SUBDIVISION IMPACTS ON GROUNDWATER QUALITY

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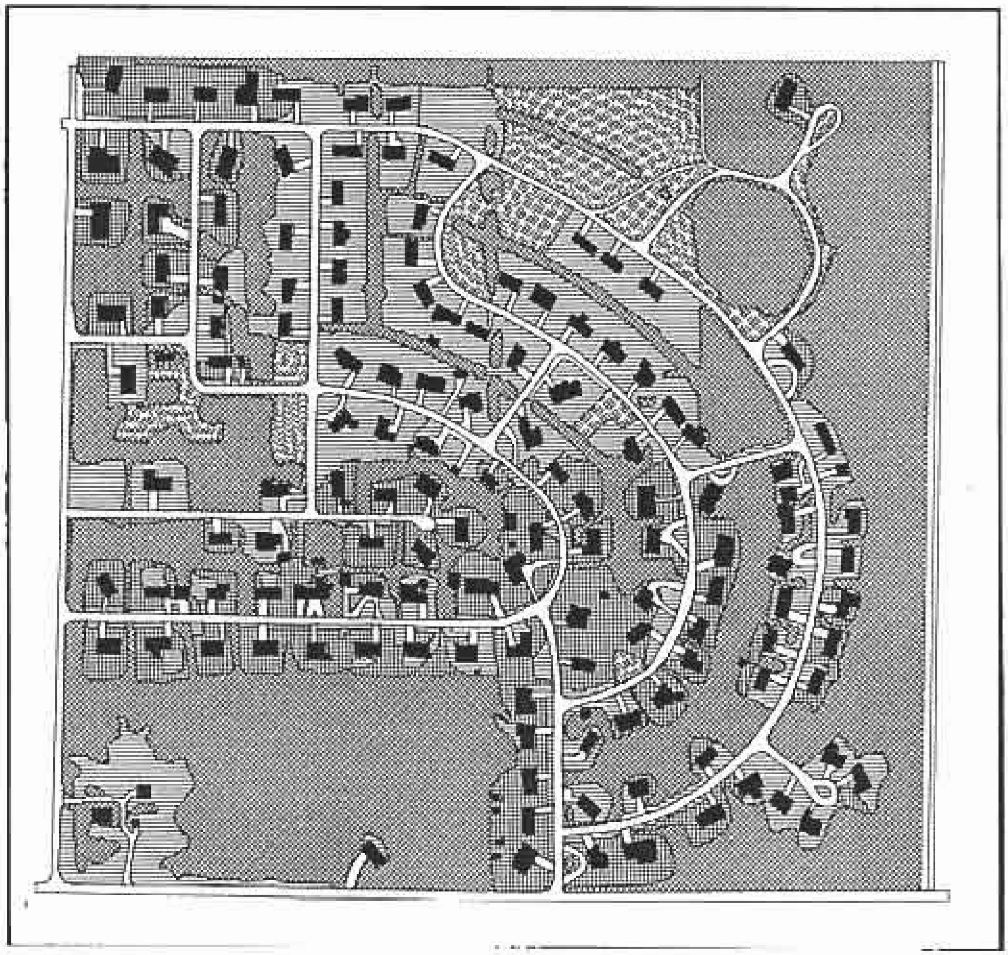

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Abstract

The impact of subdivisions on groundwater quality has become a topic of interest throughout the United States, as interest in groundwater protection has increased. Development of unsewered subdivisions adjoining municipal areas have increased as urban populations expand and people seek suburban areas.

This study was initiated in 1987 in an attempt to quantify the impacts of subdivisions on groundwater quality in the Central Sands area of Wisconsin. The project involved the installation of over 200 monitoring wells in and around two subdivisions. These wells were sampled and analyzed for a variety of chemicals over a four year period. Nitrate-N loading to groundwater was the primary focus of the project, with volatile organic chemicals, phosphorous, and several other indicator chemicals run on selected samples.

Homeowners were surveyed to determine household and lawn chemical use, and to obtain their opinions on groundwater quality. A number of individual septic systems were monitored, as were several lawns, to obtain data specific to these practices that impact groundwater quality. A Nitrogen Mass Balance model was used to test its capabilities to predict subdivision impacts.

Results of this project clearly demonstrated that subdivisions on sandy soils do impact groundwater quality with nitrate-N levels exceeding 10 mg/l. Chloride, phosphorous, sodium, and limited volatile organic chemicals were also found in elevated concentrations downgradient of the subdivisions. Septic systems contributed approximately 80 percent of the nitrate-N to groundwater for the areas studied, with lawns contributing the remaining 20 percent. Lot sizes in these subdivision were approximately 0.16 hectare, with about three homes per hectare including roads, vacant lots, and open areas.

The BURBS mass balance nitrogen Loading model provided good estimates of groundwater impacts from subdivisions.

Extensive water quality differences were observed within and downgradient of the subdivisions. Contaminant plumes from septic systems mixed slowly with groundwater, which resulted in dramatic variability of water quality both vertically and horizontally downgradient of the subdivision. This wide variability makes it very difficult to measure groundwater impacts even when a large number of multi-level wells are used. Variability seasonally and from year to year was observed in shallow monitoring wells, responding to relative amounts of groundwater recharge.

The presence of relatively undiluted contaminant plumes 30 meters downgradient of septic systems makes it extremely important to be certain private wells are not located in a groundwater flow path from drainfields, or that they are of sufficient depth to avoid the contaminant plume.

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A. Introduction

Concern over the impact of subdivisions on groundwater quality has been growing for a number of years. Increased incidence of high nitrates in private wells, concern over wellhead protection, and an awareness of groundwater protection have all led to this widespread concern. Portage County, Wisconsin has worked on a groundwater management plan since 1985. One of the most controversial parts of the plan has been the use of increased lot size to protect groundwater from onsite waste disposal. This may improve groundwater quality, but results in more expensive housing and all the problems associated with urban sprawl.

This study was initiated in 1987 to address the subdivision water quality issues and attempt to quantify the impacts of subdivisions on groundwater quality in sandy soils areas near Stevens Point. This project was directed by Dr. Byron Shaw with three M.S. graduate students at UW-Stevens Point working on various aspects of the project. Detailed results of this project are found in the M.S. theses of Peter Arntsen, Steve Henkle, and William VanRyswyk. In addition, much of a PhD thesis by Erik Harmson, UW-Madison contains information relative to the project. Fred Madison, UW-Madison assisted with several aspects of the project. Chris Mechenich of the Central Wisconsin Groundwater Center compiled the survey of homeowner practices and attitudes, this data is summarized in a report by Mechenich et. al. 1991.

Two subdivisions near Stevens Point were selected for detailed analysis in this study (Figure 1). The subdivisions were selected based on historical data indicating groundwater quality problems or the potential for groundwater quality problems.

Primary objectives of this project were as follows:

1. Determine homeowner practices that could impact groundwater quality and determine attitudes of homeowners relative to groundwater quality and protection;
2. Determine nitrate-N loading to groundwater from subdivisions and evaluate the use of BURBS nitrogen mass balance model for predicting nitrogen impact;
3. Determine nitrogen contribution from septic systems and lawns;
4. Determine the impact of individual septic systems on nitrate-N and phosphorous concentrations in groundwater downgradient of the system;
5. Determine if volatile organic chemicals (VOCs) are reaching groundwater from subdivision activities;
6. Evaluate the various monitoring systems that could be used to determine subdivision impacts on groundwater;
7. Evaluate the use of geophysical techniques for locating septic system effluent plumes.

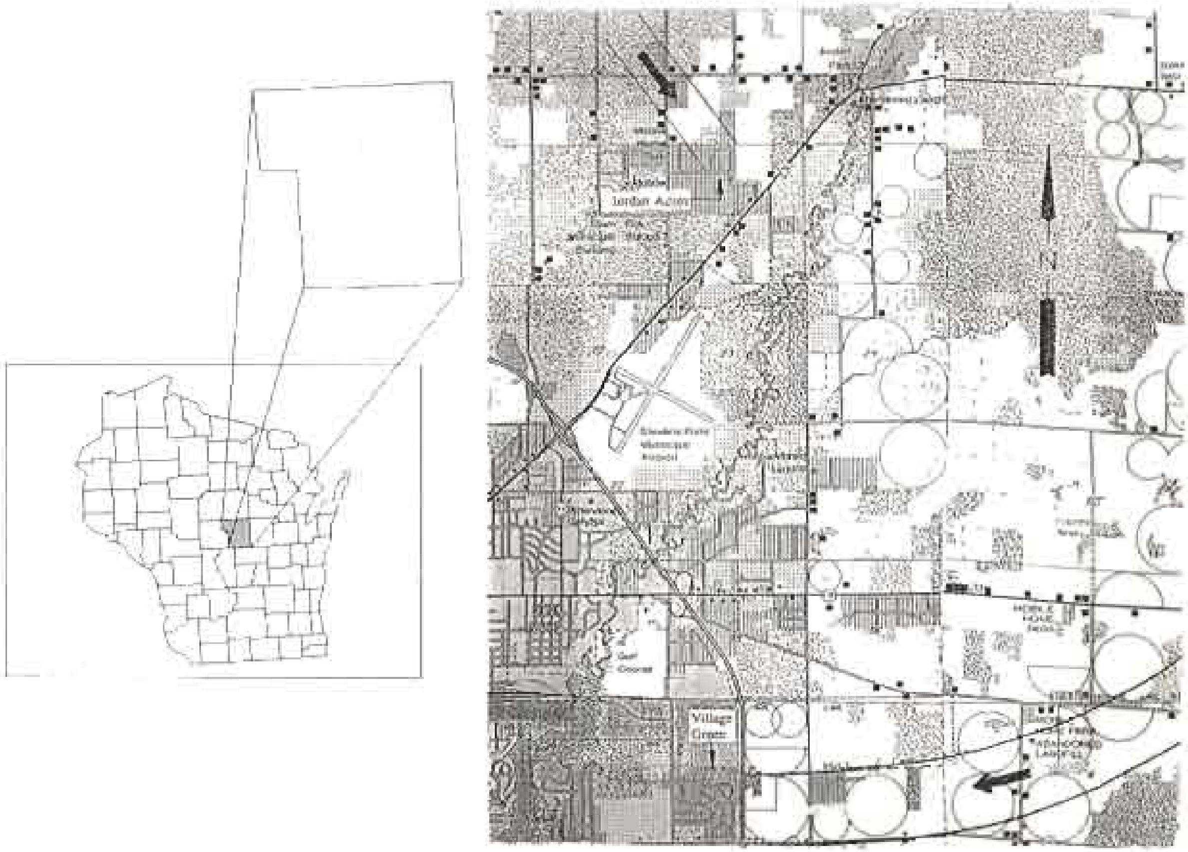

Figure 1. Upgradient land uses and locations of the subdivision study sites in Portage County, Wisconsin.

