**Potential Effects of On-Site Sewage Treatment Facilities on Surface and Ground Water Quality in Travis County, Texas.**

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## Abstract

*On-site sewage treatment facilities (OSSF), including conventional septic tank absorption field systems, may impact water quality if improperly constructed, maintained or placed without consideration of surface and ground water impacts. OSSF density, location and age were compared to existing surface and ground water quality data for nitrogen, phosphorus, chloride, sulfate and indicator bacteria in Travis County, Texas. OSSF may be impacting water quality of springs and are positively related to watershed loads for major surface water streams, though impacts from OSSF may be less than water quality impacts from leaking and failing central sewer with larger volumes of wastewater conveyance and pressurized forcemains Results from this study indicate that a priori experiments and/or risk assessment methods may be needed to determine the appropriate response to OSSF impacts.*

## 1.0 Introduction

### *1.1 Use and Distribution*

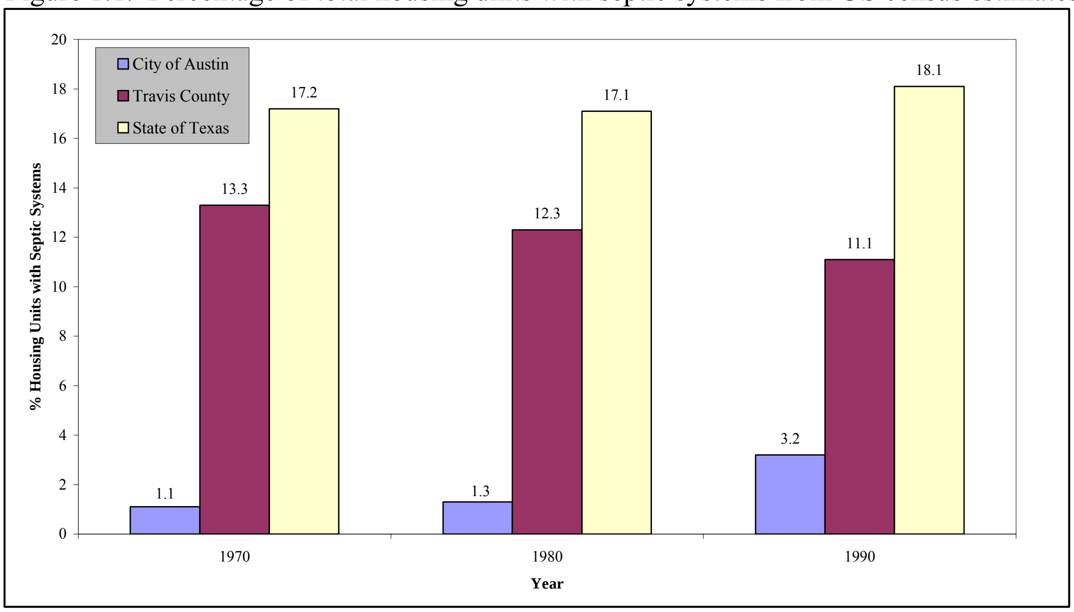
The 1990 decennial census conducted by the US Department of Commerce estimates that more than 25 million housing units, or approximately 25% of the total housing units in the United States, utilize a conventional on-site septic tank/absorption field facilities for sewage disposal (US Census 1990). Approximately 4 billion gallons of liquid wastes are discharged to On-Site Sewage Facilities (OSSF), in the US daily (EPA 2005). OSSF may be a cost-effective long-term option for meeting public health and water quality goals when adequately managed (EPA 1997). Decentralized wastewater management systems by OSSF have been proposed to limit or eliminate the financial and environmental difficulties inherent in the construction and maintenance of sanitary sewer pipe networks which typically run through the channels of creeks (Venhuizen 1997).

Failing or improperly managed OSSF, however, may pose a threat to water quality and public safety as non-point sources of pollution (Alhajjar et al 1990, EPA 2005). The US Environmental Protection Agency (EPA 1997), ranks on-site sewage facilities as one of the top five sources of ground water contamination in America.

The typical OSSF is composed of two parts: a settling or septic tank and the drain or absorption field (EPA 2005). The settling tank is where gravity and microbiological action separate and decompose human household wastes. The septic tank utilizes the same mechanisms of primary wastewater treatment (Metcalf and Eddy 1979) whereby floating scum and settlable suspended solids are separated from the liquid. Accumulated tank bottom sludge is occasionally pumped and removed by licensed contractors. A distribution box may contain a pumping apparatus but is generally responsible for dispensing the liquid into the perforated pipes or aerial sprinklers which make up the leach- or absorption-field where final treatment by soil microbes and discharge of liquid effluent occurs. Overloaded drain fields will flood discharging sewage to the ground surface (EPA 2005). Typical pollutants treated by OSSF are biochemical oxygen demand, nitrogen, phosphorus and pathogenic microorganisms (EPA 2005).

Data on method of sewage disposal was collected during the decennial census by the US Census Bureau in 1970, 1980 and 1990. While the percentage of housing units with septic systems for sewage disposal has decreased in Travis County, the percentage has increased for the Austin metropolitan area (Figure 1.1).

Figure 1.1. Percentage of total housing units with septic systems from US census estimates.



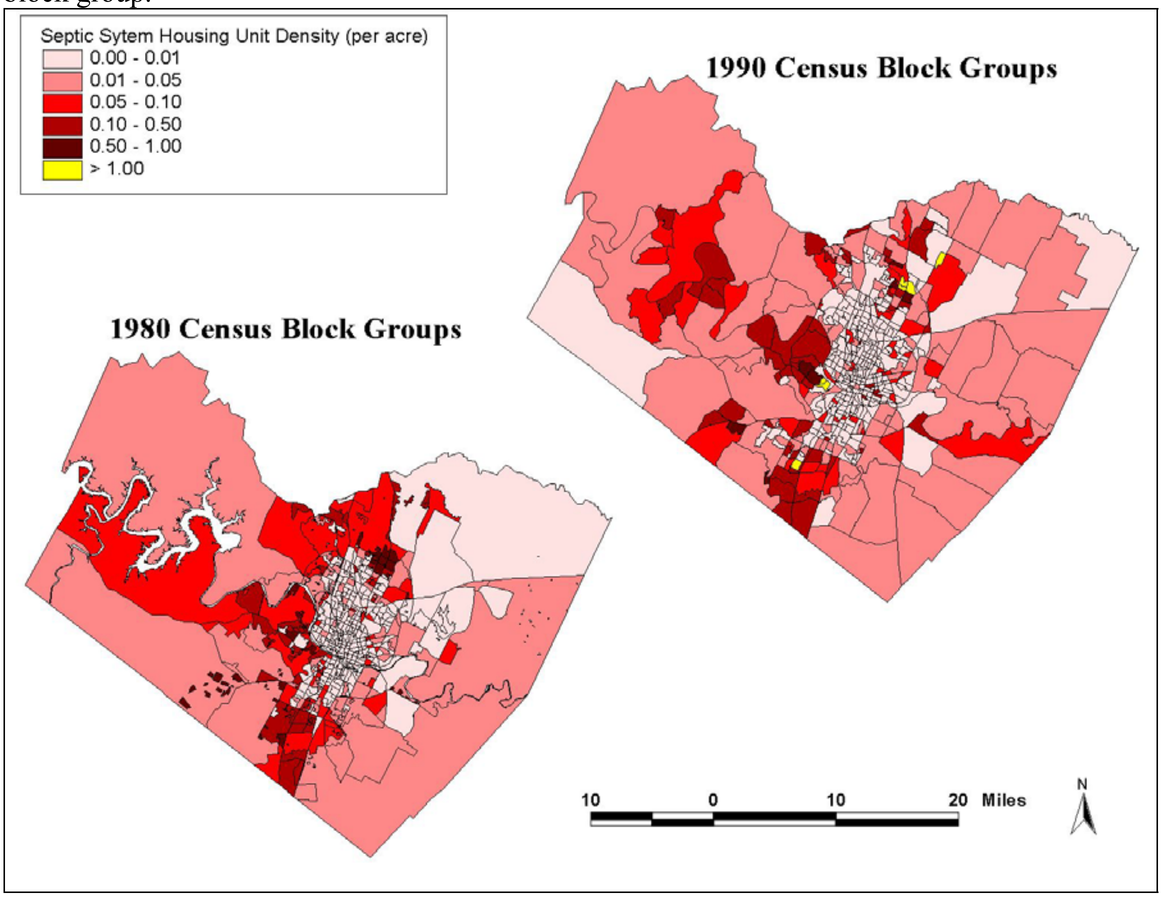

The number of housing units in Austin relative to the total number of housing units in Travis County, however, remained relatively constant (~83%) from 1980 to 1990. In comparison to Travis County and Austin, the more rural Hays County had a higher percentage of septic systems as shown in Table 1.1. However, this percentage has decreased in the ten years between surveys in Hays County with increasing extensions of central sewage collection systems..

Table 1.1. Total number of housing units by location from US Decennial Census estimates. No estimate for Hays County was available in 1970.

| | 1970 | | 1980 | | 1990 | |
|---|---|---|---|---|---|---|
| **Location** | **Septic System** | **Public Sewage** | **Septic System** | **Public Sewage** | **Septic System** | **Public Sewage** |
| Travis County | 13346 | 87356 | 21280 | 151795 | 29289 | 234131 |
| Hays County | . | . | 5586 | 8283 | 12156 | 12796 |
| City of Austin | 927 | 84686 | 1810 | 142887 | 6905 | 209551 |

Potential explanations for the sharp increase in percentages of homes utilizing OSSF in the City of Austin is that rapid expansion of home construction outpaced public sewage service extension or that new home construction was focused in more outlying areas outside of public sewage service areas. Comparison of 1980 and 1990 public sewage density at the census tract levels indicates that from 1980 to 1990 outlying areas within both the Austin city limits and Travis County experienced the largest changes in sewage disposal methods (Figure 1.2). According to American Housing Survey estimates (US Census Bureau 1993, US Census Bureau 2001), the percentage of housing units with OSSF for sewage disposal in central cities across the nation increased from 1993 (1.9% of total) to 2001 (2.3% of total), although percentages of housing units in suburbs (25.6% to 22.7%) and outside metropolitan survey areas (49.4% to 46.7%) both decreased. Thus, the observed changes in patterns of OSSF usage in the City of Austin and Travis County appear roughly in line with other major metropolitan areas across the nation.

Figure 1.2. Comparison of septic system housing unit density from 1980 to 1990 US Census by census block group.




The sewage disposal method question was not repeated on the 2000 decennial census; therefore, no recent estimates of sewage disposal methods from the US Census are available.

*1.2 Water Quality Impacts*

OSSF may cause discharge of effluent with nitrate-nitrogen (EPA 2005) concentrations to ground water greater than the currently applicable 10 mg/L drinking water standard (Wilhelm et al 1994), and may release phosphorus (Sawhney and Starr 1977) and human pathogenic bacteria or viruses (Hagedorn 1984, Corapcioglu et al 1997). Septic tank effluents may have high levels of nitrogen (70 mg/L), and may yield an annual nitrogen load of 3.5 pounds (Barrett and Charbeneau 1996b). Nitrate moves freely in solution in soil water (Alhajjar et al 1990), and nitrate removal by denitrification in aquifers underlying OSSF are rare (Wilhelm et al 1994). Thus, the contaminant loads from OSSF may be delivered to surface waters via contaminated ground water at water table intersections or through springs and seeps in streambanks. OSSF have been identified in the environmentally-sensitive Barton Creek Watershed as a potential source of increasing nitrogen from new residential developments, and nitrogen loads from OSSF to the Barton Springs segment of the Edwards Aquifer have been estimated at approximately 10% of the total (Barrett and Charbeneau 1996b). As many as 5900 septic tanks may lie over the recharge zone (Barrett and Charbeneau 1996a). In residential areas of Australia with OSSF sewage disposal, inputs of nitrogen to the soil from OSSF may account for up to 80% of total area inputs (Gerritse et al 1995). Closer to the region of the current study, ground water samples near OSSF by Lake Granbury, Texas yielded elevated levels of nutrients (TWRI 1993). Similarly, OSSFs studied in the Lake Travis and Lake Austin watersheds were found to have elevated nitrate and chloride concentrations in shallow groundwater wells (EHA, 1985).

Typical concentrations of phosphorus in sewage effluent range from 5 to 20 mg/L (Metcalf and Eddy, Inc 1979), and are much greater than the levels that may stimulate algal growth of 0.03 mg/L (Schindler 1977), especially in the generally phosphorus-limited streams in the greater Austin area (City of Austin 2004b). Although phosphates are generally removed by the soil under OSSF drain fields (Alhajjar et al 1990), relatively large phosphate plumes from mature OSSF have been measured at 10 meters from infiltration pipes (Robertson et al 1998), and thus setback distances from critical environmental features are crucial for the protection of water resources. Phosphorus actually comprises approximately 1% of the dry weight of a human body (Campbell 1993), and the average human body excretes 1 pound of phosphorus annually (EPA 1987). Although plants can only absorb and use phosphorus as orthophosphate in aquatic systems, phosphorus in freshwater is mostly in solid form adsorbed to particulates (Miertschin and Armstrong 1986). Excessive algal growth and eutrophication concerns have been identified in Austin area creeks (City of Austin 1997) and lakes (City of Austin 2004b). Additional components of wastewater effluents, such as chloride and sulfate, may be used to identify impacts of OSSF on water quality (Alhajjar et al 1990). Though not considered by this analysis, solvents used as septic tank cleaners and household hazardous wastes dumped down the drains may also pose an environmental and/or public health threat (Lee et al 1997).

Viral contamination of ground water may occur from properly functioning OSSF, and although indicator bacteria are generally not transported through soil to local ground water from OSSF effluent, virus counts may increase as distance from drain fields increase (Alhajjar et al 1988). Viruses appear to adsorb to sediments in response to high ionic strength of septic tank effluent, but may become desorbed under reduced ionic conditions farther from the drain field (Alhajjar et al 1988).

A field study by Parten and Liljestrand (1995) indicated that leaching from OSSF drain fields occurs sporadically in local soils, and monitoring wells only produced enough moisture to sample following significant rainfall events. Leaching provides a mechanism for transport of dissolved pollutants to shallow groundwater, and potentially forms preferential flow pathways through soil structure over time.

Although publicly-owned wastewater treatment plants undergo continual and stringent inspection, OSSF are subject to little inspection or monitoring after initially permitted (Wilhelm et al 1994) and the responsibilities of maintenance or ultimate shut-down are solely those of the owner (NSFC 1996). The City of Austin no longer routinely inspects OSSF after installation and final permit approval (James Grubbs, personal communication, 19 April 2005). One estimate of the national rate of failure of OSSF at 10.2% of the total (Knowles 1998) translates to approximately 2.5 million malfunctioning OSSF annually, or more than 7000 per day. Though extreme, the National Smalls Flow Clearinghouse reported that across the nation, an estimated 92,402 OSSF were repaired or replaced in 1993. Therefore, the magnitude of potential water quality impacts is large.

Health departments responsible for OSSF permitting across the nation noted in 1993 that sites that would previously have not been considered for OSSF are now being “purchased, planned and developed with on-site wastewater treatment in mind" (NSFC 1996). These marginally sited systems may be more prone to failure than previous developments.

Factors contributing to the contamination of surface and ground water resources from OSSF include inadequate design, poor construction and improper maintenance (NSFC 1993, Rupp 1998). Hydraulic failure of old systems through corrosion or breakage, inaccurate site assessments at installation, tank leakage and owner ignorance or complacence have been identified as reasons for OSSF failure and ground water degradation (Rupp 1998). Malfunction of OSSF may be caused by soil compaction from parking cars over drain fields reducing infiltration, allowing trees and shrubs to grow over drain fields so that roots clog flow paths or damage piping, or rainwater drainage systems flooding drain fields (EPA 2005). Plugging of outflow pipes may occur within the first five to fifteen years of OSSF operation if inadequate maintenance is performed, even if the OSSF were properly installed (Rupp 1998).

Hazardous household cleaners, toxic solvents and oil-based paints may inhibit proper microbial degradation of organic wastes thereby increasing effluent nutrient concentrations, in addition to the potential toxic effects of the products themselves.

The most common technical reasons for permit refusal by health departments across the nation are inadequate lot size, inadequate soils, high water table, shallow bedrock, steep slope and the availability of central sewers (NSFC 1993).

### *1.3 Regulation*

Regulation of OSSF generally involve setback requirements from wells, minimum percolation rates and absorption field size requirements to insure adequate treatment of wastewater for the protection of ambient water quality. OSSF density may be positively correlated with local water contamination (Brown and Bicki 1997). Tuthill et al (1998) demonstrated negative correlation between residential lot size and nitrate concentrations and fecal coliform bacterial counts. Review of previous work indicates that minimum lot sizes should be 0.5 to 1.0/acre, although effects on ground water at lower densities have been observed (Brown and Bicki 1997). Studies in Leon County, Florida, indicate that at densities of 0.25/acre water quality impacts were not observable from OSSF but that hydraulic failure rate of OSSF in the ¼ acre lot-size residential subdivision was high, even for soils considered well suited for OSSF (Leon County Public Health Unit 1987).

