Literature Review for Septic System-Well Study in the Hyatt Creek Watershed

by

Nancy E. Deal, BS, MS REHS
Environmental Senior Specialist, Onsite Water Protection Branch
Division of Public Health, NC Department of Health and Human Services
December 2011

AUTHOR'S NOTE

This Literature review was prepared as part of a collaborative research project entitled "Influence of Septic Systems on Ground Water Quality in a Rural, Fractured Bedrock Setting, Hyatt Creek Watershed, Haywood County, North Carolina", conducted by the NC Division of Water Quality, Haywood Waterways Association (HWWA), NC State University, Division of Environmental Health Wastewater Discharge Elimination (WaDE) Program and the US Geological Survey (USGS). Grant funds for this review were provided by the Pigeon River Fund, administered by HWWA. At the inception of the project, the author was a member of the faculty of NC State University, but continued to participate in the project through her current role of Non-Point Source Specialist in the Onsite Water Protection Branch, NC Division of Public Health.

INTRODUCTION

The water quality and health risk where private wells and onsite wastewater treatment and dispersal systems (aka septic systems) are used on the same lot or in proximity is variable and site specific. Potable water wells are subject to contamination from many sources and among these are onsite septic systems. Septic systems have the potential to introduce contaminants (e.g. chemicals, pathogens, and nutrients) into groundwater, and thus into wells. This paper presents information from the scientific literature as it relates to this subject. The initial section describes 'traditional' contaminants (chemicals and pathogens) as well as 'emerging' contaminants (pharmaceuticals and consumer products) and summarizes information regarding the known or suspected health effects associated with contamination by these constituents. The following section describes the parameters and methods used to indicate that contamination has occurred and potentially identify the source. The concluding section discusses the physical characteristics of septic systems and wells that can affect their integrity and performance with reference to appropriate research. The body of knowledge on contamination of potable water sources by human sewage encompasses a broad range of research. Wherever possible, research that focuses on specifically on potable water wells, springs and onsite wastewater treatment systems is cited.

GROUNDWATER CONTAMINANTS AND ASSOCIATED HEALTH EFFECTS

Chemical Contaminants

Excessive levels of the nitrate (NO_3^-) form of nitrogen are of particular concern relative to drinking water supplies. A drinking water standard of 10 mg/L (as N) has been set by the US EPA largely on the basis of implicated development of acquired methemoglobinemia in infants (US EPA, 2011). Other factors may be responsible for the illness and the standard may thus be set unnecessarily high (Avery, 1999). While excessive amounts of nitrate have been linked with adverse health conditions, there are many potential sources of elevated nitrate aside of septic systems, including fertilizers (Tinker, 1991; Flipse, 1972; Katz et al., 1980), animal waste (Komor and Anderson, 1993; Katz et al., 1980) atmospheric deposition (Kaushal et al., 2007), as well as sewer exfiltration and landfills (Katz et al., 1980). Properly designed and operating soil-based treatment systems facilitate the conversion of ammonium form (NH_4^+) of nitrogen to the nitrate form (NO_3^-) because of aerobic, unsaturated conditions beneath a drainfield (Wilhelm et al., 1994). The fate and transport of the nitrate produced has been widely studied (Anderson, 1998; Katz et al., 2010b; Robertson et al., 1991; Wilhelm et al., 1996; and others). Seventy-two percent of groundwater samples collected between October 1985 and September 1996 in the mid-Atlantic region contained detectable nitrate levels. Ten percent of samples exceeded the US EPA MCL and were found primarily in agricultural areas (Ator and Ferrari, 1997). Thirty potable water wells completed in bedrock aquifers in the Blue Ridge physiographic region sampled for nitrate by USGS in 1997 had a median concentration of 0.419 mg/L and only one sample exceeded the USEPA maximum contaminant level (MCL) of 10 mg/L. The primary potential sources of nitrate in the area were listed as agricultural (fertilizer and manure), septic systems and precipitation. The authors concluded that neither septic effluent nor agricultural practices constituted a significant source of contamination in this study since detection of bacteria was concurrently low (Kozar et al., 2001).

Microbial contaminants

Direct contact with effluent or indirect exposure through contaminated drinking water can pose significant health risks because of the potential presence of microorganisms such as viruses, bacteria, protozoa and helminthes (Meschke and Sobsey, 1999; Crites and Tchobanoglous, 1998). Approximately 17 million persons in the United States rely on private household wells for drinking water each year, and more than 90,000 new wells are drilled annually throughout the United States (US General Accounting Office, 1997).

Contaminated groundwater is the most commonly reported source of waterborne disease outbreaks (WBDO) in the United States, associated with 92% of outbreaks between 2001 and 2002 (Blackburn et al., 2004) and 88% of outbreaks between 2005 and 2006 (Yoder, et al., 2008). Of 23 well-related WBDO during 2001-2002, thirteen were associated with community or non-community wells (subject to EPA regulations) resulting in 846 cases or 90% of the total. Ten were associated with individual wells (not subject to EPA regulations) resulting in 52 cases or 5.5% of the total. The total number of WBDO and cases from all sources declined between the two reporting periods. These trends could be a result of a lag in reporting, since these documents are regularly updated as additional information becomes available. However, although the percentage of the US population using privately owned individual water systems increased from 6% in 2003 to 17% in 2006 (US EPA, 2003; US EPA, 2006), the number and percent of WBDO and the total number of cases attributed to these systems decreased, as shown in Table 1. Under-reporting by private water supply owners is undoubtedly possible.

Table 1: Water Borne Disease Outbreaks (WBDO) by drinking water source for two reporting periods.

	All sources		Groundwater sources		Community and non-community wells		Privately-owned individual wells or springs	
Reporting period	**WBDO (% of total)**	**Cases (% of Total)**	**WBDO (% of total)**	**Cases (% of Total)**	**WBDO (% of total)**	**Cases (% of Total)**	**WBDO (% of total)**	**Cases (% of Total)**
2001-2002[1]	25 (100)	940 (100)	23 (92.0)	898 (95.5)	13 (59.1)	846 (90.0)	10 (40.9)	52 (5.5)
2005-2006[2]	9 (100)	518 (100)	8 (89.0)	458 (88.4)	6 (75.0)	478 (83.8)	2 (25.0)	32 (6.2)

[1]Blackburn et al., 2004
[2]Yoder et al., 2008

In 1992, public health officials reported that groundwater was responsible for many cases of endemic enteric disease that were too sporadic to easily identify the infection source, but viruses were presumed to be responsible (Craun, 1992). The etiologic agents for WBDO attributed to individual water supplies during two reporting periods are shown in Table 2. The source of contamination was not reported.

Table 2: Etiologic Agent for WBDO associated with individual water supplies

	Etiologic Agent: Number of outbreaks (Number of cases)				
Reporting Period	**Bacteria**	**Viruses**	**Parasites**	**Chemicals**	**Unknown**
2001-2002[1]	2 (15)	0 (0)	2 (16)	1(2)	5 (19)
2005-2006[2]	0 (0)	1 (16)	0 (0)	0 (0)	1 (16)

[1]Blackburn et al., 2004
[2]Yoder et al., 2008

While contamination of water supply wells is possible in any geological configuration, those located in fractured rock and karst settings are potentially vulnerable because of minimal attenuation of constituents and high groundwater flow rates (Sawyer, 2008). Disease outbreaks in such settings have been documented in Alaska (Beller et al., 1997), Wisconsin (Braatz, 2004; Borchardt et al., 2010), Ohio (O'Reilly et al., 2007; Fong et al., 2007) and South Dakota (Daly, 1993 as cited in Sawyer, 2008; Manduca, 2000 as cited in Sawyer, 2008). Kozar et al., (2001) reported 38% of wells completed in fractured bedrock aquifers tested positive for total coliform bacteria in an area of the Blue Ridge physiographic region where land use includes agricultural pursuits as well as septic systems. However, none of the 30 wells sampled tested positive for *E. Coli* which is more closely linked to the intestinal tract of warm-blooded animals. The authors theorized that the presence of total coliform from a probable non-fecal source may indicate the possibility of bacterial contamination in the study area, but concluded that fecal contamination was absent on the sampling date.

Organic Wastewater Contaminants (OWCs)

The use of pharmaceuticals and personal care products (PPCPs) may introduce constituents that can potentially disrupt the human (and animal) endocrine system. Such constituents are often referred to as "organic wastewater contaminants" (OWCs) by many sources and include volatile organic compounds or VOCs. Some compounds in this group are also referred to "emerging contaminants" (ECs) because their effects on human health are not well understood and maximum contaminant levels (MCLs) have not yet been established. Discussions of regulatory issues surrounding ECs include Harvey et al., 2006 and Richardson and Ternes, 2005.

Documented health effects from exposure to pharmaceuticals in drinking water supplies are currently restricted to research on non-human subjects. Results of *in vivo* tank experiments on rainbow trout (Oncorhynchus mykiss) and adult roach (Rutilus rutilus) were reported by Routledge et al. (1998). Male roach and trout produced high plasma levels of vitellogenin when exposed to environmentally relevant concentrations (0.001 to 0.1 μg/L) of 17 β-estradiol (E1) and estrone (E2) during a three-week exposure period. Folmar et al. (1996) reported similar effects in male carp (Cyprinus carpio) found near a major wastewater treatment plant outfall located in St. Paul, Minnesota. Triebskorn et al. (2004a; 2004b) reported that environmentally relevant concentrations (1 to 500 μg/L) of diclofenac resulted in cytopathological reactions in liver, kidney and gills of rainbow trout. Further, they reported concentration-related accumulation of the drug in all organs studied.

The “environmentally relevant” values used in the studies references above were based upon levels detected in sewage treatment plant effluent and receiving surface waters. Considerably lower concentrations have been reported in surface and ground water receiving discharge from septic systems. Standley et al. (2008) reported concentrations of hormones in surface water originating in aquifers receiving discharge from septic systems in the range of 0.0014 to 0.0065 μg/L. Likewise, much lower diclofenac concentrations have been documented in effluent plumes beneath septic drainfields (<0.004 to 0.02 μg/L by Carrara et al., 2008 and Conn et al., 2006) and in drinking water wells (0.046 μg/L by Miller and Meek, 2006).

Cunningham et al. (2009) studied the potential impact to human exposure to 44 active pharmaceutical ingredients (APIs) marketed by GlaxoSmithKline (GSK) representing 22 general pharmacological classes. They found that based upon currently available data, these compounds do not appear to pose an appreciable risk to human health from potential environmental exposure from drinking water and fish consumption. This conclusion was based upon predicted no effect concentrations from environmental exposure for human health ($PNEC_{HH}$s) which were compared to predicted environmental concentrations (PECs) calculated using regional assessment models.