B. Literature Review

The following is a review of literature relevant to the movement and fate of potential groundwater contaminants from an unsewered residential subdivision in the Central Wisconsin Sand Plain. Specific sections will be devoted to Sand Plain Geology, Subdivisions and Nitrates, Septic Systems, Lawns, and Previous Work in the Study Area.

Sand Plain Geology

The geology of the Central Wisconsin Sand Plain is characterized by a relatively thick layer of highly permeable glacial sediments overlying impermeable rock (Faustini, 1985). The glacial material consists primarily of outwash sands and gravels and tends to be quite uniform in composition both laterally and vertically (Weeks et al., 1965). Though the sand plain is often assumed to be homogeneous, inconsistencies, such as layers or bands of higher or lower hydraulic conductivities, have been noted (Manser, 1983, Kimball, 1983, Stoertz, 1985).

Reported hydraulic conductivities in the sand plain range from 0.05 cm/sec (130 ft/day) (Weeks, 1969) to 0.18 cm/sec (500 ft/day) (Weeks and Stangland, 1971), with Faustini (1985) reporting an average from several sources of 0.10 cm/sec (270 ft/day). Slug tests performed in the study areas by Harmsen (1989) indicated slightly lower values of hydraulic conductivities ranging from 0.02 cm/sec to 0.07 cm/sec (57 ft/day to 198 ft/day).

Harmsen (1989) reported a range of 96.5 to 99.7 percent sand from samples taken in the upper 15 meters in the study areas. It was also noted that the sands graded to coarse sands and gravels at 23 to 25 meters below the surface.

Thicknesses of unconsolidated sediments overlying bedrock in the region ranging from 0 to 27 meters (0 to 90 ft.) were reported by Holt (1965) and by Weeks et al. (1965) during an investigation of the Little Plover River Basin. Harmsen (1989) reported an average depth to bedrock of 33 meters (108 ft.) in the Jordan Acres subdivision and an average depth to bedrock of 30 meters (98 ft.) in the Village Green subdivision. These values are estimates taken from well logs in the region of the subdivisions.

Effective porosities reported by Weeks et al. (1965) in the Little Plover River Basin ranged from 27.7 to 35.7 percent, with an average of 32.3 percent. Stoertz 1985) reported a range of 36.5 to 40.5 percent in five repacked samples taken from a site near Wisconsin Rapids.

Using an estimated average effective porosity of 0.23 and measured hydraulic gradients of 0.0025 and 0.0020 for the Jordan Acres and Village Green subdivisions respectively, Harmsen (1989) calculated average horizontal seepage velocities of 0.45 m/day (1.48 ft.) in Jordan Acres and 0.30 m/day (0.98 ft.) in Village Green.

Subdivisions and Nitrate-N

Studies evaluating the impact of rural housing on groundwater quality have been limited. The studies that have been conducted have focused primarily on the loading of nitrate-N from septic systems and to some degree lawns.

Nitrate-N is of special concern as a groundwater contaminant because it has been associated with methemoglobinemia (blue baby syndrome). Methemoglobinemia most often occurs in infants as a result of the ingestion of high nitrate-N water. The nitrate-N is converted to nitrite in the digestive system and then reacts with the hemoglobin in the blood converting it to methemoglobin (Mechenich, 1988). The methemoglobin cannot carry oxygen to the body as the normal hemoglobin can, resulting in oxygen deprivation (indicated by bluish-gray skin color) and possibly resulting in death. As an infant ages the pH in the stomach decreases and the susceptibility to the disease also seems to decrease (Mechenich, 1988). The State and Federal Standard for nitrates in drinking water is 10 mg/l. Studies have suggested that concentrations of nitrate-N as low as 13 mg/l can cause methemoglobinemia (Vigel et al., 1965).

Nitrate-N has also been associated with the potential for the formation of nitrosamines in soil (Brown et al., 1980), and in the human digestive system (Mechenich, 1988). Nitrosamines are among the most potent and broadly acting carcinogens known (Harmsen, 1989).

Numerous studies employing groundwater monitoring and modeling have demonstrated a correlation between groundwater contamination and onsite sewage disposal density (Bicki and Brown, 1991). The density of septic systems in an area is usually regulated by state or local agencies through zoning ordinances specifying setback distances. Septic system setback distances are specified minimum distances a septic tank or drainfield must be from surrounding homes, property lines, or water supply wells and often indirectly dictate the minimum lot size possible. As a result, lot size is often based upon engineering rather then environmental considerations (Perkins, 1984). According to the Environmental Protection Agency (1977), in most parts of the country septic tank density is the most important factor influencing local and regional groundwater contamination. Perkins (1984) interpreted this to indicate that drinking water well setback distances do not appear to be adequate in many regions to prevent groundwater contamination from septic system effluent.

Perkins (1984) reviewed several studies and empirical models designed to estimate the minimum lot size necessary to prevent groundwater contamination. Estimated lot sizes ranged from 0.2 to 0.4 hectares (0.5 to 1.0 acres) based on reported data and 0.3 to 0.4 hectares (0.75 to 1.0 acres) based upon theory. Bicki and Brown (1991) reviewed literature relative to septic system densities and reported that lot sizes in this range (0.2 to 0.4 hectares) are often cited as minimums for the prevention of groundwater contamination from septic system effluent. They also noted that some studies have found groundwater contamination from nitrate-N with lot sizes in this range due to site specific soil, hydrogeologic, and climatic conditions.

Bauman and Schafer (1984) present a simplified model and examine the possible groundwater quality impacts of nitrate-N loading from septic systems and the factors influencing such impacts. They also propose the addition of hydrogeologic or aquifer assessment criteria to the septic system site evaluation procedure. Included in this aquifer assessment criteria would be considerations for depth to aquifer, aquifer thickness, recharge rates, and groundwater flow velocities.

Depth to aquifer is important in the evaluation of the potential for denitrification to occur. Bauman and Schafer (1984) specify that in this evaluation of the vadose zone, specific characteristics to look for are; 1) the presence of restricting layers which may create anaerobic conditions, 2) temperature (warmer temperatures associated with shallow water tables promote metabolic activity, thereby enhancing denitrification), 3) residence time in the vadose zone (longer periods allow more time for denitrification to occur if reducing conditions exist) and 4) Dissolved organic carbon (DOC) content of the groundwater (higher concentrations stimulate bacterial activity, increasing the potential for both anaerobic conditions and denitrification).

Groundwater flow velocities become important when evaluating the dilution potential of an aquifer. Dilution is often the final process relied upon to reduce concentrations of conservative solutes to an acceptable level once they enter a groundwater system. Walker et al. (1973, II) concludes that 0.2 Ha is needed as a minimum lot size necessary to reduce groundwater nitrate-N concentration to less then 10 mg/l downgradient of on-site disposal systems in sandy Wisconsin soils, by stating that "*dilution is an unacceptable part of the waste treatment system because flow patterns are often difficult to predict*". Walker et al. (1973, II) discuss a preferable concept to relying upon dilution as the final treatment process. This concept would be to consider the water table as the lower boundary of the treatment system, thereby requiring purification of the wastes in the unsaturated zone beneath the seepage bed. Admittedly, this concept seems much more "holistic" in theory but in certain soils, such as those found in the sand plain, achieving complete purification with a conventional septic system is unlikely. Pitt et al. (1975) reported that in some aquifers with high groundwater flow velocities (often associated with sand and gravel aquifers) the dilution potential can be significant. In a sensitivity analysis performed on the model formulated by Bauman and Schafer (1984), flow velocity was established as a model sensitive variable. The model indicated that in lower velocity flow systems the effects of dilution are minimal and are therefore more susceptible to appreciable contamination.

