of the ground water. Analyses of these factors are key elements in understanding the contamination potential of individual or clustered OWTSs in a watershed or ground water recharge area.

## Receiving environments and contaminant plume transport

Most onsite systems ultimately discharge treated water to ground water. Water beneath the land surface occurs in two primary zones, the aerated or vadose zone and the saturated (groundwater) zone. Interstices in the aerated (upper) vadose zone are unsaturated, filled partially with water and partially with air. Water in this unsaturated zone is referred to as vadose water. In the saturated zone, all interstices are filled with water under hydrostatic pressure. Water in this zone is commonly referred to as ground water. Where no overlying impermeable barrier exists, the upper surface of the ground water is called the water table. Saturation extends slightly above the water table due to capillary attraction but water in this "capillary fringe" zone is held at less than atmospheric pressure.

Onsite wastewater treatment system performance should be measured by the ability of the system to discharge a treated effluent capable of meeting public health and water quality objectives established for the receiving water resource. Discharges from existing onsite systems are predominantly to ground water but they might involve direct (point source) or indirect (nonpoint source) surface water discharges in some cases. Ground water discharges usually occur through soil infiltration. Point source discharges are often discouraged by regulatory agencies because of the difficulty in regulating many small direct, permitted discharges and the potential for direct or indirect human contact with wastewater. Nonpoint source surface water discharges usually occur as base flow from ground water into watershed surface waters. In some cases regional ground water quality and drinking water wells might be at a lesser risk from OWTS discharges than nearby surface waters because of the depth of some aquifers and regional geology.

The movement of subsurface aqueous contaminant plumes is highly dependent on soil type, soil layering, underlying geology, topography, and rainfall. Some onsite system setback/separation codes are based on plume movement models or measured relationships that have not been supported by recent field data. In regions with moderate to heavy rainfall, effluent plumes descend relatively intact as the water table is recharged from above. The shape of the plume depends on the soil and geological factors noted above, the uniformity of effluent distribution in the SWIS, the orientation of the SWIS with respect to ground water flow and direction, and the preferential flow that occurs in the vadose and saturated zones (Otis, 2000).

In general, however, plumes tend to be long, narrow, and definable, exhibiting little dispersion (figure 3-9). Some studies have found SWIS plumes with nitrate levels exceeding drinking water standards (10 mg/L) extending more than 328 feet (100 meters) beyond the SWIS (Robertson, 1995). Mean effluent plume dispersion values used in a Florida study to assess subdivision SWIS nitrate loadings over 5 years were 60 feet, 15 feet, and 1.2 feet for longitudinal, lateral, and vertical disper-

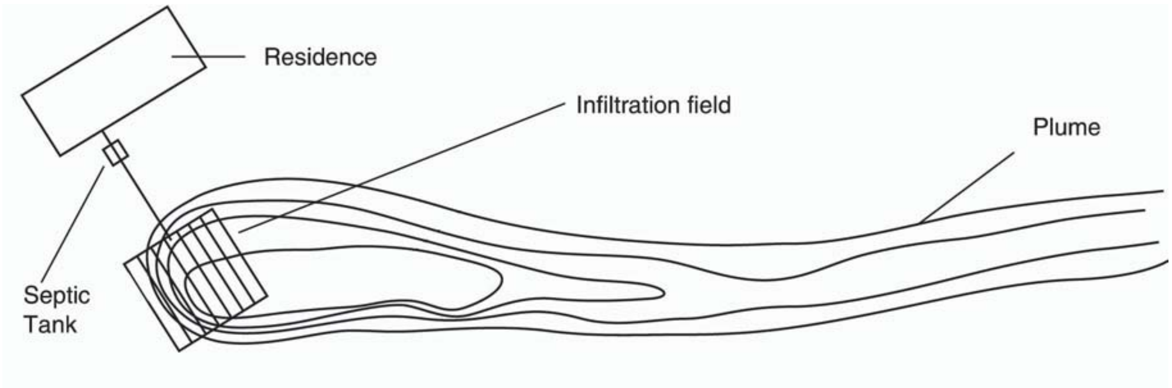



**Figure 3-9. An example of effluent plume movement**
Source: NSFC, 1998.

respectively (Florida HRS, 1993). A study that examined SWIS plume movement in a shallow, unconfined sand aquifer found that after 12 years the plume had sharp lateral and vertical boundaries, a length of 426 feet (130 meters), and a uniform width of about 32.8 feet (10 meters) (Robertson, 1991). At another site examined in that study, a SWIS constructed in a similar carbonate-depleted sand aquifer generated a plume with discrete boundaries that began discharging into a river 65.6 feet (20 meters) away after 1.5 years of system operation.

Given the tendency of OWTS effluent plumes to remain relatively intact over long distances (more than 100 meters), dilution models commonly used in the past to calculate nitrate attenuation in the vadose zone are probably unrealistic (Robertson, 1995). State codes that specify 100-foot separation distances between conventional SWIS treatment units and downgradient wells or surface waters should not be expected to always protect these resources from dissolved, highly mobile contaminants such as nitrate (Robertson, 1991). Moreover, published data indicate that viruses that reach groundwater can travel at least 220 feet (67 meters) vertically and 1,338 feet (408 meters) laterally in some porous soils and still remain infective (Gerba, 1995). One study noted that fecal coliform bacteria moved 2 feet (0.6 meter) downward and 50 feet (15 meters) longitudinally 1 hour after being injected into a shallow trench in saturated soil on a 14 percent slope in western Oregon (Cogger, 1995). Contaminant plume movement on the surface of the saturated zone can be rapid, especially under sloping conditions, but it typically slows upon penetration into ground water in the saturated zone. Travel times and distances under unsaturated conditions in more level terrain are likely much less.

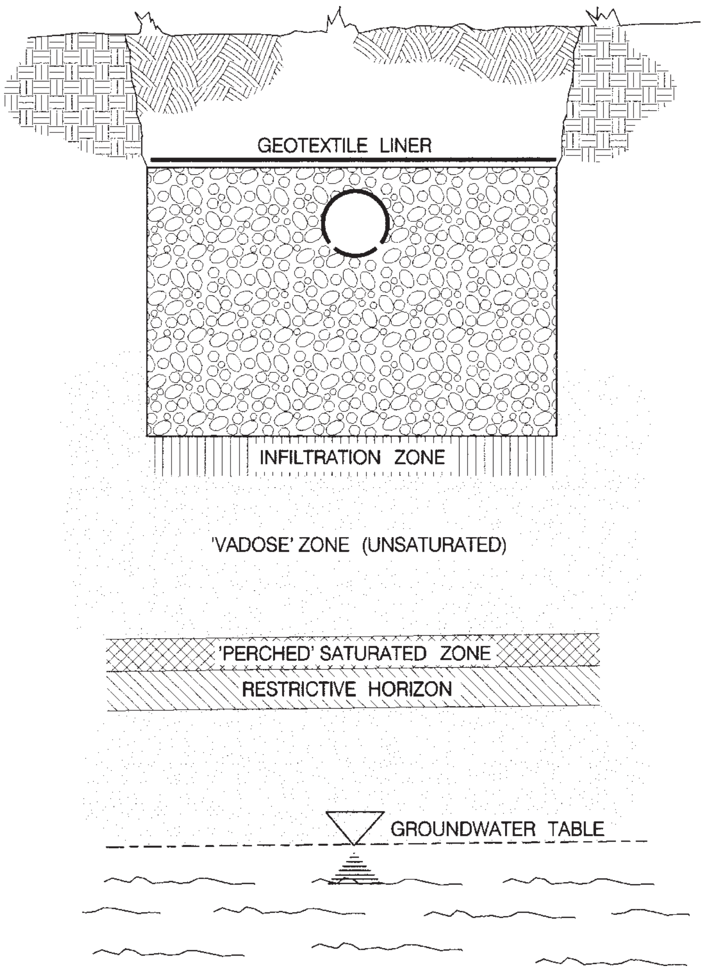
## Ground water discharge

A conventional OWTS (septic tank and SWIS) discharges to ground water and usually relies on the unsaturated or vadose zone for final polishing of the wastewater before it enters the saturated zone. The septic tank provides primary treatment of the wastewater, removing most of the settleable solids, greases, oils, and other floatable matter and anaerobic liquifaction of the retained organic solids. The biomat that forms at the infiltrative surface and within the first few centimeters of unsaturated soil below the infiltrative field provides physical, chemical, and biological treatment of the SWIS effluent as it migrates toward the ground water.

Because of the excellent treatment the SWIS provides, it is a critical component of onsite systems that discharge to ground water. Fluid transport from the infiltrative surface typically occurs through three zones, as shown in figure 3-10 (Ayres Associates, 1993a). In addition to the three zones, the figure shows a saturated zone perched above a restrictive horizon, a site feature that often occurs.

Pretreated wastewater enters the SWIS at the surface of the infiltration zone. A biomat forms in this zone, which is usually only a few centimeters thick. Most of the physical, chemical, and biological treatment of the pretreated effluent occurs in this zone and in the vadose zone. Particulate matter in the effluent accumulates on the infiltration surface and within the pores of the soil matrix, providing a

**Figure 3-10. Soil treatment zones**




Source: Ayres Associates, 1993a.

source of carbon and nutrients to the active biomass. New biomass and its metabolic by-products accumulate in this zone. The accumulated biomass, particulate matter, and metabolic by-products reduce the porosity and the infiltration rate through them. Thus, the infiltration zone is a transitional zone where fluid flow changes from saturated to unsaturated flow. The biomat controls the rate at which the pretreated wastewater moves through the infiltration zone in coarse- to medium-textured soils, but it is less likely to control the flow through fine-textured silt and clay soils because they may be more restrictive to flow than the biomat.

Below the zone of infiltration lies the unsaturated or vadose zone. Here the effluent is under a negative pressure potential (less than atmospheric) resulting from the capillary and adsorptive forces of the soil matrix. Consequently, fluid flow occurs over the surfaces of soil particles and through finer pores of the soil while larger pores usually remain air-filled. This is the most critical fluid transport zone because the unsaturated soil allows air to diffuse into the open soil pores to supply oxygen to the microbes that grow on the surface of the soil particles. The negative soil moisture potential forces the wastewater into the finer pores and over the surfaces of the soil particles, increasing retention time, absorption, filtration, and biological treatment of the wastewater.

From the vadose zone, fluid passes through the capillary fringe immediately above the ground water and enters the saturated zone, where flow occurs in response to a positive pressure gradient. Treated wastewater is transported from the site by fluid movement in the saturated zone. Mixing of treated water with ground water is somewhat limited because ground water flow usually is laminar. As a result, treated laminar water can remain as a distinct plume at the ground water interface for some distance from its source (Robertson et al., 1989). The plume might descend into the ground water as it travels from the source because of recharge from precipitation above. Dispersion occurs, but the mobility of solutes in the plume varies with the soil-solute reactivity.

Water quality-based performance requirements for ground water discharging systems are not clearly defined by current codes regulating OWTSs. Primary drinking water standards are typically required at a point of use (e.g., drinking water well) but are addressed in the codes only by requirements that the infiltration system be located a specified horizontal distance from the wellhead and vertical distance from the seasonal high water table. Nitrate-nitrogen is the common drinking water pollutant of concern that is routinely found in ground water below conventional SWISs. Regions with karst terrain or sandy soils are at particular risk for rapid movement of bacteria, viruses, nitrate-nitrogen, and other pollutants to ground water. In addition, geological conditions that support "gaining streams" (streams fed by ground water during low-flow conditions) might result in OWTS nutrient or pathogen impacts on surface waters if siting or design criteria fail to consider these conditions.

## Surface water discharge

Direct discharges to surface waters require a permit issued under the National Pollutant Discharge Elimination System (NPDES) of the Clean Water

Act. The NPDES permitting process, which is administered by all but a few states, defines discharge performance requirements in the form of numerical criteria for specific pollutants and narrative criteria for parameters like color and odor. The treated effluent should meet water quality criteria before it is discharged. Criteria-based standards may include limits for $BOD_5$, TSS, fecal coliforms, ammonia, nutrients, metals, and other pollutants, including chlorine, which is often used to disinfect treated effluent prior to discharge. The limits specified vary based on the designated use of the water resource (e.g., swimming, aquatic habitat, recreation, potable water supply), state water classification schemes (Class I, II, III, etc.), water quality criteria associated with designated uses, or the sensitivity of aquatic ecosystems—especially lakes and coastal areas—to eutrophication. Surface water discharges are often discouraged for individual onsite treatment systems, however, because of the difficulty in achieving regulatory oversight and surveillance of many small, privately operated discharges.

### Atmospheric discharge

Discharges to the atmosphere also may occur through evaporation and transpiration by plants. Evapotranspiration can release significant volumes of water into the atmosphere, but except for areas where annual evaporation exceeds precipitation (e.g., the American Southwest), evapotranspiration cannot be solely relied on for year-round discharge. However, evapotranspiration during the growing season can significantly reduce the hydraulic loading to soil infiltration systems.

### Contaminant attenuation

Performance standards for ground water discharge systems are usually applied to the treated effluent/ground water mixture at some specified point away from the treatment system (see chapter 5). This approach is significantly different from the effluent limitation approach used with surface water discharges because of the inclusion of the soil column as part of the treatment system. However, monitoring ground water quality as a performance measure is not as easily accomplished. The fate and transport of wastewater pollutants through soil should be accounted for in the design of the overall treatment system.

Contaminant attenuation (removal or inactivation through treatment processes) begins in the septic tank and continues through the distribution piping of the SWIS or other treatment unit components, the infiltrative surface biomat, the soils of the vadose zone, and the saturated zone. Raw wastewater composition was discussed in section 3.4 and summarized in table 3-7. Jantrania (1994) found that chemical, physical, and biological processes in the anaerobic environment of the septic tank produce effluents with TSS concentrations of 40 to 350 mg/L, oil and grease levels of 50 to 150 mg/L, and total coliform counts of $10^6$ to $10^8$ per 100 milliliters. Although biofilms develop on exposed surfaces as the effluent passes through piping to and within the SWIS, no significant level of treatment is provided by these growths. The next treatment site is the infiltrative zone, which contains the biomat. Filtration, microstraining, and aerobic biological decomposition processes in the biomat and infiltration zone remove more than 90 percent of the BOD and suspended solids and 99 percent of the bacteria (University of Wisconsin, 1978).

As the treated effluent passes through the biomat and into the vadose and saturated zones, other treatment processes (e.g., filtration, adsorption, precipitation, chemical reactions) occur. The following section discusses broadly the transport and fate of some of the primary pollutants of concern under the range of conditions found in North America. Table 3-18 summarizes a case study that characterized the septic tank effluent and soil water quality in the first 4 feet of a soil treatment system consisting of fine sand. Results for other soil types might be significantly different. Note that mean nitrate concentrations still exceed the 10 mg/L drinking water standard even after the wastewater has percolated through 4 feet of fine sand under unsaturated conditions.

### Biochemical oxygen demand and total suspended solids

Biodegradable organic material creates biochemical oxygen demand (BOD), which can cause low dissolved oxygen concentrations in surface water, create taste and odor problems in well water, and cause leaching of metals from soil and rock into ground water and surface waters. Total suspended solids (TSS) in system effluent can clog the infiltrative surface or soil interstices, while colloidal solids

**Table 3-18. Case study: septic tank effluent and soil water quality [a]**

| Parameter (units) | Statistics | Septic tank effluent quality | Soil water quality [b] at 0.6 meter | Soil water Quality[b] at 1.2 meters |
|---|---|---|---|---|
| BOD (mg/L) | Mean | 93.5 | <1 | <1 |
| | Range | 46–156 | <1 | <1 |
| | # samples | 11 | 6 | 6 |
| TOC (mg/L) | Mean | 47.4 | 7.8 | 8.0 |
| | Range | 31–68 | 3.7–17.0 | 3.1–25.0 |
| | # samples | 11 | 34 | 33 |
| TKN (mg/L) | Mean | 44.2 | 0.77 | 0.77 |
| | Range | 19–53 | 0.40–1.40 | 0.25–2.10 |
| | # samples | 11 | 35 | 33 |
| $NO_3$-N (mg/L) | Mean | 0.04 | 21.6 | 13.0 |
| | Range | 0.01–0.16 | 1.7–39.0 | 2.0–29.0 |
| | # samples | 11 | 35 | 32 |
| TP (mg/L) | Mean | 8.6 | 0.40 | 0.18 |
| | Range | 7.2–17.0 | 0.01–3.8 | 0.02–1.80 |
| | # samples | 11 | 35 | 33 |
| TDS (mg/L) | Mean | 497 | 448 | 355 |
| | Range | 354–610 | 184–620 | 200–592 |
| | # samples | 11 | 34 | 32 |
| Cl (mg/L) | Mean | 70 | 41 | 29 |
| | Range | 37–110 | 9–65 | 9–49 |
| | # samples | 11 | 34 | 31 |
| F. Coli (log # per 100 mL) | Mean | 4.57 | nd [c] | nd |
| | Range | 3.6–5.4 | <1 | <1 |
| | # samples | 11 | 24 | 21 |
| F. strep. (log # per 100 mL) | Mean | 3.60 | nd | nd |
| | Range | 1.9–5.3 | <1 | <1 |
| | # samples | 11 | 23 | 20 |

[a] The soil matrix consisted of a fine sand; the wastewater loading rate was 3.1 cm per day over 9 months. TOC = total organic carbon; TKN = total Kjeldahl nitrogen; TDS = total dissolved solids; Cl = chlorides;
F. coli = fecal coliforms; F. strep = fecal streptococci.
[b] Soil water quality measured in pan lysimeters at unsaturated soil depths of 2 feet (0.6 meter) and 4 feet (1.2 meters).
[c] nd = none detected.