Pharmaceuticals undergo transformation during human metabolism and these metabolites may also be endocrine disruptors. Salicylic acid and gentisic acid (resulting from hydrolysis of acetylsalicylic acid) have exerted toxic effects on embryos of zebrafish (Henschel et al., 1997) and on *Daphnia magna* and *Daphnia longispina* (Marques et al., 2004) but the half maximal effective concentration (EC50) values reported in these studies (1148 mg/L and 50 mg/L, respectively) exceed those detected in the environment by orders of magnitude. Lowe et al. (2009) reported salicylic acid concentrations in raw wastewater and STE samples ranging from non-detect to 205 μg/L, indicating that the occurrence of pharmaceutical residues from single-residential sources will be highly variable. Carrara et al., 2008 reported salicylic acid in the range of <0.1 to .03 μg/L in septic field wastewater plumes. Gibson et al. (2007) reported salicylic acid concentrations of 0.0078 ± 0.002 μg/L and .0096 ± 0.012 μg/L in a (non-potable) spring north of Mexico City on two separate sampling dates. Concentrations concurrently detected in wastewater (combined domestic, industrial, runoff and rainwater) upslope of the spring were 0.062 ± 0.14 μg/L and 19.06 ± 25.18 μg/L. Authors stated that concentrations for this compound were highly variable relative to others analyzed during a study designed to evaluate analytical methods for detection of EDC’s. Table 3 includes the referenced values.

Table 3: Salicylic acid concentration at specific sampling locations

Reference	Sampling location	Salicylic acid concentration range (µg/L)
Lowe et al., 2009	Raw WW and STE	ND - 205
Carrara et al., 2008	Septic field WW plume	<0.1 to .03
Gibson et al., 2007	WW upslope of non-potable spring	Date 1: 0.062 ± 0.14 Date 2: 19.06 ± 25.18
	Non potable spring	Date 1: 0.0078 ± 0.002 Date 2: .0096 ± 0.012

Toxic pharmaceutical derivatives may also form as a result of disinfection processes. Bedner and MacCrehan (2006) documented the transformation of acetaminophen to 1,4-benzoquinone and N-acetyl-p-benzoquinone imine as a result of chlorination of wastewater. Both of these disinfection by-products (DBPs) are toxic and the latter compound is the toxicant responsible for lethality of acetaminophen overdoses

Nonionic surfactants are present in common household products such as laundry detergents and household cleaners. Surfactants, fragrances, disinfecting agents and plasticizers are ingredients in personal care products as well. As they enter the waste stream these compounds are environmentally transformed into chemicals that can mimic estrogen and are variously referred to as *endocrine disrupting compounds* (EDCs), *endocrine active compounds* (EASs) or *pharmaceutically active compounds* (PhACs). Others are suspected carcinogens. Jobling et al. (1995) documented that many synthetic organic chemicals in effluents interfered with estradiol receptors in fish. Surfactants are chemically classified as alkylphenol polyethoxylates (APEOs) and are transformed into alkylphenol ethoxycarboxylates and the estrogenic alkylphenols known as nonylphenol and octylphenol (Ahel et al., 1987; Stanford and Weinberg, 2007). Domestic and industrial cleaning agents, fire-retardants, pesticides and drug additives also include compounds that may act as endocrine disruptors.

While documented effects on human health are lacking, two main properties of EDCs are that bioaccumulation occurs due to solubility in adipose tissues and synergistic effects may be seen when multiple compounds are present (Quan et al., 2005 as cited in Caliman and Gravrilescu, 2009). Health based guidelines are not available for most of the EDCs under study (Barnes et al., 2008; Focazio et al., 2008, Schaider et al., 2010) and the health effects of long-term exposure to low levels of these types of compounds, especially in complex mixtures, are not yet known. One study compared incidence of breast cancer to long term consumption of drinking water affected by wastewater effluent and found a positive association (Gallagher et al., 2010). The effluent plume originated from a municipal treatment works that used primary treatment followed by discharge to sand filter beds from 1937 until 1980 when secondary treatment was added. Effluent is discharged “to the groundwater”.

Three groups (plasticizers, insect repellant, and detergent metabolites) contributed about 66% of the total measured concentrations measured in groundwater samples collected downgradient from various potential sources of contamination at sites across the US (Barnes et al., 2008). Sample collection points were not necessarily drinking water wells. Measured concentrations in the majority of detections were <1 µg/L. Schaider et al. (2010) detected endocrine disruptors, including alkylphenols and other estrogenic phenolic compounds in public drinking water supplies suspected of being affected by commercial and residential septic systems on Cape Cod. Where health based guidelines applied, levels detected in samples fell below the guideline values.

Summary of Contaminants and Health Effects

Since excessive nitrate contamination may adversely affect infant health, it has been (and will continue to be) of critical importance in water quality. However, the validity of the MCL's established for nitrate on the basis of the perceived risk continues to be debated. Disease outbreaks due to microbial contamination have been more fully investigated in the relatively recent past and the sources have been more fully and specifically characterized. This has promoted a more complete understanding of the range of factors that contribute to outbreaks. While contamination by some OWCs has been well documented for decades, studies on emerging contaminants are garnering significant attention. Because of the potential for endocrine mimicry, concern over long term exposure even to low levels is justifiably increasing.

CONTAMINATION INDICATORS AND SOURCE TRACKING METHODS

Davis et al. (1998) described an ideal groundwater tracer as non-toxic, inexpensive, easy to detect, non-reactive, conservative, and not inclined to alter the natural direction of groundwater flow. Chemicals or compounds may be specifically introduced into systems to serve as tracers. Examples of introduced tracers include fluorescent dyes such as rhodamine, or uranine used to track the flow of effluent from its source (Murray et al., 2007). However, characterization of background fluorescence sources that may cause interferences (such as dissolved organic matter) must also be considered when using introduced tracers (Brown, 2009). Effluent constituents commonly present in elevated concentrations are also used to document contamination of potable water wells and (through analysis along an effluent plume) attenuation in soil. These may be non-toxic chemicals (such as chloride or bromide) or contaminants (such as pathogenic organisms or OWCs). The rest of this section focuses on indicators that enter the waste stream through 'typical use' as opposed to those which might be intentionally introduced.

Chemical Indicators

Nutrients

Nutrients are commonly used to indicate the potential presence of wastewater. Total nitrogen and nitrogen species are indicative of potential contamination due to their presence in human waste. Concentration of ammonium nitrogen (NH_4^+) can indicate sewage contamination, but may also originate from animal wastes or fertilizers (Verstraeten et al., 2005). Values ranging from 0.5 to 2 mg/L of NO_3^- (as N) have been reported to be the maximum concentration of nitrate in natural ground water unaffected by anthropogenic sources (Hallberg and Keeney, 1993; Komor and Anderson, 1993; Gallagher et al., 2010).

Chloride, pH and Specific Conductance

Chloride concentrations can be used to identify animal waste contamination of rivers and groundwater (Aravena & Robertson, 1998; Seiler, 2005). In, 1990, Alhajjar et al. (1990) compared the relative effectiveness of using Cl, pH and specific conductance as indicators of septic plumes in groundwater. Concentration of Cl decreased by 3 times with each 3m downgradient distance from the source (from 12 to 35 mg/L), a characteristic of a conservative tracer. Specific conductance decreased more rapidly than Cl with distance from the drainfield as a result of dilution and interactions with soil anions and cations. While pH decreased with distance from the drainfield, this was presumably from the acidifying effects of nitrification. Although this makes pH an acceptable tracer, Cl was deemed the most effective and specific conductance was considered semi-conservative. Chloride concentrations of 30 mg/L were used to delineate wastewater plumes under tile beds in Ontario, Canada (Carrara et al., 2008) and concentrations of 28 mg/L have been used as a baseline value to indicate fecal contamination in household wells in 7 major geologic districts in Wisconsin (Borchardt et al., 2003). The latter study found that viruses were present whenever Cl levels were elevated in the wells, but there was a concurrent low positive predictive value for the presence of enteric viruses.

Chloride concentrations at different sampling points have a wide range. Those in potable water wells have been reported from 2.7 to 886 mg/L (Brendle et al., 2006; Miller and Meek, 2006; Kozar et al., 2001; Close et al., 1989). Alhajjar et al. (1990) reported a range from 1 to 830 mg/L in STE and 0.5 to 560 mg/L in wastewater plumes. Chloride may originate from road deicing agents in cold climates and from saltwater intrusion in coastal regions. Other confounding sources of both Na and Cl include animal waste operations and municipal landfills. The importance of comparing potable water supply concentrations to those of background soil water (for inherent levels and confounding sources), septic effluent and effluent plumes is apparent. Use of additional indicators (bromine and iodide) can help to separate other potential sources from domestic wastewater (Panno et al., 2006).

Specific conductance is higher in samples influenced by Na and Cl (common human dietary components) than when Ca, Mg and HCO_3 dominate (Panno et al., 2006). Higher specific conductance can also be indicative of elevated total dissolved solids content which might be a result of sewage contamination.

Boron

Boron (B) has also been used as an indicator of contamination by domestic wastewater. Naturally occurring B originates from weathering of rock and background levels can be quantified. Boron in the form of borosilicates and borates are present in the range of 5 to 133 mg/L (5,000 to 133,000 µg/L) in soils (Brady and Weil, 2002) and confined aquifer groundwater generally contains B concentrations in the range of 30 to 150 µg/L (Ford and Tellam, 1994). At higher soil pH (between 8 and 10) adsorption to soil particles may occur (Bundschuh et al., 1993). Anthropogenic B generally enters the waste stream as an ingredient in household detergents (sodium perborate), but groundwater affected by metal-industry activities may also exhibit high B concentrations (average 745 µg/L) as reported by Ford and Tellam (1994). Clay minerals (especially illite) tend to adsorb B (Ford and Tellam, 1994) and it is thus best used as an indicator in soils with low clay content (Verstraeten et al., 2005).