The City of Austin Water Utility is the authorized agent of the Texas Commission on Environmental Quality (TCEQ) for OSSF permitting in the City of Austin full purpose jurisdiction and limited purpose with health and safety jurisdiction. The TCEQ rules for OSSF are primarily contained within Title 30 of the Texas Administrative Code, Chapter 285 On-Site Sewage Facilities, with additional regulatory authority derived from Title 5 of the Texas Health and Safety Code. For areas in the recharge zone of the Barton Springs segment of the Edwards Aquifer, additional requirements can be found in 30 TAC 213. Chapter 15-5 of the City of Austin Code of Ordinances contains additional specifications, but basically adopts state standards of system design and operation. Similarly, Chapter 30-2-198 of the Austin/Travis County combined subdivision ordinances adopts 30 TAC 285 as the standard for ETJ projects. Other local jurisdictions have some permutation of permitting authority, but primarily rely on TCEQ standards.

Permitting in the City of Austin prior to 2001 was conducted by the Austin Health Department. Although the City of Austin used to conduct annual inspections of OSSF along the shores of Lake Austin, the annual program was suspended in 1999. The Austin Health Department will still respond to health and safety complaints with on-site inspection. OSSF permitting in Travis County is conducted by the Transportation and Natural Resources Division of Travis County. The City of Austin and Travis County have regulated OSSF since 1972, although regulations adopted by TCEQ in 1997 superceded existing City of Austin and Travis County regulations. Efforts were made in 2001 to adopt an Austin specific ordinance regulating OSSF; however, disagreements between installers and environmental advocates over the need for more stringent regulations resulted in adoption of the 30 TAC 285 standards as the baseline regulation.

### *1.4 Previous City of Austin Assessment Efforts*

Water quality assessments of Barton Creek in 1993 indicated that several tributaries influenced by high density residential development or golf courses yielded higher nutrient concentrations and *Cladophora* algal blooms were observed in the mainstem of Barton Creek (COA 1993). Based on these findings, the City of Austin initiated a sampling program designed to assess the effects of golf courses and high density residential developments on small stream water quality utilizing different wastewater disposal methods known as the Canyon Study (COA 1997). Results generated in 1997 from the Canyon Study indicated that nitrate levels in residential areas with OSSF were not significantly elevated over baseline conditions, potentially due to the mitigating effects of larger lot sizes and creek buffers required by county health departments. Golf courses had strong impacts on water quality, and residential neighborhoods on central sewers generally yielded higher constituent concentrations than residential areas on OSSF. However, in comparison to undeveloped rural land uses, the Canyon Study indicated that "some form of statistically significant water quality degradation can be documented for tributaries representing any .. developed land use categories…" including OSSF-impacted residential neighborhoods (COA 1997).

Twenty-four sites from the Canyon Study were included in this analysis. Comparison of the classification of these sites based on wastewater disposal methodology and level of development for the Canyon Study with the OSSF data from the US census and the Canyon Study yielded multiple discrepancies. Three sites were listed as "Residential Septic (RS)" for the Canyon Study, but few OSSF permits were observed within site contributing drainage areas and US census information indicated that density of housing units in 1990 utilizing public sewers was greater than number of housing units with OSSF. Two sites considered "Undeveloped (UND)" for the Canyon Study had housing units with OSSF, and one site considered "Residential Central (RC)" had OSSF permit locations within 400 feet of the site in the site drainage area boundaries. Although the findings of the Canyon Study are most likely valid, more intensive research incorporating current GIS data should be utilized in future *a priori* investigations.

Recommendations from the 1997 Barton Creek Report (COA 1997) suggesting continued sampling were carried out by City of Austin staff, and additional data indicated that nitrate was increasing in Barton Creek tributaries draining residential areas and golf courses over time, although no increases were observed in undeveloped watersheds (COA 2003a). No major increase in impervious cover in these watersheds was observed after 1995, although increases in nitrate concentrations were observed after 1998 (COA 2003a).

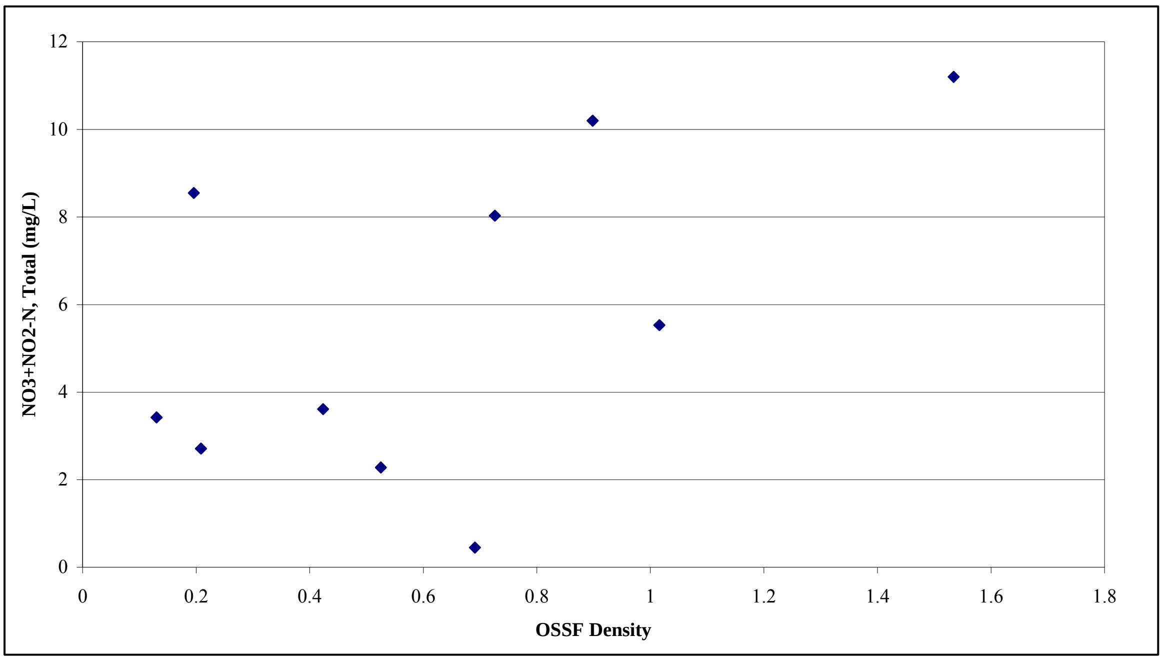
Analysis of Canyon Study data in 2002 indicated that nutrient and fecal coliform bacteria concentrations were higher in residential areas with OSSF and residential areas utilizing effluent irrigation for wastewater disposal than undeveloped areas, although tributary watersheds impacted by golf course effluent irrigation and watersheds with residential areas on central sewers yielded the highest nutrient concentrations (City of Austin 2002).

Additional analyses were undertaken in 2000, following a proposed new City of Austin ordinance. These preliminary analyses used existing water quality data in an attempt to attribute observed water quality effects to OSSF. Unpublished results from this exercise yielded statistically significant relationships between OSSF density and nitrate in surface water, although there was a clear lack of surface water sites with OSSF densities in the 0.1 to 1.0/acre range. OSSF density was determined from Austin Health Department permit record information.

Two separate investigations of potential OSSF impacts were undertaken in the Walnut Creek Watershed near the Eubank/Walnut Creek Subdivision. Data from 1996 to 1998 at two mainstem Walnut Creek sites at Lamar Boulevard and IH-35 bracketing the neighborhood with a high density of OSSF were compared. No statistically significant difference in median concentrations was observed between the upstream and downstream mainstem sites for nutrients or fecal coliform bacteria (City of Austin internal memorandum 1998). However, individual tributaries draining this high density OSSF neighborhood were compared to other rural OSSF and urban residential on central sewer neighborhoods, and the high density OSSF Eubanks/Walnut Creek Subdivision yielded statistically higher mean values of nitrate, ammonia and orthophosphorus (City of Austin internal memorandum 1999).

Additional sampling was undertaken for the current study (November 2000) in an attempt to test the validity of the relationships observed between nitrates and OSSF density. Ten sites located in the Walnut Creek, Bull Creek, Williamson Creek and Lake Austin watersheds with OSSF densities ranging from 0.13 to 1.5/acre as determined from permit database records were selected using GIS and sampled once under non-storm influenced conditions for nutrients and bacteria. The results of the November sampling event again yielded positive relationships between OSSF density and nitrate concentrations (Figure 1.3).

Figure 1.3. Results from November 2000 sampling of watersheds with intermediate density of OSSF for nitrate+nitrite and ammonia.




*1.5 Purpose of Study*

Based on these preliminary and previous studies, an intensive *a posteriori* investigation was undertaken to assess the correlations between OSSF density, age and distance from surface and ground water features using existing data for nitrate, ammonia, Kjeldahl nitrogen, orthophosphorus, phosphorus, chloride, sulfate and indicator bacteria in surface and ground water from the greater Austin/Travis County area.

## 2.0 Methods & Data Sources

*2.1 Water Quality*

Surface and ground water quality data included in the analyses were queried from the WRE Field Sampling Database (FSDB), available to the public at http://www.ci.austin.tx.us/wrequery/ . Data was collected by Water Resources Evaluation (WRE) staff as well as field staff of other regional and state agencies (Table 2.1) from 1970 through December 2004. Though data from numerous agencies was included, the primary sources of the data were the WRE and USGS.

Table 2.1. Data collection by agency.

| Agency | # Data Points | Begin Year | End Year |
|---|---|---|---|
| City of Austin - Water Resource Evaluation | 25558 | 1990 | 2004 |
| Lower Colorado River Authority | 470 | 1990 | 2003 |
| Texas Commission on Environmental Quality | 122 | 1990 | 2003 |
| United States Geological Survey | 10972 | 1970 | 2004 |

Although the FSDB contains data on more than 700 constituents in water samples across the greater Austin area, only a subset of parameters have been collected routinely over time and are expected to correlate with wastewater collection systems (Table 2.2) for both surface and ground water mediums.

Table 2.2. Parameters and analysis methods included in the analyses.

| Parameter | Unit of Measure | Filter Fraction | Analysis Methods |
|---|---|---|---|
| Ammonia as N | mg/L | Total, Dissolved | EPA 350.1, Hach 8155, SM 4500-NH3 |
| Nitrate as N | mg/L | Total, Dissolved | EPA 300, EPA 353.2, Hach 8171, Hach 8192, SM 4500-NO3 |
| Nitrate+Nitrite as N | mg/L | Total, Dissolved | EPA 353.2, SM 4500-NO3, Hach 8171 |
| Total Kjeldahl Nitrogen as N | mg/L | Total, Dissolved | EPA 351.2, EPA 351.4 |
| Orthophosphorus as P | mg/L | Total, Dissolved | EPA 300, Hach 8048, SM 4500-P |
| Phosphorus as P | mg/L | Total, Dissolved | SM 4500-P, EPA 365.4 |
| Chloride | mg/L | Total | EPA 300 |
| Sulfate | mg/L | Total | EPA 300 |
| E coli bacteria | cfu/100mL | n/a | SM 9222G, Colilert (IDEXX) |
| Fecal coliform bacteria | cfu/100mL | n/a | SM 9221E, SM 9222D |

*2.2 Data Quality Assurance*

In an attempt to maintain consistent data quality across the variety of analysis methods used to analyze the included data, available quality assurance/quality control (QA/QC) data were screened programmatically by WRE guidelines (COA 2004a). Data points were checked to verify that accuracy (lab standards, matrix spikes) and precision (laboratory split samples, field replicate samples) were within acceptable limits, that the sample result was greater than or equal to the associated laboratory blank result, and that no logical test failures occurred (i.e., a "dissolved" result was greater than a "total" result for a given sample).

Data points that failed accuracy or precision checks or logical tests were excluded from the analysis.

### *2.3 Environmental Integrity Index*

The City of Austin has created and implemented a unique method for master plan inventory and prioritization of environmental health of creeks in the greater Austin area known as the Environmental Integrity Index (EII). Formally beginning in 1996, the EII is evaluated for 47 watersheds once every three years on a rotating basis. Spatially distributed and representative sites throughout each watershed (generally 4 sites per watershed) are sampled for each component of the index, scores are calculated and indexed on a 100-point scale and then made available to the public as fact sheets via the internet (http://www.ci.austin.tx.us/watershed/learn_ws.htm). Components of the EII include water quality, sediment quality, contact recreation, aesthetics (also known as non-contact recreation), physical integrity and aquatic life (consisting of benthic macroinvertebrate rapid bioassessment and periphyton rock scrapings). Biological, aesthetic and physical integrity data samples are collected once per sampling year, and water quality samples are collected four times per sampling year according to current EII methodologies.

Average EII values and component scores for sample site and watersheds from 1996 to 2004 were included as independent variables in this analysis as a more holistic indicator of stream system health than individual water quality parameter averages alone.

The average EII values and component scores for watersheds are analyzed in addition to numerical annual pollutant loads calculated for selected City of Austin watersheds using a grid-based model developed for the City of Austin master plan process (Osborne 2000).

### *2.4 Storm Influence Determination*

Rainfall infiltration may influence OSSF contamination of surface and ground water. The determination of storm-influenced versus non-storm influenced (baseflow) sample conditions was made using the protocol adapted from the City of Austin Environmental Integrity Index sampling methodology (COA 2005). This is clearly a rough approximation, as sample times may occur at any point along the hydrograph. However, considering the large number of sample events (more than 7,700 sample events) and the lack of flow gages on many of the study watersheds, antecedent rainfall was the best method of storm flow effect separation.

Rainfall data on a minute-by-minute basis was downloaded from the City of Austin Flood Early Warning System (FEWS) as stored in the Hydstra database maintained by the City of Austin Water Quality Monitoring Section. For each site, the closest FEWS rainfall gages within the approximated individual drainage areas were identified, and rainfall totals were summed over the previous 24, 48 and 72 hours prior to sample collection. Weighted-averages for the antecedent periods were calculated using the applicable gages depending on the distance from the site to rainfall gage, and the weighted averages were compared to established values (Table 2.3) to determination storm influence.

Storm influenced samples were identified by a flow type of “S”, while non-storm (baseflow) samples were identified for analysis by a flow type value of “B.”

Table 2.3. Antecedent rainfall averages used to determine storm influence.

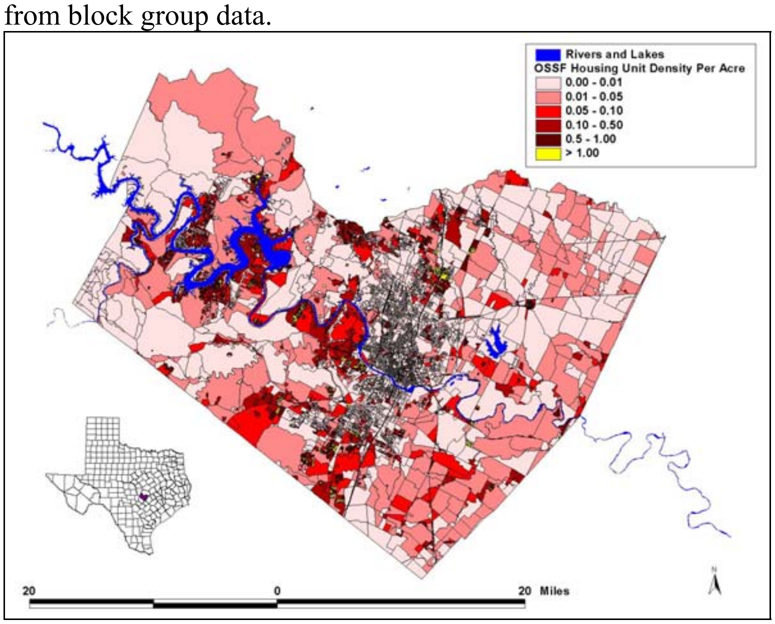
| Averaging period prior to sample collection time (hours) | Average rainfall (inch) in period prior to sampling to determine storm influence |
|---|---|
| 24 | ≥0.10 = storm |
| 48 | ≥0.25 = storm |
| 72 | >1.00 = storm |

*2.5 US Census Data*

Information on method of sewage disposal (public sewer or septic system) were obtained from the 1990 US decennial census via the US Census internet application (http://factfinder.census.gov). The level of spatial organization of census data proceeds from the tract to the block group to the block, and the aggregation of these units varies from decennial census to decennial census. The smallest spatial unit in the 1990 census with sewage disposal information publicly and electronically available is the block group.

The number of housing units with septic and sewage disposal, respectively, for each census block group in Travis County were downloaded and then joined to the 1990 Travis County census block group shapefile maintained by the City of Austin. The 1990 Travis County block group coverage with sewage disposal information was then intersected with the 1990 Travis County block coverage to yield more refined spatial estimates of the distribution of OSSF in Travis County. To estimate the number of OSSF per block, the number of persons in each block were divided by the number of persons in the corresponding block group, and this number was multiplied to the total number of OSSF housing units in the block group to yield number of OSSF housing units in each block. This estimation of number of housing units with OSSF is based on the assumptions that OSSF density within each block is directly proportional to the population density within each block, and that OSSF are equally distributed throughout each block. Examination of the disaggregated septic system housing unit density block group data at the census block level reveals that while OSSF are distributed throughout the county, clusters of high density housing unit blocks clearly occur (Figure 2.1).

Figure 2.1. Septic system housing unit density by 1990 US Census block in Travis County, generated from block group data.



To estimate the number of housing units with OSSF for each site, the individual site drainage areas delineated by the water quality extension were intersected with the block group coverage. The intersected block areas were re-calculated, and the number of OSSF housing units for each site was determined by summing for each block in the site drainage area the product of the density of OSSF housing units in the block by the new area of the block.