Source: Adapted from Anderson et al., 1994.

cause cloudiness in surface waters. TSS in direct discharges to surface waters can result in the development of sludge layers that can harm aquatic organisms (e.g., benthic macro invertebrates). Systems that fail to remove BOD and TSS and are located near surface waters or drinking water wells may present additional problems in the form of pathogens, toxic pollutants, and other pollutants.

Under proper site and operating conditions, however, OWTSs can achieve significant removal rates (i.e., greater than 95 percent) for biodegradable organic compounds and suspended solids. The risk of ground water contamination by BOD and TSS (and other pollutants associated with suspended solids) below a properly sited, designed, constructed, and maintained SWIS is slight (Anderson et al., 1994; University of Wisconsin, 1978). Most settleable and floatable solids are removed in the septic tank during pretreatment. Most particulate BOD remaining is effectively removed at the infiltrative surface and biomat. Colloidal and dissolved BOD that might pass through the biomat are removed through aerobic biological processes in the vadose zone, especially when uniform dosing and reoxygenation occur. If excessive concentrations of BOD and TSS migrate beyond the tank because of poor maintenance, the infiltrative surface can clog and surface seepage of wastewater or plumbing fixture backup can occur.

## Nitrogen

Nitrogen in raw wastewater is primarily in the form of organic matter and ammonia. After the septic tank, it is primarily (more than 85 percent) ammonia. After discharge of the effluent to the infiltrative surface, aerobic bacteria in the biomat and upper vadose zone convert the ammonia in the effluent almost entirely to nitrite and then to nitrate. Nitrogen in its nitrate form is a significant ground water pollutant. It has been detected in urban and rural ground water nationwide, sometimes at levels exceeding the USEPA drinking water standard of 10 mg/L (USGS, 1999). High concentrations of nitrate (greater than 10 mg/L) can cause methemoglobinemia or "blue baby syndrome," a disease in infants that reduces the blood's ability to carry oxygen, and problems during pregnancy. Nitrogen is also an important plant nutrient that can cause excessive algal growth in nitrogen-limited inland (fresh) waters and coastal waters, which are often limited in available nitrogen. High algal productivity can block sunlight, create nuisance or harmful algal blooms, and significantly alter aquatic ecosystems. As algae die, they are decomposed by bacteria, which can deplete available dissolved oxygen in surface waters and degrade habitat conditions.

Nitrogen contamination of ground water below infiltration fields has been documented by many investigators (Anderson et al., 1994; Andreoli et al., 1979; Ayres Associates, 1989, 1993b, c; Bouma et al., 1972; Carlile et al., 1981; Cogger and

**Table 3-19. Wastewater constituents of concern and representative concentrations in the effluent of various treatment units**

| Constituents of concern | Example direct or indirect measures (Units) | Tank-based treatment unit effluent concentrations | | | | | SWIS percolate into ground water at 3 to 5 ft depth (% removal) |
|---|---|---|---|---|---|---|---|
| | | Domestic STE[1] | Domestic STE with N-removal recycle[2] | Aerobic unit effluent | Sand filter effluent | Foam or textile filter effluent | |
| Oxygen demand | $BOD_5$ (mg/L) | 140-200 | 80-120 | 5-50 | 2-15 | 5-15 | >90% |
| Particulate solids | TSS (mg/L) | 50-100 | 50-80 | 5-100 | 5-20 | 5-10 | >90% |
| Nitrogen | Total N (mg N/L) | 40-100 | 10-30 | 25-60 | 10-50 | 30-60 | 10-20% |
| Phosphorus | Total P (mg P/L) | 5-15 | 5-15 | 4-10 | <1-10[4] | 5-15[4] | 0-100% |
| Bacteria (e.g., *Clostridium perfringens, Salmonella, Shigella*) | Fecal coliform (organisms per 100 mL) | $10^6$-$10^8$ | $10^6$-$10^8$ | $10^3$-$10^4$ | $10^1$-$10^3$ | $10^1$-$10^3$ | >99.99% |
| Virus (e.g., hepatitis, polio, echo, coxsackie, coliphage) | Specific virus (pfu/mL) | 0-$10^5$ (episodically present at high levels) | 0-$10^5$ (episodically present at high levels) | 0-$10^5$ (episodically present at high levels) | 0-$10^5$ (episodically present at high levels) | 0-$10^5$ (episodically present at high levels) | >99.9% |
| Organic chemicals (e.g., solvents, petro-chemicals, pesticides) | Specific organics or totals (µg/L) | 0 to trace levels (?) | 0 to trace levels (?) | 0 to trace levels (?) | 0 to trace levels (?) | 0 to trace levels (?) | >99% |
| Heavy metals (e.g., Pb, Cu, Ag, Hg) | Individual metals (µg/L) | 0 to trace levels | 0 to trace levels | 0 to trace levels | 0 to trace levels | 0 to trace levels | >99% |

[1] Septic tank effluent (STE) concentrations given are for domestic wastewater. However, restaurant STE is markedly higher particularly in $BOD_5$, COD, and suspended solids while concentrations in graywater STE are noticeably lower in total nitrogen.
[2] N-removal accomplished by recycling STE through a packed bed for nitrification with discharge into the influent end of the septic tank for denitrification.
[3] P-removal by adsorption/precipitation is highly dependent on media capacity, P loading, and system operation.
Source: Siegrist, 2001 (after Siegrist et al., 2000)

Source: Siegrist, 2001 (after Siegrist et al., 2000).

Carlile, 1984; Ellis and Childs, 1973; Erickson and Bastian, 1980; Gibbs, 1977a, b; Peavy and Brawner, 1979; Peavy and Groves, 1978; Polta, 1969; Preul, 1966; Reneau, 1977, 1979; Robertson et al., 1989, 1990; Shaw and Turyk, 1994; Starr and Sawhney, 1980; Tinker, 1991; Uebler, 1984; Viraraghavan and Warnock, 1976a, b, c; Walker et al., 1973a, b; Wolterink et al., 1979). Nitrate-nitrogen concentrations in ground water were usually found to exceed the drinking water standard of 10 mg/L near the infiltration field. Conventional soil-based systems can remove some nitrogen from septic tank effluent (table 3-19), but high-density installation of OWTSs can cause contamination of ground or surface water resources. When nitrate reaches the ground water, it moves freely with little retardation. Denitrification has been found to be significant in the saturated zone only in rare instances where carbon or sulfur deposits are present. Reduction of nitrate concentrations in ground water occurs primarily through dispersion or recharge of ground water supplies by precipitation (Shaw and Turyk, 1994).

Nitrogen can undergo several transformations in and below a SWIS, including adsorption, volatilization, mineralization, nitrification, and denitrification. Nitrification, the conversion of ammonium nitrogen to nitrite and then nitrate by bacteria under aerobic conditions, is the predominant transformation that occurs immediately below the infiltration zone. The negatively charged nitrate ion is very soluble and moves readily with the percolating soil water.

Biological denitrification, which converts nitrate to gaseous forms of nitrogen, can remove nitrate from percolating wastewater. Denitrification occurs under anaerobic conditions where available electron donors such as carbon or sulfur are present. Denitrifying bacteria use nitrate as a substitute for oxygen when accepting electrons. It has been generally thought that anaerobic conditions with organic matter seldom occur below soil infiltration fields. Therefore, it is has been assumed that all the nitrogen applied to infiltration fields ultimately leaches to ground water (Brown et al., 1978; Walker et al., 1973a, b). However, several studies indicate that denitrification can be significant. Jenssen and Siegrist (1990) found in their review of several laboratory and field studies that approximately 20 percent of nitrogen is lost from wastewater percolating through soil. Factors found to favor denitrification are fine-grained soils (silts and clays) and layered soils (alternating fine-grained and coarser-grained soils with distinct boundaries between the texturally different layers), particularly if the fine-grained soil layers contain organic material. Jenssen and Siegrist concluded that nitrogen removal below the infiltration field can be enhanced by placing the system high in the soil profile, where organic matter in the soil is more likely to be present, and by dosing septic tank effluent onto the infiltrative surface to create alternating wetting and drying cycles. Denitrification can also occur if ground water enters surface water bodies through organic-rich bottom sediments. Nitrogen concentrations in ground water were shown to decrease to less than 0.5 mg/L after passage through sediments in one Canadian study (Robertson et al., 1989, 1990).

It is difficult to predict removal rates for wastewater-borne nitrate or other nitrogen compounds in the soil matrix. In general, however, nitrate concentrations in SWIS effluent can and often do exceed the 10 mg/L drinking water standard. Shaw and Turyk (1994) found nitrate concentrations ranging from 21 to 108 mg/L (average of 31 to 34 mg/L) in SWIS effluent plumes analyzed as part of a study of 14 pressure-dosed drain fields in sandy soils of Wisconsin. The limited ability of conventional SWISs to achieve enhanced nitrate reductions and the difficulty in predicting soil nitrogen removal rates means that systems sited in drinking water aquifers or near sensitive aquatic areas should incorporate additional nitrogen removal technologies prior to final soil discharge.

## Phosphorus

Phosphorus is also a key plant nutrient, and like nitrogen it contributes to eutrophication and dissolved oxygen depletion in surface waters, especially fresh waters such as rivers, lakes, and ponds. Monitoring below subsurface infiltration systems has shown that the amount of phosphorus leached to ground water depends on several factors: the characteristics of the soil, the thickness of the unsaturated zone through which the wastewater percolates, the applied loading rate, and the age of the system (Bouma et al., 1972; Brandes, 1972; Carlile et al., 1981, Childs et al., 1974; Cogger and Carlile, 1984; Dudley and Stephenson, 1973; Ellis and Childs, 1973; Erickson and Bastian, 1980; Gilliom and Patmont, 1983; Harkin et al., 1979;

Jones and Lee, 1979; Whelan and Barrow, 1984). The amount of phosphorus in ground water varies from background concentrations to concentrations equal to that of septic tank effluent. However, removals have been found to continue within ground water aquifers (Carlile et al., 1981; Childs et al., 1974; Cogger and Carlile, 1984; Ellis and Childs, 1973; Gilliom and Patmont, 1983; Rea and Upchurch, 1980; Reneau, 1979; Reneau and Pettry, 1976; Robertson et al., 1990).

Retardation of phosphorus contamination of surface waters from SWISs is enhanced in fine-textured soils without continuous macropores that would allow rapid percolation. Increased distance of the system from surface waters is also an important factor in limiting phosphorus discharges because of greater and more prolonged contact with soil surfaces. The risk of phosphorus contamination, therefore, is greatest in karst regions and coarse-textured soils without significant iron, calcium, or aluminum concentrations located near surface waters.

The fate and transport of phosphorus in soils are controlled by sorption and precipitation reactions (Sikora and Corey, 1976). At low concentrations (less than 5 mg/L), the phosphate ion is chemisorbed onto the surfaces of iron and aluminum minerals in strongly acid to neutral systems and on calcium minerals in neutral to alkaline systems. As phosphorus concentrations increase, phosphate precipitates form. Some of the more important precipitate compounds formed are strengite, $FePO_4 \cdot 2H_2O$; variscite, $AlPO_4 \cdot 2H_2O$; dicalcium phosphate, $CaHPO_4 \cdot 2H_2O$; octacalcium phosphate, $Ca_4H(PO_4)_3 \cdot 3H_2O$; and hydroxyapatite, $Ca_{10}(PO_4)_6(OH_2)$. In acidic soils, phosphate sorption probably involves the aluminum and iron compounds; in calcareous or alkaline soils, calcium compounds predominate.

Estimates of the capacity of the soil to retain phosphorus are often based on sorption isotherms such as the Langmuir model (Ellis and Erickson, 1969; Sawney, 1977; Sawney and Hill, 1975; Sikora and Corey, 1976; Tofflemire and Chen, 1977). This method significantly underestimates the total retention capacity of the soil (Anderson et al., 1994; Sawney and Hill, 1975; Sikora and Corey, 1976; Tofflemire and Chen, 1977). This is because the test measures the chemi-sorption capacity but does not take into account the slower precipitation reactions that regenerate the chemi-sorption sites. These slower reactions have been shown to increase the capacity of the soil to retain phosphorus by 1.5 to 3 times the measured capacity calculated by the isotherm test (Sikora and Corey, 1976; Tofflemire and Chen, 1977). In some cases the total capacity has been shown to be as much as six times greater (Tofflemire and Chen, 1977). These reactions can take place in unsaturated or saturated soils (Ellis and Childs, 1973; Jones and Lee, 1977a, b; Reneau and Pettry, 1976; Robertson et al., 1990; Sikora and Corey, 1976).

The capacity of the soil to retain phosphorus is finite, however. With continued loading, phosphorus movement deeper into the soil profile can be expected. The ultimate retention capacity of the soil depends on several factors, including its mineralogy, particle size distribution, oxidation-reduction potential, and pH. Fine-textured soils theoretically provide more sorption sites for phosphorus. As noted above, iron, aluminum, and calcium minerals in the soil allow phosphorus precipitation reactions to occur, a process that can lead to additional phosphorus retention. Sikora and Corey (1976) estimated that phosphorus penetration into the soil below a SWIS would be 52 centimeters per year in Wisconsin sands and 10 centimeters per year in Wisconsin silt loams.

Nevertheless, knowing the retention capacity of the soil is not enough to predict the travel of phosphorus from subsurface infiltration systems. Equally important is an estimate of the total volume of soil that the wastewater will contact as it percolates to and through the ground water. Fine-textured, unstructured soils (e.g., clays, silty clays) can be expected to disperse the water and cause contact with a greater volume of soil than coarse, granular soils (e.g., sands) or highly structured fine-textured soils (e.g., clayey silts) having large continuous pores. Also, the rate of water movement and the degree to which the water's elevation fluctuates are important factors.

There are no simple methods for predicting phosphorus removal rates at the site level. However, several landscape-scale tools that provide at least some estimation of expected phosphorus loads from clusters of onsite systems are available. The MANAGE assessment method, which is profiled in section 3.9.1, is designed to estimate existing and projected future (build-out) nutrient loads and to identify "hot spots" based on land use and cover (see **http://www.epa.gov/owow/watershed/Proceed/joubert.html; http://www.edc.uri.edu/cewq/manage.html**). Such estimates provide at least some guidance in siting onsite systems and considering acceptable levels of both numbers and densities in sensitive areas.

## Pathogenic microorganisms

Pathogenic microorganisms found in domestic wastewater include a number of different bacteria, viruses, protozoa, and parasites that cause a wide range of gastrointestinal, neurological, respiratory, renal, and other diseases. Infection can occur through ingestion (drinking contaminated water; incidental ingestion while bathing, skiing, or fishing), respiration, or contact (table 3-20). The occurrence and concentration of pathogenic microorganisms in raw wastewater depend on the sources contributing to the wastewater, the existence of infected persons in the population, and environmental factors that influence pathogen survival rates. Such environmental factors include the following: initial numbers and types of organisms, temperature (microorganisms survive longer at lower temperatures), humidity (survival is longest at high humidity), amount of sunlight (solar radiation is detrimental to survival), and additional soil attenuation factors, as discussed below. Typical ranges of survival times are presented in table 3-21. Among pathogenic agents, only bacteria have any potential to reproduce and multiply between hosts (Cliver, 2000). If temperatures are between 50 and 80 degrees Fahrenheit (10 to 25 degrees Celsius)

**Table 3-20. Waterborne pathogens found in human waste and associated diseases**

| Type | Organism | Disease | Effects |
|---|---|---|---|
| Bacteria | *Escherichia coli* enteropathogenic) | Gastroenteritis | Vomiting, diarrhea, death in susceptible populations |
| | *Legionella pneumophila* | Legionellosis | Acute respiratory illness |
| | *Leptospira* | Leptospirosis | Jaundice, fever (Well's disease) |
| | *Salmonella typhi* | Typhoid fever | High fever, diarrhea, ulceration of the small intestine |
| | *Salmonella* | Salmonellosis | Diarrhea, dehydration |
| | *Shigella* | Shigellosis | Bacillary dysentery |
| | *Vibrio cholerae* | Cholera | Extremely heavy diarrhea, dehydration |
| | *Yersinia enterolitica* | Yersinosis | Diarrhea |
| Protozoans | *Balantidium coli* | Balantidiasis | Diarrhea, dysentery |
| | *Cryptosporidium* | Crypotosporidiosis | Diarrhea |
| | *Entamoeba histolytica* | Ameobiasis (amoebic dysentery) | Prolonged diarrhea with bleeding, abscesses of the liver and small intestine |
| | *Giardia lambia* | Giardiasis | Mild to severe diarrhea, nausea, indigestion |
| | *Naegleria fowleri* | Amebic Meningoencephalitis | Fatal disease; inflammation of the brain |
| Viruses | Adenovirus (31 types) | Conjunctivitis | Eye, other infections |
| | Enterovirus (67 types, e.g., polio-, echo-, and Coxsackie viruses) | Gastroenteritis | Heart anomalies, meningitis |
| | Hepatitis A | Infectious hepatitis | Jaundice, fever |
| | Norwalk agent | Gastroenteritis | Vomiting, diarrhea |
| | Reovirus | Gastroenteritis | Vomiting, diarrhea |
| | Rotavirus | Gastroenteritis | Vomiting, diarrhea |

Source: USEPA, 1999.