Swartz et al., (2006) measured B concentrations in and around a septic system consisting of a septic tank, distribution box and 3 leachate pits installed to discharge 3.0 m below land surface (bls) and 0.4 m above the water table. Upgradient wells exhibited no background B while septic tank effluent exhibited a concentration of 216 µg/L. Monitoring wells installed below the water table and downgradient of one leachate pit exhibited decreasing amounts of B with depth (declining from 181 to 17 µg/L from 3.5 m to 5.6 m) suggesting dilution by groundwater. Shallow (<30m deep) potable water wells completed in the Platte River (Nebraska) alluvium and located less than 80 m from a septic tank system had B concentrations ranging from 38 to 193 µg/L (Verstraeten et al., 2005). Boron compounds persist through biological treatment processes and are mobile once they reach the soil. If soil pH is more neutral and the role of clay and minerals is identified, elevated B concentrations are indicative of anthropogenic additions, including industrial sources (Bundschuh et al., 1993). However, unless additional indicators are also present, contamination *per se* cannot be confirmed or assumed (Verstraeten et al., 2005).

Chemical Ratios

Ratios of chemical constituents are also useful, both for documenting the presence of contamination, and identification of source. Spruill et al. (2002) determined that a Na:K ratio greater than 3.2 was most effective variable for distinguishing septic effluent from other sources of contamination (poultry litter, animal waste and crop fertilizer). Wilhelm et al. (1994) found Na:K ratios of 8 in septic wastes in Canada while ratios for swine lagoon wastes, stockpile broiler or layer litter and common fertilizers were all less than 0.5 (Zublena et al., 1991; Zublena et al., 1993). The ratio of chloride to bromide (Cl:Br) has also been used to identify septic effluent. Davis et al. (1998) reported Cl:Br ratios ranging from 300 to 600 in domestic sewage and 100 to 200 in shallow groundwater. Panno et al. (2006) measured Cl:Br ratios of precipitation, unaffected soil water and aquifers, animal waste, sea water and landfill leachate in addition to septic effluent. Ratios for the STE were among the highest measured in this study, ranging from 65 to 5404. Katz et al. (2010a) reported a median ratio of 694 in STE and 518 in groundwater beneath the associated drainfield. Authors suggested a range of Cl:Br of 400 to 1100 as a screening tool. However, in this and a subsequent study they emphasized the need to assess Cl:Br ratios in conjunction with other chemical indicators (sulfate, dissolved organic carbon [DOC], organic wastewater compounds) as well as microbiological indicators to increase certainty in discrimination among STE and other sources of contamination (Katz et al., 2010a; Katz et al., 2010b).

The ratio of nitrate to chloride (NO_3^-:Cl) in the Morongo ground water basin in California was used to determine that septage from septic tanks was the primary source of nitrate in groundwater affected by artificial recharge of imported water. Authors noted the importance of analyzing the ratio in adjacent soil to measure background concentrations. They also analyzed for pharmaceuticals, caffeine and stable isotopes of nitrogen. Native ground-water ratios ranged from 0.5 to about 1.5, imported water ratios ranged from 0.006 to less than 0.1, and septage had a ratio of 1.9. (Nishikawa et al., 2003).

Microbial indicators

Microbial indicators include coliform bacteria, fecal coliform bacteria and coliphage. Total coliform include the bacterial genera *Escherichia, Citrobacter, Klebsiella and Enterobacter* and their source may be either fecal or vegetative. Fecal coliforms (a subset of the coliform group) are indicative of contact with human or animal waste, but not necessarily an indication of the presence of enteric pathogens. *Escherichia Coli* or *E. coli* is the fecal coliform bacteria most commonly used to test for fecal contamination, but other indicators may be more effective.

Bacteriophages are viruses that infect and replicate in bacterial cells. Coliphage are viruses that specifically infect coliform bacteria. F-specific coliphage infect the F-pili of male strains of coliform bacteria while somatic coliphage infect the outer cell wall of coliform bacteria. F-specific coliphages can be further categorized as FRNA and FDNA depending upon the route of infection and can replicate in warm-blooded hosts. Somatic coliphage also replicate in warm-blooded hosts but may also replicate in natural waters (IAWPRC Study Group on Health Related Water Microbiology 1991). There are four antigenically distinct serogroups of FRNA coliphages, with groups II and III differing from those predominating in animals (groups I and IV) (Hagedorn, 2010). Verstraeten (2005) postulated that coliphage could be a better indicator than bacteria because of smaller size, longer survival and lower tendency to adsorb due to a negative charge. DeBorde et al. (1998) reported low die-off rates of seeded coliphage at ambient ground temperature. Studies of coliphage thus offer source tracking capabilities for confirming the origin of contamination.

The usefulness of microbial indicators of contamination in drinking water wells varies. If a fecal indicator is present, pathogenic viruses may be absent simply due to the lack of infections and virus shedding at the time of sampling. In one epidemiological study, children who drank from private wells that were coliform positive were not at increased risk for diarrheal disease while those who drank from private wells that were positive for fecal enterococci had 6-fold greater odds of becoming ill with diarrhea of unknown etiology (Borchardt et al., 2010). Miller and Meek (2006) detected no coliphage and limited total coliform and enterococci when pharmaceuticals and personal care products (PPCPs) were found in potable water wells in Helena Valley Montana. Of the organisms studied (total coliform, E. Coli, enterococci, male-specific coliphage and somatic coliphage) total coliform appeared to be the most reliable indicator of fecal contamination. When studying systems in a sand aquifer setting, Hinkle et al. (2005) found that coliphage detections in monitoring wells beneath systems were sporadic and not reproduced in repeated or replicate samples. Borchardt et al. (2003) reported that four virus-positive wells tested negative for FRNA coliphages, E.coli and fecal enterococci (i.e., a true-positive rate = 0%). The authors indicated that low true-positive rate for FRNA coliphage might be attributed to a small number of positive FRNA samples. In that study, total coliform had a true-positive rate of only 25%. Despite the source-specificity of FRNA coliphage, Gilpin et al. (2003) indicated that widespread use as an indicator was location-dependent as a result of antibiotic usage among the study population. Human and veterinary use of antibiotics has caused fecal microbes to develop various levels of antibiotic resistance. Bacteria from wildlife sources exhibit a lack of antibiotic resistance while bacteria from domestic animals and humans have a range of resistance to types and concentrations of commonly used antibiotics. Assessment of multiple antibiotic resistance (MAR) patterns and discriminate analysis were first used to distinguish among human and animal sources of fecal streptococci by Wiggins in 1996. Geary (2003) compared unknown fecal streptococci isolates and was able to discriminate between some sources. However, separation of human isolates was not conclusive. Through comparison of known-source isolates from the watershed being studied, human sources can be successfully separated from animal ones as demonstrated in research conducted on surface waters (Hagedorn et al., 1999).

Seasonal variations in total and fecal coliform levels (Long and Plummer, 2004) and viruses (Scandura and Sobsey, 1997) have been documented and attributed not only to variations in temperature but also amount of precipitation. WBDO reported during the period 2001-2002 peaked during higher temperatures from June to September. Sorbitol fermenting *Bifidobacteria* (a strict anaerobe) were measured and detected in areas of dense development (Long and Plummer, 2004). The percentage of detections was higher in summer than in winter (38.9% versus 10%), presumably due to higher dissolved oxygen conditions during colder weather. Due to the temporal and geographical variations inherent to microbial indicators there is general agreement that their use is most appropriate as part of a suite of determinants (Gilpin et al., 2003).

The ratio of fecal coliform to fecal streptococci concentrations of 4 or more can be used to indicate human sources of pollution (Edwards et al., 1998). However, due to variable relative survival rates of the organisms, this method is no longer advocated (APHA, 2005).

Organic Wastewater Contaminant Indicators

Because they originate from anthropogenic sources, organic wastewater contaminants can serve as indicators of contamination by septic systems. This includes the use of pharmaceuticals, caffeine (and its metabolite paraxanthine), fecal sterols/stanols, optical brighteners (OBs,also known as fluorescent whitening agents or FWAs), as well as a host of consumer products (fragrances and flavorants, flame retardants and plasticizers, disinfectants, polyaromatic hydrocarbons, and pesticides). An ideal indicator has specificity; temporal and geographic applicability; sensitivity and accuracy; repeatability; matrix independence; practicality; and rapidity of results (Hagedorn and Weisberg, 2009).

Highly sensitive analytical techniques are required for detection of OWCs because they are generally present in low concentrations. New applications of liquid chromatography and mass spectrometry continue to improve accuracy at nanogram per liter levels in both liquid and solid samples (Caliman and Gravrilescu, 2009). Methods that combine isotope dilution with gas chromatography and tandem mass spectrometry have now been used to analyze effluent collected from multiple points in a septic system treatment train (including septic tank, pump tank, aerated wetland/sand filter and associated drainfield soil) to demonstrate applicability across several matrices (Stanford and Weinberg, 2007).

Pharmaceuticals and Consumer Products

Carbamazepine, diphenydramine, coprostanol, optical brighteners and caffeine are commonly present in sufficiently high concentrations for detection in various wastewaters (Glassmeyer, et al., 2005; Hagedorn and Weisberg, 2009). Both prescription (carbamazepine, sulfamethoxazole, trimethoprim and warfarin) and non-prescription drugs (caffeine, paraxanthine and acetaminophen) have been detected in STE in Montana (Godfrey and Woessner, 2004; Godfrey et al., 2007). Carbamazepine was the most frequently detected compound in surface and ground waters (Focazio et al., 2008) and has been found to be persistent through wastewater treatment (Matamoros et al., 2009) and soil passage (Clara et al., 2004; Godfrey et al., 2007).

Carrara et al. (2008) detected 10 of 12 compounds analyzed in groundwater beneath and downgradient from three septic systems, but reported high variability among the sites in the number of detections and concentrations. Ibuprofen, gemfibrozil, and naproxen were transported through soil at the highest concentrations and greatest distances from the source and transport was enhanced within anoxic zones where treatment would be minimal. Katz et al. (2010b) detected both acetaminophen and sulfamethoxazole in STE, lysimeters and drainfield wells on sites in a karst aquifer in Florida. However, on the site where sulfamethoxazole was detected in drainfield wells, it was not detected in the associated STE. Sulfamethoxazole, carbamazepine, Dilantin and diclofenac were the most-frequently detected pharmaceuticals in potable water wells (in both bedrock and alluvial settings in Montana) located such that onsite septic system discharges could be affecting groundwater (Miller and Meek, 2006). However, sampling of associated systems was not performed. Verstraeten et al. (2005) detected trace amounts (0.05 µg/L) of antibiotics and pharmaceuticals (up to 0.129 µg/L) in potable water wells of varying construction in Nebraska, but concluded that they were not reliable indicators of contamination due to variations in amounts and types used. Hinkle et al. (2005) could not correlate concentrations of OWCs with chloride concentrations and postulated that those present either did not originate from onsite septic systems or were introduced sporadically.