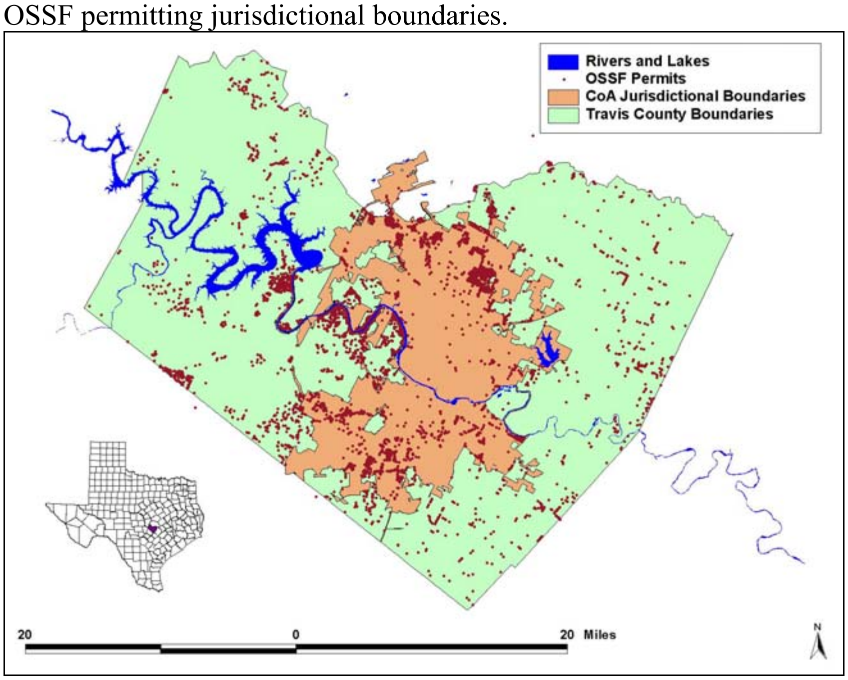
### *2.6 OSSF Permit Data*

OSSF permit data was obtained from the Austin Health Department. Regulation and permitting of OSSF was transferred from the Austin Health Department to the City of Austin Water Utility in 2001. Although the Austin Water Utility was contacted and asked to provide additional permit information, no information was received. Minimal information was included with the permit file other than the address of the permit. Permit addressed were geo-located using the “str-address.shp” ArcVIEW shape file maintained by the City of Austin and covering the extent of Travis County. Of the 6,057 permit records 5,822 (96% of total) were successfully address-matched. In order to obtain an approximate date of construction for the City permit records, information on building permits from the City of Austin PIER database was employed. Building permits were geo-located, and then the building permit issue date for the closest building permit was applied to each OSSF permit record as an approximate start date. Estimated dates for City permit records range from 1979 to 2000.

OSSF permit data was also obtained from Travis County files. Travis County OSSF permit records included address and permit issue date for 2,753 records. Although some permits contained additional information such as flow volume, tank size and drain field size, the information for these fields was inconsistently recorded and frequently absent. Permit address was successfully geo-located for Travis County permits using the “str-address.shp” coverage for all but 253 records (9% of total). Permit issue dates for Travis County records range from 1990 to 2002.

The geo-located Travis County and City of Austin OSSF permit files were merged (Figure 2.2). Again, although OSSF occur throughout the county, visible clustering of systems is evident.

Figure 2.2. City of Austin and Travis County geo-located OSSF permit information with City of Austin OSSF permitting jurisdictional boundaries.



*2.7 Statistical Methods*

All statistical analyses were completed in SAS® (version 9.1.3). Additional data manipulations were conducted using temporary tables stored in the WRE Section Oracle database. Unless otherwise specified, statistical significance is defined by a p-value $\leq 0.01$ for all analyses.

Mean values were calculated for each combination of site and parameter using SAS® (version 9.1.3). Many of the nutrient measurements were reported as censored data less than the detection limit of analyses. In order to better estimate means for datasets containing censored values (measurements reported as less than the detection limit of analysis), the Kaplan-Meier (K-M) survival analysis technique (Kaplan and Meier 1958) was employed using the LIFETEST procedure in SAS (Allison 1995). K-M estimation for censored datasets is the preferred non-parametric method for estimating summary statistics of data with less than 50% censored observations (Allison 1995, Helsel 2005), and may be applied to left-censored water quality data by transformation (Helsel 2005). Prior to K-M estimation, all data are subtracted from a constant value larger than the largest observation. Following K-M estimation, mean values are then transformed back to original scale by again subtracting from the same large constant.

By calculating site averages using K-M to deal with the censored observations, more traditional methods of correlation and regression analyses could be employed without violation of the assumption that each pair of observations must be independent. Additionally, variance components can be reduced since multiple measurements of water quality variables over time are tied to single estimates of independent variables. Site averaging also creates balanced datasets for each individual parameter.

For some nutrient parameters all measurements of a given parameter at a site were below detection limits. If all data were censored for a given parameter at a site, then the site average was listed at the lowest detection limit (Table 2.4). Although this data might typically be excluded from statistical analyses, the data was included with the substituted values to avoid biasing the results of the later correlation and regression analyses. All data for a given parameter at a site were below detection limits in only 17 of the total 1558 cases. In some cases, K-M estimation of the mean resulted in a mean site value less than the lowest detection limit. In order to avoid creating bias by identifying these sites (where values had been detected) as the lowest possible mean values, these cases were also substituted with the lowest detection limit value. Again, this occurred in a small percentage of the total number of cases and although this would tend to increase variability at the lowest detection limits for a parameter, it enables the broadest possible inclusion of data.

Table 2.4. Detection limits used for sites were all data were below detection limit by parameter.

| **Parameter** | **Detection Limit (mg/L)** |
|---|---|
| Ammonia as N | 0.01 |
| Nitrate+Nitrite as N | 0.02 |
| Orthophosphorus as P | 0.01 |
| Phosphorus as P | 0.01 |
| Kjeldahl Nitrogen | 0.04 |

Bivariate correlation analysis was conducted in SAS using the CORR procedure. Spearman's rank-order correlation coefficient ($r_S$) was chosen for correlation testing as the test is independent of distribution assumptions (Hatcher and Stepanski 1994) and considers the magnitude of the differences between data values (Helsel and Hirsch 1995). Spearman's $r_S$ is essentially the linear correlation coefficient computed on the ranks of the data (Helsel and Hirsch 1995), and measures the monotonic correlation. Spearman's $r_S$ is valid for censored datasets when the data are censored at a single detection limit (Helsel 2005), and thus should adequately handle the site means with substitution. Significance values reported for $r_S$ are computed as probability of the null hypothesis, $r_S$ equals zero, being correct. Partial Spearman correlation coefficients were calculated in some tests to adjust for effects of other variables, like percent impervious cover. Statistically, partial Spearman's correlation $r_S$ is equivalent to the Pearson correlation between the residuals of the linear regression of the variable ranks and the partialled variable ranks (SAS Institute Inc. 2004).

Univariate multiple regression was employed as a complement to correlation analysis, with water quality site means as the dependent variables versus two or more continuous independent variables (Timm and Mieczkowski 1997). Univariate multiple regression is a useful exploratory tool as the relationship between the dependent variable and the set of independent variables may be examined when taken as a group as well as individually for each independent variable while adjusting for the other independent variable effects (Hatcher and Stepanski 1994).

Comparison analyses between two independent groups was performed in SAS using the NPAR1WAY procedure to request two-tailed Wilcoxon rank sum, also known as the Mann-Whitney U-test (Gilbert 1987, Helsel and Hirsch 1995, Helsel 2005). The rank-sum test is a test for whether one group tends to produce larger observations than the second group. Significance, determined as the probability that test values are greater than chi-square ($\chi^2$), values less than the critical value lead to rejection of the null hypothesis—or, small probability values indicate the samples are from different populations. No assumptions about how the data are distributed in either group are required (Helsel and Hirsch 1995).

Comparison analyses between more than two groups was performed by non-parametric ANOVA techniques using the Kruskal-Wallis test, requested in SAS using the NPAR1WAY procedure. The Kruskal-Wallis test, an extension of the Wilcoxon rank sum test, is a non-parametric comparison based on ranks of the data testing the null hypothesis that the populations being compared have the same median value (Gilbert 1987).

Additional exploratory analysis was conducted using multiple comparison testing in SAS by requesting the REGWQ option in the GLM procedure. The Ryan-Einot-Gabriel-Welsch (Einot and Gabriel 1975), referred to as REGWQ, multiple range test was chosen as it may be one of the most powerful step-down multiple stage tests (Ramsey 1978).

### *2.8 Site Drainage Areas*

Site drainage areas were delineated using the Water Quality Extension of ArcVIEW (version 3.3) using a flow accumulation grid with a 100-foot by 100-foot cell size based on the 30 meter digital elevation model for Travis County.

Prior to delineation, surface water site locations were confirmed on the 5-acre creek grid to insure accurate watershed delineation. Surface water site locations not on the 5-acre creek grid were individually adjusted to the closest creek cell on the appropriate stream. Drainage areas for each site were visually inspected to insure correct delineation, and individual shape files were created for each site. Initial analyses sought to avoid creation of individual shape files for each site to simplify processing. The water quality extension delineates only to the next upstream site, and thus sites on the same creek have overlapping drainage areas. However, the need to use site drainage area boundaries for clipping of census information necessitated the creation of the individual files.

Ground water sites were delineated in a similar manner, although spring locations were specifically moved off the creek grid to correct bank of the creek were the discharge was observed to insure that the delineated watershed boundaries reflected the most conservative and localized surface flow patterns. Delineation of the spring "drainage area," or recharge zone, by this method is an underestimation, but useful is assessing the impacts from immediate and upslope OSSF. This method yielded very small watershed areas for each spring, with 65% of spring sites yielding watershed areas less than 1 acre and no site with a watershed area greater than 140 acres.

For initial exploratory data analyses, a cumulative drainage area grid was created from the flow accumulation grid and applied to the surface and ground water site point coverages to yield estimates of total drainage area to the sites. The permit location point coverage was converted to a grid, and then the FlowAccumulation function was applied to the grid to yield a grid with values for the cumulative number of permits upstream. However, comparison of the number of cumulative permits estimated from the calculated grid to sites with non-overlapping drainage areas indicated that the cumulative permit grid was an underestimation. Regression of estimated number of permits from the cumulative permit grid to actual number of permits for 167 of the 213 sites yielded a correction factor of 1.1979 (forcing the y-intercept to 0, with a $r^2 = 0.9783$). Use of the correction factor improved relative percent difference estimates between unadjusted and adjusted number of permits from the cumulative permit grid by 7.3%. Values for number of OSSF permits for each site estimated with the cumulative permit grid were replaced by actual number of permits when shapefiles for each individual site were generated.

Similar to the cumulative permit grid generation process, a cumulative OSSF age grid was generated for initial exploratory data analyses with the FlowAccumulation function by summing the number of days from estimated permit issue date to 31 December 2004, then dividing by the cumulative number of permits. Values for average OSSF age from the cumulative age grid were replaced with actual permit issue date averages once individual shapefiles for each site were generated.

Initially, cumulative drainage area for each site was estimated using a cumulative drainage area grid based on the flow accumulation grid, though estimates were replaced with actual values when shapefiles for each individual site were generated.

The distance from OSSF to surface water sites was estimated using the raindrop function, an Avenue script adapted for use in ArcVIEW (Petras 2000) that traces the path a raindrop would follow from a user-defined point to an outlet, based on the flow direction grid. The script was modified to graphically draw the surface flow path of every OSSF permit location along the flow direction grid. These graphics, more than 8,000 total, were selected, copied and pasted into a new line coverage referred to as the permit flow path coverage. The length of each line was calculated in feet and stored in the attribute table. The OSSF permit flow path coverage was intersected with each surface water site drainage area shapefile, and the minimum distance from an OSSF in the watershed boundaries to the site, the average distance from all OSSF in the watershed boundaries to the site and the sum of the inverse square of the distance of all OSSF to the site were calculated.

### *2.9 Golf Courses*
Fertilization practices of golf courses, especially those that utilize wastewater effluent irrigation, may mimic or mask the effects of OSSF contamination. Golf courses around Travis County were identified using the “landmark.shp” shapefile maintained by the City of Austin and generated from aerial photography. Golf courses utilizing effluent irrigation were further identified by previous City of Austin studies (COA 1997, COA 2002).

### *2.10 TCAD*
Travis County Tax Appraisal District (TCAD) spatial data, maintained by the City of Austin as the “TCAD0105.shp” ArcVIEW shapefile, for individual parcels was clipped using the surface water drainage area boundaries of sites with OSSF as the primary means of sewage disposal according to 1990 US census data. Following the assumption that one parcel equals one house, and if the majority of houses in the selected area utilize OSSF for sewage disposal, then the average lot size per watershed from TCAD

may be estimated by averaging the parcel size for each watershed drainage area. The average lot size was then related to mean water quality in an attempt to assess the water quality effects of lot size in potentially OSSF-impacted areas.

## 3.0 Surface Water Results

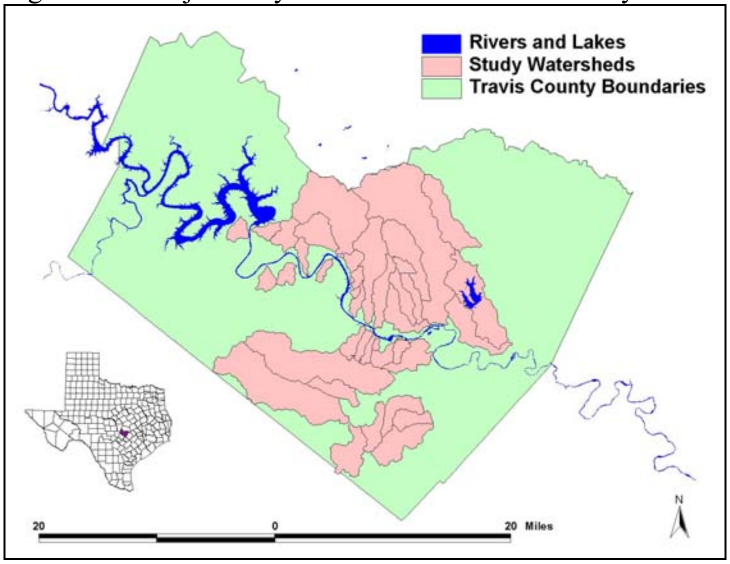
### *3.1 Site/Watershed Selection*

Surface water sites with sufficient data points and drainage areas completely enclosed by the jurisdictional boundaries of the City of Austin or Travis County were included in the analysis. Sites in Lake Austin and Town Lake were excluded, as local effects from the numerous shoreline OSSF would be diluted by upstream watersheds. Data from water quality ponds, oil/sediment treatment chambers, and other structural control sites were excluded as these do not represent ambient conditions and are likely to be more strongly influenced by other factors (direct impervious cover runoff, storage of contaminated sediment, etc) than OSSF. Sites with drainage areas lying within the jurisdictional boundaries of other entities (such as the Village of Bee Caves or City of Pflugerville) for which permit information was not available were also excluded from the analysis.

Additionally, if less than three measurements were available for any parameter at a site meeting the site selection criteria, the parameter (at that site) was excluded from the analysis.

The use of data only within the boundaries of Travis County was limited by availability of OSSF permit and US census spatial information. Although data from approximately 15 watersheds (as delineated in the City of Austin Drainage Criteria Manual) were excluded by these rules, the included watersheds were spatially diverse and covered a large portion of the available extent of data (Figure 3.1). The large percentage of housing units in Hays County with septic systems for waste disposal (48% in 1990) and the resource value of the Onion and Barton creek watersheds suggest that a renewed effort be undertaken to better quantify OSSF water quality impacts in Hays County. Watersheds not considered in this analysis and potentially deserving additional examination include: Brushy, Lake, Rattan, Buttercup, Onion (mainstem), Barton (mainstem), Bear, Little Bear, Little Barton, Gilleland, Dry Creek (East), Bee, Little Bee, and Eanes. Although small portions of the Harris Branch, Slaughter, Williamson and South Fork Dry watersheds, data from these watersheds were included in the analysis due to large number of known permitted OSSF within these drainage ways.

Figure 3.1. Major study watersheds included in analysis.



For the purposes of analysis, surface water sites were grouped (Table 3.1) on the basis of general method of sewage disposal according to US census information, influence of effluent irrigation from golf courses or residential subdivision utility districts from information in previous City of Austin Reports (COA 1997, COA 2002) and on the basis of cumulative percent impervious cover. Site groups included effluent irrigation (EI) influenced, low intensity (LI) development with low OSSF and public sewer housing unit densities, areas with high OSSF housing unit density and low public sewer housing unit density (Septic), areas with low OSSF housing unit density and high public sewer housing unit density (Sewer), areas with a fairly balanced OSSF and public sewer housing unit density (Mixed) and generally undeveloped areas with low housing unit density and low percent impervious cover (Und). For each parameter, the medians of site means were compared by Kruskal-Wallis and the means of site means by multiple comparison testing (REGWQ) for the assigned analysis groups.

Table 3.1. Surface water site groups used in analysis.

| Analysis Group ID | Group Description | # Included Sites | Avg % Impervious Cover | Avg OSSF Density | Avg Sewer Density |
|---|---|---|---|---|---|
| EI | Effluent irrigation | 8 | 6.57 | 0.02 | 0.03 |
| LI | Low intensity | 17 | 8.32 | 0.02 | 0.03 |
| Mixed | Mixed sewer/septic | 54 | 14.66 | 0.15 | 0.54 |
| Septic | Septic | 20 | 11.10 | 0.37 | 0.04 |
| Sewer | Public Sewer | 97 | 37.30 | 0.03 | 2.84 |
| Und | Undeveloped | 17 | 2.28 | 0.01 | 0.00 |

### *3.2 Watershed Results*

Initial analyses were conducted using major watersheds within jurisdictional Travis County and the City of Austin OSSF permitting boundaries to identify any potential patterns in surface water quality with OSSF factors.