Sand and gravel aquifers are often associated with high flow velocities, Robertson et al. (1991) reports that recent studies indicate that the dispersive capabilities, and therefore the contaminant dilution potential, of many sand and gravel aquifers are much less then previously thought. The study conducted by Robertson et al. (1991) in Canada found low transverse dispersion in a shallow unconfined sand aquifer downgradient of two small septic systems. The report cites several recent natural gradient tracer experiments in sands also measuring low dispersion values (ie. longitudinal dispersivity = 1 m, vertical transverse dispersivity = 0.004 m, and horizontal transverse dispersivity = 0.01 m) as reported by (Sudicky et al., 1983; Freyburg,1986; Garabedian, 1987; Moltyaner and Killey, 1988 a and b; all cited by Robertson et al. 1991). Robertson et al. conclude that the minimum well-septic system setback distances common throughout North America should not be expected to protect well-water quality in situations where mobile contaminants such as nitrate-N are not attenuated by chemical or microbiological processes.

Another important consideration in the evaluation of the impact subdivisions may have on groundwater is the effective depth of mixing occurring beneath the subdivision. The sensitivity analysis performed by Bauman and Schafer (1984) on their model indicated that in low velocity flow systems, the effective depth of mixing had little impact on nitrate-N concentration, and had only minimal effect on nitrate-N concentration in a higher velocity flow system Harmsen (1989) compares values of average flow velocity in the sand plain, 0.3 to 0.6 m/day (1-2 ft/day) (Rothchild, 1982), and an average lot size of less than 0.4 hectare (typical of those found in the study area) to the results presented by Bauman and Schafer (1984) and concludes that mixing depth is likely a model sensitive parameter in the study area.

Data pertaining to the depth of mixing occurring under subdivisions is notably absent. Wehrmann (1983) states that groundwater beneath unsewered subdivisions possessing a large number of wells "*will be mixed quite effectively*". But as noted by Harmsen (1989), no studies supporting or contradicting this theory could be found.

Bauman and Schafer (1984) also evaluate the sensitivity of their model to background nitrate-N concentrations of incoming water and found that it had little impact on the analysis. Incoming concentrations ranging from to 7 mg/l nitrate-N had very little effect on nitrate-N concentrations in a simulated subdivision with varying lot sizes. Background nitrate-N concentrations like those common in the Village Green subdivision (>20 mg/l) reported by Harmsen (1989) would likely have made more dramatic an impact on their analysis.

Tinker (1991) evaluated groundwater from five subdivisions in West Central Wisconsin using private water supply wells. Results indicate that nitrogen from septic systems and lawn fertilizer cause nitrate-N concentrations to increase in groundwater beneath the downgradient side of the subdivisions. Three of the five subdivisions had nitrate-N levels exceeding the drinking water standard of 10 mg/l. Tinker (1991) also evaluates three nitrogen mass balance models in an attempt to identify the possible sources of nitrate-N in the subdivision wells.

In a comparison of nitrogen in shallow groundwater from sewered and unsewered areas of Long Island, New York, researchers found no significant difference existing between median nitrate-N concentrations in groundwater samples from each area (Katz et al., 1980). The authors acknowledge that the lack of significant difference between the two may have been due to sampling bias, landfills and agricultural sources, and/or residual contamination from before the area was sewered. The study did find significantly lower nitrate-N concentrations in wells screened near the watertable beneath the sewered area. The results indicated that the nitrate-N concentrations were being reduced by sewering, but that the dilution process was quite slow in the Long Island aquifer.

A more conclusive study conducted in an 80 square kilometer (30 square mile) densely populated, unsewered area in East Portland, Oregon showed a significantly higher concentration of nitrate-N in groundwater samples when compared to samples taken from surrounding sewered areas (Quan et al.,1974).

A computer program developed by Cornell University and known as the BURBS model (Hughes and Pacenka, 1985) was used by Leonard (1986) in Wisconsin to determine the minimum lot size necessary to prevent nitrate-N concentrations from exceeding 10 mg/l. The model utilizes inputs from septic systems and fertilizers and performs a detailed nitrogen mass balance. Leonard's analysis was performed on two soil types common to Wisconsin, Plainfield Sand and Grays Silt Loam. Results indicated that a minimum lot size of 0.8 Ha was necessary to achieve the 10 mg/l nitrate-N concentration. Soil type was found to have little effect on the nitrate-N concentration of groundwater. Nitrate-N concentration was found to increase with housing density but at a decreasing rate. The BURBS model estimates nitrate-N concentrations in recharge water as it doesn't account for background dilution from groundwater passing under the site.

Anderson et al.(1987) developed a contaminant transport model to assist in selecting actual subdivisions for field groundwater monitoring. Models for the mean values of input parameters and for uncertain values of the input parameters were developed and solutions obtained for typical Florida groundwater conditions. The model was determined to be a "*useful tool*" in assessing the potential impact of subdivisions on groundwater quality which would likely take many years to realize in a field monitoring study.

Septic Systems and Groundwater Quality

Septic tanks contribute more than 1 trillion gallons of wastewater to the subsurface every year (OTA, 1984). This waste originates from over 22 million septic tanks in the U.S., The above statistics make septic tank systems the leading contributor of wastewater to the subsurface and the most frequently reported cause of groundwater contamination (U.S. EPA, 1977)

With statistics like these, one would expect that research in the area of septic system performance and effectiveness, and the impacts of septic systems on groundwater quality would be common and on-going. Although there has been a good deal of research evaluating the impact of conventional systems on groundwater quality, the use of these systems still dominates in many areas even where proven ineffective.

Cogger (1988) identified three primary parts of a septic system: the septic tank, the absorption area, and the surrounding soil. Wastes enter the septic tank via a gravity feed sewer line from the household. Typically no separation of gray water (water used for laundry, bathing, etc.) from blackwater (toilet wastes) is made. Once in the tank, the heavier materials and solids will sink to the bottom of the tank where decomposition will occur, thus reducing the quantity of organic material (Reneau et al., 1989). At the effluent surface, in a properly functioning tank, a scum layer of floating material containing greases and fats will form. Decomposition will also occur here.

Water levels in the tank are controlled by an inlet and an outlet located at opposite ends of the upper portion of the tank and separated by baffles. The baffles are designed to prevent the surface scum layer and bottom sludge material from escaping. In a properly functioning system, only a semi-clarified effluent from the center of the tank is allowed to discharge to the soil absorption system (Cantor and Knox, 1985).

Reneau et al. (1989) reported anaerobic digestion in the septic tank results in a reduction of sludge volume by 40%, biological oxygen demand (BOD) by 60%, suspended solids by 70%, and conversion of much of the organic-nitrogen to the ammonium form (NH_4^+).

The clarified effluent entering the absorption area will eventually cause a build up of what is termed a "biological mat" at the interface of the absorption field and the surrounding soil (Cantor and Knox, 1985). The development of a biological mat can play an important role in effluent treatment, particularly in soils with high hydraulic conductivities. This mat, sometimes called the crust layer, is a result of clogging of soil pores with organic materials and biological growth (Brown et al., 1980; Laak et al. 1975). In permeable soils the mat serves as an effective degradative filter to suspended and dissolved organic matter and tends to enhance treatment by lengthening travel times and increasing tortuosity (Brown et al., 1980; Reneau et al., 1989).

Walker et al., (1973 I) noted that below the mat, which remains anaerobic and saturated most of the time, aerobic conditions often exist.

A problem often associated with the use of conventional septic systems in highly permeable soils is uneven distribution of effluent out of the distribution pipes. This phenomenon results in elevated loading rates to a relatively small portion of the absorption area (Reneau et al., 1989) It occurs when the vast majority of effluent entering the distribution pipe discharges in one area due to the permeability of the soils below. Cogger (1988) discusses this phenomenon and notes that new systems in coarse soils may be susceptible to localized overloading resulting in poor treatment.

Due to the elevated loading rates in specific areas, the potential for groundwater contamination increases because saturated conditions prevail. Associated with these saturated conditions is an accelerated formation of the biological mat, which will then act to decrease infiltration at that location (Reneau et al., 1989). This preferential discharge usually occurs at the beginning of a distribution trench (where the effluent first encounters perforations in the distribution pipe). As the biological mat builds up in that area the discharge will be displaced further and further down along the length of the pipe. This phenomenon is well documented and is referred to as "*creeping failure*" (USEPA,1980), (Reneau et al., 1989).

Nitrogen

Many potential chemical contaminants exist in septic tank effluent but nitrogen is often thought to represent the most serious threat to human health. Nitrogen in the form of nitrate-N represents the greatest threat because of its association with methemoglobinemia in infants and because it is very soluble and chemically inactive in aerobic environments, often resulting in virtual unrestricted mobility in soil and groundwater (Reneau et al., 1989). This mobility and the fact that many land use activities are often associated with the formation or application of nitrates are the principle reasons nitrate-N is of such concern.

The fate of nitrogen in the environment is complex. It results from a variety of physical, chemical, and biological mechanisms which in turn are greatly influenced by environmental conditions (Brown et al., 1980).

Septic tank effluent typically averages 40-80 mg N/l, of which 75 percent is soluble ammonium and 25 percent organic-N (Walker et al., 1973,II; Brown et al 1980; Reneau et al., 1989). Brown et al. (1980) goes on to state that the vast majority of the organic-N is "*sorbed and transformed*" to ammonium in the anaerobic crusted zone or mat of the absorption field.

Nitrogen leaving the anaerobic biological mat zone as ammonium and entering the soil profile is often oxidized to nitrate-N. This largely biological process, shown below, is known as nitrification (Brown et al., 1980).

$$NH_4^+ + 20_2 \xrightarrow[\text{Nitrosomonas}]{\text{Nitrosococcus}} N0_2^- + H_2O + 2H^+$$

$$NO_2^- + {}^1/_2O_2 \xrightarrow{\text{Nitrobactor}} NO_3^-$$

In a properly functioning absorption system most of the nitrogen will be converted to nitrate-N in the first few inches of the aerobic soil surrounding the absorption trench (Dudley & Stephenson, 1973; Walker et al., 1973). Oxygen diffusion into the soil zone is the most rate limiting factor determining the form of nitrogen present (Reneau et al., 1989). Environmental conditions such as moisture content below the mat can indirectly control the process by restricting soil oxygen or in extremely dry conditions may result in a reduction of bacterial populations and thus limit nitrification (Brown et al., 1980).