Szabo et al. (2004) investigated the use of a variety of organic wastewater contaminants for tracing effluent from septic systems. They detected fragrances, detergent metabolites, and steroids in septic tank effluent samples. Median concentrations of the compounds were as follows: 4-nonylphenol, 26 ug/L per liter); phenol, 23 ug/L; caffeine, 31 ug/L; cotinine, 2.4 ug/L; menthol, 15 ug/L; 3-beta-coprostanol, 24 ug/L; cholesterol, 17 ug/L; and β-sitosterol, 3.5 ug/L. Analysis of samples from drive point wells downgradient from drainfields was not yet complete. Samples from associated private domestic wells were negative for all analytes except phenol which was detected in blank samples at similar levels as those in wells (maximum 0.6 ug/L).

Sources are divided on the use of caffeine as an indicator. The catabolic conversions of caffeine that occur in septic tanks as well as its persistence and mobility in soil require better understanding before its utility will be confirmed. Approximately 97% of caffeine is converted to metabolites; thus low concentrations limit its usefulness (Gilpin and Devane, 2003). However, low levels of caffeine with concurrently elevated nitrate concentrations constitute "clear and unambiguous evidence" of contamination from domestic wastewater due to system malfunction, inadequate separation, and minimal exposure to fine grained or organic sediment (Seiler et al., 1999). Swartz et al. (2006) reported that caffeine, along with paraxanthine and OBs may be useful as indicators for OWCs if complemented by boron and inorganic nitrogen species.

The type and amount of pharmaceuticals and consumer products detected in any given setting will be dependent upon the nature of use (amount and frequency) by the target population (Verstraeten et al., 2005; Godfrey et al., 2007; Conn et al., 2010b). Concurrent sampling of STE, associated groundwater and (ideally) drinking water sources would presumably be the optimum method for indicating potential contamination, but multiple studies referenced here highlight the variability in their detection at multiple locations within a given setting.

Fecal Sterols/Stanols

Cholesterol from human sources is converted primarily to coprostanol which constitutes 60% of the total sterols of human feces. Coprostanol is not found in unpolluted fresh or marine waters and 24-ethyl coprostanol is a principal herbivore indicator (MacDonald et al., 1983). Septic tank effluents were found to have higher concentrations of fecal stanols than did sewage treatment plant effluents (Gregor, 2002 as cited in Hagedorn and Weisburg, 2009). Ratios of different stanols have been put forth as indicators of contamination from various sources (Leeming, et al., 1996; Grimalt et al., 1990). Human fecal pollution typically has a ratio of coprostanol:24-ethylcoprostanol of greater than 1 (Leeming et al., 1996; Leeming and Nichols, 1996). Spatial changes in ratios have also been used. Investigation of surface water pollution found a five-fold increase in the ratio of coprostanol:24-ethylcoprostanol and a 35-fold increase in that of coprostanol:24-ethylepicoprostanol in surface water samples collected downstream from known sources. Although the ratios seemed to provide the clearest indication of surface water pollution when compared to OBs, microbial indicators and chemical constituents, authors stressed the need to use multiple indicators for increased accuracy (Gilpin et al., 2003).

Impediments to using fecal sterols as indicators include a lack of standard analytical methods (Bull et al., 2002); generation of data that is complex to interpret (Gilpin and Devane, 2003), and; concerns over dilution in surface water (Pond et al., 2004).

Optical Brighteners

Optical brighteners (OBs) are sometimes referred to as fluorescent whitening agents or FWAs. OBs are organic additives to household detergents designed to whiten clothing, but may also be present in other products such as toilet paper (Hagedorn et al., 2005). While part of the OBs in detergents binds to clothing during the wash cycles, some is discharged as wastewater (Murray et al., 2007). OBs absorb long-wave ultraviolet light in the range of 365 nm and reemit it within the blue portion of the visible spectrum. The range of reemission has been reported as 415 to 435 nm (Hagedorn and Weisburg, 2009) and 400 to 500 nm (Fay et al., 1995 as cited in Murray et al., 2007).

Use of OBs as an indicator is subject to potential interferences from soil organic matter (Jourdainnais and Stanford, 1985; Alhajjar et al., 1990) or other aromatic compounds that resemble OBs (Hagedorn et al., 2005). Alhajjar et al. (1990) found fluorescence of OBs to be unreliable as an indicator because they did not detect higher levels in a septic plume relative to background wells, presumably because of naturally occurring humic and fulvic acids in the soil. In this case, Cl was found to be the best indicator. Jourdannais and Stanford (1985) could not separate relative fluorescence of optical brighteners from that of naturally occurring compounds in littoral areas of Flathead Lake in Montana. Instead, they calculated the ratio of fluorescence to DOC. Ratios exceeded 380 were indicative of malfunctioning septic systems. Gilpin et al. (2003) investigated use of OBs, microbial and chemical indicators along with ratios of stanols in a river setting. Results indicated that individually, the indicators were unlikely to provide a full account of fecal pollution sources and should instead be used collectively. Thus, although the importance of using confirmatory sampling for other constituents in addition to OBs is demonstrated, their use as a screening tool is nonetheless advocated (Hagedorn and Weisberg, 2009).

There are many analytical methods that are employed for analysis of OBs. A simple and cost effective approach is to place dye-free cotton pads in the sample liquid for a period of time (2–3 days), allowing for any OBs present to bind to the fibers in the pads. Subsequent exposure to UV light causes the pads to fluoresce (Dixon et al., 2005). Murray et al. (2007) used polyethersulfon (PES) membranes to filter water samples and retain solids that potentially included OBs. The membranes were exposed to light and a spectrofluorophotometer (i.e., spectrometer) was then used to measure the wavelength of light emitted. High performance liquid chromatography or mass spectrometry is another option (Shu and Ding, 2005). Hand held fluorometers are now available for use in the field (Hartel et al., 2007). Each method has advantages and limitations related to specificity, accuracy and repeatability in addition to relative cost and required technical expertise (Hartel et al., 2007). Various laboratory methods have been used to overcome interferences from both natural and anthropogenic organic constituents. These include double extraction of samples into 1,2-dichloroethane followed by flourometric detection (Close et al., 1989) and measuring the change in fluorometric values before and after UV exposure (Hartel et al., 2007). The latter method takes advantage of the fact that unlike natural organic compounds, OBs degrade predictably when exposed to UV light.

Stable Isotopes

The percentage of two stable isotopes of Nitrogen (^{14}N and ^{15}N) is nearly constant in the atmosphere at 0.366%. Biochemical and physical processes preferentially use the lighter isotope, and the heavier form becomes enriched in the remaining reactants. Variations in isotopic ratios can help identify the source of nitrate. The ratio of the stable isotopes is typically expressed in delta (δ) units. The equation for N isotopes is:

$$\delta^{15}N\ (‰) = \frac{(\delta^{15}N/\delta^{14}N)_{Sample} - (\delta^{15}N/\delta^{14}N)_{Standard}}{(\delta^{15}N/\delta^{14}N)_{Standard}} \times 1000$$

The ratio has units of “per mil” (‰). Positive values are indicative of enrichment and negative values are depleted. A parallel equation can be used for all stable isotopes. $\delta^{15}N$ values in NO_3^- are reported relative to atmospheric air (AIR) and $d^{18}O$ values in NO_3^- are reported relative to Vienna Standard Mean Ocean Water (VSMOW).

The variation among ratios of stable isotopes of nitrogen is a result of bacterially-mediated reactions which may be unpredictable (Heaton, 1986; Kendall et al., 2007; Xue et al., 2009). $\delta^{15}N$ of various sources of NO_3^- have been reported over the years with considerable overlap of ratios from known sources. The most recent and extensive compilation of reported ratios illustrates the significant overlap in ranges reported for sewage (+5 to +20‰) and manure (+6 to +26‰) (Xue et al., 2009). Synthetic fertilizers exhibit ratios in the range of -4 to +4‰ (Kendall et al., 2007). Like those of nitrogen, isotopes of oxygen are enriched or depleted on the basis of microbial transformations. Synthetic fertilizers exhibit $\delta^{18}O_{NO}{}^{3-}$ values in the range of atmospheric oxygen (+23.5‰) while $\delta^{18}O$ values for nitrate derived from human wastes should be depleted relative to atmospheric sources, but enriched relative to soil nitrate (i.e., between -10‰ and + 23.5‰). Unlike stable isotopes of nitrogen and oxygen that occur in nitrogen species, isotopes of Boron (B) are unaffected by microbial processes and are thus conservative. Adsorption to clay minerals is unlikely at pH lower than about 9. Anthropogenic sources of B (from detergents) are isotopically distinct from natural B and ratios of +7 to 25‰ have been documented in groundwater contaminated with sewage (Vengosh et al., 1994; Bassett et al., 1995).

The use of a multi-isotope and multi-tracer approach has been strongly advocated (Kendall et al., 2007) and implemented (Aravena et al., 1993; Komor and Anderson, 1993; Aravena and Robertson, 1998; Widowry et al., 2004; Seiler, 2005; Verstraeten et al., 2005). Multi-variate statistical analysis of isotopic, chemical and physical parameters has successfully distinguished among sources (Spruill et al., 2002). Most field investigations combine analysis of multiple isotopes with investigations of traditional indicators (nitrogen species, chloride, bromide, and microbial) to increase accuracy of source tracking. The multi-isotopic tracer approach also reveals additional information about the chemical and microbial transformations that occur in soil and water (Komor and Anderson, 1993; Aravena and Robertson, 1998; Seiler, 2005).

Spatial and temporal variation seen with other indicators also applies to the use of stable isotopes (Komor and Anderson, 1993; Aravena and Robertson, 1998; Verstraeten et al., 2005). Enriched $\delta^{15}N$ with increasing depth may indicate either denitrification or changes in proportions of nitrate from various sources. Seasonal variation may occur due to changing rates of denitrification or changing sources at any given time. Concurrent measurement of other analytes such as caffeine (Seiler, 2005), Cl:Br ratio (Vengosh et al., 1994) has been reported to define the source of contamination when isotopic evidence was inconclusive. Fogg et al. (1998) emphasized the importance of measuring the specific $\delta^{15}N$ of suspected sources of nitrate contamination for the particular area of interest since diet, use of hormones or antibiotics and a multitude of other site-specific parameters can affect baseline values. The same applies to background concentrations of other isotopes as well.

Methods for isotopic analysis have been well documented (Sigman et al., 2001; Casciotti et al., 2002; Czerwieniec and Tomaszek, 2007; Coplen et al., 2004).