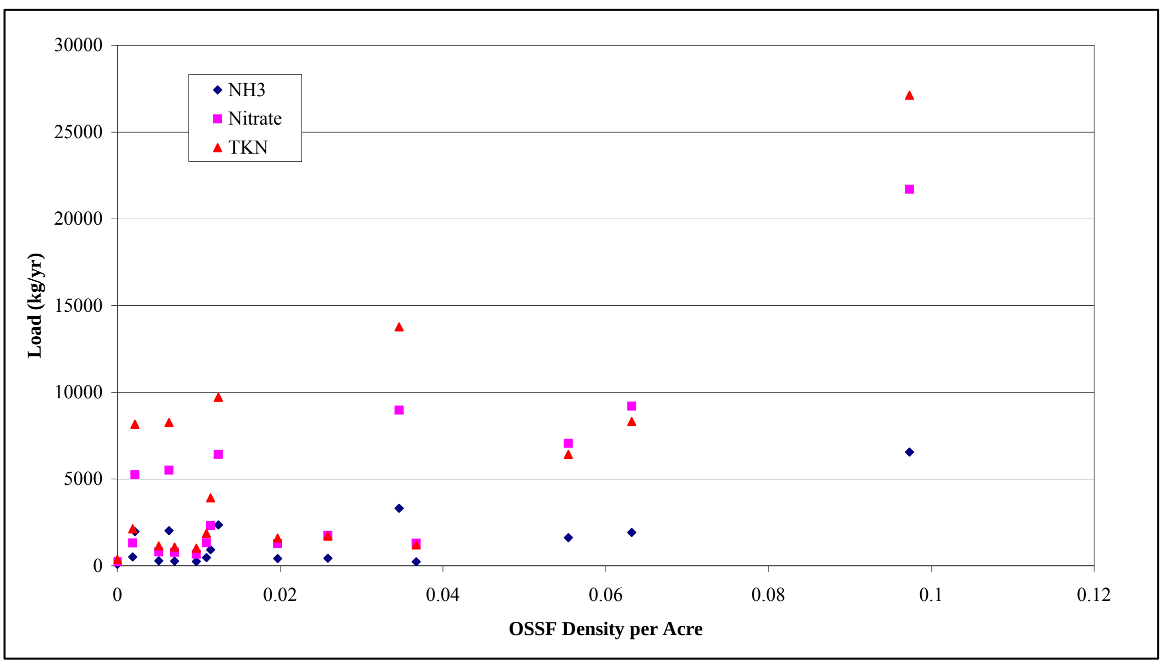
Contaminant loading estimates generated for the City of Austin Drainage Utility Master Plan Process (Osborne 2000) were correlated to OSSF density for 17 watersheds. Although OSSF density estimates generated from City of Austin and Travis County permit records yielded no statistically significant Spearman correlation coefficients with watershed loads, OSSF densities from 1990 US Census block groups clipped by watershed yielded statistically significant Spearman correlation coefficients for total suspended solids, total organic carbon, total Kjeldahl nitrogen, total nitrate and total nitrogen (Table 3.2). Relationships between watershed loads and OSSF density are generally weak, with OSSF density explaining no more than 40% of the variation in watershed load for any parameter. No statistically significant correlations between public sewer housing unit density and watershed loads was observed.

Table 3.2. Spearman correlation coefficients ($r_S$) and associated probability values for OSSF density from 1990 US Census data to watershed load estimates (N=17).

| **Parameter** | **$r_S$** | **$p(r_S>0)$** |
|---|---|---|
| Total suspended solids | 0.51 | 0.0376 |
| Biochemical Oxygen Demand | 0.37 | 0.1408 |
| Chemical oxygen demand | 0.37 | 0.1408 |
| Total organic carbon | 0.59 | 0.0135 |
| Phosphorus, Dissolved | 0.37 | 0.1408 |
| Phosphorus, Total | 0.35 | 0.1743 |
| Ammonia, Total | 0.37 | 0.1408 |
| Kjeldahl Nitrogen, Total | 0.49 | 0.0445 |
| Nitrate, Total | 0.64 | 0.0059 |
| Nitrogen, Total | 0.56 | 0.0184 |
| Copper, Total | 0.37 | 0.1408 |
| Lead, Total | 0.36 | 0.1524 |
| Zinc, Total | 0.34 | 0.1876 |

The Walnut Creek watershed, with the largest estimate OSSF permit density as calculated from the 1990 US Census block group information (density = 0.09/acre), strongly influences the correlation for nitrogenous nutrients (Figure 3.2). After removing the Walnut Creek watershed data and repeating the correlation analysis, only the relationship between OSSF density and nitrate is statistically significant ($r_S$=0.56, p=0.02).

Figure 3.2. Relationship between watershed loads and estimated OSSF density from 1990 US Census block group information for ammonia, nitrate and total Kjeldahl nitrogen (TKN).




OSSF density in major watershed units is inversely and significantly related to watershed percent impervious cover ($r_S$ = -0.69, p <0.01). Number of housing units on public sewer in major watersheds are significantly and positively correlated to watershed percent impervious cover ($r_S = 0.95$, $p<0.01$). OSSF density is significantly and positively related to watershed drainage area ($r_S = 0.78$, $p=0.0001$)—there is no statistically significant correlation between drainage area and number of housing units on public sewer.

To remove the potential masking effects of urbanization, a partial Spearman correlation analysis of watershed loads to OSSF density as estimated from 1990 US census block groups and adjusting for the effects of urbanization using percent impervious cover yields statistically significant correlations ($r_S$ partial with percent impervious cover $\geq 0.65$, $p\leq 0.01$) for all parameters considered (Table 3.3).

Multiple regression of OSSF density from 1990 US census block groups with watershed load estimates controlling for the effects of percent impervious cover again yield statistically significant ($p<0.01$) positive relationships between OSSF density and water quality constituent loads (Table 3.3) with reasonable $r^2$ values. Regression analysis of watershed loads against impervious cover values alone yield no statistically significant relationships, and no model for any parameter yielded an $r^2$ value greater than 11%.

Table 3.3. Spearman partial correlation with percent impervious cover and multiple regression controlling for impervious cover results of OSSF density from 1990 census information and watershed load estimates.

| **Parameter** | **Spearman correlation, partial with %IC ($p<0.01$)** | **Multiple Regression $r^2$, adjusting for % IC ($p<0.01$)** |
|---|---|---|
| Biochemical Oxygen Demand | 0.65 | 0.81 |
| Chemical Oxygen Demand | 0.65 | 0.82 |
| Total Organic Carbon | 0.71 | 0.80 |
| Total Suspended Solids | 0.65 | 0.74 |
| Ammonia, Total | 0.65 | 0.83 |
| Nitrate, Total | 0.73 | 0.89 |
| Kjeldahl Nitrogen, Total | 0.69 | 0.83 |
| Nitrogen, Total | 0.72 | 0.81 |
| Phosphorus, Dissolved | 0.65 | 0.82 |
| Phosphorus, Total | 0.64 | 0.68 |
| Copper, Total | 0.65 | 0.82 |
| Lead, Total | 0.66 | 0.78 |
| Zinc, Total | 0.66 | 0.77 |

Additionally, watershed loads were compared to total number of housing units from 1990 census block information as a surrogate for total population density by Spearman correlation analysis. All watershed loads were negatively correlated with total number of housing units. Thus, OSSF effects on watershed loads appear to be independent of total population density effects.

Sub-indices of the EII were compared to OSSF density estimates by 1990 US census block group information (Table 3.4), adjusting for variation in percent impervious cover for indices which exhibited a statistically significant correlation with percent impervious cover (Sediment, Contact Recreation and Total EII). Weak, though statistically significant positive relationships between OSSF density and EII sub-index scores were observed for the Water Quality, Sediment Quality, Benthic Macroinvertebrate and Total Watershed EII values. In comparison, density of housing units utilizing public sewer as estimated from 1990 US census information indicates statistically significant inverse relationships for most sub-index values, though $r^2$ values for the correlation between public sewer density and EII sub-indices are weak. Only the contact recreation sub-index yields $r^2$ values above 50%.

Table 3.4. Spearman correlation coefficients ($r_S$) and associated probability values for EII watershed sub-index values versus OSSF densities from US census information. Starred indices (*) indicate partial correlation with percent impervious cover used.

| SubIndices | OSSF Density | | Public Sewer Density | |
|---|---|---|---|---|
| | $r_S$ | p | $r_S$ | p |
| Water Quality | 0.43 | 0.0148 | -0.18 | 0.3277 |
| Sediment Quality* | 0.37 | 0.0400 | -0.53 | 0.0017 |
| Contact Recreation* | -0.19 | 0.3000 | -0.83 | 0.0000 |
| Non-Contact Recreation | 0.16 | 0.3809 | -0.03 | 0.8860 |
| Benthic Macroinvertebrates | 0.36 | 0.0436 | 0.05 | 0.7725 |
| Diatoms | 0.28 | 0.1184 | -0.01 | 0.9571 |
| Total EII* | 0.41 | 0.0200 | -0.54 | 0.0015 |

Individual components of the three sub-indices that exhibit statistically significant relationships with OSSF density (Water Quality, Sediment Quality, and Benthic Macroinvertebrates) were examined by correlation analysis to determine if identifiable elements within each sub-index may be driving the observed correlations. For the individual components, only three were significantly correlated with OSSF density on an individual basis (Table 3.5), though $r^2$ values were less than 40%. The Hilsenhoff Biotic Index, or HBI, (Hilsenhoff 1982) and the percent predator score positive correlation with OSSF density may be driven by nutrient enrichment phenomena as described by the intermediate disturbance hypothesis (Connell 1978, Ward and Stanford 1983) in which slight increases in nutrient concentrations drive increases in community diversity and abundance. Average OSSF age, determined from City of Austin and Travis County permit records, yielded no statistically significant correlation with watershed load estimates or EII sub-index values.

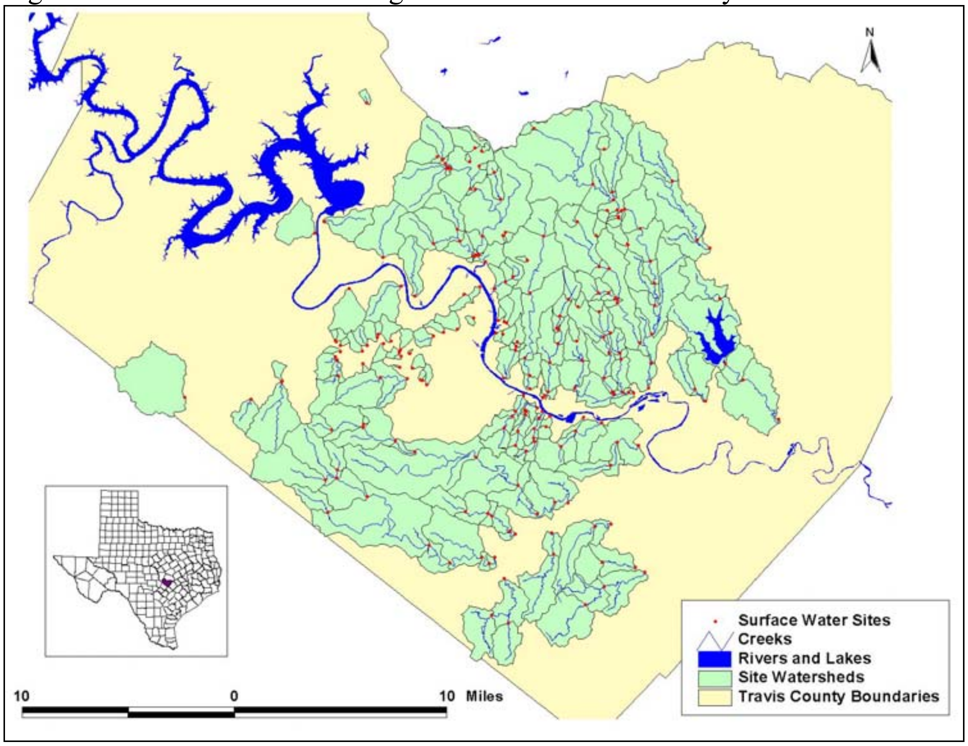
Table 3.5. Spearman correlation coefficients and associated probability values for EII watershed component scores versus septic and sewer housing unit densities.

| Component | Sub-Index | Septic Density | | Sewer Density | |
|---|---|---|---|---|---|
| | | $r_S$ | p | $r_S$ | p |
| HBI | Benthic Macroinvertebrates | 0.6450 | 0.0038 | 0.0857 | 0.7354 |
| % Predators | Benthic Macroinvertebrates | 0.5615 | 0.0153 | -0.6401 | 0.0042 |
| Bacteria | Water Quality | 0.4696 | 0.0493 | -0.8679 | 0.0000 |

## *3.3 Site Results*

Delineated surface water drainage areas included in the study (Figure 3.3) were distributed across the central portions of Travis County along flow paths generally terminating in Town Lake or the Colorado River below Austin in conjunction with City of Austin and Travis County permit jurisdictional boundaries.

Figure 3.3. Surface water drainage areas included in the analyses.




Site water quality parameter averages were compared to drainage area size in acres (*acreage*), average OSSF age in years from permit information (*avg_age*), minimum distance to OSSF in feet (*min_dist*), average distance to OSSF in feet (*avg_dist*), average adjusted distance calculated as $\Sigma$distance$^{-2}$ (*avg_adj_dist*), percent impervious cover (*perc_IC*), OSSF density per acre from permit information (*permit_density*), OSSF density per acre from US census information (*septicHU_density*), and public sewer housing unit density per acre from US census information (*sewerHU_density*).

Because site mean values are generally correlated with percent impervious cover and percent impervious cover is generally not correlated to OSSF housing unit density per acre, some correlation and regression analyses were adjusted for impervious cover variation to remove the effects of urbanization and thereby more isolate potential effects of OSSF. Additionally, sites with golf courses or residential neighborhoods with effluent irrigation in the immediate area were also excluded from correlation and regression analysis to further isolate potential OSSF water quality effects, removing 22 affected sites.

Each of these indicator variables were correlated with each other by bivariate Spearman correlation analysis. Most of the indicator variables were significantly inter-correlated, though only six relationships yielded absolute values for Spearman correlation coefficients greater than 0.70, translating to $r^2$ values greater than 50% (Table 3.6). As expected, the average adjusted distance correlates inversely with average distance and minimum distance. Average distance and average adjusted distance respond to drainage basin acreage. Permit density from City of Austin and Travis County records correlate positively with OSSF density from census records. Note the relatively strong positive correlation ($r^2$=73%) between percent impervious cover, an urbanization indicator frequently used by the City of Austin in regulation of development, and density of housing units on public sewer from US census information.

Table 3.6. Indicator variables correlated with themselves by Spearman correlation. Only relationships with $r_S$>0.70 presented.

| Variable | With | $r_S$ | p | # Obs |
|---|---|---|---|---|
| acreage | avg_adj_dist | -0.71 | <0.0001 | 1623 |
| acreage | avg_dist | 0.92 | <0.0001 | 1623 |
| avg_adj_dist | avg_dist | -0.84 | <0.0001 | 1623 |
| avg_adj_dist | min_dist | -0.73 | <0.0001 | 1623 |
| perc_IC | sewerHU_density | 0.85 | <0.0001 | 2035 |
| permit_density | septicHU_density | 0.72 | <0.0001 | 2035 |

Percent impervious cover was positively correlated with nearly all water quality site mean values (Table 3.7) in non-storm flow conditions. Percent impervious cover values for the surface water site drainages range from 0 to 66.35%, with more than half of the 213 sites having 19% impervious cover or greater in the contributing watershed. However, correlations between surface water quality site mean values and percent impervious cover were generally weak, and percent impervious cover explained less than 50% of variation (as estimated by $r^2$) for all variables except dissolved non-storm ammonia and non-storm chloride. Drainage area acreage was generally not correlated with water quality site mean concentrations, especially during non-storm flow conditions.

Table 3.7. Spearman correlation coefficients ($r_S$) and associated probability values for percent impervious cover versus surface water quality site means for statistically significant relationships by flow type (B=baseflow, S=storm flow).

| Parameter | FlowType | Variable | $r_S$ | p | # Obs |
|---|---|---|---|---|---|
| Ammonia as N, Dissolved | B | perc_IC | 0.86 | 0.0065 | 8 |
| Ammonia as N, Total | B | perc_IC | 0.40 | 0.0000 | 184 |
| Ammonia as N, Total | S | perc_IC | 0.41 | 0.0005 | 70 |
| Chloride, Total | B | perc_IC | 0.72 | 0.0000 | 31 |
| E coli bacteria, N/A | B | perc_IC | 0.70 | 0.0000 | 33 |
| Fecal coliform bacteria, N/A | B | perc_IC | 0.69 | 0.0000 | 170 |
| Nitrate as n, Total | B | perc_IC | 0.39 | 0.0000 | 173 |
| Nitrate as n, Total | S | perc_IC | 0.31 | 0.0127 | 63 |
| Nitrate+Nitrite as N, Total | B | perc_IC | 0.36 | 0.0007 | 84 |
| Nitrate+Nitrite as N, Total | S | perc_IC | 0.49 | 0.0156 | 24 |
| Orthophosphorus as P, Dissolved | S | perc_IC | 0.58 | 0.0155 | 17 |
| Orthophosphorus as P, Dissolved | B | perc_IC | 0.28 | 0.0243 | 65 |
| Orthophosphorus as P, Total | B | perc_IC | 0.51 | 0.0000 | 181 |
| Orthophosphorus as P, Total | S | perc_IC | 0.44 | 0.0005 | 60 |
| Phosphorus as P, Total | S | perc_IC | 0.39 | 0.0342 | 29 |
| Phosphorus as P, Total | B | perc_IC | 0.27 | 0.0400 | 57 |
| Sulfate, Total | B | perc_IC | 0.47 | 0.0007 | 49 |
| Kjeldahl Nitrogen as N, Total | B | perc_IC | 0.31 | 0.0198 | 55 |

Each of the OSSF-related indicator variables were bivariately correlated to water quality site means by Spearman correlation in a preliminary investigation of OSSF effects on water quality. Sites with golf courses or residential neighborhoods with effluent irrigation in the immediate area were excluded from the analysis to further isolate potential OSSF water quality effects, removing 22 affected sites.