In an evaluation of the potential for nitrification to occur in the sandy inorganic soils of the New Jersey Pine Barrens, Brown at. al. (1980) noted that the low pH and base status of the native soils may discourage oxidation of ammonium, but then commented that the near neutral wastewater would probably increase soil pH to an acceptable range overtime. Although this may be the case, once nitrification began to occur there would likely be a subsequent decrease in pH as noted by Reneau et al. (1989) and Alhajjar et al. (1990) and discussed below.

Brown et al. (1980) also report that cooler temperatures associated with the northeastern regions of the United States may inhibit the activity of nitrifying bacteria resulting in the movement of ammonium to groundwater. However, other investigators (Viraraghavan and Warncock, 1976; Viraraghaven, 1985) found that winter conditions posed no threat to septic system operation and cited studies in Alaska where septic systems performed satisfactorily.

The primary mechanism for removal of nitrogen from soils is denitrification. Denitrification is the reduction of nitrates to gaseous nitrogen by bacteria under anaerobic conditions in the soil (Cogger, 1988). This reaction is depicted in the following equation, where CH_2O represents organic matter as a carbon source (Robertson et al., 1991).

$$\mathbf{{}^{4}/_{5}NO_3^- + CH_2O \quad > {}^{2}/_{5}N_2 + HCO_3^- + {}^{1}/_{5}H^+ + {}^{2}/_{5}H_2O}$$

A properly functioning septic system in sandy well aerated soils (such as those found in the study areas) will have minimal denitrification, and then only in anaerobic microsites (Bouma, 1979; Reneau et al.,1989).

A study conducted by Alhajjar et al. (1989) in Wisconsin evaluated the impact of phosphate built versus carbonate-built laundry detergents on groundwater quality downgradient of septic systems. The authors concluded that the use of phosphate-built laundry detergents improved the efficiency of nitrogen removal during effluent percolation through septic system drainfields and reduced the nitrate-N level in downgradient groundwater plumes without any significant effect on phosphorus concentrations. They theorize that the greater amounts of phosphorus reaching the soil from the phosphate-built detergents stimulated "*prolific growth*" of denitrifying bacteria in the clogging mat and soil, thus enhancing the removal of nitrogen.

Cogger and Carlile (1984) provided indirect evidence of denitrification around septic systems but found that it varied from one system to another, seasonally, and was most effective in wet soils which were otherwise unsatisfactory for wastewater treatment.

Denitrification may be significant in soils with restricted drainage but nitrification of ammonium must occur first, then denitrifying bacteria and a carbon source must also be present. Robertson, et al. (1991), in a study conducted in Canada, reported nitrate-N concentrations decreasing from 20 mg/l to less than 0.5 mg/l in the last meter of flow before discharging into the Muskoka River. The nitrate-N had traveled 20 meters from a septic system before "*vigorous denitrification* *occurred in the riverbed sediments as a result of anaerobic conditions existing there*". Carbon was also abundant in these sediments.

Acidity produced from the nitrification of ammonium resulted in depressed pH levels in the plumes of both systems studied by Robertson and has been noted by other investigators. A study conducted in Australia (Whelan, 1988) measured a significant reduction in pH (9.0 to 5.5) caused by the nitrification process below a soak well. Reneau et al. (1990) point out that the lowering of pH to this level could adversely affect the activity of denitrifying bacteria. Alhajjar et al. (1990) also noted a substantial reduction in the pH of groundwater impacted by septic leachate.

These data indicate that well drained soils, traditionally considered to be ideally suited for conventional septic systems, are very susceptible to groundwater contamination from nitrates due to the limited potential for denitrification. The most probable mechanism for the reduction of nitrates under these conditions is dilution by groundwater (Reneau, et al, 1989). Table 1 summarizes some relevant data from septic system studies.

Reference	System Age (yrs)	Effluent Nitrogen (mg/l)	Groundwater Nitrate-N (mg/l)	Depth to Groundwater (m)	Distance Moved (m)
Ellis & Childs *	15		8.0	1.5-1.8	100
Ellis & Childs*	8			0.9-1.2	9
Dudley & Stephenson*	5	27.1-33.8	15.5	3-4	6.1
Dudley & Stephenson*	8	27.1-33.8	2.4-20.3	4	9.1
Dudley & Stephenson*	9	27.1-33.8	13.8	17.1	0
Dudley & Stephenson*	1	27.1-33.8	2.4-11.4	7.5	0.9
Walker et al. 1973*			40	5-6	0
Walker et al. 1973*			10	5-6	70
Walker et al. 1973*			12		35
Shaw and Turyk	5-10	46-105	15-101	3-7	5-13
Virarghaven & Warncock 1976	New	77-111***	0.4	2-3	12
Rea & Upchurch (1980)	50		10	1	25
Robertson et al. 1991	12	30**	33***	2.5	0
Robertson et al. 1991	1-2	59**	39	3	0

* As cited by Brown and Associates, 1980, p.51
** Reported value of ammonia nitrogen in septic tank effluent
*** Reported background nitrate-N of 27 mg/l

Table 1. Summary of field studies of nitrate-N movement from septic systems in groundwater.

Phosphorus

Literature relative to phosphorus movement away from septic systems is less consistent then that of nitrogen. Soils appear to vary greatly in their ability to adsorb soluble phosphate ions (Brown et al., 1980). The greatest environmental concern associated with phosphorus movement away from septic systems is the eutrophication of surface water bodies (Cogger, 1988). Phosphorus is often the limiting nutrient in aquatic ecosystems. Excessive additions can cause nuisance algae blooms and enhanced growth of aquatic macrophytes, often resulting in oxygen depletion.

Phosphorus in septic tank effluent originates primarily from human wastes and detergents (Brown et al., 1980). The contribution from the latter has likely decreased in Wisconsin in recent years since the use of phosphate based laundry detergents has been restricted. However, phosphates are still a component of many non-laundry household detergents and cleaners (Shaw, 1988). Phosphate movement through most soils is limited, and seems to be controlled primarily by adsorption and precipitation type reactions (Reneau et al., 1989).

Phosphate precipitation in the soil is primarily dependant upon the pH of the soil and the presence of aluminum, iron, calcium, and organic colloids (Laak et al., 1975). Laak et al. (1975) also report that phosphorus fixation is at a minimum at near neutral pH and tends to be at a maximum at pH extremes. In soils where iron and aluminum are present (usually associated with lower pH's) phosphates can be chemically adsorbed by hydrous oxides of aluminum and iron forming an extremely insoluble gel complex (Kuo and Mikkelsen, 1979). In calcareous sandy soils such as those found in the study area, precipitation reactions with compounds containing phosphorus and calcium would likely dominate (Reneau et al., 1989) although iron and aluminum precipitation and/or sorption may also occur.

Childs et al., (1974) evaluated effluent migration away from several septic systems surrounding Houghton Lake, Michigan. The study reported phosphorus mobility equivalent to that of nitrates and chlorides in some situations while at other nearby sites very little phosphorus movement was noted. The difference in phosphorus mobility from site to site was attributed to variations in adsorptive capacity between soil types and loading rate variations.

Nagpal (1986) reported that phosphorus sorption is more affected by an increase in hydraulic loading then by phosphorus concentration in the effluent. Nagpal (1986) also suggests that measures to control hydraulic loading at any one time would be more effective at reducing phosphorus movement through the soil then controlling soil type or phosphorus concentration in the effluent. Lance (1977) also reported that phosphorus removal from effluent was proportional to loading rates. Reneau (1978; as cited by Reneau et al., 1989) suggested that a low pressure dosing system would greatly reduce phosphorus movement in some situations by achieving uniform effluent distribution and allowing the system to be placed at a shallower depth, thus maximizing the unsaturated zone.

In a recent Canadian study, Robertson et al. (1991) evaluated phosphorus movement in sandy aquifers from two septic systems. Although phosphorus concentrations in the tile effluent of about 10 mg/l PO_4-P were reported at both sites, significant subsurface attenuation was noted. At one site no detectable PO_4-P was observed in the groundwater, and the other site indicated very little attenuation in the unsaturated zone, while significant attenuation (>5 mg/l to <0.02 mg/l) occurred after several meters of flow in the saturated zone. The authors attribute the phosphate removal in the unsaturated zone at the first site (pH = 5.1, system age 4 yrs.) to sorption or precipitation with iron or aluminum. Phosphate attenuation at the second site (pH 7.0, system age 14 yrs.) was believed to be controlled by precipitation with Ca^{+2} to form hydroxylapatite ($Ca_{10}(PO_4)_6(OH)_2$) in the saturated zone (Robertson et al., 1991).

A field investigation of the efficiency of a septic system on a relatively fine textured soil (sandy loam and silty loam), conducted by Viraraghaven and Warncock

1976), reported concentrations of phosphate-P in the groundwater of 5 mg/l to 10 mg/l approximately 15 meters (50 feet) from the tile bed. The authors noted that the phosphate reduction achieved in the study was low but offered no explanation as to why. The drain tile was a new addition to an existing system so the attenuation capacity of the soil should not have been exhausted from previous loading. Near ground level water tables were noted during the spring snow melt at the study site.