Summary of Contamination Indicators and Source Tracking Methods

The literature describes many different options for documentation of contamination and source identification. However, research has not concluded that individual indicators are effective. Ratios of stable isotopes must be interpreted carefully and supported by chemical and/or microbial evidence. Optical brighteners are advocated as an economical and practical screening tool, but potential interference from organic fractions may skew results. Chloride concentrations in excess of 28 mg/L can indicate fecal contamination since viruses have been concurrently detected. However, a low positive predictive value for the presence of enteric viruses is of concern. Additionally, Cl concentrations must be compared to background levels as well as effluent and groundwater plume concentrations to be definitive.

The nature of soil-based septic system treatment and dispersal is that treated water rejoins the hydrologic cycle. An indicator must be detectable at the source and respond to treatment processes in a similar fashion as contaminants. Reliable indicators should appear in well water only if contamination from untreated wastewater is present and not simply because treated water recharge has occurred. The presence of B in potable wells may simply indicate that recharge has occurred since it generally survives treatment processes that would normally remove contaminants. The fact that carbamazepine has been found to be persistent in both pretreatment components and soil by many researchers is less indicative of its usefulness as an indicator than of the difficulty of removing it as a constituent. Pharmaceuticals and consumer products are most reliable when soil conditions are such that oxidation of organic constituents does not occur, i.e., when treatment is inadequate due either to excessive hydraulic loading or insufficient separation to the limiting condition. These are conditions which increase the potential for contamination. Under these circumstances, PPCPs could be an effective indicator. However, the variability of use (amount and frequency) limits the applicability of these for broad use and analyses may be expensive. Carrara et al (2008) reported that Cl traced a wastewater stream farther downgradient than pharmaceuticals. The most reliable evidence will emerge from studies that include sampling for multiple indicators from the background, the source, the effluent plume as well as potentially affected potable water wells.

The presence of multiple indicators at multiple locations is strong evidence of contamination provided that the flowpath of groundwater can be verified. Location of monitoring points relative to dispersal fields is an important consideration because assuming a single wastewater plume from the source may be misleading. Rea and Upchurch (1980) documented multiple bifurcating plumes resulting from variations in geological characteristics and local hydrologic characteristics. Local and seasonal variation in hydrology was demonstrated by O'Driscoll, et al. (2010) in research on a system in eastern NC. Despite extensive mapping of the wastewater plume, seasonal variation in water table elevations indicated the potential for reverse groundwater flow. Transport time is also critical. Sherlock et al. (2002) reported that concentrations of Cl tracers vertically beneath and 3 m downslope of a septic system drainfield peaked 125 and 200 days (respectively) after injection.

Ultimately, multi-tracer approaches are advocated, particularly because of the temporal and spatial variability associated with use of septic systems. From a practical perspective, it makes sense to consider economic factors when choosing indicators. Inexpensive on-site screening techniques can be used to measure nitrate, ammonium, dissolved oxygen, chemical ratios and optical brighteners. Screening for a basic suite of PPCPs could be incorporated if funds allow and if the nature and frequency of use can be documented. If screening methods reveal potential contamination, microbial testing should be performed.

WELL, SPRING AND SEPTIC SYSTEM CHARACTERISTICS RELATIVE TO CONTAMINATION

Well construction and spring development methods are of obvious importance in protecting drinking water sources and groundwater in general. Although proper construction/development does not guarantee that contamination will be prevented, it does offer some protection. If wastewater treatment systems are not sound and if wells and springs are vulnerable, contamination may occur and disease may result. When assessing attenuation of contaminants, septic system design, siting, installation, use and maintenance must all be considered. Older systems tend to be installed at a greater depth and the integrity of components may be questionable. Older systems have historically received little or no maintenance. Increased understanding of system function and performance has resulted in additional regulatory scrutiny and oversight in recent decades, but prescriptive standards still prevail and management programs are still evolving.

Soil and Site Considerations in Outbreak Investigations

The specific factors associated with waterborne disease outbreaks of microbial origin are important to consider. An investigation of the circumstances may reveal a direct connection between a poorly constructed septic "pit" and a single well as reported in Beller et al. (2007). Dye tests indicated that effluent from the septic pit was detected in the well within 24 hours. Alternately, a combination of contamination sources may be implicated as when a norovirus outbreak occurred on Ohio's South Bass Island in 2006 (O'Reilly et al., 2007). Thin soil overlying a karst aquifer has little treatment capacity for the septic systems used outside the village. The municipal system outfall discharges to Lake Erie which is likely connected to ground water via fractures. Cross connections between the public and private drinking water supplies were discovered during the investigation. This combination of factors resulted in illness for 1450 (reported) persons.

However, newly constructed systems that meet current BMPs for siting and installation have also been implicated in outbreaks. A norovirus outbreak in Wisconsin provided definitive proof of drinking water well contamination by a new septic system that was designed and installed according to code. The recharge time of 15 days between septic system and drinking water source was inadequate to attenuate the viruses introduced to the system by ill employees (Borchardt et al., 2010). Although the soil has tremendous capacity to treat pathogens (sometimes within a matter of feet) there is tremendous variability in pathogen concentrations among different individuals in the human population and variability in concentrations within individual people over time (Hinkle et al., 2005).

Contaminant Attenuation in Pretreatment Components and Soil

Matamoros et al. (2009) reported 80% removal of pharmaceutical and consumer products in biofilters, sand filters and constructed wetland pretreatment components in Denmark. Additional removal of OWCs occurs as wastewater moves through soils. This has been demonstrated in areas where irrigated wastewater recharges aquifers (Cheftez et al., 2008; Gibson et al., 2007). Attenuation in septic systems has also been documented. Concentrations of contaminants measured beneath septic system drainfields and at downgradient monitoring points suggested significant removal of OWCs in Massachusetts (Swartz et al., 2006), Ontario (Carrara et al., 2008), Florida, Minnesota and Colorado (Conn et al., 2010b). Removal is highly dependent upon oxidation state of the soil, thus indicating the importance of adequate separation distance to limiting conditions (such as seasonal high groundwater or bedrock) and appropriate hydraulic loading rates. Katz et al. (2010b) reported that microbial indicators, viruses, OWCs and pharmaceuticals were highly attenuated or removed from STE that percolated through 5 to 7 m of unsaturated soil. However, higher detections of fecal indicators, enteric viruses, OWCs, and pharmaceuticals were found in groundwater adjacent to systems where depth to limestone was the shallowest and average daily water use was highest. Carrara et al. (2008) studied systems on three different sites and found greater removal efficiencies in those that incorporated aerobic treatment and used lower hydraulic loading rates. Conn et al. (2010a) reported 90% removal of most target constituents after movement through 2.4 m of sandy loam regardless of level of effluent treatment (primary or secondary) or hydraulic loading rate. However, Godfrey et al. (2007) detected carbamazepine, sulfamethoxazole and nicotine after effluent filtration through 2 m of sand.

The content of organic carbon in a soil also affects attenuation rates. Sun et al. (2006 as cited in Caliman and Gravrilescu, 2009) indicated that sorption to humic substances affected their reactivity and bioavalability as well as their transport. Carrara et al. (2008) suggested that the higher organic soil matter content of one site may limit constituent transport. Swartz et al. (2006) suggested that the redox chemistry promotes greater sorption of OWCs to organic matter along oxic flow lines and intimates that if sorption were reversible, breakthrough of more soluble contaminants is possible as conditions change. The mobility of carbamazepine and diclofenac from reclaimed wastewater was significantly lower in organically rich soil layers than in organically poor ones (Cheftez et al., 2008).

System density

Since septic tanks are a major contributor of contaminated water to the subsurface (Robertson et al., 1991), increased density of septic systems results in higher amounts of contaminants in groundwater (Yates, 1985, USEPA, 1986). Brendle et al. (2004) reported that as the density of septic systems increased in northeast Colorado, concentrations of NO_3^-, Cl and B in drinking water wells increased. However, only 7% (2 samples) exceeded the USEPA primary drinking water standard for NO_3^- (10 mg/L as N). Schaider et al. (2010) reported that wells located in more highly populated areas tend to have more frequent detections and higher levels of emerging contaminants. However, the actual number or nature of septic systems in the vicinity of wells that were sampled was not specified.

Well Construction and Spring Development Characteristics

In general, lower concentrations (Barnes et al., 2008) and lower frequencies of detection (Focazio et al., 2008) of OWCs have been reported in groundwater sources than in surface water. However, groundwater sources are more often cited in WBDO investigations (Blackburn et.al., 2004; Yoder et al., 2008). Factors affecting potential for well contamination include underlying geology, well depth, well casing depth, grouting of annular space and proximity/orientation to potential sources of contaminants.

Underlying geology affects the nature of groundwater flow, including the transit/residence time and amount of filtration that occurs during transit. While Zimmerman et al., (2001) found no significant difference in total coliform concentrations between wells constructed in areas underlain by carbonate bedrock and those underlain by non-carbonate bedrock in Pennsylvania, all of the *E. coli* detections were in wells underlain by carbonate bedrock. Similarly, Braatz (2004) found the highest percentage of coliform detections and 100% of *E. coli* detections in wells completed in areas underlain by carbonate bedrock in Wisconsin.

Well depth generally affects the nature and amount of potential contamination of water supply wells. Hallberg and Keeney (1993) and Panno (1996) documented significantly less NO_3^- contamination in deeper wells. Barnes et al. (2008) found fewer pharmaceutical and other compounds with increasing well depth and Kolpin (1995) found fewer pesticides. Conversely, one study found that the presence of MTBE was positively correlated with well depth (Ayotte et al., 2005). Total coliform concentrations decreased with increasing depth to water-bearing zone in sanitary wells completed in non-carbonate bedrock in Wisconsin. The reverse was true for sanitary wells in carbonate bedrock, presumably because of conduit flow that reduces filtration and residence time (Braatz, 2004; Ator and Ferrari, 1997).

Zimmerman et al. (2001) compared total and median concentrations of total coliform in wells that had been grouted along the entire annulus of the casing with values from ungrouted wells and did not find statistically significant variation. However, 75% of the detections of *E. coli* were in water from wells of unsanitary construction. Uncased sand point wells less than 14 m deep exhibited the highest concentrations of DOC, Coliphage, nitrogen species and pharmaceuticals in Nebraska (Verstraeten et al., 2005). While non-carbonate aquifers and deeper drilling appear to provide safe supplies, underlying geology and depth to available water on a given site are not alterable. However, techniques for well construction and spring development are within the realm of human control. Cased and grouted wells and properly developed springs offer the highest level of protection for drinking water supplies.