Average adjusted distance was significantly correlated with only three water quality variables, and no correlation coefficient was greater than 0.40, indicating that average adjusted distance does not explain a large portion of the variance in mean site concentrations. Few statistically significant correlations between water quality site means and average distance were observed (Table 3.8).

Table 3.8. Spearman correlation coefficient (rS) and associated probability values for bivariate correlation of water quality site means with average adjusted distance and average distance from OSSF.

| Parameter | FlowType | Variable | rS | p | n |
|---|---|---|---|---|---|
| Ammonia as N, Total | S | avg_adj_dist | -0.30 | 0.0395 | 47 |
| Fecal coliform bacteria | S | avg_adj_dist | -0.40 | 0.0057 | 46 |
| Nitrate as N, Total | B | avg_adj_dist | 0.24 | 0.0085 | 120 |
| Ammonia as N, Total | S | avg_dist | 0.36 | 0.0136 | 47 |
| Chloride | S | avg_dist | 0.64 | 0.0129 | 14 |
| E coli bacteria | B | avg_dist | -0.39 | 0.0471 | 26 |
| Fecal coliform bacteria | S | avg_dist | 0.51 | 0.0003 | 46 |

To account for additional sources of variation in site means, mean values were regressed on average adjusted distance and average distance using percent impervious cover and acreage as covariates. Because of the inter-correlation between percent impervious cover and acreage, the effect of each covariate was assessed individually. From the multiple regression analysis, only total nitrate+nitrite during storm flow was significantly and positively related to average adjusted distance using percent impervious cover (p=0.0006, r2=0.56). The positive relationship would imply that total nitrate+nitrite concentrations increase as distance from OSSF permits increase. After adjusting for percent impervious cover and acreage, only fecal coliform mean counts in non-storm flow conditions yield statistically significant (p=0.0176, r2=0.25) positive relationships with average distance. Again, this implies that as average distance increases, indicator bacteria increase. These observed relationships indicate that average adjusted distance and average distance are not valid predictors of OSSF effects on water quality site mean values, or that some other unknown independent variable must be identified before the effects of average adjusted distance or average distance of OSSF on water quality may be quantified.

Few statistically significant bivariate correlations with minimum distance to closest known OSSF permit were observed for the water quality site mean values by Spearman correlation analysis, even when adjusting for the effects of impervious cover by partial correlation (Table 3.9). The relationship between dissolved nitrate+nitrite, though strongly significant, is based on a small number of observations.

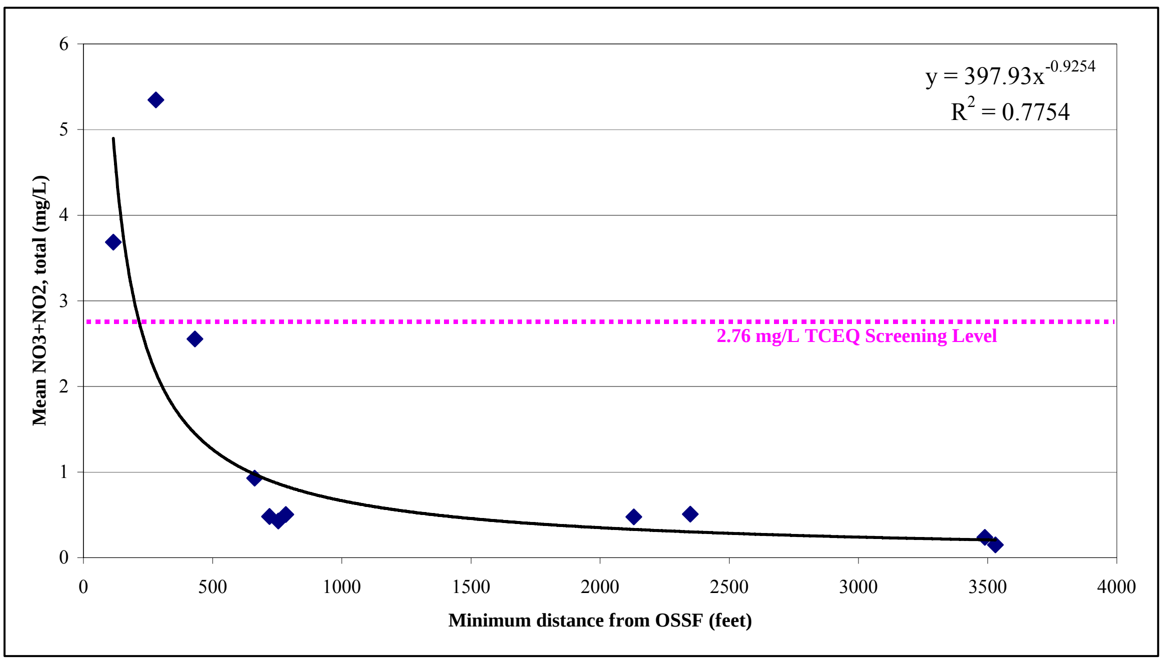
Table 3.9. Spearman correlation coefficients and associated probabilities for bivariate correlation and correlation partial with percent impervious cover for minimum distance and site means by parameter (only statistically significant correlations presented).

| Parameter | FlowType | bivariate correlation | | | partial w/ %IC | | |
|---|---|---|---|---|---|---|---|
| | | $r_S$ | p | N | $r_S$ | p | N |
| Ammonia as N, Total | B | 0.18 | 0.0379 | 129 | 0.19 | 0.0198 | 144 |
| Nitrate as N, Total | B | -0.30 | 0.0009 | 120 | -0.25 | 0.0039 | 134 |
| Nitrate+Nitrite as N, Dissolved | B | 0.95 | 0.0003 | 8 | 0.95 | 0.0003 | 8 |

More rigorous analysis was conducted using univariate multiple regression of water quality site means on minimum distance. Only fecal coliform bacteria yielded statistically significant relationships with minimum distance to OSSF. Even after adjusting for percent impervious cover, mean site fecal coliform bacteria in both non-storm (p=0.0008, r2=0.21, n=119) and storm flow conditions (p=0.0089, r2=0.19, n=45) were significantly and positively related to minimum distance from OSSF. Regression of fecal coliform bacteria and minimum distance adjusting for variation in drainage basin acreage yielded similar statistically significant positive relationships. The observed pattern in non-storm flow was strongly influenced by a single outlier value with a fecal coliform site mean of approximately 9,700 cfu/100mL and a minimum distance from OSSF of approximately 10,000 feet. Removal of this data point and repeat of the multiple regression did not change the result, again indicating that some other unidentified independent factor must be included in the model to identify the effects of OSSF on water quality site means.

To further the investigation of the effects of distance of OSSF permit locations on water quality, only sites with OSSF within 5000 feet were included in a repeat of the initial analyses on minimum distance. Only total nitrate+nitrite in storm flow ($p=0.0118$, $r^2=0.83$, $n=7$) yielded a statistically significant inverse relationship with minimum distance although both were based on a small number of data points. Plots of the mean total nitrate+nitrite values in storm flow conditions reveal an exponentially decreasing relationship with minimum distance from OSSF (Figure 3.4). A power regression line fit the data well, and although this type of equation would clearly over predict nitrate+nitrite concentrations at small minimum distances, predicted nitrate+nitrite values would not fall below the 2002 TCEQ screening level of 2.76 mg/L until minimum distances increase above 216 feet.

Figure 3.4. Total nitrate+nitrite in storm flow conditions for all sites with a minimum distance to OSSF less than 5,000 feet against 2002 TCEQ screening levels for nitrate+nitrite.

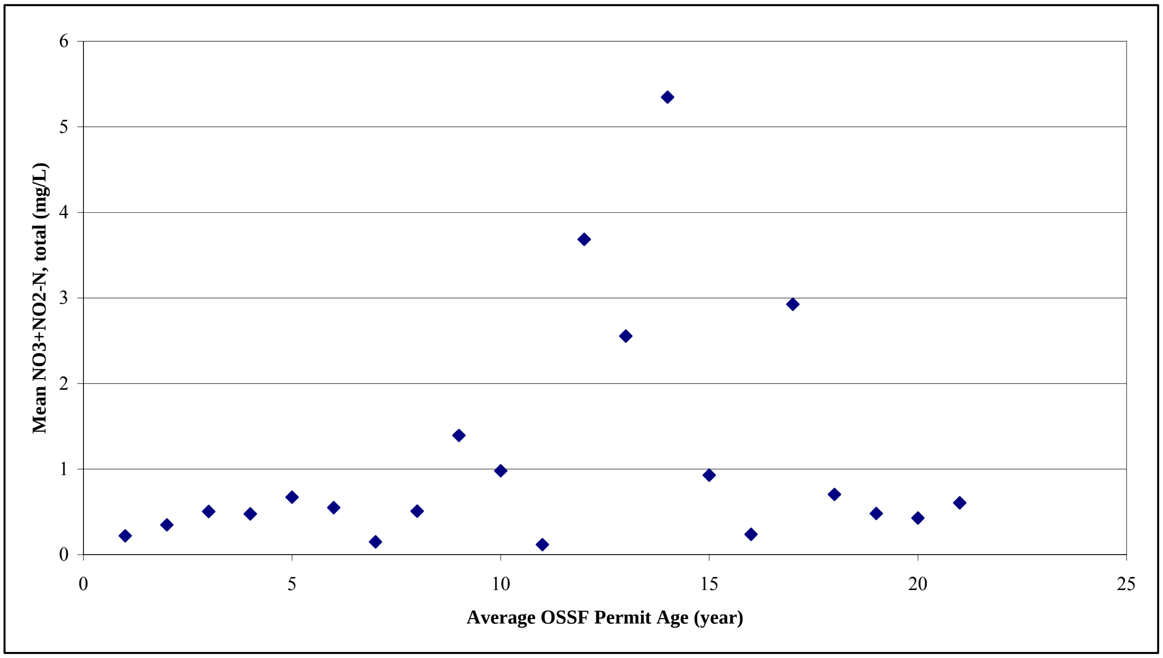



Average age of OSSF permits in the drainage areas of each site when compared to mean site concentrations yielded few parameters with statistically significant correlations, even when partial for impervious cover (Table 3.10). Multiple regression analysis of water quality mean values with average age indicated that only nitrate+nitrite in storm flow was significantly related to average age ($p=0.03$, $r2=0.27$, $n=20$) only when adjusting for drainage area size. No multiple regression relationship was statistically significant when adjusting for percent impervious cover. Values of mean nitrate+nitrite were plotted versus average OSSF permit age (Figure 3.5), and a spike in mean nitrate+nitrite values approaching or above the 2.76 mg/L 2002 TCEQ screening level are observed between the 12 and 15 year average age period, followed by a return to low levels in later years. It is postulated that this may reflect an average time to replacement; but no corroborating evidence, even anecdotal, was found supporting this.

Table 3.10. Spearman correlation coefficients ($r_S$) and associated probabilities of average age partial with percent impervious cover (statistically significant correlations presented).

| Parameter | FlowType | $r_S$ | p | # obs |
|---|---|---|---|---|
| Ammonia as N, Total | B | 0.22 | 0.0091 | 143 |
| Fecal coliform bacteria | B | 0.26 | 0.0022 | 134 |
| Nitrate+Nitrite as N, Total | S | 0.52 | 0.0114 | 23 |
| Orthophosphorus as P, Total | B | 0.20 | 0.0183 | 141 |
| Orthophosphorus as P, Total | S | 0.45 | 0.0024 | 44 |

Figure 3.5. Nitrate+nitrite in storm flow conditions versus average OSSF permit age.




OSSF density per acre, calculated from US census information and permit records, is inversely correlated to ammonia, fecal coliform bacteria and orthophosphorus concentrations (Table 3.11). When adjusted for percent impervious cover as a partial variable, the conclusions of the Spearman correlation analysis for OSSF density do not change. Multiple regression analyses of site means on OSSF density adjusting for the effects of percent impervious cover and acreage yield no evident statistically significant positive relationships between OSSF density and water quality. Multiple regression analysis of number of OSSF permits and number of OSSF housing units individually on water quality site means adjusting for drainage acreage also yields no statistically significant relationships between OSSF density and water quality site means. Reducing the number of included sites to only those with10 or more measurements per parameter and performing the multiple regression again adjusting for impervious still yielded no statistically significant positive relationships between OSSF density and water quality. However, the average OSSF density from US census information is 0.086 OSSF/acre, with the median density of 0.024 OSSF/acre and more than 80% of sites with an OSSF density of less than 0.1 OSSF/acre. At this low density, impacts may not yet appear even within a larger regional dataset such as this.

Table 3.11. Spearman correlation coefficients (rS) and associated probabilities for OSSF density.

| **Parameter** | **FlowType** | **Var** | **rS** | **p** | **n** |
|---|---|---|---|---|---|
| Ammonia as N, Dissolved | B | permit_density | -0.76 | 0.0280 | 8 |
| Ammonia as N, Total | B | permit_density | -0.25 | 0.0011 | 164 |
| Fecal coliform bacteria | B | permit_density | -0.19 | 0.0191 | 152 |
| Orthophosphorus as P, Dissolved | B | permit_density | -0.35 | 0.0085 | 56 |
| Orthophosphorus as P, Dissolved | S | permit_density | -0.69 | 0.0021 | 17 |
| Orthophosphorus as P, Total | B | permit_density | -0.24 | 0.0017 | 163 |
| Orthophosphorus as P, Total | S | permit_density | -0.31 | 0.0243 | 52 |
| | | | | | |
| Ammonia as N, Dissolved | B | septicHU_density | -0.76 | 0.0280 | 8 |
| Ammonia as N, Dissolved | S | septicHU_density | -0.67 | 0.0130 | 13 |
| Ammonia as N, Total | B | septicHU_density | -0.31 | 0.0001 | 164 |
| Ammonia as N, Total | S | septicHU_density | -0.47 | 0.0001 | 60 |
| Fecal coliform bacteria | B | septicHU_density | -0.38 | 0.0000 | 152 |
| Orthophosphorus as P, Dissolved | S | septicHU_density | -0.67 | 0.0031 | 17 |
| Orthophosphorus as P, Total | B | septicHU_density | -0.28 | 0.0003 | 163 |
| Orthophosphorus as P, Total | S | septicHU_density | -0.42 | 0.0019 | 52 |

For the septic site group, when analyzed separately, mean site water quality values did not significantly correlate with average lot size estimated from TCAD parcel information by Spearman correlation for any parameter under any flow condition. OSSF density from US census information also did not significantly correlate with any water quality parameter mean value for sites in the septic group by Spearman bivariate correlation. Multiple regression analysis of water quality site means with OSSF density from US census information adjusting for average lot size again yields no statistically significant relationship between OSSF density and water quality mean values.

Site groups were compared by the non-parametric Kruskal-Wallis analysis, with exploration of the differences between individual site group mean values by the REGWQ multiple range test (Table 3.12). Two-sample comparison tests were performed for the septic versus the sewer group and for the septic versus the undeveloped group. For no parameter with sufficient data were septic group means statistically different from the effluent irrigation group.