Cogger (1988), in a review of literature relative to septic systems and groundwater contamination, points out that phosphate movement is usually associated with soils having limited fixation capacities and is especially prevalent around old or heavily loaded systems with shallow water tables. This is consistent with the results of a soil column study conducted by Sawhney (1977). The study concluded that soils have a finite ability to remove phosphorus if continuously dosed. Once phosphorus breakthrough occurred, increasingly larger amounts of phosphorus appeared in the column effluent. Consequently, after prolonged use of a soil, especially a soil of low sorption capacity, subsurface waters could be expected to contain high concentrations of phosphorus.

Numerous investigators have documented that phosphorus moves rather freely once it enters the saturated zone (Childs et al. 1974; Viraraghaven and Warncock, 1976; Reneau, 1979). Other studies have indicated that significant attenuation can occur in the saturated zone (Robertson et al., 1991). Table 2 summarizes some relevant data from septic system studies. The mechanisms controlling phosphorus movement will be greatly influenced by loading rates and the geochemical conditions

Reference	System Age (yrs)	PO4 in Effluent (mg/l)	PO4 in Groundwater (mg/l)	Depth to Groundwater (m)	Distance Moved (m)
Ellis & Childs, 1973 *	15		0.099	1.5-1.8	100
Ellis & Childs, 1973 *	8	11.5	11.6	0.9-1.2	9
Childs et al. 1973	10		up to 8	shallow	16
Childs et al. 1973	10		up to 8	shallow	30
Dudley & Stephenson 1973*	5	27.1-33.8	0.05	3-4	6.1
Dudley & Stephenson 1973*	8		0.65	4	
Dudley & Stephenson 1973*	9		up to 5.5	17.1	12.2
Dudley & Stephenson 1973*		13.16	0.05-0.28	7.5	18
Viraraghavan & Warncock	new	6.25-30.00	up to 5	2-3	12
Reneau 1977 *		10.8	0.01-0.55		10.4
Rea & Upchurch 1980	50		up to 5	1	8
Robertson et al. 1991	12	8	4	2	0
Robertson et al. 1991	1-2	13	0.01	3	0

* As cited by Brown and Associates, 1980, p.51

Table 2. Summary of field studies of phosphate movement from septic systems in groundwater.

existing in the unsaturated and the saturated zone. Phosphorus movement in the coarse soils of the study areas is likely, especially where heavy loading and poor effluent distribution is occurring or in old systems.

Bacteria

Bacteriological contamination of groundwater from septic systems is well documented but is not a focus of this project. For a comprehensive discussion of bacteriological and viral contamination of groundwater from septic systems refer to Yates and Yates, 1989; Yates, 1985; and Reneau et al., 1989.

Chlorides and Other Potential Contaminants

Chloride is a naturally occurring anion in surface and ground waters, which is usually present at low concentrations. It is also a common constituent in animal and human wastes, and often a component of road de-icing agents. As a result, elevated concentrations of chlorides are often indicative of contamination from man-made sources. Concentrations of chloride in septic effluent vary with human diet and with the quality of the water supply source (Alhajjar et al., 1990). Septic systems do not effectively remove chloride due to it's anionic form and conservative nature. As a result, it is often used as an indicator of contamination (Alhajjar et al. , 1990)

Alhajjar et al., (1990) statistically evaluated the use of four groundwater chemical characteristics to determine which were best suited as indicators of groundwater contamination from septic systems. Results indicated that of the four chemical characteristics evaluated (Cl^-, electrical conductivity, pH, and fluorescence) only chloride was considered a conservative tracer, and thus the best indicator. Electrical conductivity and pH were classified as semi-conservative and were only "acceptable" as indicators. Fluorescence, originating primarily from optical brighteners in laundry detergents, was considered a poor indicator of septic contaminated groundwater. The authors go on to state that "*septic systems are not sources of fluorescence to groundwater, and fluorescence is not a reliable indicator of organic pollutants in groundwater in the vicinity of septic systems*" (Alhajjar et al. 1990). However, results of this study do not support this conclusion

Lawn Studies

Since 1970, pesticide and fertilizer use on private home lawns has steadily increased (Watshke, 1983 as cited by Morton et al., 1988). With this increased chemical usage has come increased threats to surface and groundwater resources. In-ground home lawn irrigation systems are also becoming more common, especially in areas with well drained soils such as the Central Wisconsin Sand Plain. Home lawn irrigation water is often applied with little regard for the moisture status or water holding capacity of the soil, which often results in over-watering (Morton et al., 1988). Irrigation resulting in over-watering has been shown to significantly increase nitrate-N leaching (Endelman et al., 1974; Rieke and Ellis, 1974).

Petrovic (1990) reviews current literature on the fate of nitrogenous fertilizers applied to turf grass. The report concludes that the leaching of fertilizer nitrogen applied to turf grass is dependant upon soil texture, type and amount of nitrogen applied, timing, and irrigation/precipitation events. Suggested practices for minimizing the impact of nitrogen to groundwater include using irrigation water only to replace the amount of water used by plants, using slow release nitrogen sources, and avoiding fertilization and irrigation on sandy soils (Petrovic, 1990).

In a sand and gravel aquifer on Long Island, New York, Flipse et al. (1984) evaluated nitrate-N concentrations in groundwater beneath a sewered subdivision. The analysis indicated a significant regional increase in nitrate-N concentrations (0.22 mg/l/yr) over a seven year period. The principle source of this nitrate-N was attributed to fertilizers from lawns.

Gold et al., (1990) compared nitrate-N losses to groundwater from agricultural and suburban land uses. Using ceramic suction lysimeters, the study compared soil water percolate from the following land uses:

1) Urea-fertilized silage corn with a rye cover crop.
2) Urea-fertilized silage corn with no cover crop.
3) Manure-fertilized silage corn with a rye cover crop.
4) Fertilized home lawn.
5) Unfertilized home lawn.

6) Mature, mixed oak-pine forest.
7) Conventional septic system from a three person home.
8) Forested area.

All treatments were located on well drained, silty or sandy loam soils over highly permeable, stratified drift deposits of sands and gravels.

The septic system achieved an estimated dissolved inorganic nitrogen (DIN) removal of 21 percent in the septic tank and absorption area. This percentage was based on a measured nitrogen loading rate of 9.5 kg/yr (21 lbs./yr) in drainfield percolate compared with the U.S. Environmental Protection Agency (1980) estimated average of 12 kg/yr (26.4 lbs/yr) for a three person home.

The urea fertilized home lawn treatment received as much nitrogen as the urea fertilized silage corn (200-250 kg/ha/yr) but resulted in much lower nitrate-N percolate. Most of the nitrate-N flux observed in the lawn plot occurred during the spring thaw (Gold et al., 1990).

The urea fertilizer was applied to the lawn in small increments throughout the growing season. This seemed to minimize leaching of nitrogen from the root zone. However, the authors note that substantial nitrogen leaching can be expected from turf grass when nitrate-N forms of fertilizer are applied and when over-watering occurs citing Morton et al.,1990 and Rieke and Ellis, 1974.

These researchers conclude that replacing production agriculture with un-sewered residential subdivisions will not markedly reduce nitrate-N concentrations in groundwater (Gold et al., 1990).

Previous Studies in the Project Area

Harmsen (1989) evaluated the nitrate-N distribution occurring under both the

Jordan Acres and Village Green subdivisions. Nitrate-N distributions were determined via two multilevel well transects placed parallel to groundwater flow in each subdivision.

At Jordan Acres the affect of the subdivisions on groundwater quality was apparent. Elevated nitrate-N concentrations in downgradient wells were attributed to septic systems and lawn fertilizers.

The Village Green Subdivision showed less conclusively the impact attributable to subdivision activities. Nitrate-N concentrations increased with depth at this subdivision, and actually tended to decrease at the downgradient end of the subdivision. The elevated background concentrations of nitrate-N at depth was attributed to upgradient agricultural activities. The two subdivisions represent two extreme cases, one with high, the other with low background nitrate-N concentrations, but neither are atypical of the sand plain region

Harmsen (1989) also noted that spatial nitrate-N distribution appeared to be highly variable in the vertical and horizontal planes, and plumes originating in the subdivisions were vertically thin and some seemed to exhibit vertical bifurcation. Sharp concentration contrasts measured in the horizontal and vertical planes suggest that mixing associated with hydrodynamic dispersion was minimal (Harmsen, 1989).

Henkel (1992) evaluated water from monitoring wells downgradient of individual septic systems within the subdivisions for organic compounds. Results indicated that organic compounds are present in groundwater in both subdivisions, but in relatively small quantities as a result of homeowner product use and disposal practices. Several detects of VOC's were confirmed, but most were at very low concentrations. The highest concentration of a VOC detected was 21.6 ppb of 1,1,1-Trichloroethane (111-TCA). The state Preventative Action Limit for 111-TCA is 40 ppb (Henkel, 1992).

Jonas (1990) conducted toxicity tests on groundwater from subdivision monitoring and private wells using Ceriodaphnia dubia. Three wells from the Jordan Acres subdivision (1 monitoring, 2 private) and six wells from the Village Green subdivision (2 monitoring, 4 private) were evaluated. Wells which displayed elevated concentrations of nitrate-N during previous testing were selected. Results indicated that one private well from each subdivision appeared to be toxic to Ceriodaphnia. The author suggests that the results of these tests are probably more reflective of inconsistent laboratory procedures (feeding regimes and dilution water) then toxic water quality problems, but offers no clear explanation.