**Table 3-21. Typical pathogen survival times at 20 to 30 °C**

| Pathogen | Typical survival times in days: In fresh water & sewage | Typical survival times in days: In unsaturated soils |
|---|---|---|
| Viruses[a] | | |
| Enteroviruses[b] | < 120 but usually < 50 | < 100 but usually < 20 |
| Bacteria | | |
| Fecal coliforms[a] | < 60 but usually < 30 | < 70 but usually < 20 |
| *Salmonella* spp.[a] | < 60 but usually < 30 | < 70 but usually < 20 |
| *Shigella* spp.[a] | < 30 but usually < 10 | |
| Protozoa | | |
| *Entamoeba histolytica* cysts | < 30 but usually < 15 | < 20 but usually < 10 |
| Helminths | | |
| *Ascaris lumbricoides* eggs | Many months | Many months |

[a] In seawater, viral survival is less and bacterial survival is very much less than in fresh water.
[b] Includes polio-, echo-, and Coxsackie viruses.

Source: Adapted from Feacham et al., 1983, cited in UNDP-World Bank, 1992.

and nutrients are available, bacterial numbers may increase 10- to 100-fold. However, such multiplication is usually limited by competition from other, better-adapted organisms (Cliver, 2000).

Enteric bacteria are those associated with human and animal wastes. Once the bacteria enter a soil, they are subjected to life process stresses not encountered in the host. In most nontropical regions of the United States, temperatures are typically much lower; the quantity and availability of nutrients and energy sources are likely to be appreciably lower; and pH, moisture, and oxygen conditions are not as likely to be conducive to long-term survival. Survival times of enteric bacteria in the soil are generally reduced by higher temperatures, lower nutrient and organic matter content, acidic conditions (pH values of 3 to 5), lower moisture conditions, and the presence of indigenous soil microflora (Gerba et al., 1975). Potentially pathogenic bacteria are eliminated faster at high temperatures, pH values of about 7, low oxygen content, and high dissolved organic substance content (Pekdeger, 1984). The rate of bacterial die-off approximately doubles with each 10-degree increase of temperature between 5 and 30 °C (Tchobanoglous and Burton, 1991). Observed survival rates for various potential pathogenic bacteria have been found to be extremely variable. Survival times of longer than 6 months can occur at greater depths in unsaturated soils where oligotrophic (low-nutrient) conditions exist (Pekdeger, 1984).

The main methods of bacterial retention in unsaturated soil are filtration, sedimentation, and adsorption (Bicki et al., 1984; Cantor and Knox, 1985; Gerba et al., 1975). Filtration accounts for the most retention. The sizes of bacteria range from 0.2 to 5 microns (µm) (Pekdeger, 1984; Tchobanoglous and Burton, 1991); thus, physical removal through filtration occurs when soil micropores and surface water film interstices are smaller than this. Filtration of bacteria is enhanced by slow permeability rates, which can be caused by fine soil textures, unsaturated conditions, uniform wastewater distribution to soils, and periodic treatment system resting. Adsorption of bacteria onto clay and organic colloids occurs within a soil solution that has high ionic strength and neutral to slightly acid pH values (Canter and Knox, 1985).

Normal operation of septic tank/subsurface infiltration systems results in retention and die-off of most, if not all, observed pathogenic bacterial indicators within 2 to 3 feet (60 to 90 centimeters) of the infiltrative surface (Anderson et al., 1994; Ayres Associates, 1993a, c; Bouma et al., 1972; McGauhey and Krone, 1967). With a mature biomat at the infiltrative surface of coarser soils, most bacteria are removed within the first 1 foot (30 centimeters) vertically or horizontally from the trench-soil interface (University of Wisconsin, 1978). Hydraulic loading rates of less than 2 inches/day (5 centimeters/day) have also been found to promote better removal of bacteria in septic tank effluent (Ziebell et al., 1975). Biomat formation and lower hydraulic loading rates promote unsaturated flow, which is one key to soil-based removal of bacteria from wastewater. The retention behavior of actual pathogens in unsaturated soil might be different from that of the indicators (e.g., fecal coliforms) that have been measured in most studies.

Failure to properly site, design, install, and/or operate and maintain subsurface infiltration systems can result in the introduction of potentially pathogenic bacteria into ground water or surface waters. Literature reviews prepared by Hagedorn (1982) and Bicki et al. (1984) identify a number of references that provide evidence that infiltrative surfaces improperly constructed below the ground water surface or too near fractured bedrock correlate with such contamination. Karst geology and seasonally high water tables that rise into the infiltrative field can also move bacteria into ground water zones. Once in ground water, bacteria from septic tank effluent have been observed to survive for considerable lengths of time (7 hours to 63 days), and they can travel up to and beyond 100 feet (30 meters) (Gerba et al., 1975).

Viruses are not a normal part of the fecal flora. They occur in infected persons, and they appear in septic tank effluent intermittently, in varying numbers, reflecting the combined infection and carrier status of OWTS users (Berg, 1973). It is estimated that less than 1 to 2 percent of the stools excreted in the United States contain enteric viruses (University of Wisconsin, 1978). Therefore, such viruses are difficult to monitor and little is known about their frequency of occurrence and rate of survival in traditional septic tank systems. Once an infection (clinical or subclinical) has occurred, however, it is estimated that feces may contain $10^6$ to $10^{10}$ viral particles per gram (Kowal, 1982). Consequently, when enteric viruses are present in septic tank effluent, they might be present in significant numbers (Anderson et al., 1991; Hain and O'Brien, 1979; Harkin et al., 1979; Vaughn and Landry, 1977; Yeager and O'Brien, 1977).

Some reduction (less than 1 log) of virus concentrations in wastewater occurs in the septic tank. Higgins et al. (2000) reported a 74 percent decrease in MS2 coliphage densities, findings that concurs with those of other studies (Payment et al., 1986; Roa, 1981). Viruses can be both retained and inactivated in soil; however, they can also be retained but not inactivated. If not inactivated, viruses can accumulate in soil and subsequently be released due to changing conditions, such as prolonged peak OWTS flows or heavy rains. The result could be contamination of ground water. Soil factors that decrease survival include warm temperatures, low moisture content, and high organic content. Soil factors that increase retention include small particle size, high moisture content, low organic content, and low pH. Sobsey (1983) presents a thorough review of these factors. Virus removal below the vadose zone might be negligible in some geologic settings. (Cliver, 2000).

Most studies of the fate and transport of viruses in soils have been columnar studies using a specific serotype, typically poliovirus 1, or bacteriophages (Bitton et al., 1979; Burge and Enkiri, 1978; Drewry, 1969, 1973; Drewry and Eliassen, 1968; Duboise et al., 1976; Goldsmith et al., 1973; Green and Cliver, 1975; Hori et al., 1971; Lance et al., 1976; Lance et al., 1982; Lance and Gerba, 1980; Lefler and Kott, 1973, 1974; Nestor and Costin, 1971; Robeck et al., 1962; Schaub and Sorber, 1977; Sobsey et al., 1980; Young and Burbank, 1973; University of Wisconsin, 1978). The generalized results of these studies indicate that adsorption is the principal mechanism of virus retention in soil. Increasing the ionic strength of the wastewater enhances adsorption. Once viruses have been retained, inactivation rates range from 30 to 40 percent per day.

Various investigations have monitored the transport of viruses through unsaturated soil below the infiltration surface has been monitored by (Anderson et al., 1991; Hain and O'Brien, 1979; Jansons et al., 1989; Schaub and Sorber, 1977; Vaughn and Landry, 1980; Vaughn et al., 1981; Vaughn et al., 1982, 1983; Wellings et al., 1975). The majority of these studies focused on indigenous viruses in the wastewater and results were mixed. Some serotypes were found to move more freely than others. In most cases viruses were found to penetrate more than 10 feet (3 meters) through unsaturated soils. Viruses are less affected by filtration than bacteria (Bechdol et al., 1994) and are more resistant than bacteria to inactivation by disinfection (USEPA, 1990). Viruses have been known to persist in soil for up to 125 days and travel in ground water for distances of up to 1,339 feet (408 meters). However, monitoring of eight conventional individual home septic tank systems in Florida indicated that 2 feet (60 centimeters) of fine sand effectively removed viruses (Anderson et al., 1991; Ayres Associates, 1993c). Higgins (2000) reported 99 percent removal of virus particles within the first 1 foot (30.5 centimeters) of soil.

Recent laboratory and field studies of existing onsite systems using conservative tracers (e.g., bromide ions) and microbial surrogate measures (e.g., viruses, bacteria) found that episodic break-throughs of virus and bacteria can occur in the SWIS, particularly during early operation (Van Cuyk et al., 2001). Significant (e.g., 3-log) removal of viruses and near complete removal of fecal bacteria can be reasonably achieved in 60 to 90 centimeters of sandy media (Van Cuyk et al., 2001).

Inactivation of pathogens through other physical, chemical, or biological mechanisms varies considerably. Protozoan cysts or oocysts are generally killed when they freeze, but viruses are not. Ultraviolet light, extremes of pH, and strong oxidizing agents (e.g., hypochlorite, chlorine dioxide, ozone) are also effective in killing or inactivating most pathogens (Cliver, 2000). Korich (1990) found that in demand-free water, ozone was slightly more effective than chlorine dioxide against *Cryptosporidium parvum* oocysts, and both were much more effective than chlorine or monochloramine. *C. parvum* oocysts were found to be 30 times more resistant to ozone and 14 times more resistant to chlorine dioxide than are *Giardia lamblia* cysts (Korich et al., 1990).

## Toxic organic compounds

A number of toxic organic compounds that can cause neurological, developmental, or other problems in humans and interfere with biological processes in the environment can be found in septic tank effluent. Table 3-22 provides information on potential health effects from selected organic chemicals, along with USEPA maximum containment levels for these pollutants in drinking water. The toxic organics that have been found to be the most prevalent in wastewater are 1,4-dichlorobenzene, methylbenzene (toluene), dimethylbenzenes (xylenes), 1,1-dichloroethane, 1,1,1-trichloroethane, and dimethylketone (acetone). These compounds are usually found in household products like solvents and cleaners.

No known studies have been conducted to determine toxic organic treatment efficiency in single-family home septic tanks. A study of toxic organics in domestic wastewater and effluent from a community septic tank found that removal of low-molecular-weight alkylated benzenes (e.g., toluene,

**Table 3-22. Maximum contaminant levels (MCLs) for selected organic chemicals in drinking water**

| Contaminant | MCL (mg/L) | Potential health effects |
|---|---|---|
| Benzene | 0.005 | Anemia; decrease in blood platelets; increased risk of cancer |
| Chlordane | 0.002 | Liver or nervous system problems; increased risk of cancer |
| Chlorobenzene | 0.1 | Liver or kidney problems |
| 2,4-D | 0.07 | Liver, kidney, or adrenal gland problems |
| o-Dichlorobenzene | 0.6 | Liver, kidney, or circulatory system problems |
| 1,2-Dichloroethane | 0.005 | Increased risk of cancer |
| Dichloromethane | 0.005 | Liver problems, increased risk of cancer |
| Dioxin | 0.00000003 | Reproductive difficulties; increased risk of cancer |
| Ethylbenzene | 0.7 | Liver or kidney problems |
| Hexachlorobenzene | 0.001 | Liver or kidney problems; reproductive difficulties; increased risk of cancer |
| Lindane | 0.0002 | Liver or kidney problems |
| Toluene | 1.0 | Nervous system, kidney, or liver problems |
| Trichloroethylene | 0.005 | Liver problems; increased risk of cancer |
| Vinyl chloride | 0.002 | Increased risk of cancer |
| Xylenes (total) | 10 | Nervous system damage |

Source: USEPA, 2000a.

xylene) was noticeable, whereas virtually no removal was noted for higher-molecular-weight compounds (DeWalle et al., 1985). Removal efficiency was observed to be directly related to tank detention time, which is directly related to settling efficiency.

The behavior of toxic organic compounds in unsaturated soil is not well documented. The avenues of mobility available to toxic organics include those which can transport organics in both gaseous and liquid phases. In the gaseous phase toxic organics diffuse outward in any direction within unobstructed soil voids; in the liquid phase they follow the movement of the soil solution. Because of their nonpolar nature, certain toxic organics are not electrochemically retained in unsaturated soil. Toxic organics can be transformed into less innocuous forms in the soil by indigenous or introduced microorganisms. The biodegradability of many organic compounds in the soil depends on oxygen availability. Halogenated straight-chain compounds, such as many chlorinated solvents, are usually biodegraded under anaerobic conditions when carbon dioxide replaces oxygen (Wilhelm, 1998). Aromatic organic compounds like benzene and toluene, however, are biodegraded primarily under aerobic conditions. As for physical removal, organic contaminants are adsorbed by solid organic matter. Accumulated organic solids in the tank and in the soil profile, therefore, might be important retainers of organic contaminants. In addition, because many of the organic contaminants found in domestic wastewater are relatively volatile, unsaturated conditions in drain fields likely facilitate the release of these compounds through gaseous diffusion and volatilization (Wilhelm, 1998).

Rates of movement for the gaseous and liquid phases depend on soil and toxic organic compound type. Soils having fine textures, abrupt interfaces of distinctly different textural layers, a lack of fissures and other continuous macropores, and low moisture content retard toxic organic movement (Hillel, 1989). If gaseous exchange between soil and atmosphere is sufficient, however, appreciable losses of low-molecular-weight alkylated benzenes such as toluene and dimethylbenzene (xylene) can be expected because of their relatively high vapor pressure (Bauman, 1989). Toxic organics that are relatively miscible in water (e.g., methyl tertiary butyl ether, tetrachloroethane, benzene, xylene) can be expected to move with soil water. Nonmiscible toxic organics that remain in liquid or solid phases (chlorinated solvents, gasoline, oils) can become tightly bound to soil particles (Preslo et al., 1989). Biodegradation appears to be an efficient removal mechanism for many volatile organic compounds. Nearly complete or complete removal of toxic organics below infiltration systems was found in several studies (Ayres Associates, 1993a, c; Robertson, 1991; Sauer and Tyler, 1991).

Some investigations have documented toxic organic contamination of surficial aquifers by domestic wastewater discharged from community infiltration fields (Tomson et al., 1984). Of the volatile organic compounds detected in ground water samples collected in the vicinity of subsurface infiltration systems, Kolega (1989) found trichloromethane, toluene, and 1,1,1-trichloroethane most frequently and in some of the highest concentrations. Xylenes, dichloroethane, and dichloromethane were also detected.

**Table 3-23. Case study: concentration of metals in septic tank effluent[a]**

| Metal constituent | Mean concentration (µg/L) | Range (µg/L) |
|---|---|---|
| Arsenic | 37 (5)[b] | 6–59 |
| Barium | 890 (5) | 400–1310 |
| Cadmium | 83 (7) | 30–330 |
| Chromium | 320 (7) | 60–1400 |
| Lead | 2700 (1) | - |
| Mercury | 2 (2) | 1–3 |
| Nickel | 4000 (1) | - |
| Selenium | 15 (6) | 3–39 |

[a] Samples collected from the outlet of nine septic tanks.
[b] Number in parentheses indicates number of septic tanks in which metals were detected.
Source: Florida HRS, 1993, after Watkins, 1991.