Table 3.12. Comparison of surface water site groups by Kruskal-Wallis by parameter and flow type (B=baseflow, S=storm flow).

| Parameter | Flowtype | # groups | chi-square | p > chi-square |
|---|---|---|---|---|
| Ammonia as N, Dissolved | B | 3 | 4.08 | 0.1298 |
| Ammonia as N, Dissolved | S | 3 | 7.74 | 0.0209 |
| Ammonia as N, Total | B | 6 | 28.02 | <.0001 |
| Ammonia as N, Total | S | 6 | 15.39 | 0.0088 |
| Nitrate as N, Dissolved | B | 0 | | |
| Nitrate as N, Dissolved | S | 3 | 0.29 | 0.8637 |
| Nitrate as N, Total | B | 6 | 48.17 | <.0001 |
| Nitrate as N, Total | S | 6 | 16.50 | 0.0056 |
| Nitrate/Nitrite as N, Dissolved | B | 3 | 3.62 | 0.1635 |
| Nitrate/Nitrite as N, Dissolved | S | 3 | 1.88 | 0.3915 |
| Nitrate/Nitrite as N, Total | B | 6 | 19.71 | 0.0014 |
| Nitrate/Nitrite as N, Total | S | 5 | 9.94 | 0.0414 |
| Kjeldahl Nitrogen as N, Dissolved | B | 0 | | |
| Kjeldahl Nitrogen as N, Dissolved | S | 3 | 0.24 | 0.8878 |
| Kjeldahl Nitrogen as N, Total | B | 6 | 9.80 | 0.0812 |
| Kjeldahl Nitrogen as N, Total | S | 5 | 10.49 | 0.0329 |
| Orthophosphorus as P, Dissolved | B | 6 | 6.22 | 0.2853 |
| Orthophosphorus as P, Dissolved | S | 3 | 8.38 | 0.0151 |
| Orthophosphorus as P, Total | B | 6 | 53.93 | <.0001 |
| Orthophosphorus as P, Total | S | 6 | 17.44 | 0.0037 |
| Phosphorus as P, Dissolved | B | 3 | 1.68 | 0.4317 |
| Phosphorus as P, Dissolved | S | 3 | 1.64 | 0.4412 |
| Phosphorus as P, Total | B | 6 | 9.37 | 0.0951 |
| Phosphorus as P, Total | S | 5 | 15.28 | 0.0042 |
| E coli bacteria, N/A | B | 5 | 14.05 | 0.0071 |
| E coli bacteria, N/A | S | 2 | 0.43 | 0.5127 |
| Fecal coliform bacteria, N/A | B | 6 | 76.08 | <.0001 |
| Fecal coliform bacteria, N/A | S | 6 | 12.68 | 0.0265 |
| Chloride, Total | B | 5 | 15.51 | 0.0038 |
| Chloride, Total | S | 3 | 1.93 | 0.3813 |
| Sulfate, Total | B | 5 | 14.41 | 0.0061 |
| Sulfate, Total | S | 3 | 2.06 | 0.3575 |

Total ammonia in both storm and non-storm flow conditions (Figure 3.6) yielded higher mean values in the sewer group. Mean sewer group values for total ammonia in both storm and non-storm flow are significantly greater than mean septic group values by Wilcoxon rank-sum test (p=0.0035). The total ammonia mean value in non-storm flow conditions was numerically higher in the septic group than the effluent irrigation group and the undeveloped group, though one-to-one comparison by Wilcoxon rank-sum test indicated that the difference was not statistically significant. If the sewer group is removed from the analysis, then all other site groups are not statistically different by Kruskal-Wallis test. Non-storm influenced total Kjeldahl nitrogen values were not different between groups.

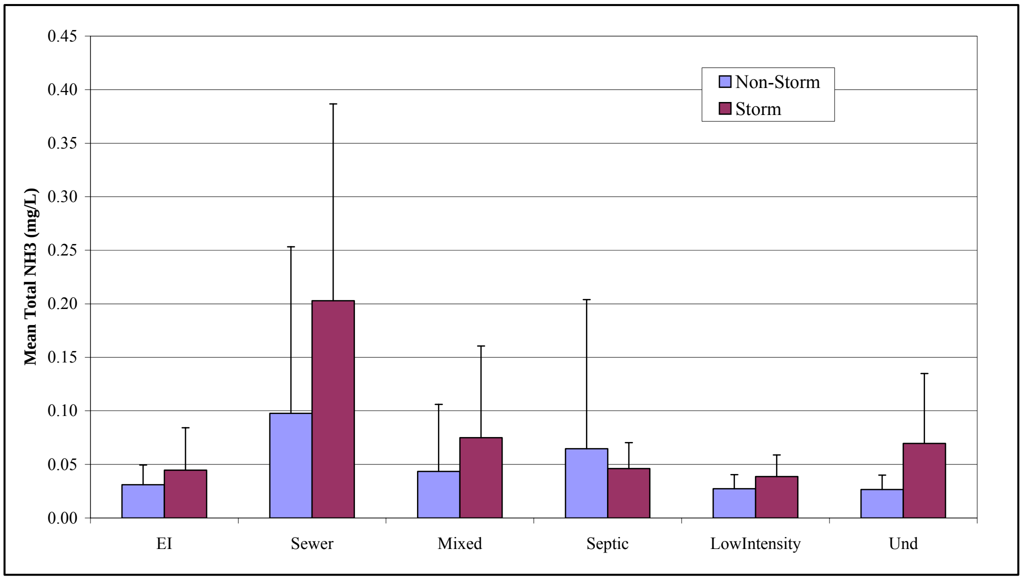
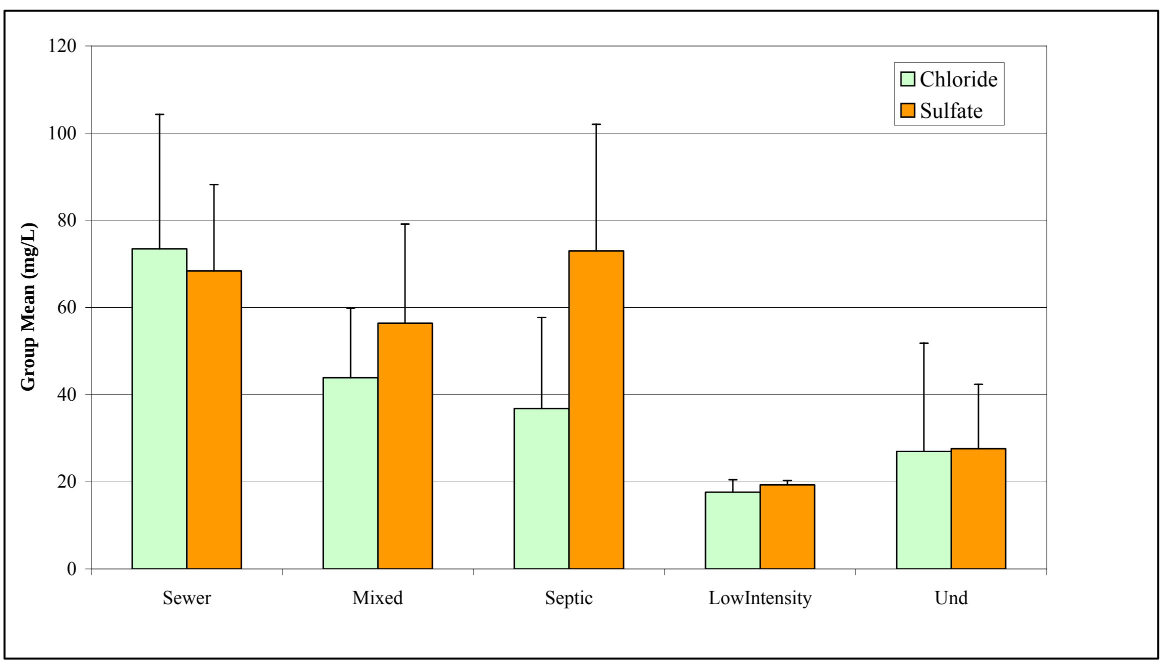



Figure 3.6. Mean total ammonia in non-storm and storm flow by site group with standard deviation error bars.

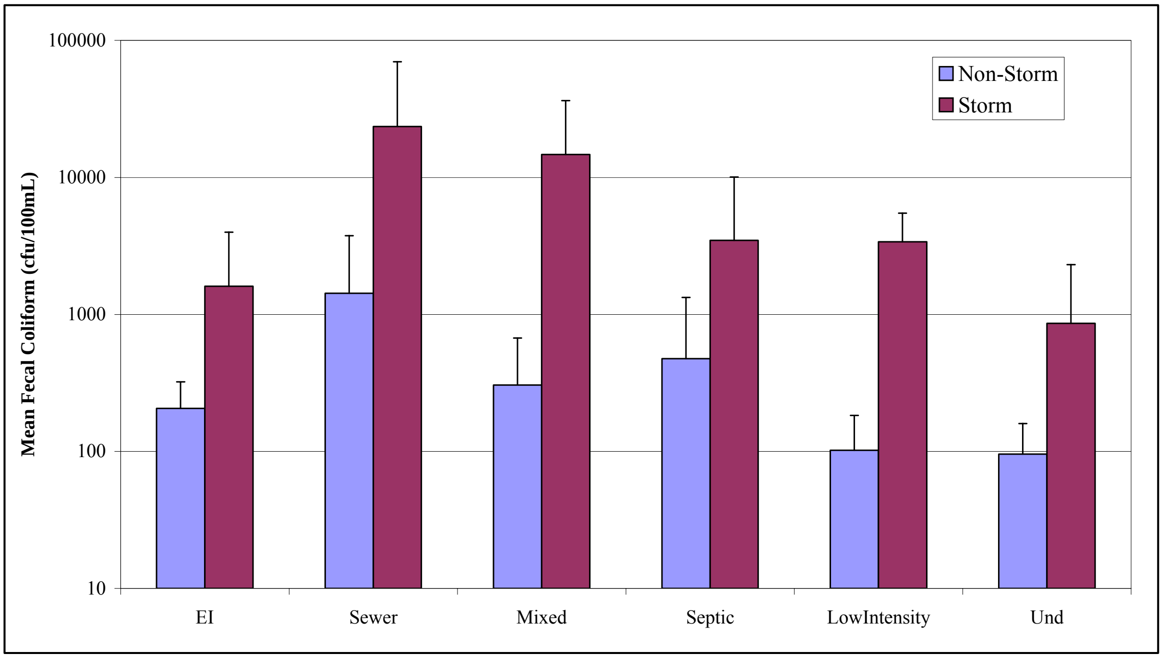
Chloride and sulfate (Figure 3.7) are significantly higher in the sewer, mixed and septic groups combined than the low intensity and undeveloped groups by Wilcoxon rank-sum test. However, neither chloride nor sulfate is significantly higher in the septic group than the undeveloped group. Chloride and sulfate are generally higher in the more urbanized sewer, septic and mixed groups than the lesser developed low intensity and undeveloped groups, though no statistically significant difference between the sewer and septic groups or the septic and undeveloped groups under non-storm flow conditions are observed by these analyses.

Figure 3.7. Chloride and sulfate group means in non-storm flow conditions with standard deviation error bars. No EI group data available.



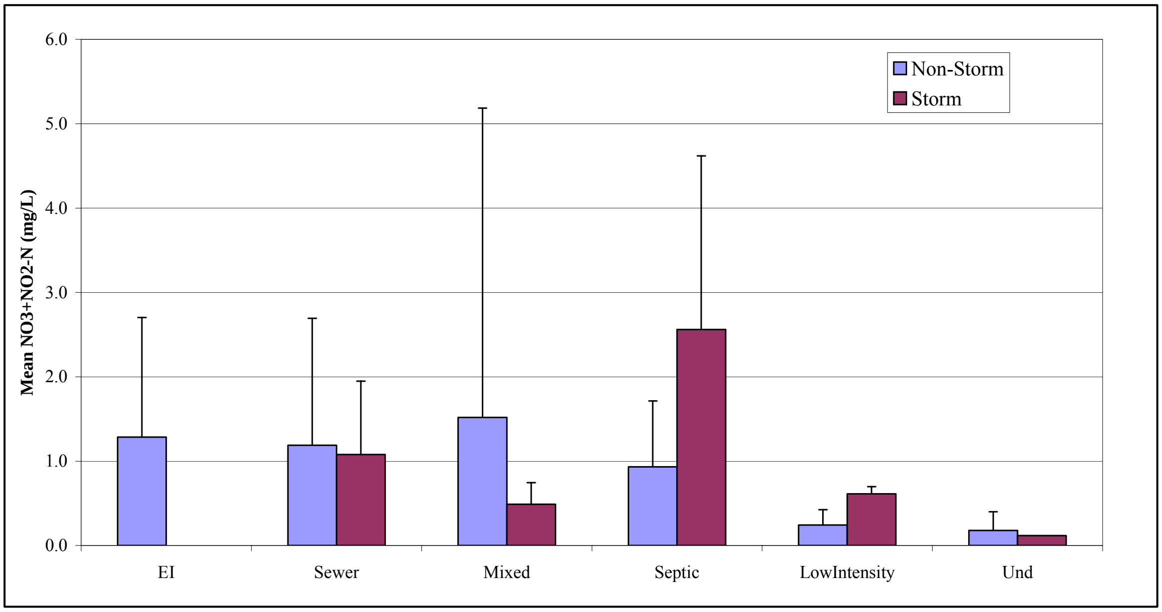
Fecal coliform bacteria were significantly greater in the sewer group than the septic group in both non-storm (p=0.0002) and storm influenced (p=0.0327) conditions. The septic group was significantly greater than the undeveloped group in non-storm flow conditions (p=0.0119). Fecal coliform group means (Figure 3.8) and group means for E coli bacteria responded similarly, though more data for fecal coliform was available for comparison.

Figure 3.8. Fecal coliform bacteria group means in storm and non-storm flow conditions (error bars indicate one standard deviation of the mean).



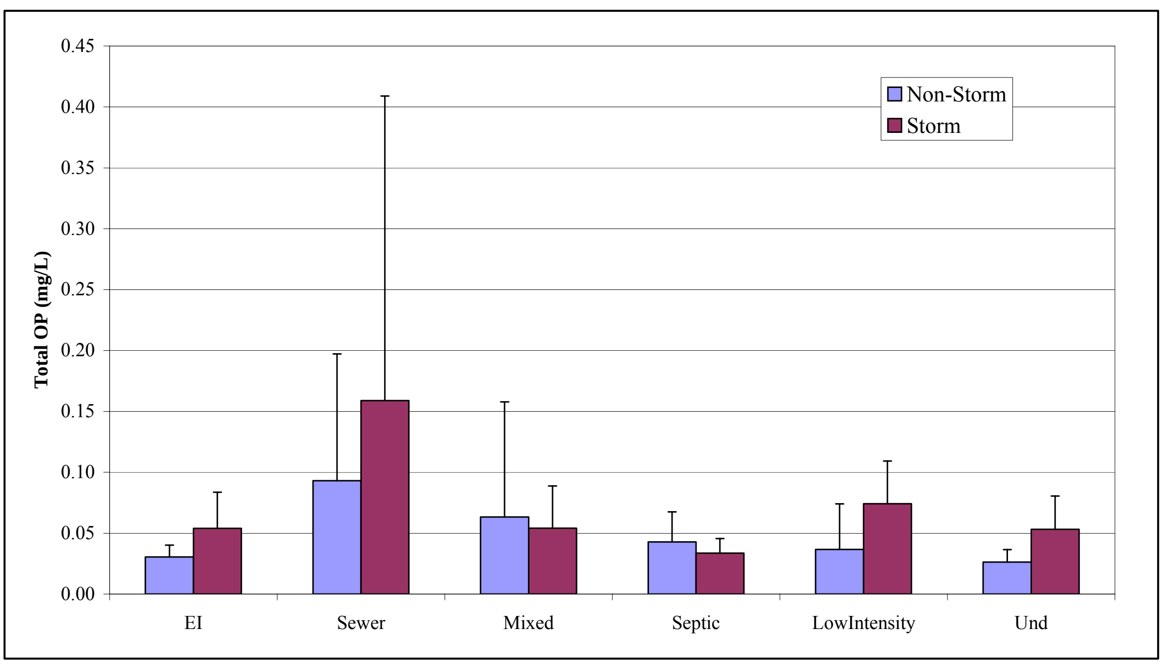

Nitrate+nitrite was not statistically different between sewer and septic groups in either storm or non-storm flow conditions, although mean total nitrate+nitrite for septic sites approaches the 2.76 mg/L TCEQ (2002) screening level in storm flow conditions (Figure 3.9). Total nitrate+nitrite was statistically greater than in the septic group than the undeveloped (p=0.0372) group in non-storm flow conditions. Again, nitrate+nitrite was not significantly different between the effluent irrigated group and the septic site group. Similar patterns are observed for total nitrate, and although septic and sewer groups are not significantly different for total nitrate, the septic group is significantly greater than the undeveloped group (p=0.0018) for total nitrate in non-storm flow conditions.

Figure 3.9. Total nitrate+nitrite group mean values in non-storm and storm conditions (error bars indicate one standard deviation of the mean).




Sewer group means were significantly greater than septic group means for total orthophosphorus (Figure 3.10) in both storm (p=0.0011) and non-storm (p=0.0138) flow conditions. Septic group means were significantly greater than undeveloped sites in non-storm flow conditions (p=0.0280) for total orthophosphorus. Group means for total orthophosphorus site groups decrease with decrease mean site group impervious cover, although again septic group values are not significantly different from effluent irrigation mean values. Non-storm total phosphorus values were not different between groups.

Figure 3.10. Total orthophosphorus group mean values in storm and non-storm conditions (error bars indicate one standard deviation of the mean).