C. Methods

Two subdivisions in the Stevens Point area were selected and instrumented with a large number of monitoring wells during the period from 1987 to 1991 The selection of the subdivisions was based upon local private well water quality information obtained from the Environmental Task Force (ETF) at the University of Wisconsin-Stevens Point. Areas with differing upgradient land uses were selected in an attempt to 1) represent subdivisions typical of the region, and 2) evaluate the effects of subdivision land use activities relative to upgradient land use activities in the same groundwater watershed.

The following is a description of the methods, techniques and procedures used in the study.

Survey of Homeowners

During the spring of 1987, a survey was conducted of all households in both subdivisions (see Appendix C) to collect information relative to homeowner chemical usage, waste disposal patterns, and fertilizer/pesticide usage (Mechenich, et. al., 1991). The survey was conducted with the assistance of the Central Wisconsin Groundwater Center. A personal interview was also conducted with many of the respondents, at which time they were asked to sketch well and drainfield locations in their yards. Individuals interested in having monitoring wells placed in their yards were also identified at this time. Henkel (1992) summarizes some of the results of this survey.

The two subdivisions chosen for the study are part of a larger research effort evaluating impacts of unsewered subdivisions on groundwater quality. Names of property owners in these subdivisions were obtained from the Portage County Land Records office. Vacant parcels were eliminated; only those actually living in the subdivisions were included. One hundred eighty-four (184) potential participants were identified.

A two-part questionnaire was developed and is included as Appendix C. The first part, eight pages focusing on chemical use and disposal practices, was mailed to all subdivision residents. A cover letter explained the study objectives. Residents were asked to complete the questionnaire and hold it for a personal visit from researchers.

Two weeks later, residents were called to set up a personal interview. Researchers visited each home and collected and reviewed the first part of the questionnaire. They then conducted the second part of the survey, a three-page questionnaire focusing on attitudes and opinions about the causes and severity of groundwater contamination and the acceptability of potential solutions. A water sample was also taken during the home visit and analyzed for nitrate-N, chloride, hardness, alkalinity, pH, specific conductance, and corrosivity index as part of the larger research effort. Results of chemical analyses are included in Appendix A.

The residents of 21 homes refused to participate, and another 24 could not be contacted during the time frame of the study. Participation rates were 89 percent in the Jordan Acres subdivision and 70 percent in the Village Green subdivision. In total, 139 surveys were conducted.

Data analysis was conducted using the dBase III+ data base software (Ashton-Tate Corporation, Torrence, CA) and SPSS-X statistical software package (SPSS, Chicago, IL). Frequencies were calculated in quartiles for pesticide use and household chemical use. Chi-square analysis (CROSSTABS) and cluster analysis (CLUSTER) procedures were used in SPSS-X to search for significant relationships between and among questionnaire parameters.

Monitoring Well Installation and Design

Four piezometers (survey wells) were installed around the perimeter of each subdivision during the summer of 1987. The wells were constructed of 3.18 cm ($1^1/_4$ in.) PVC (polyvinyl chloride) and were fitted with 30.48 cm (1 ft.) slotted, 0.0254 cm (0.01 in.) slot size screens. The screened intervals were positioned slightly below the watertable to account for water level fluctuations while still reflecting near watertable conditions. The wells were then surveyed with respect to an arbitrary datum of 30.48 m. (100.00 ft). Surveying errors were less then 0.006 and 0.012 m. for the Jordan Acres and Village Green Subdivisions respectively (Harmsen, 1989) Water levels were then measured in the wells using a fiberglass reinforced tape with an attached popper. Local hydraulic gradient and principle groundwater flow direction were determined from this information.

Two transects parallel to groundwater flow were then established in each subdivision. Along each transect four multiport wells were installed to monitor changes in groundwater quality as water passed from one end of the subdivision to the er.

Multiport well construction was based on a design by Bradbury and Bahr (1987). The wells consisted of a 1.27 cm (0.50 in.) PVC spine surrounded by up to eight, 0.635 cm inside diameter polypropylene tubes. The tubes were attached to the PVC center spine with nylon reinforced tape. An attempt was made to screen the spine with a slotted PVC screened interval at the watertable. Each tube extended to a different depth in the aquifer and was perforated with 0.32 cm ($^1/_8$ in) holes over its last 15.25 cm (6 in.) and wrapped with a nylon fabric. This fabric served as a screen to exclude the finer textured materials from entering the well port. This well design (see Figure 2) allowed discrete samples to be taken from various depths in the aquifer. When installed in transects parallel to flow, these samples helped to distinguish between subdivision impacted water and upgradient water as the water moved from one end of the subdivision to the other. Sampling ports were placed at approximately 0.75, 1.5, 3.0, 4.5, 6.0, and 7.5 m below the watertable. The up and downgradient wells of one transect at each subdivision had additional sampling ports at approximately 9.0, 12.0, and 15 m below the watertable (Harmsen, 1989).

In the Jordan Acres subdivision the east transect contained five wells, instead of the typical four. The furthest downgradient well in this transect (E5) was located on a small knoll. The well was not constructed to account for the change in topography, causing the upper two sampling ports to be located above the watertable throughout the duration of the project. As a result, no water samples were collected from those ports. Another multiport well in the Jordan Acres Subdivision (JA-C) was located at the downgradient end of the subdivision between the two transects. Figures 3 and 4 show the basic subdivision layouts and well locations for the Jordan Acres and Village Green Subdivisions respectively.

The multiport wells were then surveyed to the same arbitrary datum as the survey wells, so all elevations were relative. From this, a more detailed flow map for the subdivisions could be constructed. Water levels were measured in the multiport wells with the use of an electric ohm meter and coaxial cable. The two leads from the circuit tester were connected to the separate wires of the coaxial cable, and the cable was inserted down the center PVC spine. When the end of the cable reached the watertable the circuit was completed and registered a deflection on the meter. The cable was then removed, and the distance from the end of the cable to the point located at well top datum was measured. This distance corresponds to the depth to water.

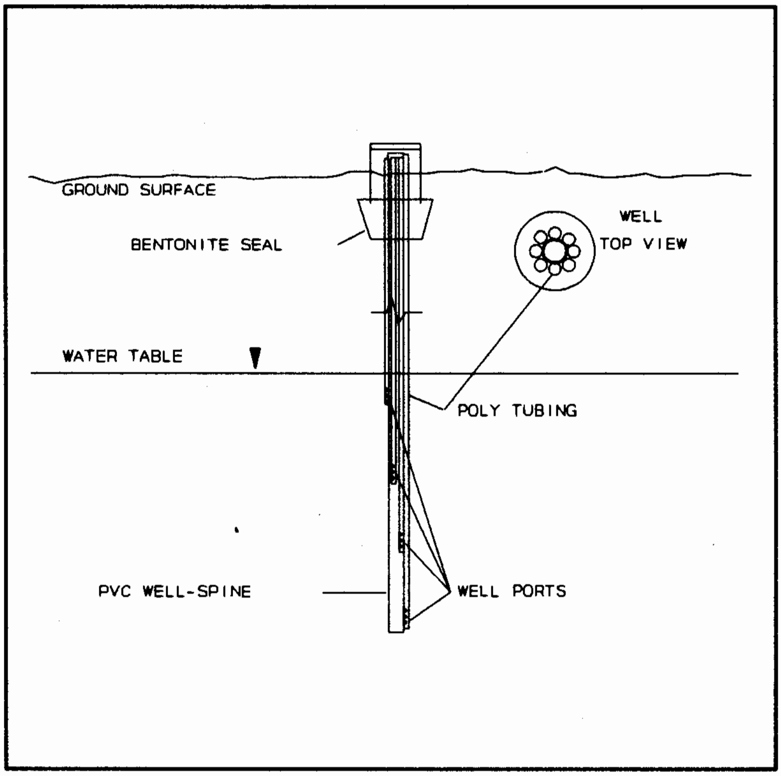

Figure 2. Original Subdivision project multiport monitoring well design.