Summary of System, Well and Spring Characteristics

As stated previously, the risk associated with use of septic systems and private wells is highly variable and site specific. Assessment of the effect of dispersed effluent and associated constituents on groundwater and potable water wells is fundamental to protecting public health and the environment. While natural attenuation processes alone may not remove or decrease contaminants to non-detectable levels, proper system management will help to maximize protection of drinking water sources. This includes use of construction BMPs, observance of adequate vertical and horizontal separation between water supplies and wastewater treatment systems as well as application of appropriate hydraulic loading rates to those systems. Diligent implementation of performance standards and associated monitoring and maintenance is highly recommended.

REFERENCES

APHA. 2005. Standard Methods for the Examination of Water and Wastewater, 21st ed. Washington, DC: APHA, AWWA, WEF.

Ahel, M., T. Conrad, and W. Giger. 1987. Persistent Organic Chemicals in Sewage Effluents. 3. Determinations of Nonylphenoxy Carboxylic Acids by High-Resolution Gas Chromatography/Mass Spectrometry and High-Performance Liquid Chromatography. Environ. Sci. Technol. 21:697-703.

Alhajjar, B.J. G. Chesters and J.M. Harkin. 1990. Indicators of Chemical Pollution from Septic Systems. Ground Water. 28(4):559-568.

Anderson, D. L. 1998. Natural Denitrification In Groundwater Impacted by Onsite Wastewater Treatment Systems. In: On-Site Wastewater Treatment, Vol. 8. D. M. Sievers, Ed. ASAE Publ. 03-98, American Society of Agricultural Engineers, St. Joseph, MI. pp. 336-345.

Aravena, R., M.L. Evans, and J.A. Cherry. 1993. Stable Isotopes of Oxygen and Nitrogen in Source Identification of Nitrate from Septic Systems. Ground Water. 31(2): 180-186.

Aravena, R., and W.D. Robertson. 1998. Use of Multiple Isotope Tracers to Evaluate Denitrification in Ground Water: Study of Nitrate from a Large-Flux Septic System Plume. Ground Water. 36(6):975-982.

Ator, S. W. and M. J. Ferrari. 1997. Nitrate and Selected Pesticides in Groundwater of the Mid-Atlantic Region. http://md.water.usgs.gov/publications/wrir-97-4139/html.htm. Verified January 12, 2011.

Avery, A.A. 1999. Infantile Methemoglobinemia: Reexamining the Role of Drinking Water Nitrates. Environmental Health Perspectives. 107(7): 583-586.

Ayotte, J. D., D. M. Argue, F. J. McGarry. 2005. Methyl tert-Butyl Ether Occurrence and Related Factors in Public and Private Wells in Southeast New Hampshire. Environ. Sci. Technol. 39 (1), 9-16.

Barnes, K, D.W. Kolpin, E.T. Furlong, S.D. Zaugg, M.T. Meyer, L.B. Barber. 2008. A national reconnaissance of pharmaceuticals and other organic wastewater contaminants in the United States — I) Groundwater. Science of the Total Environment. 402:192-200.

Bassett, R.L., P.M. Buszka, G.R Davidson, and D. Chong-Diaz. 1995. Identification of groundwater solute sources using boron isotopic composition. Environ. Sci. Technol. 29, 2915–2922.

Bedner, M. and W.A. MacCrehan. 2006. Transformation of acetaminophen by chlorination produces the toxicants 1,4-benzoquinone and N-acetyl-p-benzoquinone imine. Environ Sci Technol 40:516–522.

Beller, M., A. Ellis, S.H. Lee, M.A. Drebot, S.A. Jenkerson, E. Funk, M.D. Sobsey, O.D. Simmons III, S.S. Monroe, T. Ando, J. Noel, M. Petric, J.P. Middaugh, and J.S. Spika. 1997. Outbreak of viral gastroenteritis due to a contaminated well. Journal of the American Medical Association 278, no. 7: 563–568.

Blackburn B.G., Craun G.F., Yoder J.S., Hill V., Calderon R.L., Chen N. 2004. Surveillance for waterborne-disease outbreaks associated with drinking water-United States, 2001-2002. MMWR Surveill Summ 53(8):23-4.

Borchardt, M. A., K.R. Bradbury, E.C. Alexander, Jr., R.J. Kolberg, S.C. Alexander, J.R. Archer, L.A. Braatz, B.M. Forest, J.A. Green, and S.K. Spencer. 2010. Norovirus Outbreak Caused by a New Septic System in a Dolomite Aquifer. Ground Water.

Borchardt, Mark A. P.D. Bertz, S.K. Spencer, and D.A. Battigelli. 2003. Incidence of Enteric Viruses in Groundwater from Household Wells in Wisconsin. Applied and Environmental Microbiology. 69 (2):1172–1180.

Braatz, L. 2004. A Study of Fecal Indicators and Other Factors Impacting Water Quality in Private Wells in Door County, Wisconsin.

Brady, N.C. and R.R. Weil. 2002 The Nature and Property of Soils, 13th Edition. Prentice Hall. Upper Saddle River, NJ. Pp. 643-644.

Brendle, D.L. 2004. Potential Effects of Individual Sewage Disposal System Density on Ground-Water Quality in the Fractured-Rock Aquifer in the Vicinity of Bailey, Park County, Colorado, 2001-2002. USGS Fact Sheet 2004-3009.

Brown, T.L. 2009 Fluorescence Characterization of Karst Aquifers in East Tennessee. Master's Thesis, University of Tennessee, Knoxville.

Bundschuh, J., K.D. Balke, G. Baudino, A. Fuertes and R. Garcia. 1993. Testing Boron as an environmental Tracer and Indicator in Lerma Valley, Argentina. Tracers in Hydrology (Proceedings of the Yokohama Symposium, July 1993. IAHS Publ. no 215, 1993.

Caliman, C.A. and M. Gavrilescu. 2009. Pharmaceuticals, Personal Care Products and Edocrine Disrupting Agents in the Environment - A Review. Clean. 37(4 – 5):277 – 303.

Carrara C, C.J. Ptacek, W. D. Robertson, D.W. Blowes, M.C. Moncur, E. Sverko, S. Backus. 2008. Fate of pharmaceutical and trace organic compounds in three septic system plumes, Ontario, Canada. Environ Sci Technol 42:2805–2811.

Casciotti, K.L., D.M. Sigman, M.G. Hastings, J.K. Böhlke, and A. Hilkert (2002). Measurement of the oxygen isotopic composition of nitrate in seawater and freshwater using the denitrifier method. Analytical Chemistry, 74: 4905-4912.

Cheftez, B., T. Mualem, J. Ben-Ari. 2008. Sorption and Mobility of Pharmaceutical Compounds in Soil Irrigated with Reclaimed Wastewater, Chemosphere. 73:1335.

Close, M.E. L.R. Hodgson and G. Tod. 1989. Field evaluation of fluorescent whitening agents and sodium tripolyphosphate as indicators of septic tank contamination in domestic wells. New Zealand Journal of Marine and Freshwater Research. 23: 563-568.

Conn, K.E., L.B. Barber, G.K. Brown, and R.L. Siegrist. 2006. Occurrence and fate of organic contaminants during onsite wastewater treatment. Environ. Sci. Tech. 40(23):7358–7366.

Conn, K.E., R.L. Siegrist, L.B. Barber and M.T. Meyers. 2010a. Fate of trace organic compounds during vadose zone soil treatment in an onsite wastewater system. Environmental Toxicology and Chemistry. 29(2): 285-293.

Conn, K.E., K.S. Lowe, J.E. Drewes, C. Hoppe-Jones and M.B. Tucholke. 2010b. Occurrence of pharmaceuticals and consumer product chemicals in raw wastewater and septic tank effluent from single-family homes. Environmental Engineering Science. 27(4):347-356.

Coplen, T. B. , J.K. Bohlke and K. L. Casciotti. 2004. Using dual-bacterial denitrification to improve d15N determinations of nitrates containing mass-independent ^{17}O. Rapid Communications in Mass Spectrometry. 2004; 18: 245–250.

Craun, G. F. 1992. Waterborne disease outbreaks in the United States of America: causes and prevention. World Health Stat. Q. 45:192–199.

Crites, R., and G. Tchobanoglous. 1998. Small and Decentralized Wastewater Management Systems. McGraw-Hill, Boston, USA.

Cunningham, V. L., S. P. Binks, M. J. Olson. 2009. Human Health Risk Assessment from the Presence of Human Pharmaceuticals in the Aquatic Environment, Regul. Toxicol. Pharmacol. 53:39.

Czerwieniec, E.W. and J.A. Tomaszek. 2007. Stable isotopes of oxygen and nitrogen in nitrate identification measuring techniques. Environment Protection Engineering. 33(2):97-105.

Davis, S.N., D.O. Whittemore, and J. Fabryka-Martin. 1998. Uses of Chloride/Bromide Ratios in Studies of Potable Water. Ground Water. 36(2):338-350.

DeBorde, D.C., W.W. Woessner, B. Lauerman and P.N. Ball. 1998. Virus Occurrence and Transport in a School Septic System and Unconfined Aquifer. Ground Water. 36(5): 825-834.

Dixon, L.K., H.M. Taylor, E. Staugler, J.O. Scudera. 2005. Development of a fluorescence method to detect optical brighteners in the presence of varying concentrations of fluorescent humic substances: identifying regions in influenced by OSTDS in the estuarine waters of Charlotte Harbor. Mote Marine Laboratory Technical Report No. 1045.

Edwards, D.D., G.A. McFeters, M.I. Venkatesan. 1998. Distribution of Clostridium perfringens and fecal sterols in a benthic coastal marine environment influenced by the sewage outfall from McMurdo Station. Antarctica Appl Environ Microbiol 64:2596–2600.

Fay, S.R., Spong, R.C. Alexander S.C., Alexander E.C.J. 1995. Optical brighteners: sorption behavior, detection, septic system tracer applications. Paper presented at the International association of hydrogeologists XXVI international congress, Edmonton, Alberta.

Flipse, W. J. Jr., B.G. Katz, J. B. Lindner, R. Markel . 1984. Sources of Nitrate in Ground Water in a Sewered Housing Development, Central Long Island, New York. Ground Water. 22(4):418-426.

Focazio, M.J., D.W. Kolpin, K.K. Barnes, E.T. Furlong, M.T. Meyer, S.D. Zaugg. 2008. A national reconnaissance for pharmaceuticals and other organic wastewater contaminants the United States— II) untreated drinking water sources. Sci Total Environ. 402:201–16.