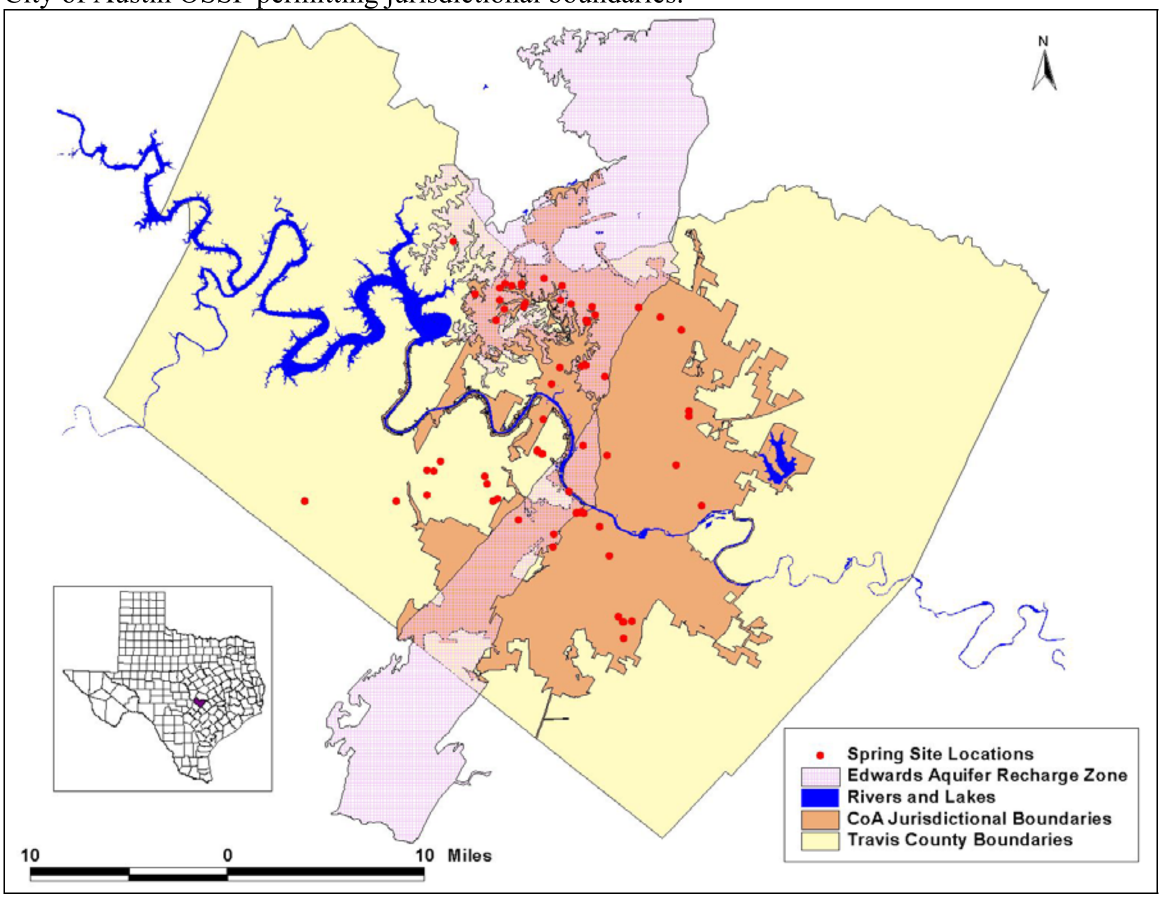


## 4.0 Ground Water Results

*4.1 Spring Site Selection*

Similar to the selection of surface water sites for analysis, ground water spring discharge sites with three or more data points per parameter, at least one sample event after January 1990, and spatially located within the jurisdictional boundaries of the City of Austin or Travis County were included in the analysis (Figure 4.1). One additional exclusion for spring sites, however, was eliminating results for 5 major springs (Barton Spring, Cold Spring, Old Mill Spring, Eliza Spring and Upper Barton Springs), located in or near Zilker Park in central Austin. Recharging areas for these watersheds are known to be drastically larger based on discharge rate and dye tracing studies (COA 1997), and thus are not directly comparable to data from other smaller springs in the Austin area.

Figure 4.1. Spring locations included in the analyses presented with Edwards Aquifer Recharge Zone and City of Austin OSSF permitting jurisdictional boundaries.




Using the selection criteria specified, data from 61 springs were utilized in the analysis. The majority of the data was collected by WRE personnel (6,116 samples), with an additional small percentage of samples collected by the USGS (9.7% of total).

Spring sites occurred in seven different geologic formations, ranging from Edwards limestone to terrace deposits and alluvium. ANOVA comparisons of site averages from different formation yield only two parameters with statistically significant differences between formations. E coli and dissolved phosphorus in baseflow conditions are significantly higher in Glen Rose springs than all other formations. However, due to the general lack of differences between formations, no further site separation on the basis of geologic formation was employed.

Intersection of the delineated spring watershed coverage with the OSSF permit locations generated from City of Austin and Travis County permit records yielded only one site with any known permitted OSSF in the watershed area: Bronc Spring, with 51 permits located within the 45 acre watershed with an average age of 14 years. No known permitted OSSF is located within 150 linear feet (the minimum setback distance required by TCEQ) of any spring location. Only 8 known permit locations fall within 300 linear feet of any springs (Bronc Spring, Wild Basin Ledge Spring, Shelf Spring and Tufa Spring).

To further identify sites that may be impacted by OSSF even though no OSSF appear in the immediate watershed area, sites were grouped into "potentially impacted by OSSF" and "no projected impacts by OSSF" visually using the permit location coverage in combination with topographic and flow direction information. If a spring was located downslope and within surface water drainage area boundaries of permitted OSSF, then the spring was labeled as "potentially impacted by OSSF."

## *4.2 Spring Results*

Several springs are known to be impacted from golf course irrigation and fertilization practices (COA 2003b), and golf course impacted springs were identified for this analysis using GIS. Site averages for golf course-impacted springs and non-impacted springs were compared by Wilcoxon rank-sum test. Golf course impacted springs yielded significantly higher concentrations of chloride, total nitrate+nitrite and sulfate during non-storm flow conditions. Due to these observed differences, golf course springs were separated from other springs in further analyses.

Site means for the "OSSF-impacted" and "non-impacted" site groups identified visually were compared by Wilcoxon rank-sum test, removing golf-course impacted springs prior to analysis. Few statistically significant differences between potentially-impacted and non-impacted springs were observed. However, total orthophosphorus ($p=0.02$) and total Kjeldahl nitrogen ($p=0.02$) in non-storm flow conditions were significantly higher in impacted springs. Total phosphorus ($p=0.08$) in non-storm flow may be higher in impacted springs.

OSSF density estimates from 1990 US census block information were used to determine more rigorously quantitative site groups. Spring watersheds were intersected with the census block coverage, and any spring with estimated OSSF density greater than zero was labeled as "impacted" while all zero density spring watersheds were labeled as "unimpacted." By Wilcoxon rank-sum analysis, no statistically significant differences were observed between impacted and unimpacted site groups with or without golf course springs included in the analysis.

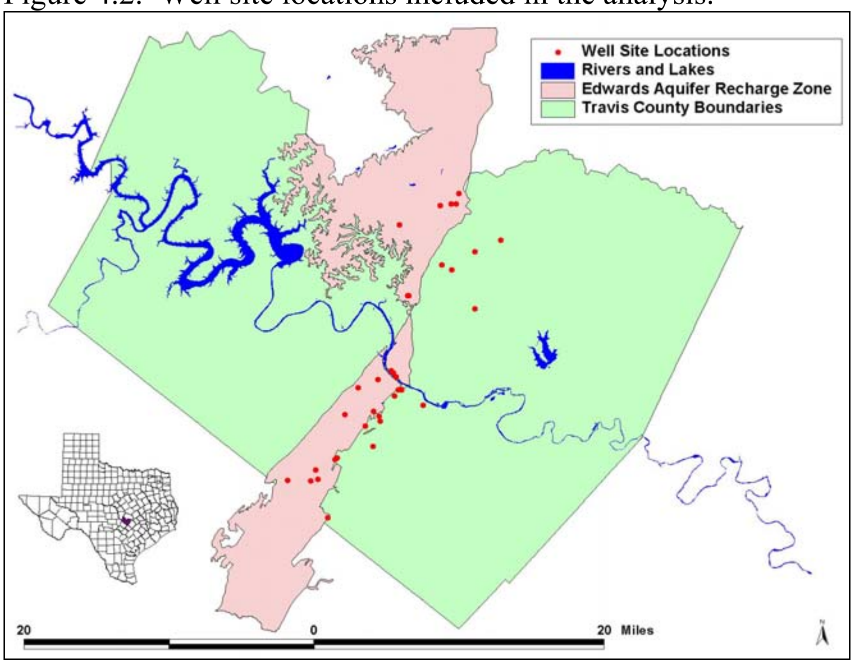
Spearman correlation analysis was performed on spring site averages versus OSSF and public sewer housing unit density from US census information and percent impervious cover in the spring watershed. Percent impervious cover was positively correlated with chloride, sulfate and E coli in non-storm flow conditions. Percent impervious cover was negatively correlated with mean spring ammonia concentrations ($r_S=-0.41, p<0.01, n=46$). Sewer housing unit density was positively correlated with chloride in both storm ($r_S=0.85$, $p=0.03$, $n=6$) and non-storm flow ($r_S=0.34$, $p=0.04$, $n=38$) conditions. OSSF density was not significantly correlated with any parameter in storm or non-storm flow conditions. The Spearman correlation was run again, partial with percent impervious cover, and no significant correlations between OSSF density and mean spring parameter concentrations was observed in either storm or non-storm flow conditions.

Only 4 spring locations (Bronc Spring, Wild Basin Ledge Spring, Shelf Spring and Tufa Spring) are within 300 linear feet, twice the legal setback, of a known permitted OSSF. Site averages for these four springs were compared to the average of all other springs by Kruskal-Wallis. Again, few statistically significant differences were observed, although mean total phosphorus ($p>\chi^2=0.04$) and total Kjeldahl nitrogen ($p>\chi^2=0.01$) were significantly greater during non-storm conditions in the 4-spring group than the average of all other spring sites. Mean total orthophosphorus in non-storm conditions was also greater in the potentially impacted 4-spring group, although the difference was slightly non-significant ($p>\chi^2=0.0594$).

### *4.3 Well Site Selection*

Ground water well sites with three or more data points per parameter, at least one sample event after January 1990, and spatially located within the jurisdictional boundaries of the City of Austin or Travis County were included in the analysis (Figure 4.2). Well ground water quality data as queried from the WRE Field Sampling Database for the 35 sites included in the analysis was collected primarily by the USGS (625 samples), with the City of Austin collecting an additional small percentage of samples (4% of total). Well sample locations were spatially diverse along the north-south axis of the Northern and Barton Springs portions of the Edwards Aquifer through Travis County. The majority of wells occurred in Edwards limestone.

Figure 4.2. Well site locations included in the analysis.




### *4.4 Well Results*

Contaminated wells were identified using the 2002 TCEQ screening levels for nutrient parameters (TCEQ 2004) and the year 2000 segment-specific Texas state water quality standards for chloride and sulfate (TAC) for segments within the Edwards Aquifer recharge zone (Table 4.1). All well site average phosphorus nutrient levels were below TCEQ screening levels, and thus no wells were considered as contaminated for phosphorus nutrients. All well site averages were between 1 and 10 cfu/100mL, and below TCEQ contact recreation standards for fecal coliform, and thus not considered contaminated for bacteria. Nitrogenous nutrient values did exceed TCEQ screening levels for some well locations, and contaminated versus uncontaminated wells were compared for ammonia and nitrate.

Table 4.1. Average site concentrations used to determine well contamination by parameter derived from TCEQ screening levels and state standards.

| Parameter | Contamination Level (mg/L) | Source |
|---|---|---|
| Ammonia as N | 0.17 | 2002 Screening Level |
| Nitrate+Nitrite as N | 2.76 | 2002 Screening Level |
| Chloride | 50 | Standard |
| Sulfate | 50 | Standard |

Average values for septic housing unit density, public sewer housing unit density and average minimum distance from OSSF permits were calculated for contaminated and uncontaminated wells, and compared by Wilcoxon rank sum test (Table 4.2). Average septic density was statistically greater for wells uncontaminated by chloride (p=0.03), and average sewer housing unit density was statistically greater in wells contaminated by sulfate (p=0.01). No clear patterns between contaminated and uncontaminated wells for septic housing unit density or average minimum distance are evident between contaminated and uncontaminated wells.

Table 4.2. Average septic housing unit density, average public sewer housing unit density and average minimum distance for contaminated and uncontaminated wells by parameter with associated probabilities values from Wilcoxon rank-sum test.

| | Avg Septic Density | | | Avg Sewer Density | | | Avg Min Dist | | |
|---|---|---|---|---|---|---|---|---|---|
| | Cont. | Uncont. | p | Cont. | Uncont. | p | Cont. | Uncont. | p |
| Nitrate+Nitrite as N | 0.055 | 0.055 | 0.98 | 1.892 | 1.213 | 0.32 | 1688 | 2384 | 0.55 |
| Ammonia as N | 0.016 | 0.068 | 0.09 | 1.531 | 1.076 | 0.16 | 3221 | 1403 | 0.43 |
| Chloride | 0.032 | 0.063 | 0.03 | 1.789 | 0.951 | 0.52 | 2366 | 2047 | 0.54 |
| Sulfate | 0.029 | 0.071 | 0.32 | 1.851 | 0.739 | 0.01 | 3028 | 1597 | 0.15 |

Spearman correlation analysis was performed on the well site average concentrations versus the density of septic and public sewage housing units from 1990 US census estimates as well as the minimum distance (in feet) from the nearest mapped OSSF permit location (Table 4.3). Few parameters exhibited statistically significant correlations with the census-derived sewage disposal method or OSSF permit distance. Ammonia was inversely related to septic housing unit density. Both chloride and sulfate were directly related to housing units with public sewer density.

Table 4.3. Spearman correlation coefficient ($r_S$), associated probability and number of site averages for wells versus septic housing unit density, sewer housing unit density and minimum distance from OSSF permit. Statistically significant correlations highlighted.

| Parameter | Variable | $r_S$ | p | N |
|---|---|---|---|---|
| NH3 | septic_density | -0.38 | 0.0492 | 27 |
| NH3 | sewer_density | 0.34 | 0.0834 | 27 |
| NH3 | min_dist | 0.21 | 0.3010 | 27 |
| NO3+NO2 | septic_density | 0.18 | 0.3504 | 29 |
| NO3+NO2 | sewer_density | -0.10 | 0.6088 | 29 |
| NO3+NO2 | min_dist | 0.00 | 0.9879 | 29 |
| OP | septic_density | -0.07 | 0.8314 | 11 |
| OP | sewer_density | 0.19 | 0.5685 | 11 |
| OP | min_dist | 0.34 | 0.3117 | 11 |
| TP | septic_density | 0.04 | 0.8355 | 29 |
| TP | sewer_density | 0.17 | 0.3760 | 29 |
| TP | min_dist | 0.08 | 0.6665 | 29 |
| Chloride | septic_density | 0.19 | 0.3065 | 30 |
| Chloride | sewer_density | 0.50 | 0.0054 | 30 |
| Chloride | min_dist | 0.05 | 0.7964 | 30 |
| Sulfate | septic_density | -0.02 | 0.8973 | 30 |
| Sulfate | sewer_density | 0.38 | 0.0366 | 30 |
| Sulfate | min_dist | 0.35 | 0.0573 | 30 |
| fecals | septic_density | -0.27 | 0.3073 | 16 |
| fecals | sewer_density | 0.10 | 0.7077 | 16 |
| fecals | min_dist | 0.15 | 0.5773 | 16 |

The relationship between septic housing unit density and ammonia was strongly affected by three extreme data points (Figure 4.3). One well (site #685) in the Barton Springs Zone/Edwards Aquifer limestone formation with high septic housing unit density (1 septic/acre) yielded low average ammonia levels, while two sites with zero septic housing unit density yielded high average ammonia values. Removal of these three points yields a non-significant Spearman correlation ($p=0.31$, $n=24$) between site average ammonia and septic housing unit density. The regression between mean ammonia and septic housing unit density was non-significant ($p=0.62$, $r^2<0.01$).

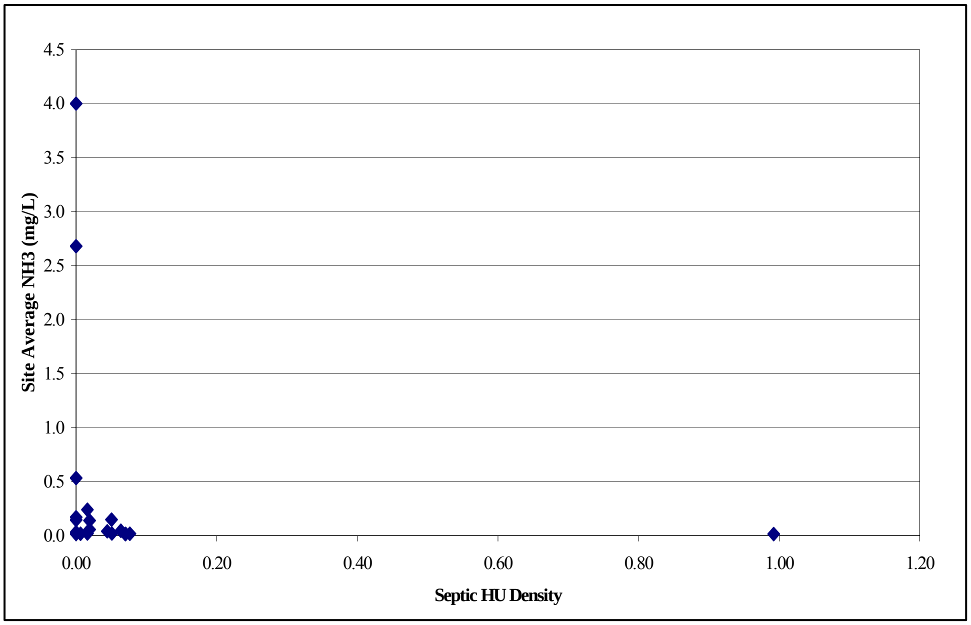



Figure 4.3. Septic housing unit density versus average well concentration.

Correlations between sulfate and chloride versus public sewer housing unit density, though statistically significant, are weak (Figure 4.4), and yield non-significant linear regressions with $r^2$ values unacceptable for prediction.