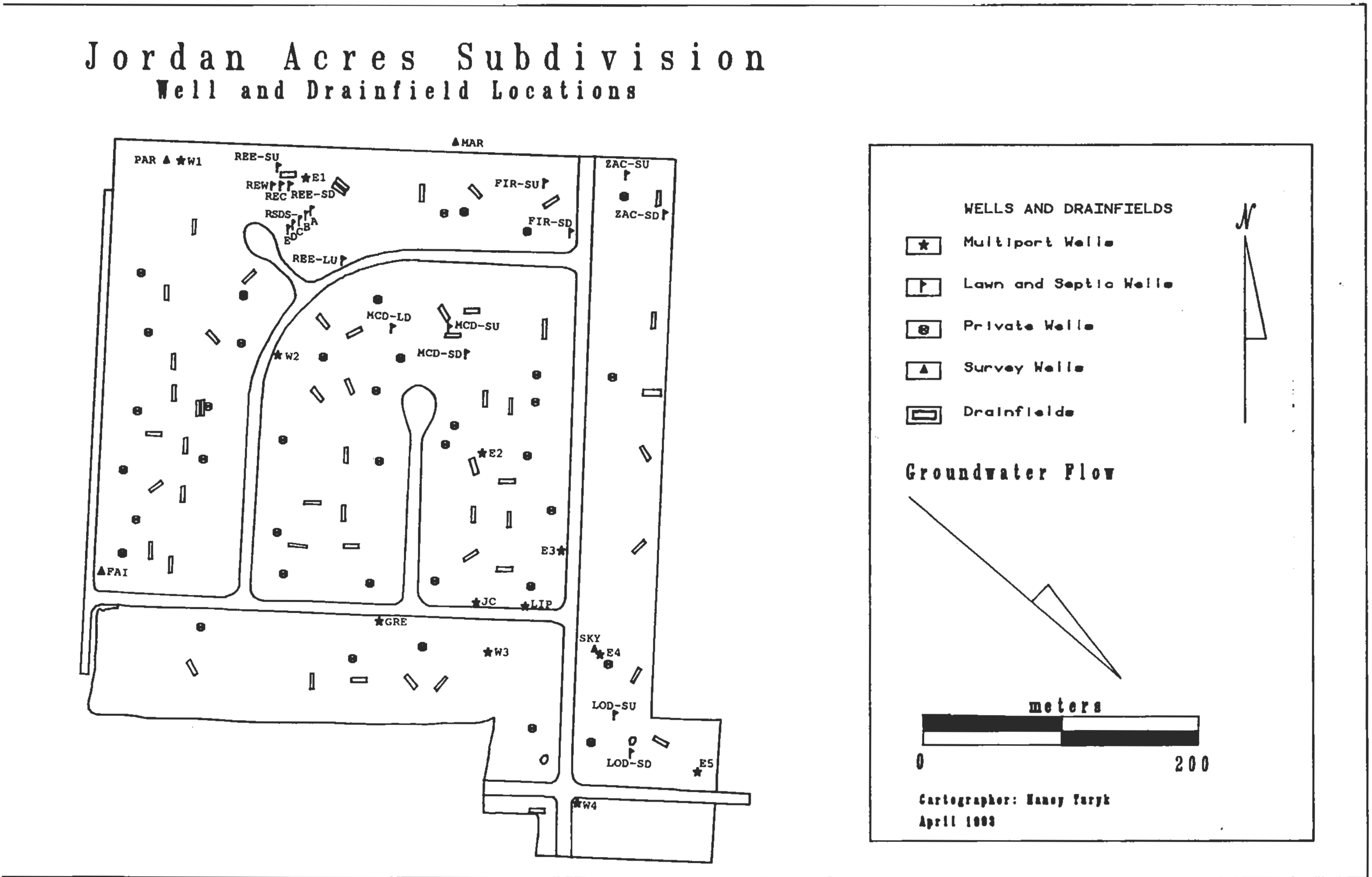

Figure 3. Jordan Acres monitoring well network, showing the location of multilevel, septic, lawn and survey monitoring wells.

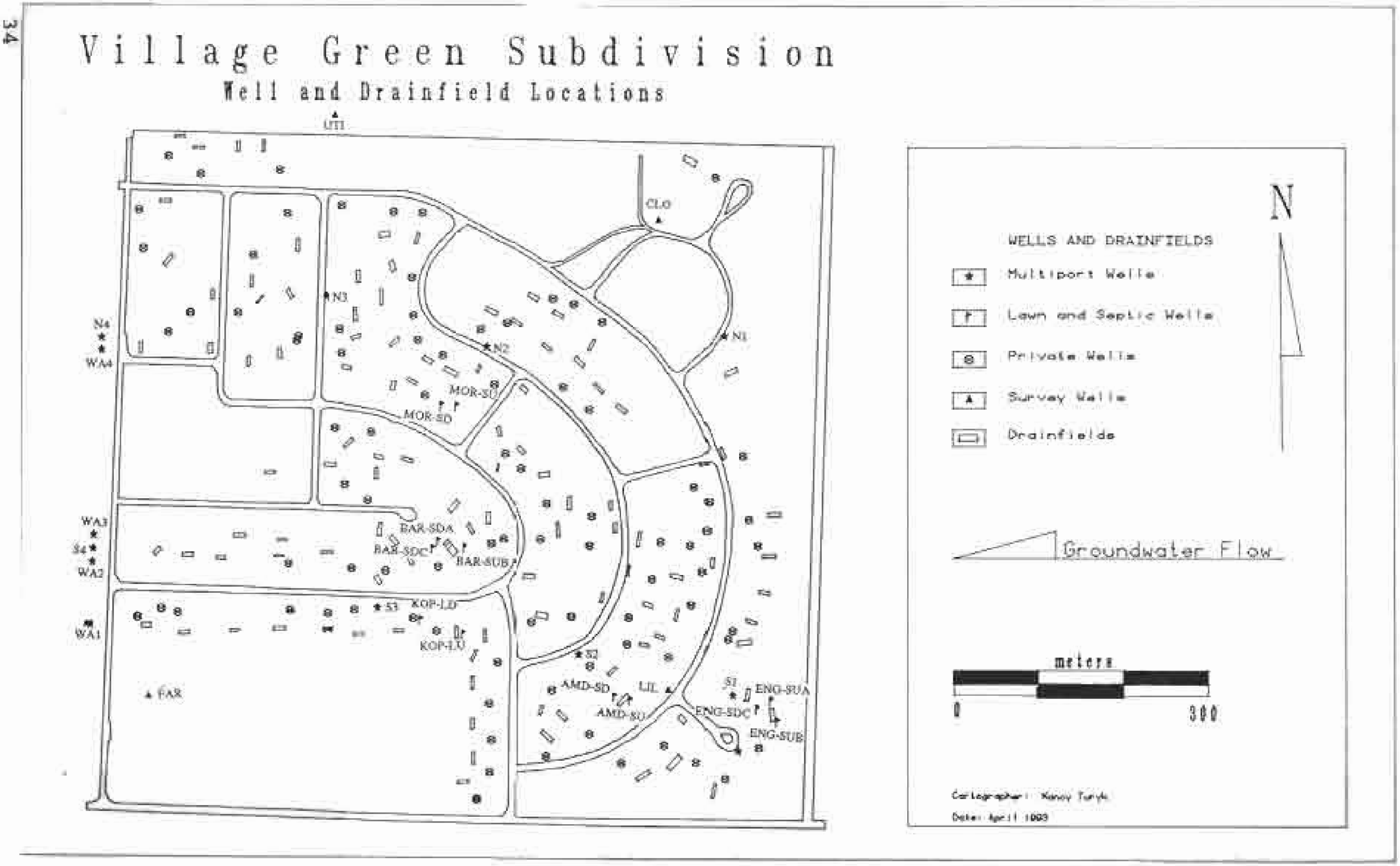

Figure 4. Village Green monitoring well network, showing the location of multilevel, septic, lawn and survey monitoring wells.

During the summer of 1988 several additional wells were installed in both subdivisions. These wells were installed in an attempt to quantify the impact individual septic systems and lawns were having on groundwater quality. This information was determined to be necessary for better estimation of the total nitrogen input for a mass balance computer model, BURBS (Hughes and Pacenka, 1985), being used for the subdivisions.

Five septic systems and one lawn from each subdivision were selected for detailed monitoring. Each septic system and lawn was instrumented with an upgradient and at least one downgradient well, with respect to groundwater flow. These wells were of similar construction to the survey wells except that the 3.18 cm ($1^1/_4$ in) PVC pipe had threaded, rather than solvent welded joints. Threaded joints were determined necessary to avoid potential VOC contamination associated with the solvent welding technique. The well screens used in the construction of these wells were also longer, 91.44 cm (36 in), and were positioned to intercept the water table in most instances. Downgradient septic and lawn wells were positioned as close to the septic drainfield or lawn as the geographic location and the homeowner would allow, generally within 6 m (20 ft).

During the summer of 1989 several more monitoring wells were installed in both subdivisions. The wells were positioned at key locations where additional water quality information was determined to be beneficial to the objectives of the study.

In Village Green five more multiport wells were installed, four in a transect perpendicular to groundwater flow at the downgradient end of the subdivision (WA-1 through-4), and one upgradient (LC) to better define incoming and exiting water quality. Figure 4 shows the location of these wells.

Two additional multiport wells were also installed on the downgradient end of the Jordan Acres subdivision. These wells (GRE and LIP) were installed to better quantify the impact the subdivision was having on groundwater quality. Figure 3 shows the location of these wells. These multiport wells were constructed in a similar fashion to the original multiport wells except that the screened intervals of the polypropylene tubes were wrapped with TYPAR rather then nylon. The wells also differ in that the center spine of 1.27 cm (0.5 in) was screened over its last one foot interval instead of a five foot section near the watertable. The wells were all approximately 21.3 m (70 ft) deep with 8 or 9 poly tube ports and the one foot screened port at 21.3 m, as shown in Figure 5.