Once toxic organics reach an aquifer, their movement generally follows the direction of ground water movement. The behavior of each within an aquifer, however, can be different. Some stay near the surface of the aquifer and experience much lateral movement. Others, such as aliphatic chlorinated hydrocarbons, experience greater vertical movement because of their heavier molecular weight (Dagan and Bresler, 1984). Based on this observation, 1,4-dichlorobenzene, toluene, and xylenes in septic tank effluent would be expected to experience more lateral than vertical movement in an aquifer; 1,1-dichloroethane, 1,1,1-trichloroethane, dichloromethane, and trichloromethane would be expected to show more vertical movement. Movement of toxic organic compounds is also affected by their degree of solubility in water. Acetone, dichloromethane, trichloromethane, and 1,1-dichloroethane are quite soluble in water and are expected to be very highly mobile; 1,1,1-trichloroethane, toluene, and 1,2-dimethylbenzene (o-xylene) are expected to be moderately mobile; and 1,3-dimethylbenzene (m-xylene), 1,4-dimethylbenzene (p-xylene), and 1,4-dichlorobenzene are expected to have low mobility (Fetter, 1988).

System design considerations for removing toxic organic compounds include increasing tank retention time (especially for halogenated, straight-chain compounds like organic solvents), ensuring greater vadose zone depths below the SWIS, and placing the infiltration system high in the soil profile, where higher concentrations of organic matter and oxygen can aid the volatilization and treatment of aromatic compounds. It should be noted that significantly high levels of toxic organic compounds can cause die-off of tank and biomat microorganisms, which could reduce treatment performance. Onsite systems that discharge high amounts of toxic organic compounds might be subject to USEPA's Class V Underground Injection Control Program (see **http://www.epa.gov/safewater.uic.html**).

## Metals

Metals like lead, mercury, cadmium, copper, and chromium can cause physical and mental developmental delays, kidney disease, gastrointestinal illnesses, and neurological problems. Some information is available regarding metals in septic tank effluent (DeWalle et. al. 1985). Metals can be present in raw household wastewater because many commonly used household products contain metals. Aging interior plumbing systems can contribute lead, cadmium, and copper (Canter and Knox, 1985). Other sources of metals include vegetable matter and human excreta. Several metals have been found in domestic septage, confirming their presence in wastewater. They primarily include cadmium, copper, lead, and zinc (Bennett et al., 1977; Feige et al., 1975; Segall et al., 1979). OWTSs serving nonresidential facilities (e.g., rural health care facilities, small industrial facilities) can also experience metal loadings. Several USEPA priority pollutant metals have been found in domestic septic tank effluent (Whelan and Titmanis, 1982). The most prominent metals were nickel, lead, copper,

**Table 3-24. Maximum contaminant levels (MCLs) for selected inorganic chemicals in drinking water**

| Contaminant | MCL (mg/L) | Potential health effects |
|---|---|---|
| Arsenic | 0.05[1] | Increase in blood cholesterol; decrease in blood glucose |
| Cadmium | 0.005 | Kidney damage |
| Chromium | 0.1 | Possible allergic dermatitis after long exposures |
| Copper | 1.3 (action level) | Gastrointestinal distress with short-term exposure; liver or kidney damage possible with long-term exposure |
| Lead | 0.015 (action level) | Physical and mental developmental delays in children; kidney problems, high blood pressure for adults |
| Inorganic mercury | 0.002 | Kidney damage |
| Nitrate-nitrogen | 10.0 | Methemoglobinemia (blue baby syndrome) |
| Nitrite-nitrogen | 1.0 | Methemoglobinemia (blue baby syndrome) |
| Selenium | 0.05 | Hair or fingernail loss; numbness in fingers or toes; circulatory problems |

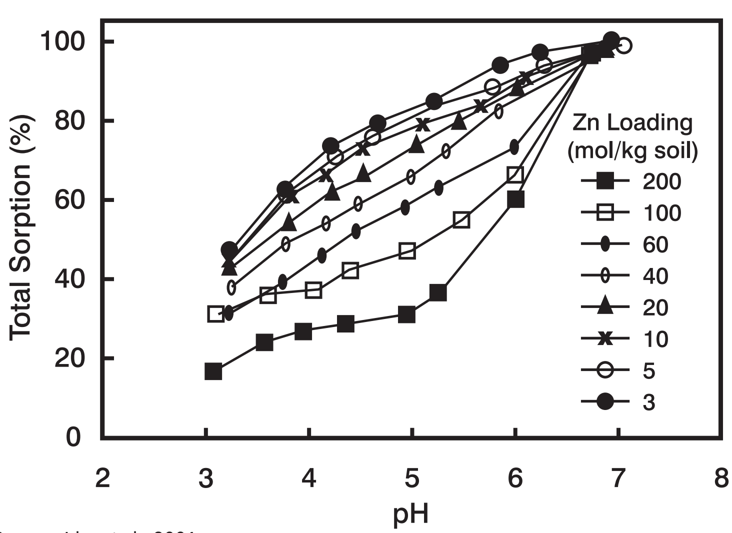
[1] The MCL for arsenic is currently under review by USEPA.
Source: USEPA, 2000a.

zinc, barium, and chromium. A comparison of mean concentrations of metals in septic tank effluent as found in one study (table 3-23) with the USEPA maximum contaminant levels for drinking water noted in table 3-24 reveals a potential for contamination that might exceed drinking water standards in some cases.

The fate of metals in soil is dependent on complex physical, chemical, and biochemical reactions and interactions. The primary processes controlling the fixation/mobility potential of metals in subsurface infiltration systems are adsorption on soil particles and interaction with organic molecules. Because the amount of naturally occurring organic matter in the soil below the infiltrative surface is typically low, the cation exchange capacity of the soil and soil solution pH control the mobility of metals below the infiltrative surface. Acidic conditions can reduce the sorption of metals in soils, leading to increased risk of ground water contamination (Evanko, 1997; Lim et al., 2001). (See figure 3-11.) It is likely that movement of metals through the unsaturated zone, if it occurs at all, is accomplished by movement of organic ligand complexes formed at or near the infiltrative surface (Canter and Knox, 1985; Matthess, 1984).

Information regarding the transport and fate of metals in ground water can be found in hazardous waste and soil remediation literature (see **http://www.gwrtac.org/html/Tech_eval.html#METALS**). One study attempted to link septic tank systems to metal contamination of rural potable water supplies, but only a weak correlation was found (Sandhu et al., 1977). Removal of sources of metals from the wastewater stream by altering user habits and implementing alternative disposal practices is recommended. In addition, the literature suggests that improving treatment processes by increasing septic tank detention times, ensuring greater unsaturated soil depths, and improving dose and rest cycles may decrease risks associated with metal loadings from onsite systems (Chang, 1985; Evanko, 1997; Lim et al., 2001).

**Figure 3-11. Zinc sorption by clay as a function of pH at various loading concentrations (in 0.05 M NaCl medium)**




Source: Lim et al., 2001.

## Surfactants

Surfactants are commonly used in laundry detergents and other soaps to decrease the surface tension of water and increase wetting and emulsification. Surfactants are the largest class of anthropogenic organic compounds present in raw domestic wastewater (Dental et al., 1993). Surfactants that survive treatment processes in the septic tank and subsequent treatment train can enter the soil and mobilize otherwise insoluble organic pollutants. Surfactants have been shown to decrease adsorption — and even actively desorb — the pollutant trichlorobenzene from soils (Dental, 1993). Surfactants can also change soil structure and alter wastewater infiltration rates.

Surfactant molecules contain both strongly hydrophobic and strongly hydrophilic properties and thus tend to concentrate at interfaces of the aqueous system including air, oily material, and particles. Surfactants can be found in most domestic septic tank effluents. Since 1970 the most common anionic surfactant used in household laundry detergent is linear alkylbenzenesulfonate, or LAS. Whelan and Titmanis (1982) found a range of LAS concentrations from 1.2 to 6.5 mg/L in septic tank effluent. Dental (1993) cited studies finding concentrations of LAS in raw wastewater ranging from 3 mg/L to 21 mg/L.

Because surfactants in wastewater are associated with particulate matter and oils and tend to concentrate in sludges in wastewater treatment plants (Dental, 1993), increasing detention times in the tank might aid in their removal. The behavior of surfactants in unsaturated soil is dependent on surfactant type. It is expected that minimal retention of anionic and nonionic surfactants occurs in unsaturated soils having low organic matter content. However, the degree of mobility is subject to soil solution chemistry, organic matter content of the soil, and rate of degradation by soil microorganisms. Soils with high organic matter should favor retention of surfactants because of the lipophilic component of surfactants. Surfactants are readily biodegraded under aerobic conditions and are more stable under anaerobic conditions. Substantial attenuation of LAS in unsaturated soil beneath a subsurface infiltration system has been demonstrated (Anderson et al., 1994; Robertson et al., 1989; Shimp et al., 1991). Cationic surfactants strongly sorb to cation exchange sites of soil particles and organic matter (McAvoy et al., 1991). Thus, fine-textured soils and soils having high organic matter content will generally favor retention of these surfactants.

Some investigations have identified the occurrence of methylene blue active substance (MBAS) in ground water (Perlmutter and Koch, 1971; Thurman et al., 1986). The type of anionic surfactant was not specifically identified. However, it was surmised that the higher concentrations noted at the time of the study were probably due to use of alkylbenzenesulfonate (ABS), which is degraded by microorganisms at a much slower rate than LAS. There has also been research demonstrating that all types of surfactants might be degraded by microorganisms in saturated sediments (Federle and Pastwa, 1988). No investigations have been found that identify cationic or nonionic surfactants in ground water that originated from subsurface wastewater infiltration systems. However, because of concerns over the use of alkylphenol polyethoxylates, studies of fate and transport of this class of endocrine disrupters are in progress.

## Summary

Subsurface wastewater infiltration systems are designed to provide wastewater treatment and dispersal through soil purification processes and ground water recharge. Satisfactory performance is dependent on the treatment efficiency of the pretreatment system, the method of wastewater distribution and loading to the soil infiltrative surface, and the properties of the vadose and saturated zones underlying the infiltrative surface. The soil should have adequate pore characteristics, size distribution, and continuity to accept the daily volume of wastewater and provide sufficient soil-water contact and retention time for treatment before the effluent percolates into the ground water.

Ground water monitoring below properly sited, designed, constructed, and operated subsurface infiltration systems has shown carbonaceous biochemical oxygen demand (CBOD), suspended solids (TSS), fecal indicators, metals, and surfactants can be effectively removed by the first 2 to 5 feet of soil under unsaturated, aerobic conditions. Phosphorus and metals can be removed through adsorption, ion exchange, and precipitation reactions, but the capacity of soil to retain these ions is finite and varies with soil mineralogy, organic content, pH, reduction-oxidation potential, and cation exchange capacity. Nitrogen removal rates vary significantly, but most conventional SWISs do not achieve drinking water standards (i.e., 10 mg/L) for nitrate concentrations in effluent plumes. Evidence is growing that some types of viruses are able to leach with wastewater from subsurface infiltration systems to ground water. Longer retention times associated with virus removal are achieved with fine-texture soil, low hydraulic loadings, uniform dosing and resting, aerobic subsoils, and high temperatures. Toxic organics appear to be removed in subsoils, but further study of the fate and transport of these compounds is needed.

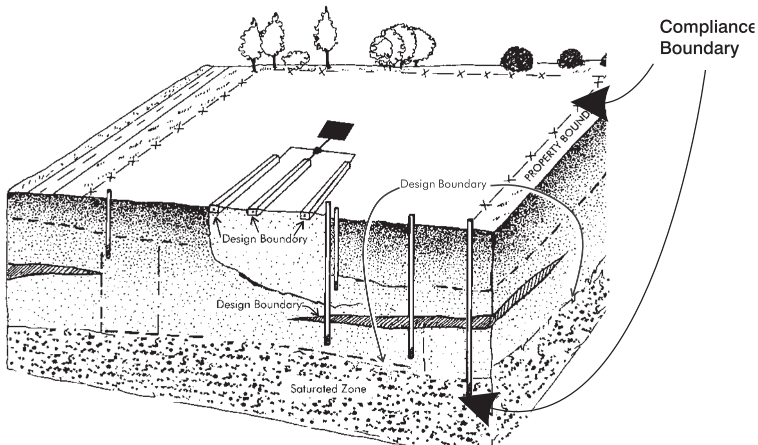
Subsurface wastewater infiltration systems do affect ground water quality and therefore have the potential to affect surface water quality (in areas with gaining streams, large macropore soils, or karst terrain or in coastal regions). Studies have shown that after the treated percolate enters ground water it can remain as a distinct plume for as much as several hundred feet. Concentrations of nitrate, dissolved solids, and other soluble contaminants can remain above ambient ground water concentrations within the plume. Attenuation of solute concentrations is dependent on the quantity of natural recharge and travel distance from the source, among other factors. Organic bottom sediments of surface waters appear to provide some retention or removal of wastewater contaminants if the ground water seeps through those sediments to enter the surface water. These bottom sediments might be effective in removing trace organic compounds, endotoxins, nitrate, and pathogenic agents through biochemical activity, but few data regarding the effectiveness and significance of removal by bottom sediments are available.

Public health and environmental risks from properly sited, designed, constructed, and operated septic tank systems appear to be low. However, soils with excessive permeability (coarse-texture soil or soil with large and continuous pores), low organic matter, low pH, low cation exchange capacities, low oxygen-reduction potential, high moisture content, and low temperatures can increase health and environmental risks under certain circumstances.

## 3.8 Establishing performance requirements

As noted in chapter 2, the OWTS regulatory authority and/or management entity establishes performance requirements to ensure future compliance with the public health and environmental objectives of the community. Performance requirements are based on broad goals such as eliminating health threats from contact with effluent or direct/indirect ingestion of effluent contaminants. They are intended to meet standards for water quality and public health protection and can be both quantitative (total mass load or concentration) or qualitative (e.g., no odors or color in discharges to surface waters). Compliance with performance requirements is measured at a specified performance boundary (see chapter 5), which can be a physical boundary or a property boundary. Figure 3-12 illustrates performance and compliance boundaries and potential monitoring sites in a cutaway view of a SWIS.

**Figure 3-12. Example of compliance boundaries for onsite wastewater treatment systems**




Design boundaries are where conditions abruptly change. A design boundary can be at the intersection of unit processes or between saturated and unsaturated soil conditions (e.g., the delineation between the infiltrative, vadose, and ground water zones) or at another designated location, such as a drinking water well, nearby surface water, or property boundary.

Performance requirements for onsite treatment systems should be established based on water quality standards for the receiving resource and the assimilative capacity of the environment between the point of the wastewater release to the receiving environment and the performance boundary designated by the management entity or regulatory authority. Typically, the assimilative capacity of the receiving environment is considered part of the treatment system to limit costs in reaching the desired performance requirement or water quality goals (see figure 3-12). The performance boundary is usually a specified distance from the point of release, such as a property boundary, or a point of use, such as a drinking water well or surface water with designated uses specified by the state water agency.

Achievement of water quality objectives requires that treatment system performance consider the assimilative capacity of the receiving environment. If the assimilative capacity of the receiving environment is overlooked because of increases in pollutant loadings, the treatment performance of onsite systems before discharge to the soil should increase. OWTSs serving high-density clusters of homes or located near sensitive receiving waters might be the subject of more stringent requirements than those serving lower-density housing farther from sensitive water resources.

Performance requirements for onsite systems should be based on risk assessments that consider the hazards of each potential pollutant in the wastewater to be treated, its transport and fate, potential exposure opportunities, and projected effects on humans and environmental resources. A variety of governmental agencies have already established water quality standards for a wide range of surface water uses. These include standards for protecting waters used for recreation, aquatic life support, shellfish propagation and habitat, and drinking water. In general, these standards are based on risk assessment processes and procedures that consider the designated uses of receiving waters, the hazard and toxicity of the pollutants,

### Nitrogen contributions from onsite systems

The San Lorenzo River basin in California is served primarily by onsite wastewater treatment systems. Since 1985 the Santa Cruz County Environmental Health Service has been working with local stakeholders to develop a program for inspecting all onsite systems, assessing pollutant loads from those systems, and correcting identified problems. Studies conducted through this initiative included calculations of nutrient inputs to the river from onsite systems. According to the analyses performed by the county and its contractors, 55 to 60 percent of the nitrate load in the San Lorenzo River during the summer months came from onsite system effluent. Assumptions incorporated into the calculations included an average septic tank effluent total nitrogen concentration of 50 mg/L, per capita wastewater generation of 70 gallons per day, and an average house occupancy of 2.8 persons. Nitrogen removal was estimated at 15 percent for SWISs in sandy soils and 25 percent for SWISs in other soils.

*Source: Ricker et al., 1994.*

### Performance requirements of Wisconsin's ground water quality rule

Wisconsin was one of the first states to promulgate ground water standards. Promulgated in 1985, Wisconsin's ground water quality rule establishes both public health and public welfare ground water quality standards for substances detected in or having a reasonable probability of entering the ground water resources of the state. Preventive action and enforcement limits are established for each parameter included in the rule. The preventive action limits (PALs) inform the Department of Natural Resources (DNR) of potential threats to ground water quality. When a PAL is exceeded, the Department is required to take action to control the contamination so that the enforcement limit is not reached. For example, nitrate-nitrogen is regulated through a public health standard. The PAL for nitrate is 2 mg/L (nitrogen), and its enforcement limit is 10 mg/L (nitrogen). If the PAL is exceeded, the DNR requires a specific control response based on an assessment of the cause and significance of the elevated concentration. Various responses may be required, including no action, increased monitoring, revision of operational procedures at the facility, remedial action, closure, or other appropriate actions that will prevent further ground water contamination.