Fogg. G.E., D.E. Rolston, D.L. Decker, D.T. Louie and M.E. Grismer. 1998. Spatial Variation in Nitrogen Isotope Values Beneath Nitrate Contamination Sources. Ground Water. 36(3):418-426.

Folmar, L. C., N.D. Denslow, V. Rao, M. Chow, A.D. Crain, J. Enblom, J. Marcino, and L.J. Guillette, Jr. 1996. Vitellogenin Induction and Reduced Serum Testosterone Concentrations in Feral Male Carp(Cyprinus carpio) Captured near a Major Metropolitan Sewage Treatment Plant. Environ. Health Perspect. 104:1096-1101.

Fong,T-T. L.S. Mansfield, D.L. Wilson, D. J. Schwab, S.L. Molloy, J.B. Rose. 2007. Massive Microbiological Groundwater Contamination Associated with a Waterborne Outbreak in Lake Erie, South Bass Island, Ohio. Environ.Health Perspect. 115(6): 856-864.

Ford, M. and J.H. Tellam. 1994. Source, type and extent of inorganic contamination within the Birmingham urban aquifer system, UK. Journal of Hydrology. 156:101 – 135.

Gallagher L.G., T.F. Webster, A. Aschengrau, V.M. Vieira. 2010. Using Residential History and Groundwater Modeling to Examine Drinking Water Exposure and Breast Cancer. Environ Health Perspect 118:749-755.

Geary, P.M. 2003. On-site Treatment System Failure and Shellfish Contamination in Port Stephens, NSW. Enhancement Grant Project E08.

Gibson, R., E. Becerril-Bravo, V. Silva-Castro and B. Jimenez. 2007. Determination of acidic pharmaceuticals and potential endocrine disrupting compounds in wastewaters and spring waters by selective elution and analysis by gas chromatography–mass spectrometry. J Chromatogr A. 1169:31–39.

Gilpin, B., T. James, F. Nourozi, D. Saunders, P. Scholes and M. Savill. 2003. The use of chemical and molecular microbial indicators for faecal source identification. Water Science and Technology. 47(3):39–43

Gilpin, B. and M. Devane. 2003. Advanced indicators for the identification of faecal pollution sources. Institute of Environmental Scienc and Research Limited. Client Report FW0346. June 2003.

Glassmeyer, S.T., E.T. Furlong, D.W. Kolpin, J. D. Cahill, S.D. Zaugg, S.L. Werner, M.T. Meyer, D.D. Kryak. 2005 Transport of chemical and microbial compounds from known wastewater discharges: potential for use as indicators of human fecal contamination. Environ Sci Technol 39:5157–5169.

Godfrey, E., and W.W. Woessner. 2004. Screening level study of pharmaceuticals in septic tank effluent and city sewage. In Proceedings of the 4th International Conference on Pharmaceuticals and Endocrine Disrupting Chemicals in Water. Westerville, Ohio: National Ground Water Association. 296–308. http://info.ngwa.org/GWOL/pdf/042380190.pdf.

Gregor, J. N. Garrett, B. Gilpin, C. Randall and D. Saunders. 2002. Use of classification and regression tree (CART) analysis with chemical faecal indicators to determine sources of contamination. NZ J Mar Freshw Res 36:387–398.

Grimalt, J.O., P. Fernandez, J. M. Bayona, and J. Albaige. 1990. Assessment of fecal sterols and ketones as indicators of urban sewage inputs to coastal waters. Environ Sci. Technol. 24:357–363.

Hagedorn, C., S.L. Robinson, J.R. Filtz, S.M. Grubbs, T.A. Angier and R.B. Reneau, Jr. 1999. Determining Sources of Fecal Pollution in a Rural Virginia Watershed with Antibiotic Resistance Patterns in Fecal Streptococci. Applied and Environmental Microbiology. 65(12):5522-5531.

Hagedorn, C., M. Saluta, A. Hassall, and J. Dickerson. 2005. Fluorometric detection of optical brighteners as an indicator of human sources of water pollution. Part I. Description and detection of optical brighteners. Crop & Soil Environmental News, online. http://filebox.vt.edu/users/chagedor/biol_4684/BST/OB%20Article-I.pdf. Accessed 12 Jan 2011.

Hagedorn, C. and S.B. Weisberg. 2009. Chemical-based fecal source tracking methods: current status and guidelines for evaluation. Rev Environ Sci Biotechnol. 8:275–287.

Hagedorn, C. 2010. Bacterial Source Tracking (BST) Methods. Online: http://filebox.vt.edu/users/chagedor/biol_4684/BST/BSTmeth.html. accessed 27 Jan 2011.

Hallberg, G.R., and D.R. Keeney. 1993. Nitrate. In Regional Ground-Water Quality, ed. W.M. Alley, 297–322. New York: Van Nostrand Reinhold.

Hartel, P.G., C. Hagedorn, J.L. McDonald, J.A. Fisher, M.A. Saluta, J.W. Dickerson, L.C. Gentit, S.L. Smith, N.S. Mantripragada, K.J. Ritter, C.N. Belcher. 2007. Exposing water samples to ultraviolet light improves fluorometry for detecting human fecal contamination. Water Res 41:3629–3642.

Harvey, P.W. and D. J. Everett, Regulation of Endocrine-Disrupting Chemicals: Critical Overview and Deficiencies in Toxicology and Risk Assessment for Human Health, Best Practice Res. Clin. Endocrinol. Metab. 2006, 20, 145.

Heaton, T.H.E., 1986. Isotopic studies of nitrogen pollution in the hydrosphere and atmosphere: a review. Chem. Geol. 59:87–102.

Henschel, K.P., A. Wenzel, M. Diedrich, and A. Fliedner. 1997. Environmental hazard assessment of pharmaceuticals. Regul Toxicol Pharmacol 25:220–225.

Hinkle, S.R., R.J. Weick, J.M. Johnson, J.D. Cahill, S.G. Smith, and B.J. Rich. 2005. Organic wastewater compounds, pharmaceuticals, and coliphage in ground water receiving discharge from onsite wastewater treatment systems near La Pine, Oregon—Occurrence and implications for transport: U.S. Geological Survey Scientific Investigations Report 2005-5055.

Jobling, S., T. Reynolds, R. White, M.G. Parker, and J. Sumpter, J. 1993. A variety of environmentally persistent chemicals, including some phthalate plasticizers, are weakly estrogenic. P. Environ. Health Perspect. 103, 582-587.

Jourdonnais, J. H. and J. A. Stanford. 1985. Verification of shoreline sewage leachates in Flathead Lake, Montana. Montana Water Resources Research Center, Montana State Univ.-Bozeman. 38 pp.

Katz, B.G., J.B. Lindner, and S. E. Ragone. 1980. A Comparison of Nitrogen in Shallow Ground Water from Sewered and Unsewered Areas, Nassau County, New York, from 1952 through 1976. Ground Water. 18(6):607-616.

Katz, B.G. S. M Eberts and L.J. Kauffman. 2010a. Using Cl/Br ratios and other indicators to assess potential impacts on groundwater quality from septic systems: A review and examples from principal aquifers in the United States. Journal of Hydrology. 398:161-166.

Katz, B.G, D.W. Griffin, and P.B. McMahon, H.S. Harden, E. Wade and R.W. Hicks and J.P. Chanton. 2010b. Fate of Effluent-Borne Contaminants beneath Septic Tank Drainfields Overlying a Karst Aquifer. J. Environ. Qual. 39:1181–1195.

Kaushal, S.S, W.M Lewis, Jr., and J.H. McCuthchan, Jr. 2006. Land use change and Nitrogen Enrichment of a Rocky Mountain Watershed. Ecological Applications. 16(1): 299–312.

Kendall, C., S. D. Wankel, and E. M. Elliott (2007), Tracing anthropogenic inputs of nitrogen to ecosystems, in Stable Isotopes in Ecology and Environmental Science, edited by R. H. Michener and K. Lajtha, pp. 375–449, Blackwell Sci., Malden, Mass.

Kolpin, D.W., D.A. Goolsby, E.M. Thurman. 1995. Pesticides in near-surface aquifers: an assessment using highly sensitive analytical methods and tritium. J Environ Qual. 24:1125–32.

Komor, S.C and H.W. Anderson. 1993. Nitrogen isotopes as Indicators of Nitrate Sources in Minnesota Sand-Plain Aquifers. Ground Water. 31(2):260-270.

Kozar, M.D., C.H. Sheets, and C.A. Hughes. 2001 Ground-water Quality and Geohydrology of the Blue Ridge Physiographic Province, New River Basin, Virginia and North Carolina. USGS Water Resources Investigations Report 00-4270.

Leeming, R., and P.D. Nichols. 1996. Concentration of coprostanol that correspond to existing bacterial indicator guideline limits. Water Res 30:2997–3006.

Leeming, R., A. Ball, N. Ashbolt, and P. Nichols. 1996. Using faecal sterols from humans and animals to distinguish faecal pollution in receiving waters. Water Res 30:2893–2900.

Long, S. C. and J.D. Plummer. 2004. Assessing Land Use Impacts on Water Quality Using Microbial Source Tracking. Journal of the American Water Resources Association, 40: 1433–1448.

MacDonald, I.A., V. D. Bokkenheuser, J. Winter, A. M. McLernon, E.H. Mosbach .1983. Degradation of fecal sterols in the human gut. J Lipid Res. 24:675–694.

Manduca, C.A., 2000, Living with karst: Maintaining a clean water supply in Olmsted County, Minnesota, in Schneiderman, J.S., ed., The earth around us: Maintaining a livable planet: New York, N.Y., W.H. Freeman and Company, p. 268-282.

Marques, C.R., N. Abrantes and F. Goncalves F. 2004. Life-history traits of standard and autochthonous cladocerans: II. Acute and chronic effects of acetylsalicylic acid metabolites. Environ Toxicol 19:527–540.

Meschke, J. and M. Sobsey. 1999. Microbial pathogens and on-site soil treatment Systems. North Carolina State University Cooperative Extension Service, Raleigh, NC.

Miller, K. and J. Meek. 2006. Helena Valley Ground Water: Pharmaceuticals, Personal Care Products, Endocrine Disruptors (PPCPs), and Microbial Indicators of Fecal Contamination. Montana Bureau of Mines and Geology. Open-File Report 532.

Murray, K.E., D.R. Straud and W.W. Hammond. 2007. Characterizing groundwater flow in a faulted karst system using optical brighteners from septic systems as tracers. Environ Geol (2007) 53:769–776.

Nishikawa,T., J.N. Densmore, P. Martin and J. Matti. 2003. Evaluation of the Source and Transport of High Nitrate Concentrations in Ground Water, Warren Subbasin, California. USGS Water-Resources Investigations Report 03-4009.