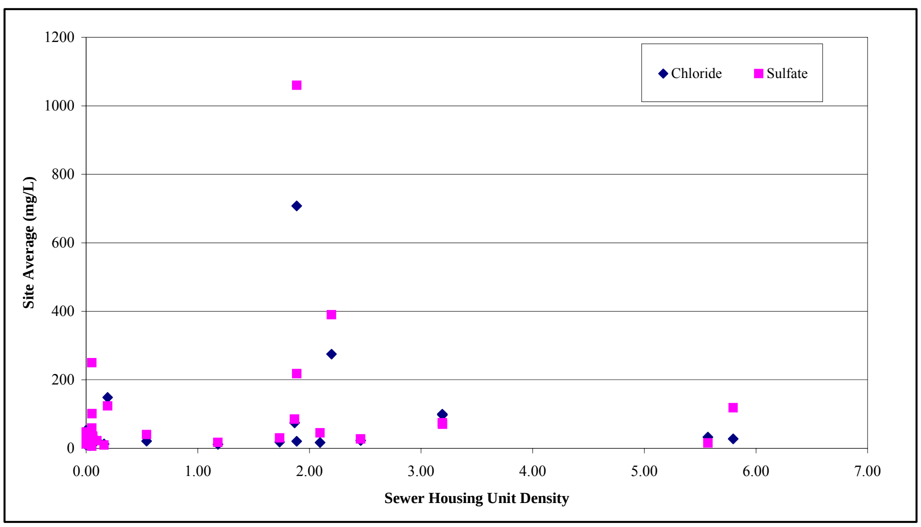



Figure 4.4. Well site average chloride and sulfate concentrations versus public sewer density.

In a final attempt to elucidate differences between wells in correlation to observed differences in OSSF distribution, well site locations were visually inspected in comparison to OSSF permit locations and the 5-acre creek grid to group well site local drainage areas into three categories: potentially affected locally by OSSF (Yes), no OSSF in local area (No), and unknown (Maybe). Average well site concentrations by parameter were compared for these groups by Kruskal-Wallis, with additional exploratory analysis using the REGWQ multiple range test. Then, the "yes" group was directly compared to the "no" group by

Wilcoxon rank-sum test (Table 4.4). Few parameters were significantly different between all three groups, and in no case were average parameter concentrations greater in wells with drainage areas potentially including local OSSF. Dissolved phosphorus and total sulfate actually yielded higher concentrations in the unaffected (No) group.

Table 4.4. Comparison of well site groups by parameter.

| **param** | **K-W p-value for comparison of all three site groups (p-value)** | **Wilcoxon rank-sum for Yes vs. No (p-value)** | **Max mean group concentration in…** |
|---|---|---|---|
| Ammonia as N, Dissolved | 0.01 | 0.06 | Maybe |
| Ammonia as N, Total | 0.70 | 0.64 | No |
| Chloride, Total | 0.21 | 0.36 | No |
| Fecal coliform bacteria | 0.74 | 0.35 | Maybe |
| Nitrate as N, Dissolved | 0.12 | 0.42 | No |
| Nitrate as N, Total | 0.02 | 0.23 | No |
| Nitrate+Nitrite as N, Dissolved | 0.31 | 0.42 | Yes |
| Nitrate+Nitrite as N, Total | 0.18 | 0.24 | No |
| Orthophosphorus as P, Diss | 0.96 | 0.39 | No |
| Orthophosphorus as P, Total | 0.21 | 0.23 | Maybe |
| Phosphorus as P, Dissolved | 0.21 | 0.04 | No |
| Phosphorus as P, Total | 0.38 | 0.32 | Maybe |
| Sulfate, Total | 0.05 | 0.03 | No |
| Kjeldahl Nitrogen as N, Dissolved | 0.07 | 0.12 | Maybe |
| Kjeldahl Nitrogen as N, Total | 0.72 | 0.41 | No |

## 5.0 Conclusions

Although the number of housing units utilizing OSSF for waste disposal increased in Travis County and the City of Austin from 1970 to 1990, the percentage of OSSF housing units in the City of Austin more than doubled in the same time period. As Austin continues to annex new areas and development in outlying zones previously outside wastewater service areas, the number of OSSF in Austin will increase. Increasing population density may also result in OSSF placement in unsuitable site areas. Water quality impacts from OSSF are expected to increase in magnitude and spatial extent in the City of Austin and Travis County in the future.

OSSF density estimates in previous efforts by the City of Austin were limited by the usefulness of the permit record information from the City of Austin and Travis County. The permit database is incomplete, and the US census provides better estimates of the number and location of OSSF in the Austin/Travis County area. Estimates of OSSF from the US census are not available after 1990, though still provide a better spatial estimation of OSSF location. Permit records may be a source of additional explanatory variables such as type of system, size of drain field and soil percolation rates if a complete query can be obtained from City of Austin Water Utility. OSSF information from the US census was disaggregated from the block group level to the block level. Better spatial disaggregation may be available in the future.

Annual watershed loading estimates of total suspended solids, total organic carbon, total Kjeldahl nitrogen, total nitrate and total nitrogen are positively correlated to OSSF density. Multiple regression of OSSF density and watershed loads adjusting for watershed percent impervious cover yield statistically significant positive relationships between OSSF density and water quality, with model $r^2$ values ranging from 68% to 82%.

Average adjusted distance and average distance are not useful predictors of OSSF effects on surface water quality site mean value, or the effects of these variables are masked by other explanatory variables not quantified in this analysis. Minimum distance from OSSF may be inversely related to nitrate+nitrite concentrations in storm flow conditions, and the estimated relationship suggests that distances greater than 216 feet may be necessary to maintain acceptable receiving surface water stream nitrate levels.

Average age explained little variation in surface water quality site mean values, although nitrate+nitrite in storm flow may be positively related to average age of OSSF in contributing watersheds. High mean nitrate+nitrite values are observed as sites where average age of OSSF are between 12 and 15 years, suggesting a window for potentially increased frequency of malfunction and need for replacement.

OSSF densities explained little variation in surface water or ground water quality site mean values. However, average densities for surface water site drainage areas were generally low, with the average OSSF density of all sites estimated at 0.086 OSSF/acre and a median density of only 0.024 OSSF/acre. More than 80% of sites considered in surface water analyses yielded OSSF densities less than 0.1/acre. Spring watersheds were small, and most likely an underestimate due to the scale of grids used in the watershed delineation process. Identification of OSSF-impacted springs may be inaccurate with the data currently available.

Lot size for potentially OSSF-impacted sites did not correlate with observed patterns in surface water quality site mean values.

Comparison of average concentrations indicate that while OSSF-impacted surface water sites are generally similar to areas utilizing effluent irrigation, mean values of OSSF-impacted surface water sites are generally less than values observed for sites in areas served by public central sewers. Mean indicator bacteria, nitrate and orthophosphorus levels from OSSF-impacted surface water sites were higher than sites in undeveloped areas.

Mean ground water quality of selected springs did not vary by the geologic formation in which the springs occurred. Spring mean total orthophosphorus, Kjeldahl nitrogen and total phosphorus may be higher during non-storm flow conditions in OSSF-impacted springs than unimpacted springs. No OSSF impacts were observed on monitored wells included in this analysis.

## 6.0 Discussion

Following the assumption that one parcel equals one house used to calculate average lot size, the number of parcels for each surface water site watershed in areas that primarily utilize OSSF for sewage disposal may be used to determine the relative accuracy of OSSF density as calculated from the US census estimates and the City of Austin and Travis County permit records. Although the assumption that number of parcels equals number of housing units is most likely an overestimation, the inadequacy of the permit record information is evident by comparing the number of parcels to the number of permits for each of the 20 site watersheds identified as primarily OSSF. The number of parcels is on average 23 times greater than the number of permits from City of Austin and Travis County records, even though the permit information was current as of 2001. However, the number of parcels is only on average 3 times greater than the number of estimated OSSF housing units from 1990 US census information. Considering that Travis County has experienced approximately 37% growth in total number of housing units from 1990 to 2002, and the TCAD information was current as of January 2005 while the septic housing unit information was generated from the 1990 US census, it is unclear how the number of OSSF housing units is more accurately estimated from the US census information than the number of permit records from the City of Austin and Travis County. However, due to the age of the data, the number of septic systems estimated from 1990 US census information is still an underestimation of the total number of housing units currently utilizing OSSF in Travis County.

An illustration of the differences between the City of Austin and Travis County permit records versus estimates based on US census blocks can be found for the Granada Hills Subdivision in the Slaughter Creek Watershed in southwestern Travis County. The 555 acre watershed was assessed as a residential with septic site in previous City of Austin studies, and is included in this analysis as the primarily septic group. Permit records indicate that there are only 7 permitted OSSF within the tributary drainage area boundaries. US census information indicates a total of 234 OSSF housing units in the tributary drainage area boundaries, while TCAD yields a total of 355 single-family housing sized (approximately 1-acre) parcels.

The use of US census data to water quality impact investigation may be further extended by evaluation of number of housing units on public sewers. Impervious cover is frequently used as the surrogate measure for the quantification of development and urbanization on water quality (Schueller 1995), although accurate estimates of impervious cover via GIS are not always available in large spatial extents. The strong correlations between both impervious cover and surface water quality site mean values with housing units on public sewer density from US census estimates suggest that public sewer housing unit density may be used as a surrogate for impervious cover in predicting or explaining water quality impacts when impervious cover is not available.

Failing OSSF are frequently targeted as explanations for water quality impairments. Fecal bacterial contamination of recreational areas on the Guadalupe River near Kerr County, Texas, initially targeted near-shore OSSF of area campgrounds as the source. Extensive testing and bacterial source tracking by the Upper Guadalupe River Authority in 1993; however, indicated camps near the river were not the source of the bacterial impairment (TWRI 1995). Although correlations between water quality and OSSF distribution are observed in Travis County, OSSF do not appear to exert undue impacts on area surface and ground water resources.

The importance of scale in studies of OSSF impacts to water quality has been identified in previous studies (Brown and Bicki 1987), and should always be considered when evaluating impacts for the formulation of regulations or management strategies. OSSF effects in previous studies were noted at densities of 0.5 OSSF/acre and 1 OSSF/acre, within the density limits of 0.5 OSSF/acre to 1.3 OSSF/acre that could result from existing City of Austin or TCEQ regulations. However, actual densities observed for the small and large watersheds considered were low, and generally less than 0.1 OSSF/acre. Correlations between large watershed water quality loading estimates and OSSF density were observed at low density, though few statistically significant correlations were observed between OSSF density and small watershed mean water quality parameter values. Use of questionable permit data for comparison to regional water quality data is problematic in this application.

Average age of OSSF did not explain the observed variation in water quality, although effects were observed for nitrate+nitrite following storm events. TCEQ indicates that OSSF greater than 15 years of age may need replacement, although the lifetime of a properly maintained OSSF may be much greater. Average age may be useful in the future to determine the inspection frequency of OSSF if area regulatory agencies decide to move from a complaint-response driven process to a more temporally scheduled inspection process. Focusing on OSSF systems above a certain age may provide a means of prioritization of inspections to better focus resources.

Distance of OSSF from sites generally did not correlate with observed patterns of water quality. It should be noted that the raindrop function used to estimate distance should result in greater estimates of distance than the linear distance requirements of current regulations. Minimum distance estimates for plumes for phosphorus from the literature were measured at 10meters, less than setback distances generally required by current regulations in Travis County. Only four surface water sites had OSSF within 150 feet, although the average distance from OSSF at the watershed scale for three of these four sites ranged from 1500 to 37000 feet. Comparison of these four surface water sites to other sites (excluding golf course and effluent irrigation impacted sites) yielded no statistically significant differences by Wilcoxon rank-sum test. The importance of setbacks for the protection of environmentally-sensitive recharge features and creeks has been previously noted (COA 1997), and although current regulations appear adequate, future increases should be carefully monitored.

Although assessed OSSF factors did not explain variation in observed water quality of included wells, drinking water wells may be most sensitive to OSSF impacts and pose the most direct threat to public safety. Previous similar studies were also unable to relate OSSF site factors to well water quality (Stark et al 1999). The complexity of ground water movement in combination with the difficulties inherent in sampling and normalizing well water quality (including both water table height and well casing information) suggest that further focused assessments be utilized to better identify the impacts of OSSF on wells in Travis County.

Although current regulations may have been adequate to protect water quality in the past, increasing residential developments utilizing OSSF for waste disposal will increase. US census data was found to better estimate numbers of OSSF systems that permit records, although US census information is not available after 1990. Extrapolation of US census OSSF information to the year 2000 and beyond could be accomplished by modeling of US census data based on population using relative percentages of means of sewage disposal, although these percentages have changed over time. Another more spatially representative method could be achieved by combining census information on population with TCAD parcels or by utilizing arbitrary buffer zones around new roads, identifying parcels or road buffers outside current central sewer service areas (if that data was available spatially) as utilizing OSSF. Home owners who choose OSSF for sewage disposal inside existing wastewater service areas could be identified by regulatory permit agency records.

The *a posteriori* analysis approach utilized increases the inherent value of the large size and extent of water quality data collected by the City of Austin and other agencies in the Travis County area. However, the correlative relationships identified cannot be substituted as causal relationships. Only more focused *a priori* experimental designs can clearly quantify the relationships between OSSF distribution and water quality. Watersheds in Hays County, though not considered in this analysis, may be a useful area for future studies due to the higher percentages of housing units with OSSF—48% in Hays County in 1990 in comparison to only 11% in Travis County—and less area impacted by large urban development.

Modeling the effects of OSSF on water quality, particularly in the formulation or review of regulation, could be accomplished by artificial neural networks (ANN). ANNs are a useful modeling tool in that large numbers of variables may be considered without explicit knowledge of the interactions between the variables (Caudill and Butler 1990). Studies from the Lake Granbury region of Texas indicated that underlying soil types and geology may be the most critical factors in determining potential impacts from OSSF, even more than lot size (TWRI 1993). In addition to the density, distance, lot size and age factors considered in this analysis, an artificial neural network could be utilized to relate these (and more) predictor variables to model effects of OSSF on water quality. Resulting models could be used to evaluate effectiveness of alternative regulations in comparison to current restrictions.

Water quality monitoring of seeps and springs may serve as an early warning indicator of contamination threatening surface water resources from OSSF in high-density OSSF areas. This approach has been adopted by the Upper Guadalupe River Authority in the hill county areas of Kerr County, Texas (Jensen 1999).

Dye tracing using fluorescent rhodamine has been used by the Upper Guadalupe River Authority to identify sites where OSSF contaminants were flowing into area streams from homes and campgrounds (Jensen 1999). Brighteners from laundry detergents fluoresce under UV light, and have also been used to identify OSSF as sources of contamination where nutrient enrichment and contact recreation impairment was occurring in the Brazos River Watershed (TWRI 1999). Studies in coarse-textured soils in Wisconsin; however, indicated that fluorescence of naturally-occurring humic and fulvic acids may mimic brighteners, and optical brighteners may not be passing through OSSF drain fields (Alhajjar et al 1990). Tracers have not been used locally to examine OSSF pollutant transport in the subsurface; but may be considered if the appropriate conditions occur.

The percentage of housing units with OSSF in the City of Austin increased from 1970 to 1990. As development continues into the future, the projected loads from OSSF will also increase. Ground water systems may be particularly vulnerable to increasing pressure, as creek recharge may be limited to a maximum value (Barrett and Charbeneau 1996a) while nutrient inputs both from stormwater runoff from impervious surfaces and through diffuse recharge from OSSF can increase without limit. Additional development may even reduce recharge to the Edwards Aquifer (Barrett and Charbeneau 1996a) , further compounding the problems of additional nutrient inputs. More nutrient input and less diluting recharge will most likely result in increasing nutrient concentrations in Austin springs and wells. The effects of increased nutrient output from major springs, like Barton Springs, will result in tangible increases in algae growth and eutrophication problems in Austin-area creeks and Town Lake.

Lake Austin, with high concentrations of OSSF along the shorelines, maintains statistically significant high concentrations of photosynthetic plankton counts than Town Lake based on data collected from the City of Austin drinking water treatment plants located on both lakes (City of Austin 2004b). Increased nutrient input from shoreline or near shore OSSF may sustain the higher Lake Austin algal levels. Other lakes in Texas have been considered of concern for impacts from OSSF, including Lake Granbury (TWRI 1993). High concentrations of nutrients and indicator bacteria were recorded in many isolated coves of Lake Granbury where reduced wind action and water currents reduced mixing with the mainstem waters of the lake (TWRI 1993). The City of Austin Health Department suspended annual inspections of the OSSF along Lake Austin in 1999. Additional focused monitoring may be necessary to determine if nutrient enrichment from OSSF effluent is sustaining increased biomass in Lake Austin, and if regular regulatory inspections should be resumed.

Although current regulations appear to be partially mitigating the large-scale impacts of OSSF on water quality in Travis County, public education may be an important component to the continued successful management of OSSF water quality impacts. Regulatory officials should encourage water conservation for OSSF home owners by providing incentives for low-water use appliances to limit the amounts of wastewater discharged to drain fields, adding that water conservation practices will increase the life of the OSSF for the home owner. Use of composting, not in-sink disposal, by home owners can minimize the amounts of suspended solids and organic matter input to the OSSF. Public education efforts advocating the avoidance of hazardous chemicals to clean OSSF systems and disposal of any hazardous chemical into the OSSF can be presented to home owners not only as a measure to protect the health of aquatic ecosystems in the area but also as a public drinking water safety initiative. While beneficial from a public relations standpoint, the funding of these initiatives should be weighed against the evidence that OSSF constitute an immediate major threat to water quality which has yet to be shown through *a priori* cause-effect studies locally. At a minimum a risk assessment of OSSF versus central wastewater collection should be undertaken.

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