*Source: State of Wisconsin Administrative Code, Chapter NR 140.*

the potential for human and ecosystem exposure, and the estimated impacts of exposure. Although federally mandated ground water quality standards (maximum contaminant levels; see tables in section 3.8) are currently applicable only to drinking water supply sources, some states have adopted similar local ground water quality standards (see sidebar).

Local needs or goals need to be considered when performance requirements are established. Watershed- or site-specific conditions might warrant lower pollutant discharge concentrations or mass pollutant limits than those required by existing water quality standards. However, existing water quality standards provide a good starting point for selecting appropriate OWTS performance requirements. The mass of pollutants that should be removed by onsite treatment systems can be determined by estimating the mass of cumulative OWTS pollutants discharged to the receiving waters and calculating the assimilative capacity of the receiving waters. Mass pollutant loads are usually apportioned among the onsite systems and other loading sources (e.g., urban yards and landscaped areas, row crop lands, animal feeding operations) in a ground water aquifer or watershed.

### 3.8.1 Assessing resource vulnerability and receiving water capacity

Historically, conventional onsite systems have been designed primarily to protect human health. Land use planning has affected system oversight requirements, but environmental protection has been a

### Massachusetts' requirements for nitrogen-sensitive areas

Nitrogen-sensitive areas are defined in state rules as occurring within Interim Wellhead Protection Areas, 1-year recharge areas of public water supplies, nitrogen-sensitive embayments, and other areas that are designated as nitrogen-sensitive based on scientific evaluations of the affected water body (310 Code of Massachusetts Regulations 15.000, 1996). Any new construction using onsite wastewater treatment in these designated areas must abide by prescriptive standards that limit design flows to a maximum of 440 gallons per day of aggregated flows per acre. Exceptions are permitted for treatment systems with enhanced nitrogen removal capability. With enhanced removal, the maximum design flow may be increased. If the system is an approved alternative system or a treatment unit with a ground water discharge permit that produces an effluent with no more than 10 mg/L of nitrate, the design flow restrictions do not apply.

*Source: Title V, Massachusetts Environmental Code.*

tertiary objective, at best, for most regulatory programs. Human health protection is assumed (but not always ensured) by infiltrating septic tank effluent at sufficiently low rates into moderately permeable, unsaturated soils downgradient and at specified distances from water supply wells. Site evaluations are performed to assess the suitability of proposed locations for the installation of conventional systems. Criteria typically used are estimated soil permeability (through soil analysis or percolation tests), unsaturated soil depth above the seasonally high water table, and horizontal setback distances from wells, property lines, and dwellings (see chapter 5).

OWTS codes have not normally considered increased pollutant loads to a ground water resource (aquifer) due to higher housing densities, potential contamination of water supplies by nitrates, or the environmental impacts of nutrients and pathogens on nearby surface waters. Preserving and protecting water quality require more comprehensive evaluations of development sites proposed to be served by onsite systems. A broader range of water contaminants and their potential mobility in the environment should be considered at scales that consider both spatial (site vs. region) and temporal (existing vs. planned development) issues (see tables 3-20 to 3-24). Some watershed analyses are driven by TMDLs (Total Maximum Daily Loads established under section 303 of the Clean Water Act) for interconnected surface waters, while others are driven by sole source aquifer or drinking water standards.

## Site suitability assessments

Some states have incorporated stricter site suitability and performance requirements into their OWTS permit programs. Generally, the stricter requirements were established in response to concerns over nitrate contamination of water supplies or nutrient inputs to surface waters. For example, in Massachusetts the Department of Environmental Protection has designated "nitrogen-sensitive areas" in which new nitrogen discharges must be limited. Designation of these areas is based on ecological sensitivity and relative risk of threats to drinking water wells.

## Multivariate rating approaches: DRASTIC

Other approaches are used that typically involve regional assessments that inventory surface and ground water resources and rate them according to their sensitivity to wastewater impacts. The ratings are based on various criteria that define vulnerability. One such method is DRASTIC (see sidebar). DRASTIC is a standardized system developed by USEPA to rate broad-scale ground water vulnerability using hydrogeologic settings (Aller et al., 1987). The acronym identifies the hydrogeologic factors considered: depth to ground water, (net) recharge, aquifer media, soil media, topography (slope), impact of the vadose zone media, and (hydraulic) conductivity of the aquifer. This method is well suited to geographic information system (GIS) applications but requires substantial amounts of information regarding the natural resources of a region to produce meaningful results. Landscape scale methods and models are excellent planning tools but might have limited utility at the site scale. These approaches should be

**Using GIS tools to characterize potential water quality threats in Colorado**

Summit County, Colorado, developed a GIS to identify impacts that OWTS-generated nitrates might have on water quality in the upper Blue River watershed. The GIS was developed in response to concerns that increasing residential development in the basin might increase nutrient loadings into the Dillon Reservoir. Database components entered into the GIS included geologic maps, soil survey maps, topographic features, land parcel maps, domestic well sampling data, onsite system permitting data, well logs, and assessors' data. The database can be updated with new water quality data, system maintenance records, property records, and onsite system construction permit and repair information. The database is linked to the DRASTIC ground water vulnerability rating. The approach is being used to identify areas that have a potential for excessive contamination by nitrate-nitrogen from OWTSs. These assessments could support onsite system placement and removal decisions and help prioritize water quality improvement projects.

*Source: Stark et al., 1999.*

supported and complemented by other information collected during the site evaluation (see chapter 5).

## GIS overlay analysis: MANAGE

A simpler GIS-based method was developed by the University of Rhode Island Cooperative Extension Service (see **http://www.edc.uri.edu/cewq/manage.html**). The Method for Assessment, Nutrient-loading, and Geographic Evaluation (MANAGE) uses a combination of map analyses that incorporates landscape features, computer-generated GIS and other maps, and a spreadsheet to estimate relative pollution risks of proposed land uses (Joubert et al., 1999; Kellogg et al., 1997). MANAGE is a screening-level tool designed for areawide assessment of entire aquifers, wellhead protection areas, or small watersheds (figure 3-13). Local knowledge and input are needed to identify critical resource areas, refine the map data, and select management options for analysis. Community decision makers participate actively in the assessment process (see sidebar).

The spreadsheet from the MANAGE application extracts spatial and attribute data from the national Soil Survey Geographic (SSURGO) database (USDA, 1995; see **http://www.ftw.nrcs.usda.gov/ssur_data.html**) and Anderson Level III Land Cover data (Anderson, 1976) through the Rhode Island GIS system. The soils are combined into hydrologic groups representing the capability of the soils to accept water infiltration, the depth to the water table, and the presence of hydraulically restrictive horizons. Estimates of nutrient loadings are made using published data and simplifying assumptions. The spreadsheet estimates relative pollutant availability, surface water runoff pollutant concentrations, and pollutant migration to ground water zones without attempting to model fate and transport mechanisms, which are highly uncertain. From these data the spreadsheet calculates a hydrologic budget, estimates nutrient loading, and summarizes indicators of watershed health to create a comprehensive risk assessment for wastewater management planning. (For mapping products available from the U.S. Geological Survey, see **http://www.nmd.usgs.gov/**.)

MANAGE generates three types of assessment results that can be displayed in both map and chart form: (1) pollution "hot spot" mapping of potential high-risk areas, (2) watershed indicators based on land use characteristics (e.g., percent of impervious area and forest cover), and (3) nutrient loading in the watershed based on estimates from current research of sources, and generally assumed fates of nitrogen and phosphorus (Joubert et al., 1999).

It is important to note that before rules, ordinances, or overlay zones based on models are enacted or established, the models should be calibrated and verified with local monitoring information collected over a year or more. Only models that accurately and consistently approximate actual event-response relationships should serve as the basis for management action. Also, the affected population must accept the model as the basis for both compliance and possible penalties.

## Value analysis and vulnerability assessment

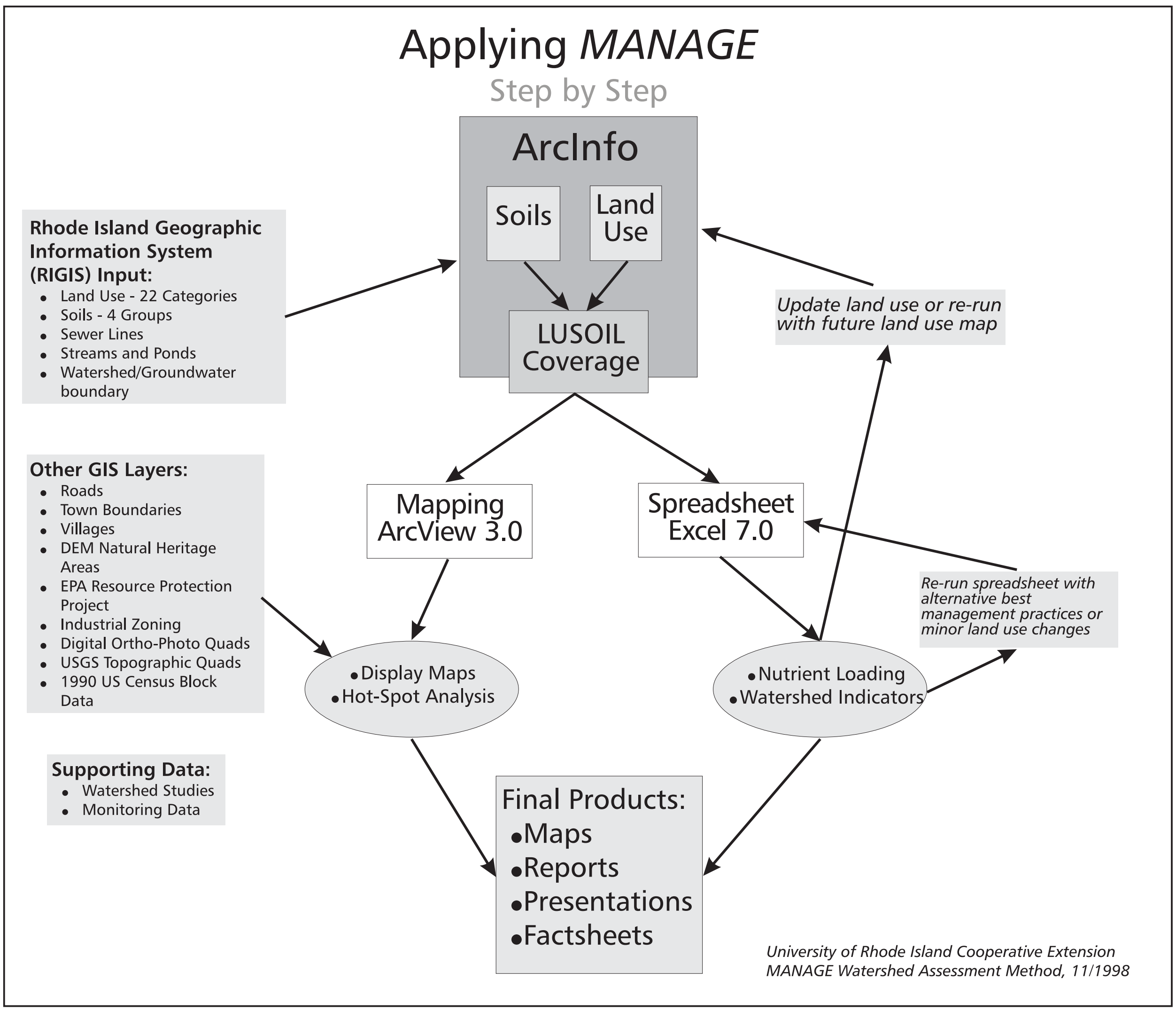
Hoover et al. (1998) has proposed a more subjective vulnerability assessment method that emphasizes public input. This approach considers risk assessment methods and management control strategies for both ground waters and surface waters. It uses three components of risk assessment and management, including consideration of

- Value of ground and surface water as a public water supply or resource
- Vulnerability of the water supply or resource
- Control measures for addressing hazards

The first part of the onsite risk assessment and management approach involves a listing of all the ground water and surface water resources in a region or community (table 3-26). Through community meetings consensus is developed on the relative perceived value of each identified resource and the potential perceived consequences of contamination. For example, a community might determine that shellfish waters that are open to public harvesting are less important than public drinking water supply areas but more important than secondary recreational waters that might be used for body contact sports. This ranking is used to create a table that shows the relative importance of each resource (table 3-26 and case study).

The second part of this risk assessment process is development of a vulnerability assessment matrix. One potential measure of pollution vulnerability is the ability of pollutants to move vertically from the point of release to the water table or bedrock.

**Figure 3-13. Input and output components of the MANAGE assessment method**




Source: Kellogg et al., 1997.

## Application of the MANAGE tool to establish performance requirements

The town of New Shoreham, Rhode Island, is a popular vacation resort on a 6,400-acre island 10 miles off the southern coast of the state. The permanent population is approximately 800, but during the summer the population swells to as many as 10,000 overnight visitors and another 3,000 daily tourists. Proper wastewater management is a serious concern on the island. A publicly owned treatment works serves the town's harbor/commercial/business district, but 85 percent of the permanent residents and 54 percent of the summer population are served by OWTSs, many of which ultimately discharge to the island's sole source aquifer. Protection of this critical water resource is vital to the island's residents and tourism-based economy.

The University of Rhode Island (URI) Cooperative Extension Service's MANAGE risk analysis model was used to identify potential sources of ground water contamination (Kellogg et al., 1997). The model was also used to analyze potential ground water impacts at build-out assuming current zoning. This projection was used to compare the relative change in pollution risk under future development scenarios including the use of alternative technologies that provide better removal of nitrogen and pathogens. Onsite treatment systems were estimated to contribute approximately 72 percent of the nitrogen entering ground water recharge areas. The model indicated that nitrogen removal treatment technologies could effectively maintain nitrogen inputs at close to existing levels even with continued growth. It also showed that nitrogen removal technologies were not necessary throughout the island but would be most beneficial in "hot spots" where the risk of system failure and pollutant delivery to sensitive areas was the greatest.

The town adopted a wastewater management ordinance that mandated regular inspections of onsite systems by a town inspector (Town of New Shoreham, 1996, 1998). It also established septic tank pumping schedules and other maintenance requirements based on inspection results. Inspection schedules have the highest priority in public drinking water supply reservoirs, community wellhead protection zones, and "hot spots" such as wetland buffers. Because the town expected to uncover failed and substandard systems, zoning standards were developed for conventional and alternative OWTS technologies to ensure that new and reconstructed systems would be appropriate for difficult sites and critical resource areas (Town of New Shoreham, 1998). A type of site vulnerability matrix was developed in cooperation with URI Cooperative Extension using key site characteristics—depth to seasonally high water table, presence of restrictive layers, and excessively permeable soils (Loomis et al., 1999). The matrix was used to create a vulnerability rating that is used to establish the level of treatment needed to protect water quality in that watershed or critical resource area.

Three treatment levels were established: T1, primary treatment with watertight septic tanks and effluent screens; T2N, nitrogen removal required to meet $\leq$ 19 mg/L; and T2C, fecal coliform removal $\leq$ 1,000 MPN/100 mL (table 3-25). The town provides a list of specific state-approved treatment technologies considered capable of meeting these standards. By the year 2005, cesspools and failing systems must be upgraded to specified standards. In addition, all septic tanks must be retrofitted with tank access risers and effluent screens.

*Source: Loomis et al., 1999.*

**Table 3-25. Treatment performance requirements for New Shoreham, Rhode Island**

| Treatment level zone | Tested & certified water-tight septic tank | Water-tight access risers to grade | Effluent filter & tipping D-box | Effluent BOD & TSS (mg/L) | TN removal percent | TN effluent (mg/L) | Fecal coliforms (CFUs per 100 mL) |
|---|---|---|---|---|---|---|---|
| T1 | ✔[a] | ☑ | ✔ | NS[b] | NS | NS | NS |
| T2N[c] | ☑ | ✔ | ☑[d] | $\leq$ 30[e] | $\geq$ 50 | $\leq$ 19 | NS |
| T2C[c] | ✔ | ☑ | ✔[d] | $\leq$ 10 | NS | NS | $\leq$ 1000 |

[a]Required by town ordinance.
[b]NS = not specified by town ordinance.
[c]Shallow pressure-dosed drain fields may be required when soil suitability rating is poor, when site vulnerability rating is high to extreme, or when the proposed system is in a wetland buffer, or where other constraints exist.
[d]Required if feasible.
[e]All concentrations and reductions are determined and measured at the outlet of the treatment unit prior to discharge to a drain field.