O'Driscoll, M. C., Humphrey, N. Deal, K. Henry, S. Thieme, D. Lindbo, T. Kelley. 2010. Building Capacity To Investigate The Link Between Wastewater, Groundwater Contamination, and Human Health Draft Project Report. August 12, 2010

O'Reilly, C.E., A.B. Bowen, N.E. Perez, J.P. Sarisky, C.A. Shepherd, M.D. Miller, B.C. Hubbard, M. Herring, S.D. Buchanan, C.C. Fitzgerald, V. Hill, M.J. Arrowood, L.X. Xiao, R.M. Hoekstra, E.D. Mintz, M.F. Lynch. 2004. The Outbreak Working Group. 2007. A waterborne outbreak of gastroenteritis with multiple etiologies among resort island visitors and residents: Ohio. Clinical Infectious Diseases. 44, 506–512.

Panno, S. V. K.C. Hackley, H.H. Hwang, S.E. Greenberg, I.G. Krapac, S. Landsberger, and D.J. O'Kelly. 2006. Characterization and Identification of Na-Cl Sources in Ground Water. Ground Water 44(2): 176-187.

Pond, K.R., R. Rangdale, W.G. Meijer, J. Brandao, L. Falcao, A. Rince, B. Masterson, J. Greaves, A. Gawler, E. McDonnell, A. Cronin, S. Pedley. 2004. Workshop report: developing pollution source tracking for recreational and shellfish waters. Environ Forensics 5:237–247.

Quan, C.S., Q. Liu, W.J. Tian, J. Kikychi and S.D. Fan. 2005 Biodegradation of Endocrine-Disrupting Chemical, Di-ethylhexyl Phthalate, by Bacillus subtilis. Appl. Microbiol. Biotechnol. 66: 702-710.

Rea, R.A., S.B. Upchurch. 1980. Influence of Regolith Properties on Migration of Septic Tank Effluent. Ground Water. 18(2):118-125.

Richardson, S.D. and T.A. Ternes. 2005. Water Analysis: Emerging Contaminants and Current Issues. Anal. Chem. 2005, 77, 3807-3838.

Robertson, W.D., J.A. Cherry and E.A. Sudicky. 1991. Ground-Water Contamination from Two Small Septic Systems on Sand Aquifers. Ground Water. 29(1):82-92.

Routledge, E. J., D. Sheahan, C. Desbrow, G.C. Brighty, M. Waldock, J.P. Sumpter. 1998. Identification of Estrogenic Chemicals in STW Effluent. 2. In Vivo Responses in Trout and Roach. Environ. Sci. Technol. 1998, 32 (11), 1559-1565.

Sawyer, J.F. 2008. Water Quality near Wastewater Treatment Systems in Alluvial and Karst Hydrogeologic Settings, Black Hills, South Dakota. Report of Investigations 117. Geological Survey Program, South Dakota Department of Environment and Natural Resources.

Scandura, J.E., and M.D. Sobsey. 1997. Viral and Bacterial Contamination of Groundwater from On-site Sewage Treatment Systems. Water Science and Technology. 35 (11-12): 141-146.

Schaider, LA, R.A. Rudel, S. Dunagan, J. Ackerman, L. Perovich, J. Brody. 2010. Emerging contaminants in Cape Cod Drinking Water.

Seiler, R.L., S.D. Zaugg, J. M. Thomas, and D.L. Howcroft. 1999, Caffeine and pharmaceuticals as indicators of waste water contamination in wells: Ground Water, v. 37, p. 405–410.

Seiler, R.L. 2005. Combined use of ^{15}N and ^{18}O of nitrate and ^{11}B to evaluate nitrate contamination in groundwater. Applied Geochemistry. 20:1626–1636

Sherlock, M.D., J.J. McDonnell, D.S. Curry, A.T. Zumbuhl. 2002. Physical controls on septic leachate movement in the vadose zone at the hillslope scale, Putnam County, New York, USA. Hydrological Processes. 16(13): 2559-2575.

Shu W.C. and W.H. Ding. 2005. Determination of fluorescent whitening agents in laundry detergents and surface waters by solid-phase extraction and ion-pair high-performance liquid chromatography. J Chromatogr A 1–2:218–223Sigman

Sigman, D.M., K.L. Casciotti, M. Andreani, C. Barford, M. Galanter, and J.K. Böhlke (2001). A bacterial method for the nitrogen isotopic analysis of nitrate in seawater and freshwater. Analytical Chemistry 73: 4145-4153.

Spruill, T.B., W.J. Showers, and S.S. Howe. 2002. Application of Classification-Tree Methods to Identify Nitrate Sources in Ground Water. Journal of Environmental Quality. 31: 1538-1549.

Standley L.J., R.A. Rudel, C.H. Swartz, K.R. Attfield, J. Christian. 2008. Wastewater contaminated groundwater as a source of endogenous hormones and pharmaceuticals to surface water ecosystems. Environ Toxicol Chem, 27: 2457.

Stanford, B.D., H.S. Weinberg. 2007. Isotope dilution for quantitation of steroid estrogens and onylphenols by gas chromatography with tandem mass spectrometry in septic, soil, and groundwater matrices. Journal of Chromatography A, 1176 (2007) 26–36.

W. L. Sun, W.L., J. R. Ni, and T. T. Liu. 2006. Effect of Sediment Humic Substances on Sorption of Selected Endocrine Disruptors, Water, Air, Soil Pollut. 6:583.

Swartz, C. H., S. Reddy, M.J. Benotti, H.F. Yin, L. B. Barber, B. J. Brownawell, R. A. Rudel. 2006. Steroid estrogens, nonylphenol ethoxylate metabolites, and other wastewater contaminants in groundwater affected by a residential septic system on Cape Cod, MA. Environ. Sci. Technol. 40:4894–4902.Szabo

Szabo, Z. E. Jacobsen and T.J. Reilly. 2004. Preliminary Evaluation of Organic Wastewater Contaminants in Septic Tanks for Possible Use as Effluent Tracers in Shallow Ground Water. Abstract. Online at: http://info.ngwa.org/GWOL/pdf/042379974.pdf. Accessed 2 Mar 2011.

Tinker, J.R. 1991. An Analysis of Nitrate-Nitrogen in Ground Water Beneath Unsewered Subdivisions Ground Water Monitoring & Remediation. 11(1):141-150.

Triebskorn, R., H. Casper, A. Heyd, R. Eilemper, and H. R. Kohler. 2004a. Toxic effects of the non-steroidal anti-inflammatory drug diclofenac Part I: histopathological alterations and bioaccumulation in rainbow trout. J. Aquat. Toxicol. 2004, 68 (2), 141-150.

Triebskorn, R., H. Casper, A. Heyd, R. Eilemper, H. R. Kohler. and J. Schwaiger. 2004b. Toxic effects of the non-steroidal anti-inflammatory drug diclofenac: Part II. Cytological effects in liver, kidney, gills and intestine of rainbow trout (Oncorhynchus mykiss). J. Aquat. Toxicol. 68(2):151-166.

US EPA. 2010. Drinking Water Contaminants: List of contaminants and their maximum contaminant levels (MCLs). Available online at http://water.epa.gov/drink/contaminants/index.cfm#List. Verified January 12, 2011.

US EPA. 2006. Private drinking water wells. Available at http://www.epa.gov/safewater/privatewells/index2.html. Verified March 29, 2011.

US EPA. 2003. Factoids: drinking water and ground water statistics for 2003. Washington, DC: Environmental Protection Agency, Office of Water. EPA publication no. 816K03001.

US General Accounting Office. 1997. Drinking water: information on the quality of water found at community water systems and private wells. GAO publication no. GAO/RCED-97-123.

Vengosh, A., K.G. Heumann, S. Juraske, and R. Kasher, R. 1994. Boron isotope application for tracing sources of contamination in groundwater. Environ. Sci., Technol. 28:1968-1974.

Verstraeten, I.M., G.S. Fetterman, M.T. Meyer, T. Bullen, and S.K. Sebree. 2005. Use of tracers and isotopes to evaluate vulnerability of water in domestic wells to septic waste. Ground Water Monitoring and Remediation. 25(2):107-117.

Widowry, D., W. Kloppmann and, L. Chery. 2004. Nitrate in groundwater: an isotopic multi-tracer approach. Journal of Contaminant Hydrology. 72:165–188.

Wiggins, B. A. 1996. Discriminant analysis of antibiotic resistance patterns in fecal streptococci, a method to differentiate human and animal sources of fecal pollution in natural waters. Appl. Environ. Microbiol. 62:3997–4002.

Wilhelm, S. R., S. L. Schiff, and J. A. Cherry. 1994. Biogeochemical Evolution of Domestic Waste Water in Septic Systems: 1. Conceptual Model. Ground Water. 32(6):905-916.

Wilhelm, S.R., S.L. Schiff and W.D. Robertson. 1996. Biogeochemical Evolution of Domestic Waste Water in Septic Systems: 2. Application of Conceptual Model in Sandy Aquifers. Ground Water. 34(5): 853-864.

Xue, D. , J. Botte, B. DeBaets, F. Accoe, A. Nestler, P. Taylor, O. Van Cleemput, M. Berglund, and P. Boeckx. 2009. Present limitations and future prospects of stable isotope methods for nitrate source identification in surface- and groundwater. Water Research. 43:1159-1170.

Zimmerman, T.M., M.L. Zimmerman, and B.D. Lindsey. 2001. Relation between selected well construction characteristics and occurrence of bacteria in private household supply wells, south-central and southeastern Pennsylvania. USGS Water-Resources Investigations Report 01-4206.

Yates, M.V. 1985. Septic tank density and ground water contamination. Ground Water. 23(5):586–591.

Yoder, J., V. Roberts, G.F. Craun, V. Hill, L. Hicks, N.T. Alexander, V. Radke, R.L. Calderon, M.C. Hlavsa, M.J. Beach, and S.L. Roy. 2008. Surveillance for waterborne disease and outbreaks associated with drinking water and water not intended for drinking, United States, 2005–2006. Surveillance Summaries 57(9): 39–62.

Zublena. J.P., J.C Barker, and T.A Carter. 1993. Soil facts. Poultry manure as a fertilizer source. Publ. AG-439-5. North Carolina Coop. Ext. Serv. Raleigh.

Zublena, J.P., J.V. Baird. and J.P. Lilly. 1991. Soilfacts. Nutrient content of fertilizer and organic materials. Pub. AG-439-18. North Carolina Coop. Ext. Serv. Raleigh.