Source: Adapted from Loomis, 2000.

Table 3-26. Resource listing, value ranking, and wastewater management schematic

| | | Water supply | | Water resource | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Ground water | | GW | Surface water | | | | GW |
| | | **Site** | **Critical area** | **Regionally important aquifer** | **Primary recreation** | **Shellfish waters** | **Nutrient-sensitive** | **Secondary recreation** | **Poor aquifer** |
| | | Directly next to wellfield | Wellfield capture zone | Outwash sand & gravel | Beaches used for swimming | Commercial open waters | Lakes, ponds, rivers, etc. | Other surface waters | Unproductive confined aquifers |
| | | Sites of community wellfields and source areas within 10 days' time of travel in the ground water to the community wellfields | Inner and outer critical areas that are within the ground water capture zones for the community wellfields | High-yielding surficial aquifers of regional importance that are used for many individual wells and that have rapid recharge | Source areas within 200 feet of frequently used swimming beaches | Source areas within 200 feet of shellfish waters that are open to public harvesting | Source areas for nutrient-sensitive surface waters that are susceptible to eutrophication or to loss of shellfish or finfish nursery areas due to nutrient inputs | Source areas within 100 feet of secondary recreational waters that are used for swimming on an unorganized basis | Poor, unproductive glacial till aquifers or productive aquifers isolated from the surface or not used for many private wells |
| **Vulnerability Rating** | **High** | R5 | R4 | R3 | R3 | R3 | R2b | R2a | R1 |
| | **Mod.** | R5 | R4 | R3 | R3 | R3 | R2b | R1 | R1 |
| | **Low** | R5 | R3 | R3 | R2a | R2a | R2b | R1 | R1 |

**Highest Value Resource** → **Lowest Value Resource**

Source: Hoover et al., 1998.

**Resource value ranking and wastewater management**

A northern U.S. unsewered coastal community was concerned about the impacts onsite treatment systems might have on its ground water resources (Hoover et al., 1998). Public water in the community is derived exclusively from ground water. The extended recharge zone for the community well fields is also a water supply source in the community. Other resources in the community include regionally important sand and gravel glacial outwash aquifers, public beaches, shellfish habitat in shallow surface waters, nutrient-sensitive surface waters, low-yield glacial till aquifers, and other surface waters used as secondary recreational waters.

Through public meetings, the community identified and ranked the various water resources according to their perceived value. After ranking, the vulnerability of each resource to pollution from onsite treatment systems was estimated. The vulnerability ratings were based on the thickness of the unsaturated zone in the soil, the rate of water movement through the soil, and the capability of the soil to attenuate pollutants (table 3-25). For each rating, a control zone designation was assigned (R5, R4, R3, R2, or R1). The criteria used for the vulnerability ratings were documented in the community's wastewater management plan. Control measures were established for each control zone. In this instance, specific wastewater treatment trains were prescribed for use in each control zone based on the depth of the unsaturated soil zone (tables 3-26 and 3-27). The treatment standards are TS1 = primary treatment, TS2 = secondary treatment, TS3 = tertiary treatment, TS4 = nutrient reduction, and TS5 = tertiary treatment with disinfection.

Important criteria considered include the thickness of the unsaturated soil layer and the properties of the soil. The vulnerability assessment matrix (table 3-26) identifies areas of low, moderate, high, or extreme vulnerability depending on soil conditions. For example, vulnerability might be "extreme" for coarse or sandy soils with less than 2 feet of vertical separation between the ground surface and the water table or bedrock. Vulnerability might be "low" for clay-loam soils with a vertical separation of greater than 6 feet and low permeability. Each resource specified in the first part of the risk assessment process can be associated with each vulnerability category. A more detailed discussion of ground water vulnerability assessment is provided in *Groundwater Vulnerability Assessment: Predicting Relative Contamination Potential under Conditions of Uncertainty* (National Research Council, 1993).

The third and final part of the risk assessment process is developing a management matrix that specifies a control measure for each vulnerability category relative to each resource (tables 3-27, 3-28). Several categories of management control measures (e.g., stricter performance requirements for OWTSs) might be referenced depending on the value and vulnerability of the resource. Generally, each management control measure would define

- Management entity requirements for each control measure
- System performance and resource impact monitoring requirements for each vulnerable category
- Types of acceptable control measures based on the vulnerability and value of the resource
- Siting flexibility allowed for each control measure
- Performance monitoring requirements for each control measure and vulnerability category

## Probability of impact approach

Otis (1999) has proposed a simplified "probability of environmental impact" approach. This method was developed for use when resource data are insufficient and mapping data are unavailable for a more rigorous assessment. The approach is presented in the form of a decision tree that considers mass loadings to the receiving environment (ground water or surface water), population density, and the fate and transport of potential pollutants to a point of use (see following case study and figure 3-14). The decision tree (figure 3-14) estimates the relative probability of water resource impacts from wastewater discharges generated by sources in the watershed. Depending on the existing or expected use of the water resource, discharge standards for the treatment systems can be established. The system designer can use these discharge standards to assemble an appropriate treatment train.

**Table 3-27. Proposed onsite system treatment performance standards in various control zones**

| Standard | BOD (mg/L) | TSS (mg/L) | $PO_4$-P (mg/L) | $NH_4$-N (mg/L) | $NO_3$-N (mg/L) | Total N (% removed)[a] | Fecal coliforms (CFU/1000 mL) |
|---|---|---|---|---|---|---|---|
| TS1 - primary treatment | | | | | | | |
| TS1u – unfiltered | 300 | 300 | 15 | 80 | NA | NA | 10,000,000 |
| TS1f – filtered | 200 | 80 | 15 | 80 | NA | NA | 10,000,000 |
| TS2 - secondary treatment | 30 | 30 | 15 | 10 | NA | NA | 50,000 |
| TS3 - tertiary treatment | 10 | 10 | 15 | 10 | NA | NA | 10,000 |
| TS4 - nutrient reduction | | | | | | | |
| TS4n - nitrogen reduction | 10 | 10 | 15 | 5 | NA | 50% | 10,000 |
| TS4p - phosphorus reduction | 10 | 10 | 2 | 10 | NA | 25% | 10,000 |
| TS4np - N & P reduction | 10 | 10 | 2 | 5 | NA | 50% | 10,000 |
| TS5 - bodily contact disinfection | 10 | 10 | 15 | 10 | NA | 25% | 200 |
| TS6 - wastewater reuse | 5 | 5 | 15 | 5 | NA | 50% | 14 |
| TS7 - near drinking water | 5 | 5 | 1 | 5 | 10 | 75% | <1[b] |

NA = not available.

[a] Minimum percentage reduction of total nitrogen (as nitrate -nitrogen plus ammonium nitrogen) concentration in the raw, untreated wastewater.

[b] Total coliform colony densities < 50 per 100 mL of effluent.

Source: Hoover et al., 1998.

**Table 3-28. Treatment performance standards in various control zones**

| Vertical separation distance (feet) | Control zone (with management entity) | | | | | |
|---|---|---|---|---|---|---|
| | R1 | R2a | R2b | R3 | R4 | R5 |
| | Treatment performance standard | | | | | |
| >4 | TS1 | TS1 | TS1 OR TS4 | TS1 | TS2 | TS4 |
| 3 to 4 | TS1 | TS1 | TS1 OR TS4 | TS2 | TS2 | TS5 |
| 2 to 3 | TS1 | TS2 | TS2 OR TS4 | TS3 | TS3 | NA |
| 1 to 3 | TS2 | TS3 | TS3 OR TS4 | TS4 | TS4 | NA |
| <1 | TS3 | TS4 | TS4 | TS5 | TS5 | NA |

Increasing Resource Value →

Increasing Vulnerability ↑

## Assessment and modeling through quantitative analysis

Numeric performance requirements for onsite wastewater treatment systems can be derived by quantifying the total pollutant assimilative capacity of the receiving waters, estimating mass pollutant loads from non-OWTS sources, and distributing the remaining assimilative capacity among onsite systems discharging to the receiving waters. Consideration of future growth, land use and management practices, and a margin of safety should be included in the calculations to ensure that estimation errors favor protection of human health and the environment.

Assimilative capacity is a volume-based (parts of pollutant per volume of water) measurement of the ability of water to decrease pollutant impacts through dilution. Threshold effects levels are usually established by state, federal, or tribal water quality standards, which assign maximum concentrations of various pollutants linked to designated uses of the receiving waters (e.g., aquatic habitat, drinking water source, recreational waters). Because wastewater pollutants of concern (e.g., nitrogen compounds, pathogens, phosphorus) can come from a variety of non-OWTS sources, characterization of all pollutant sources and potential pathways to receiving waters provides important information to managers seeking to control or reduce elevated levels of contaminants in those

### Establishing performance requirements by assessing the probability of impact

The "probability of impact" method estimates the probability that treated water discharged from an onsite system will reach an existing or future point of use in an identified water resource. By considering the relative probability of impact based on existing water quality standards (e.g., drinking water, shellfish water, recreational water), acceptable treatment performance standards can be established. The pollutants and their concentrations or mass limits to be stipulated in the performance requirements will vary with the relative probability of impact estimated, the potential use of the water resource, and the fate and transport characteristics of the pollutant.

As an example, the assessment indicates that a ground water supply well that provides water for drinking without treatment might be adversely affected by an onsite system discharge. Soils are assumed to be of acceptable texture and structure, with a soil depth of 3 feet. Nitrate-nitrogen and fecal coliforms are two wastewater pollutants that should be addressed by the performance requirements for the treatment system (i.e., constructed components plus soil). With a relative probability of impact estimated to be "high," the regulatory authority considers it reasonable to require the treatment system to achieve drinking water standards for nitrate and fecal coliforms before discharge to the saturated zone. The drinking water standards for nitrate and fecal coliforms in drinking water are 10 mg/L for nitrate and zero for fecal coliforms. Considering the fate of nitrogen in the soil, it can be expected that any of the nitrogen discharged by the pretreatment system will be converted to nitrate in the unsaturated zone of the soil except for 2 to 3 mg/L of refractory organic nitrogen. Because nitrate is very soluble and conditions for biological denitrification in the soil cannot be relied on, the performance standard for the onsite system is 12 mg/L of total nitrogen (10 mg/L of nitrite + 2 mg/L of refractory organic nitrogen) prior to soil discharge. In the case of fecal coliforms, the natural soil is very effective in removing fecal indicators where greater than 2 feet of unsaturated natural soil is present. Therefore, no fecal coliform standard is placed on the pretreatment (i.e., constructed) system discharge because the standard will be met after soil treatment and before final discharge to the saturated zone.

If the probability of impact is estimated to be "moderate" or "low," only the nitrogen treatment standard would change. If the probability of impact is "moderate" because travel time to the point of use is long, dispersion and dilution of the nitrate in the ground water is expected to reduce the concentration in the discharge substantially. Therefore, the treatment standard for total nitrogen can be safely raised, perhaps to 20 to 30 mg/L of nitrogen. If the probability of impact is "low," no treatment standard for nitrogen is necessary.

If the probability of impact is "high" but the point of ground water use at risk is an agricultural irrigation well, no specific pollutants in residential wastewater are of concern. Therefore, the treatment required need be no more than that provided by a septic tank.

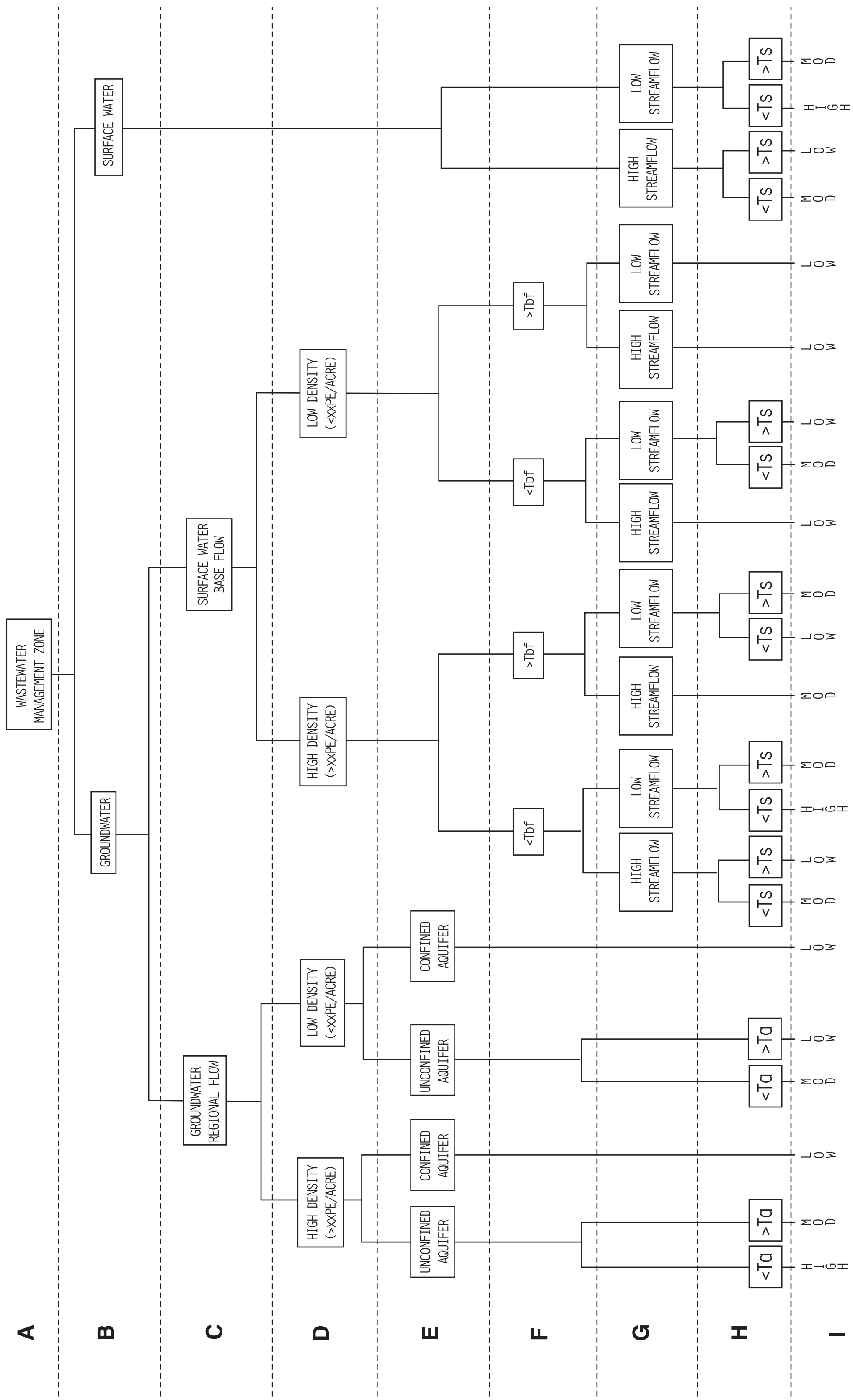
*Source: Otis, 1999.*

waters. For example, the mass balance equation used to predict nitrate-nitrogen (or other soluble pollutant) concentrations in ground water and surface waters is

$$\text{Nitrate-nitrogen (mg/L)} = \frac{\text{Annual nitrogen loading from all sources in lb/yr} \times 454{,}000\ \text{mg/lb}}{\text{Annual water recharge volume from all sources in liters}}$$

As the examples above indicate, there are a wide range of approaches for assessing water resource vulnerability and susceptibility to impacts from onsite wastewater treatment systems. Other methodologies include risk matrices similar to those summarized above and complex contaminant transport models, including Qual2E, SWMM, and BASINS, the EPA-developed methodology for integrating point and nonpoint source pollution assessments (see **http://www.epa.gov/ow/compendium/toc.htm** for more information on BASINS and other water quality modeling programs).

**Figure 3-14. Probability of environmental impact decision tree (see key)**




Source: Otis, 1999.

**Environmental sensitivity assessment key (for figure 3-14).**

| | |
|---|---|
| **A** | **Wastewater management zone**<br>Includes the entire service area of the district. |
| **B** | **Receiving environment**<br>Receiving water to which the wastewater is discharged. |
| **C** | **Fate of ground water discharge**<br>The treated discharge to ground water may enter the regional flow or become base flow to surface water. Ground water flow direction can be roughly estimated from ground surface topography if other sources of information are not available. In some instances both regional flow and base flow routes should be assessed to determine the controlling point of use. |
| **D** | **Planning area density (population equivalents per acre)**<br>The risk of higher contaminate concentrations in the ground water from ground water-discharging treatment facilities will increase with increasing numbers of people served. Where building lots are served by individual infiltration systems, the population served divided by the total area composed by contiguous existing and planned lots would determine population equivalents per acre (p.e./acre). For a large cluster system, the p.e./acre would be determined by the population served divided by the area of the infiltration surface of the cluster system. |
| **E** | **Well construction**<br>Wells developed in an unconfined aquifer with direct hydraulic connections to the wastewater discharge have a higher probability of impact from the wastewater discharge than wells developed in a confined aquifer. Wells that are considered within the zone of influence from the wastewater discharge should be identified and their construction determined from well logs. |
| **F** | **Travel time to base flow discharge, $T_{bf}$**<br>Treated wastewater discharges in ground water can affect surface waters through base flow. The potential impacts of base flows are inversely proportional to the travel time in the ground water, $T_{bf}$, because of the dispersion and dilution (except in karst areas) that will occur. Where aquifer characteristics necessary to estimate travel times are unknown, distance can be substituted as a measure. If travel time, $T_{bf}$, is greater than time to a ground water point of use, $T_a$, the ground water should be assumed to be the receiving environment. |
| **G** | **Stream flow**<br>Stream flow will provide dilution of the wastewater discharges. The mixing and dilution provided are directly proportional to the stream flow. Stream flow could be based on the 7-day, 10-year low-flow condition (${}_7Q_{10}$) as a worst case. “High” and “low” stream flow values would be defined by the ratio of the ${}_7Q_{10}$ to the daily wastewater discharge. For example, ratios greater than 100:1 might be “high,” whereas those less than 100:1 might be “low.” Stream flow based on the watershed area might also be used (cfs/acre). |
| **H** | **Travel time to aquifer or surface water point of use, $T_a$ or $T_s$**<br>The potential impacts of wastewater discharges on points of use (wells, coastal embayments, recreational areas, etc.) are inversely proportional to the travel time. Except for karst areas, distance could be used as a substitute for travel time if aquifer or stream characteristics necessary to estimate travel times are unknown. |
| **I** | **Relative probability of impact**<br>The relative probability of impact is a qualitative estimate of expected impact from a wastewater discharge on a point of use. The risk posed by the impact will vary with the intended use of the water resource and the nature of contaminants of concern. |

Source: Otis, 1999.

## Estimating nitrogen loadings and impacts for Buttermilk Bay, Massachusetts

In Buttermilk Bay, a 530-acre shallow coastal bay at the northern end of Buzzards Bay in Massachusetts, elevated nitrogen levels associated with onsite systems and land use in the watershed have contributed to nuisance algal growth and declines in eelgrass beds in some areas. An investigation in the early 1990s supported by the New England Interstate Water Pollution Control Commission and USEPA established a critical (maximum allowable) nitrogen loading rate of 115,600 pounds per year by identifying an appropriate ecological effects threshold (the nitrogen concentration associated with significant ecological impacts, or 0.24 mg/L in nitrogen-sensitive Buttermilk Bay) and considering both the size and recharge rate of the bay:

**Critical Loading Rate (pounds per year) =**

Threshold nitrogen concentration x volume x number of annual water body recharges =

240 milligrams of N per cubic meter x 2,996,000 cubic meters x 73 annual recharges =

52,489,920,000 milligrams of N / 454,000 milligrams in one pound =

**115,617 pounds per year = critical loading rate for nitrogen**

After establishing the critical nitrogen loading rate, the watershed assessment team sought to quantify annual nitrogen loads discharged to the bay under existing conditions. Loading values for various sources of nitrogen in the watershed were estimated and are presented in table 3-29. For the purposes of estimating nitrogen contributions from onsite systems, it was assumed that the total nitrogen concentration in onsite treated effluent was 40 mg/L and the per capita flow was 55 gallons per day. [It should be noted that nitrogen concentrations in onsite system treated effluent commonly range between 25 and 45 mg/L for soil-based systems, though some researcher have found higher effluent concentrations. In general, SWIS nitrogen removal rates range between 10 and 20 percent (Van Cuyk et al., 2001) for soil-based systems. Mechanized systems designed for nitrogen removal can achieve final effluent N concentrations as low as 10-25 mg/L.]

Using the research-based assumptions and estimates summarized in the table, the assessment team estimated that total current nitrogen loadings totaled about 91,053 lb/yr. Onsite wastewater treatment systems represented a significant source (74 percent) of the overall nitrogen input, followed by lawn fertilizers (15 percent) and cranberry bogs (7 percent).

The final part of the Buttermilk Bay analysis involved projecting the impact of residential build-out on nitrogen loads to the bay. With a critical (maximum allowable) nitrogen loading rate of 115,617 lb/yr and an existing loading rate of 91,053 lb/yr, planners had only a 24,564 lb/yr cushion with which to work. Full residential build-out projections generated nitrogen loading rates that ranged from 96,800 lb/yr to 157,500 lb/yr. Regional planners used this information to consider approaches for limiting nitrogen loadings to a level that could be safely assimilated by the bay. Among a variety of options that could be considered under this scenario are increasing performance requirements for onsite systems, decreasing system densities, limiting the total number of new residences with onsite systems in the bay watershed, and reducing nitrogen inputs from other sources (e.g., lawn fertilizers, cranberry bogs).

**Table 3-29. Nitrogen loading values used in the Buttermilk Bay assessment**

| Nitrogen source | Nitrogen concentration | Loading rate | Flow/recharge | Total loading |
|---|---|---|---|---|
| Onsite wastewater systems | 40 mg/L | 6.72 lb N/person/yr | 55 gal/person/day (165 gal/dwelling) | 66,940 lb |
| Fertilizers–lawns | NA | 0.9 lb N /1000 $ft^2$/yr | 18 in./yr | 13,721 lb |
| Fertilizers–cranberry bogs | NA | 15.8 lb N/1000 $ft^2$/yr | NA | 6,378 lb |
| Pavement runoff | 2.0 mg/L | 0.42 lb N/1000 $ft^2$/yr | 40 in./year | 1,723 lb |
| Roof runoff | 0.75 mg/L | 0.15 lb N/1000 $ft^2$/yr | 40 in./year | 686 lb |
| Atmospheric deposition | 0.3 mg DIN/L | 3.03 lb N/acre | NA | 1,606 lb |
| Total | | | | 91,053 lb |

NA = not available.

Source: Horsley Witten Hegemann, 1991, after Nelson et al., 1988.

### 3.8.2 Establishing narrative or numerical performance requirements

Performance requirements should reflect acceptable environmental impacts and public health risks based on assessment methods such as those described in the preceding section. They should specify observable or measurable requirements in narrative or numerical form. Conventional onsite treatment systems (septic tanks with SWISs) have used narrative requirements such as prohibitions on wastewater backup in plumbing fixtures or effluent pooling on the ground surface. These requirements are measurable through observation but address only some specific public health issues. An example of a narrative performance requirement that addresses potential environmental impacts is the Town of Shoreham's requirement for specifically approved treatment trains for environmentally sensitive areas (see sidebar and table 3-26 in preceding section). Compliance is determined by whether the required treatment processes are in place; water quality monitoring is not involved. The regulating agencies assume that the water quality objectives are achieved if these narrative performance requirements are met. Although there is merit in this approach, some additional steps (e.g., operation and maintenance monitoring, targeted water quality monitoring) would be included in a more comprehensive program.

Numerical performance requirements specify the critical parameters of concern (e.g., nitrate, phosphorus, fecal coliforms), the maximum allowable concentration or mass pollutant/flow discharge permitted per day, and the point at which the requirements apply. Examples of numerical performance requirements include Massachusetts' requirement for limited volume discharges (measured in gallons per day) in designated nitrogen-sensitive areas or a water quality standard for nitrogen of 25 mg/L, to be met at the property boundary. Unlike the narrative requirements, numerical performance requirements provide more assurance that the public health and water quality goals are being met.

## 3.9 Monitoring system operation and performance

Performance monitoring of onsite treatment systems serves several purposes. Its primary purpose is to ensure that treatment systems are operated and maintained in compliance with the performance requirements. It also provides performance data useful in making corrective action decisions and evaluating areawide environmental impacts for land use and wastewater planning. Historically, performance monitoring of onsite treatment systems has not been required. Regulatory agencies typically limit their regulatory

**Onsite system inspection/maintenance guidance for Rhode Island**

The Rhode Island Department of Environmental Management published in 2000 the *Septic System Checkup,* an inclusive guide to inspecting and maintaining septic systems. The handbook, available to the public, is written for both lay people and professionals in the field. The guide is an easy-to-understand, detailed protocol for inspection and maintenance and includes newly developed state standards for septic system inspection and maintenance. It describes two types of inspections: a maintenance inspection to determine the need for pumping and minor repairs, and a functional inspection for use during property transfers. The handbook also includes detailed instructions for locating septic system components, diagnosing in-home plumbing problems, flow testing and dye tracing, and scheduling inspections. Several Rhode Island communities, including New Shoreham, North Kingstown and Glocester, currently use *Septic System Checkup* as their inspection standard. The University of Rhode Island offers a training course for professionals interested in becoming certified in the inspection procedures.

The handbook is available free on-line at **http://www.state.ri.us/dem/regs/water/isdsbook.pdf**. Individual spiral-bound copies can be purchased for $10 with inspection report forms or $7 for the manual without forms from DEM's Office of Technical and Customer Assistance at 401.222.6822.

*Source: Rhode Island Department of Environmental Management.*

control primarily to system siting, design, and construction and certification of site evaluators, designers, and other service providers. System performance is largely ignored by the regulatory authority or management entity or addressed through sometimes weak owner education and voluntary compliance programs until a hydraulic failure is reported or observed (see chapters 2 and 5).

OWTS oversight agencies typically exert regulatory control by conducting the site evaluation and reviewing the proposed design for compliance with administrative code prescriptions for proven systems. If the site characteristics and selected system design meet the prescriptions in the code, a construction permit is issued for installation by a certified contractor. The regulatory authority or management entity usually performs a pre-coverup inspection before final approval is given to use the system. At that point the regulatory authority typically relinquishes any further oversight of the system until a hydraulic failure is observed or reported. The owner may be given educational materials and instructions describing the system and what maintenance should be performed, but routine operation and maintenance is left up to the owner. Tank pumping or other routine maintenance tasks are seldom required or even tracked by the regulatory authority or management entity for information purposes. Regular inspections of systems are usually not mandated.

This regulatory approach might be adequate for the degree of risk to human health and the environment posed by isolated and occasional hydraulic failures. Where onsite treatment is used in moderate-to-high-density suburban and seasonal developments, however, it has not proven to be adequate, particularly where treatment failures can be expected to significantly affect ground water and surface water quality. Onsite system failure rates across the nation range as high as 10 percent or more in some areas (see Section 1.3). In cases where high system densities or system age indicates the likelihood of ground or surface water contamination, incorporation of mandated performance monitoring into OWTS management programs is strongly recommended. In 2000 USEPA issued suggested guidelines for onsite system management programs. *Draft Guidelines for Management of Onsite/ Decentralized Wastewater Systems* (USEPA, 2000b) provides an excellent framework for developing a comprehensive management program that considers the full range of issues involved in OWTS planning, siting, design, installation, operation, maintenance, monitoring, and remediation (see chapter 2).

Local OWTS regulatory and management agencies in many areas are embracing more rigorous operation, maintenance, and inspection programs to deal with problems caused by aging systems serving developments built before 1970, poor maintenance due to homeowner indifference or ignorance, and regional hydraulic or pollutant overloads related to high-density OWTS installations. Operation and maintenance management programs adopted by these agencies consist mostly of an integrated performance assurance system that inventories new and existing systems, establishes monitoring or inspection approaches, requires action when systems fail to operate properly, and tracks all activities to ensure accountability among regulatory program staff and system owners. (See chapter 2 and *Draft Guidelines for Management of Onsite/ Decentralized Wastewater Systems* at **http://www.epa.gov/owm/decent/index.htm** for more information and examples.)

## 3.9.1 Operating permits

Periodic review of system performance is necessary to ensure that systems remain in compliance with established performance requirements after they are installed. Thus, regulatory agencies need to maintain rigorous, perpetual oversight of systems to ensure periodic tank pumping, maintenance of system components, and prompt response to problems that may present threats to human health or water resources. Some jurisdictions are fulfilling this responsibility by issuing renewable/revocable operating permits. The permit stipulates conditions that the system must meet before the permit can be renewed (see sidebar). The duration of such permits might vary. For example, shorter-term permits might be issued for complex treatment systems that require more operator attention or to technologies that are less proven (or with which the regulatory authority has less comfort). The owner is responsible for documenting and certifying that permit conditions have been met. If permit conditions have not been met, a temporary permit containing a compliance schedule for taking appropriate actions may be issued. Failure to meet the compliance schedule can result in fines or penalties.

### Onsite system operating permits in St. Louis County, Minnesota

St. Louis County, located in the northeastern region of Minnesota, extends from the southwestern tip of Lake Superior north to the Canadian border. The physical characteristics of the region are poorly suited for application of traditional onsite treatment systems. Many of the soils are very slowly permeable lacustrine clays, shallow to bedrock, and often near saturation. The existing state minimum code restricts onsite systems to sites featuring permeable soils with sufficient unsaturated depths to maintain a 3-foot separation distance to the saturated zone. To allow the use of onsite treatment, the county has adopted performance requirements that may be followed in lieu of the prescriptive requirements where less than 3 feet of unsaturated, permeable soils are present. In such cases the county requires that the owner continuously demonstrate and certify that the system is meeting the performance requirements. This is achieved through the issuance of renewable operating permits for higher-performance alternative treatment systems. The operating permit is based on evaluation of system performance rather than design prescription and includes the following:

- ✔ System description
- ✔ Environmental description
- ✔ Site evaluation documentation
- ✔ Performance requirements
- ✔ System design, construction plan, specifications, and construction drawings
- ✔ Maintenance requirements
- ✔ Monitoring requirements (frequency, protocol, and reporting)
- ✔ Contingency plan to be implemented if the system fails to perform to requirements
- ✔ Enforcement and penalty provisions

The permit is issued for a limited term, typically 5 years. Renewal requires that the owner document that the permit requirements have been met. If the documentation is not provided, a temporary permit is issued with a compliance schedule. If the compliance schedule is not met, the county has the option of reissuing the temporary permit and/or assessing penalties. The permit program is self-supporting through permit fees.

## 3.9.2 Monitoring programs

Monitoring individual or regional onsite system performance may include performance inspections (see Chapter 2 and *Draft Management Guidelines for Onsite/Decentralized Wastewater Systems*), water quality sampling at performance boundaries, drinking water well monitoring, and assessment of problem pollutant concentrations (pathogens, nitrate, phosphorus) in nearby surface waters. In general, monitoring of system performance seeks to ascertain if onsite systems are meeting performance requirements, i.e., protecting public health and water quality. Assessing the sensitivity of water resources to potential pollutant loadings from onsite systems helps in developing performance requirements and the monitoring methods and sampling locations that might be used.

Monitoring system performance through water quality sampling is difficult for conventional onsite systems because the infiltration field and underlying soil are part of the treatment system. The percolate that enters the ground water from the infiltration system does not readily mix and disperse in the ground water. It can remain as a distinct, narrow plume for extended distances from the system (Robertson et al., 1991). Locating this plume for water quality sampling is extremely difficult, and the cost involved probably does not warrant this type of monitoring except for large systems that serve many households or commercial systems constructed over or near sensitive ground water and surface water resources (see chapter 5). Monitoring of onsite treatment systems is enhanced considerably by the inclusion of inspection and sampling ports at performance boundaries (e.g., between treatment unit components) and the final discharge point. Other methods of monitoring such as simple inspections of treatment system operation or documentation of required system maintenance

### Monitoring requirements in Washington

The Department of Health of the state of Washington has adopted a number of monitoring requirements that OWTS owners must meet (Washington Department of Health, 1994). Because such requirements place additional oversight responsibilities on management agencies, additional resources are needed to ensure compliance. Among the requirements are the following:

The system owner is responsible for properly operating and maintaining the system and must

- Determine the level of solids and scum in the septic tank once every 3 years.
- Employ an approved pumping service provider to remove the septage from the tank when the level of solids and scum indicates that removal is necessary.
- Protect the system area and the reserve area from cover by structures or impervious material, surface drainage, soil compaction (for example, by vehicular traffic or livestock), and damage by soil removal and grade alteration.
- Keep the flow of sewage to the system at or below the approved design both in quantity and waste strength.
- Operate and maintain alternative systems as directed by the local health officer.
- Direct drains, such as footing or roof drains away from the area where the system is located.

Local health officers in Washington also perform monitoring duties, including the following;

- Providing operation and maintenance information to the system owner upon approval of any installation, repair, or alteration of a system.
- Developing and implementing plans to monitor all system performance within areas of special concern[1]; initiating periodic monitoring of each system by no later than January 1, 2000, to ensure that each system owner properly maintains and operates the system in accordance with applicable operation and maintenance requirements; disseminating relevant operation and maintenance information to system owners through effective means routinely and upon request; and assisting in distributing educational materials to system owners.

Finally, local health officers may require the owner of the system to perform specified monitoring, operation, or maintenance tasks, including the following:

- Using one or more of the following management methods or another method consistent with the following management methods for proper operation and maintenance: obtain and comply with the conditions of a renewable or operational permit; employ a public entity eligible under Washington state statutes to directly or indirectly manage the onsite system; or employ a private management entity, guaranteed by a public entity eligible under Washington state statutes or sufficient financial resources, to manage the onsite system.
- Evaluating any effects the onsite system might have on ground water or surface water.
- Dedicating easements for inspections, maintenance, and potential future expansion of the onsite system.

[1] "Areas of special concern" are areas where the health officer or department determines additional requirements might be necessary to reduce system failures or minimize potential impacts upon public health. Examples include shellfish habitat, sole source aquifers, public water supply protection areas, watersheds of recreational waters, wetlands used in food production, and areas that are frequently flooded.

*Source: Washington Department of Health, 1994.*

might be sufficient and more cost-effective than water quality sampling at a performance boundary.

The Critical Point Monitoring (CPM) approach being developed in Washington State provides a systematic approach to choosing critical locations to monitor specific water quality parameters (Eliasson et al., 2001). The program is most suitable for responsible management entities operating comprehensive management programs. CPM provides an appropriate framework for monitoring treatment train components, though it should be recognized that evaluations of overall system effectiveness—and compliance with

### State of Massachusetts' onsite treatment system inspection program

Massachusetts in 1996 mandated inspections of OWTSs to identify and address problems posed by failing systems (310 CMR 15.300, 1996). The intent of the program is to ensure the proper operation and maintenance of all systems. A significant part of the program is the annual production of educational materials for distribution to the public describing the importance of proper maintenance and operation of onsite systems and the impact systems can have on public health and the environment.

Inspections are required at the time of property transfer, a change in use of the building, or an increase in discharges to the system. Systems with design flows equal to or greater than 10,000 gpd require annual inspections. Inspections are to be performed only by persons approved by the state. The inspection criteria are established by code and must include

- ✔ A general description of system components, their physical layout, and horizontal setback distances from property lines, buildings, wells, and surface waters.
- ✔ Description of the type of wastewater processed by the system (domestic, commercial, or industrial).
- ✔ System design flow and daily water use, if metered.
- ✔ Description of the septic tank, including age, size, internal and external condition, water level, etc.
- ✔ Description of distribution box, dosing siphon, or distribution pump, including evidence of solids carryover, clear water infiltration, and equal flow division, and evidence of backup, if any.
- ✔ Description of the infiltration system, including signs of hydraulic failure, condition of surface vegetation, level of ponding above the infiltration surface, other sources of hydraulic loading, depth to seasonally high water table, etc.

A system is deemed to be failing to protect public health, safety, and the environment if the septic tank is made of steel, if the OWTS is found to be backing up, if it is discharging directly or indirectly onto the surface of the ground, if the infiltration system elevation is below the high ground water level elevation, or if the system components encroach on established horizontal setback distances.

The owner must make the appropriate upgrades to the system within 2 years of discovery. The owner's failure to have the system inspected as required or to make the necessary repairs constitutes a violation of the code.

*Source: Title V, Massachusetts Environmental Code.*

performance requirements—should be based on monitoring *at the performance boundaries* (see chapter 5).

## Elements of a monitoring program

Any monitoring program should be developed carefully to ensure that its components consider public health and water quality objectives, regulatory authority / management entity administrative and operational capacity, and the local political, social, and economic climate. Critical elements for a monitoring program include

- Clear definition of the parameters to be monitored and measurable standards against which the monitoring results will be compared.
- Strict protocols that identify when, where, and how monitoring will be done, how results will be analyzed, the format in which the results will be presented, and how data will be stored.
- Quality assurance and quality control measures that should be followed to ensure credible data.

## System inspections

Mandatory inspections are an effective method for identifying system failures or systems in need of corrective actions. Inspections may be required at regular intervals, at times of property transfer or changes in use of the property, or as a condition to obtain a building permit for remodeling or expansion. Twenty-three states now require some form of inspection for existing OWTSs (NSFC, 1999). The OWTS regulatory authority or management entity

### Effluent quality requirements in Minnesota

St. Louis County, Minnesota, has established effluent standards for onsite systems installed on sites that do not have soils meeting the state's minimum requirements. Many of the soils in the county do not meet the minimum 3-foot unsaturated soil depth required by the state code. To allow for development the county has adopted a performance code that establishes effluent requirements for systems installed where the minimums cannot be met. Where the natural soil has an unsaturated depth of less than 3 feet but more than 1 foot, the effluent discharged to the soil must have no more than 10,000 fecal coliform colonies per 100 mL. On sites with 1 foot of unsaturated soil or less, the effluent must have no more than 200 fecal coliform colonies per 100 mL. These effluent limits are monitored prior to final discharge at the infiltrative surface but recognize treatment provided by the soil. If hydraulic failure occurs, the county considers the potential risk within acceptable limits. The expectation is that any discharges to the surface will meet at least the primary contact water quality requirements of 200 fecal coliform colonies per 100 mL. Other requirements, such as nutrient limitations, may be established for systems installed in environmentally sensitive areas.

### Documenting wastewater migration to streams in Northern Virginia

The Northern Virginia Planning District Commission uses commercially available ultraviolet light bulbs and cotton swatches to screen for possible migration of residential wastewater into area streams. The methodology is based on the presence of optical brighteners in laundry detergents, which are invisible to the naked eye but glow under "black" lights. The brighteners are very stable in the environment and are added to most laundry soaps. They are readily absorbed onto cotton balls or cloth swatches, which can be left in the field for up to two weeks. Users must ensure that the absorbent medium is free from optical brighteners prior to use.

Although the methodology is acceptable for screening-level analysis, it does not detect wastewater inputs from buildings that do not have laundry facilities and does not verify the presence of other potential contaminants (e.g., bacteria, nitrogen compounds). Despite these shortcomings, the approach is inexpensive, effective, and a good tool for screening and public education.

*Source: Northern Virginia Regional Commission, 1999.*

should collect information on new systems (system owner, contact information, system type, location, design life and capacity, recommended service schedule) at the time of permitting and installation. Inventories of existing systems can be developed by consulting wastewater treatment plant service area maps, identifying areas not served by publicly owned treatment works (POTWs), and working with public and private utilities (drinking water, electricity, and solid waste service providers) to develop a database of residents and contact information. Telephone, door-to-door, or mail surveys can be used to gather information on system type, tank capacity, installation date, last date of service (e.g., pumping, repair), problem incidents, and other relevant information.

Minnesota, Massachusetts, Wisconsin, and a number of counties and other jurisdictions require disclosure of system condition or assurances that they are functioning properly at the time of property transfer (see sidebar). Assurances are often in the form of inspection certificates issued by county health departments, which have regulatory jurisdiction over OWTSs. Clermont County, Ohio, developed an OWTS owner database by cross-referencing water line and sewer service customers. Contact information from the database was used for a mass mailing of information on system operation and maintenance and the county's new inspection program to 70 percent of the target audience. Other approaches used in the Clermont County outreach program included advisory groups, homeowner education meetings, news media releases and interview programs, meetings with real estate agents, presentations at farm bureau meetings, displays at public events, and targeted publications (Caudill, 1998).

**Biochemical application of a bacterial source tracking methodology**

Researchers from Virginia Tech analyzed antibiotic resistance in fecal streptococci to determine the sources of bacteria found in streams in rural Virginia. The team first developed a database of antibiotic resistance patterns for 7,058 fecal streptococcus isolates from known human, livestock, and wildlife sources in Montgomery County, Virginia. Correct fecal streptococcus source identification averaged 87 percent for the entire database and ranged from 84 percent for deer isolates to 93 percent for human isolates. A field test of the database yielded an overall bacteria source accuracy rate of 88 percent, with an accuracy rate of at least 95 percent for differentiation between human and animal sources.

The approach was applied to a watershed improvement project on Page Brook in Clarke County, Virginia, to determine the impacts of a cattle exclusion fencing and alternative stock watering project. Pre-project bacterial analyses showed heavy bacteria contamination from cattle sources (more than 78 percent), with smaller proportions from waterfowl, deer, and unidentified sources (about 7 percent each). After the fencing and alternative stock watering stations were installed, fecal coliform levels from all sources declined by an average of 94 percent, from 15,900/100 mL to 960/100 mL. Analysis of bacteria conducted after the project also found that cattle-linked isolates decreased to less than 45 percent of the total.

*Source: Hagedorn et al., 1999.*

The Town of Shoreham, Rhode Island, adopted a similar inspection program by ordinance in 1996 (Loomis et al., 1999). The ordinance mandates regular inspection of all systems by a town inspector. Septage pumping schedules and other maintenance requirements are based on the results of the inspection. Factors considered in the inspections include site characteristics, system technology and design, system use, and condition. The ordinance allows the town to prioritize inspection schedules in critical resource areas such as public wellheads and high-risk areas determined to be prone to onsite system failure. It also authorizes the town to assess fees, levy fines, and track the inspections.

### Prescribed maintenance

Where specific unit processes or treatment trains have satisfactorily demonstrated reliable performance through a credible testing program, some programs assume that identical processes or treatment trains will perform similarly if installed under similar site-specific conditions. The system would need to be managed according to requirements of the designer/manufacturer as outlined in the operation and maintenance manual to maximize the potential for assured performance. Therefore, some states monitor system maintenance as an alternative to water quality-based performance monitoring. The method of monitoring varies. In several states the owner must contract with the equipment manufacturer or certified operator to provide operation and maintenance services. If the owner severs the contract, the contractor is obligated to notify the state regulatory authority or other management entity. Failure to maintain a contract with an operator is a violation of the law. Other states require that the owner provide certified documentation that required maintenance has been performed in accordance with the system management plan. Requiring the owner to provide periodic documentation helps to reinforce the notion that the owner is responsible for the performance of the system. Chapter 2 provides additional information on prescriptive and other approaches to monitoring, operation, and maintenance.

### Water quality sampling and bacterial source tracking

OWTS effluent quality sampling is a rigorous and expensive method of onsite system compliance monitoring. Such programs require that certain water quality criteria be met at designated locations after each treatment unit (see chapter 5). Sampling pretreated effluent before discharge to the soil requires an assumption of the degree of treatment that will occur in the soil. Therefore, the performance requirements used to determine compliance should be adjusted to credit soil treatment. Unfortunately, some incomplete or inaccurate data equate travel time in all types of soil to pollutant removals under various conditions. Even when better data are available, it is often difficult to match conditions at the site from which the data were derived to the soils, geology, water resources, slopes, topography, climate, and other conditions present at the site under consideration. Effluent monitoring should be undertaken only when the potential risk to human health and the environment from system failure is great enough to warrant the cost of sampling and analysis or when assessment information is needed to establish performance requirements or identify technologies capable of protecting valued water resources.

Ground water sampling is the most direct method of compliance monitoring. However, because of the difficulty of locating monitoring wells in the effluent plume it has historically been used only for compliance monitoring of large infiltration systems. If performance standards are to be used in the future, ground water monitoring will become more commonplace despite its cost because it is the only true determinant of compliance with risk assessment criteria and values. Installing small-diameter drop tubes at various depths at strategic downgradient locations can provide a cost-effective approach for continuous sampling.

Monitoring of the unsaturated zone has been conducted as an alternative to ground water monitoring. This method avoids the problem of locating narrow contaminant plumes downgradient of the infiltration system, but allowances should be made in parameter limits to account for dispersion and treatment that could occur in the saturated zone. To obtain samples, suction lysimeters are used. Porous cups are installed in the soil at the desired sample depth, and a vacuum is applied to extract the sample. This type of sampling works reasonably well for some dissolved inorganic chemical species but is not suitable for fecal indicators (Parizek and Lane, 1970; Peters and Healy, 1988). Use of this method should be based on a careful evaluation of whether the method is appropriate for the parameters to be monitored because it is extremely expensive and proper implementation requires highly skilled personnel.

Water quality sampling of lakes, rivers, streams, wetlands, and coastal embayments in areas served by OWTSs can provide information on potential resource impacts caused by onsite systems. Concentrations of nitrogen, phosphorus, total and fecal coliforms, and fecal streptococci are often measured to determine possible impacts from system effluent. Unless comprehensive source sampling that characterizes OWTS pollutant contributions is in place, however, it is usually difficult to attribute elevated measurements of these parameters directly to individual or clustered OWTSs. Despite this difficulty, high pollutant concentrations often generate public interest and provide the impetus necessary for remedial actions (e.g., tank pumping; voluntary water use reduction; comprehensive system inspections; system repairs, upgrades, replacements) that might be of significant benefit.

Tracer dye tests of individual systems, infrared photography, and thermal imaging are used in many jurisdictions to confirm direct movement of treated or partially treated wastewater into surface waters. Infrared and thermal photography can show areas of elevated temperature and increased chlorophyll concentrations from wastewater discharges. Areas with warmer water during cold months or high chlorophyll during warm months give cause for further investigation (Rouge River National Wet Weather Demonstration Project, 1998). The Arkansas Health Department has experimented with helicopter-mounted infrared imaging equipment to detect illicit discharges and failed systems around Lake Conway with some success (Eddy, 2000), though these and other monitoring approaches (e.g., using tracers such as surfactants, laundry whiteners, and caffeine) are not typical and are still undergoing technical review.

Recently, some success has been demonstrated by advanced bacterial source tracking (BST) methodologies, which identify bacteria sources (humans, cattle, dogs, cats, wildlife) through molecular or biochemical analysis. Molecular (genotype) assessments match bacteria collected at selected sampling points with bacteria from known mammalian sources using ribotype profiles, intergenetic DNA sequencing, ribosomal DNA genetic marker profile analyses, and other approaches (Bernhard and Field, 2000; Dombek et al., 2000; Parveen et al., 1999). Biochemical (phenotype) assessments of bacteria sources conduct similar comparisons through analysis of antibiotic resistance in known and unknown sources of fecal streptococci (Hagedorn et al., 1999), coliphage serological differentiation, nutritional pattern analysis, and other methods. In general, molecular methods seem to offer the most precise identification of specific types of sources (animal species), but are costly, time-consuming, and not yet suitable for large-scale use. The precision of most biochemical approaches appears to be somewhat less than molecular methods, but analyte costs are lower, processing times are shorter, and large numbers of samples can be assayed in shorter time periods (Virginia Tech, 2001). It has been suggested that biochemical methods be used to screen large numbers of bacterial isolates for likely sources followed by an analysis of a subset of the isolates through molecular approaches to validate the findings. (For more information, see **http://www.bsi.vt.edu/biol_4684/BST/BST.html**).

Finally, some OWTS management agencies use fecal coliform/fecal streptococci (FC/FS) ratios as a screening tool to detect the migration of poorly treated effluent to inland surface waters. Under this approach, which is effective only if samples are taken near the source of contamination, the number of fecal coliforms in a sample volume is divided by the number of fecal streptococci in an equal sample volume. If the quotient is below 0.7, the bacteria sources are most likely animals. Quotients above 4.0 indicate a greater likelihood of human sources of bacteria, while values between 0.7 and 4.0 indicate a mix of human and animal sources. Several factors should be considered when using the FC/FS screening approach:

- Bacterial concentrations can be highly variable if the pH is outside the 4.0 to 9.0 range
- Faster die-off rates of fecal coliforms will alter the ratio as time and distance from contaminant sources increase
- Pollution from several sources can alter the ratio and confuse the findings
- Ratios are of limited value in assessing bays, estuaries, marine waters, and irrigation return waters

Sampling and analysis costs vary widely across the nation and are influenced by factors such as the number of samples to be collected and assessed, local business competition, and sample collection, handling, and transport details. Because of variability in price and the capacity of local agencies to handle sample collection, transport, and analysis, several cost estimates should be solicited. Some example analytical costs are provided in table 3-30.

**Table 3-30. Typical laboratory costs for water quality analysis**

| Parameter | Cost range per sample (in dollars) | Typical cost per sample (in dollars) |
|---|---|---|
| $BOD_5$ | 15–50 | 35 |
| $NO_2$ | 10–25 | 20 |
| $NO_3$ | 10–25 | 20 |
| Fecal coliform | 15–50 | 30 |
| TKN | 4–50 | 35 |
| Total phosphorus | 5–35 | 25 |
| TSS | 8–25 | 15 |

Source: Tetra Tech, 2000.

Because of the cost and difficulty of monitoring, underfunded management agencies have often opted to focus their limited resources on ensuring that existing systems are properly operated and maintained and new systems are appropriately planned, designed, installed, operated, and maintained. They have relied on limited water quality monitoring of regional ground water and surface waters to provide an indication of regional onsite system performance. Additional site-specific monitoring is recommended, however, where drinking water or valued surface water resources are threatened.

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