The multiport wells were installed with the assistance of the Wisconsin Geological

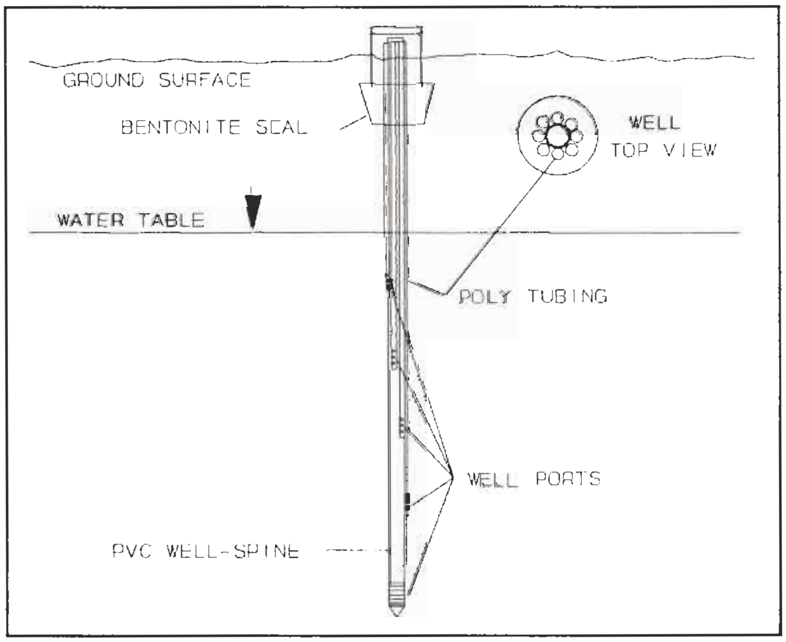

Figure 5. Design of 23 meter deep multiport monitoring wells.

and Natural History Survey crew and drill rig, a truck mounted rotary drill rig utilizing a 10.16 cm (4 in) I.D. hollow core auger. The wells were constructed at the site and were inserted into the hollow stem auger once the proper depth was obtained. The well was then used to tap out a plastic plug at the tip of the lead auger. The plug was necessary to keep cuttings from entering the hollow portion of the auger during the drilling process, and was left in the bore hole when the augers were removed. The annular space between the inside of the auger and the well was kept full of water during auger removal to prevent saturated aquifer material from surging up into the auger. Water was obtained from nearby private wells at the Jordan Acres well sites, and at the upgradient site in Village Green. A separate 5.08 cm (2 in) well was installed to supply water at the downgradient sites in Village Green. This well (WLR) was screened with a 91.44 cm (36 in) slotted (0.0254 cm) screen which was positioned approximately m below the watertable. A Stevens model water level recorder was later installed at this location to continuously monitor watertable fluctuations. As the auger was removed from the bore hole, the aquifer material collapsed inward around the well up to the watertable. The bore hole was back-filled with sand removed during the drilling process from the watertable to within 1-2 m of the surface. The last 1-2 m of the bore hole was sealed with a powdered bentonite clay.

Once installed, the wells were protected by driving a 1 m long, 15.25 cm diameter galvanized steel culvert down around them. Typically 0.3 meters was left protruding above ground level and the culvert was secured with a locking cap.

In addition to the above mentioned multiport wells, two nested wells (REC and

REW) were installed at a septic study site (REE) in Jordan Acres during the summer of 1989. The wells were installed downgradient of a septic system which had been instrumented with up and downgradient wells the previous summer. Water samples from the initial wells had shown little difference between the septic up and septic downgradient water chemistry. This was the case for four of the five septic system monitoring well sites at Jordan Acres. This site was selected for additional monitoring because of its location on the upgradient end of the subdivision, homeowner cooperation, and ample space for the installation of more wells. These two wells (REC & REW) were installed in an east-west transect with the existing downgradient well, 4.9 m (16 ft.) away from and parallel to the downgradient edge of the drainfield, as shown in Figure 6. It was believed that these wells would show whether or not preferential percolation was occurring out of this system, or if strong vertical flow components were transporting contamination deeper into the aquifer and below the existing monitoring well.

These wells were of a different design then any of the wells installed in the subdivisions to this point . The wells consisted of three 1.91 cm ($^3/_4$ in) PVC pipes taped together with nylon reinforced tape. The threaded joint pipes were screened with 30.48 cm (1 ft) slotted, 0.025 cm (0.10 in) slot size, PVC points. The screens were positioned at 15.24 cm (6 in) intervals, with the lower portion of the uppermost screen being placed at the watertable, as shown in Figure 7. This well design proved very effective at accounting for seasonal watertable fluctuations and changing plume configurations.

During the summer of 1990, five more multilevel monitoring wells were installed

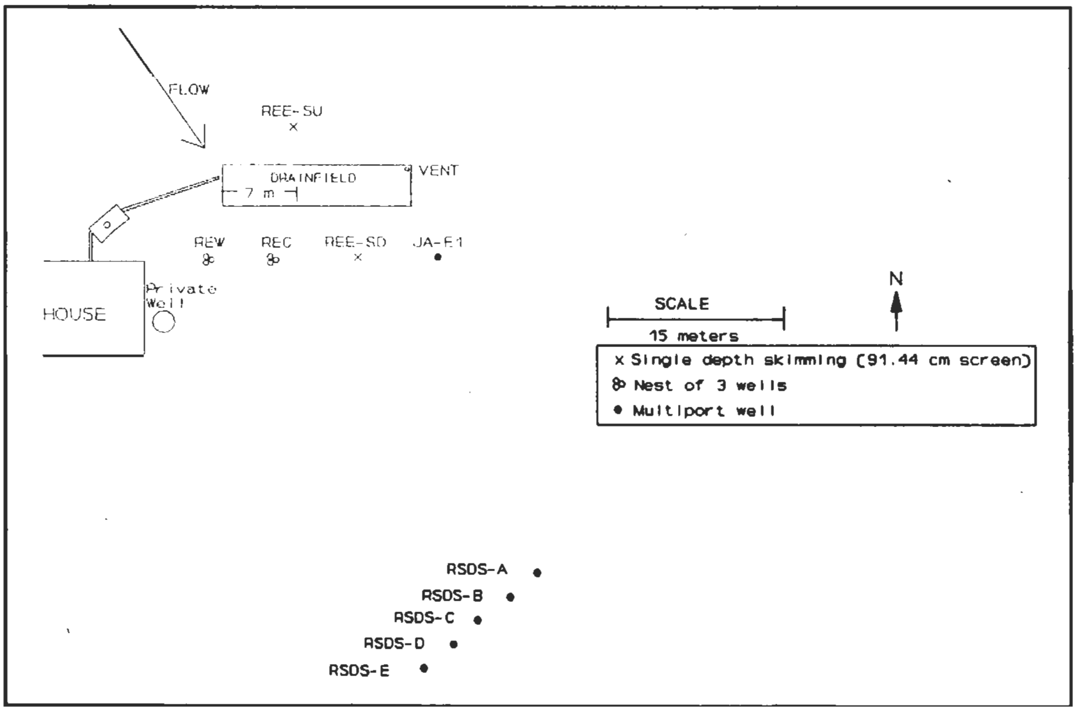

Figure 6. Location of wells at Jordan Acres septic study site REE.

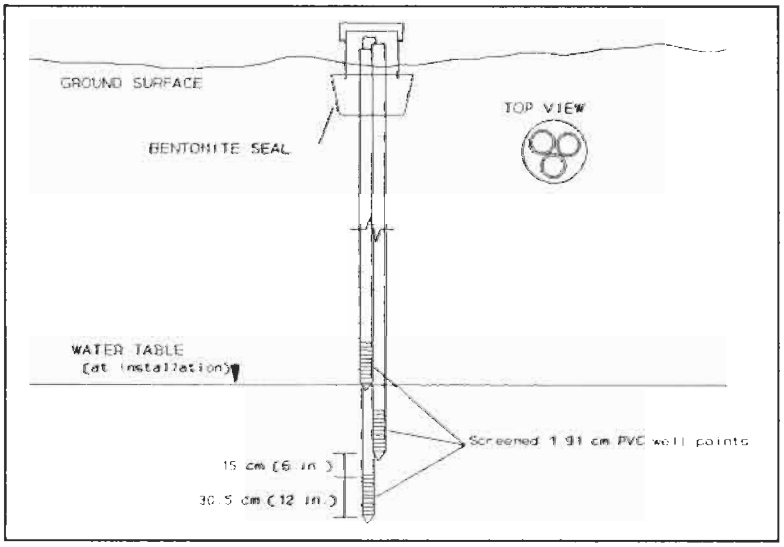

Figure 7. REC and REW well design, includes shallow, medium and deep ports, located 4.6 m downgradient of the site REE drainfield.

at site REC. These wells were installed in a transect perpendicular to groundwater flow, with well "B" being positioned 33.5 meters (110 ft) downgradient of well REC, with 3.05 m (10 ft) of separation between each of the five wells as shown in Figure 6. The wells were constructed similar to the multiport wells except a 1.91 cm ($^3/_4$ in) spine was used to allow water-level measurements to be made with a tape and popper. As with the multi-ports, the spine was screened over its last 0.3 m (1 ft) interval with a 30.48 cm (1 ft) slotted point with 0.025 cm openings. The polypropylene tubes were perforated and screened with TYPAR over a 25.4 cm (10 in) section at the bottom of each tube. Four of the wells (A,C,D,E) have five sampling ports, including the spine, at 30.48 cm (1 ft) intervals. This equates to 5.08 cm (2 in) separations between the screened intervals. The upper most screened interval was positioned at or just below the watertable, so the wells were capable of sampling the upper 1.5 m (5 ft) of the aquifer at 30.48 cm (1 ft) intervals over a 12.2 meter wide transect as shown in Figures 6 and 8. Well "B" had two additional sampling ports as shown in Figure 8.

During the summer of 1991 one additional well (KEP) was installed in the Village Green subdivision. The purpose of this well was to determine if saturated zone attenuation of phosphorus and fluorescence was occurring and to evaluate the nitrate-N:chloride ratio in the plume at this location. The well was constructed similar in style to the above mentioned RSDS wells except a 3.17 cm ($1^1/_4$ in) spine was used with a 91.44 cm (3 ft) slotted screen having 0.025 cm (0.10 in) openings. Three polypropylene tubes were perforated over 15.24 cm (6 in) intervals and wrapped with TYPAR fabric. These screens were positioned at intervals of 15.24

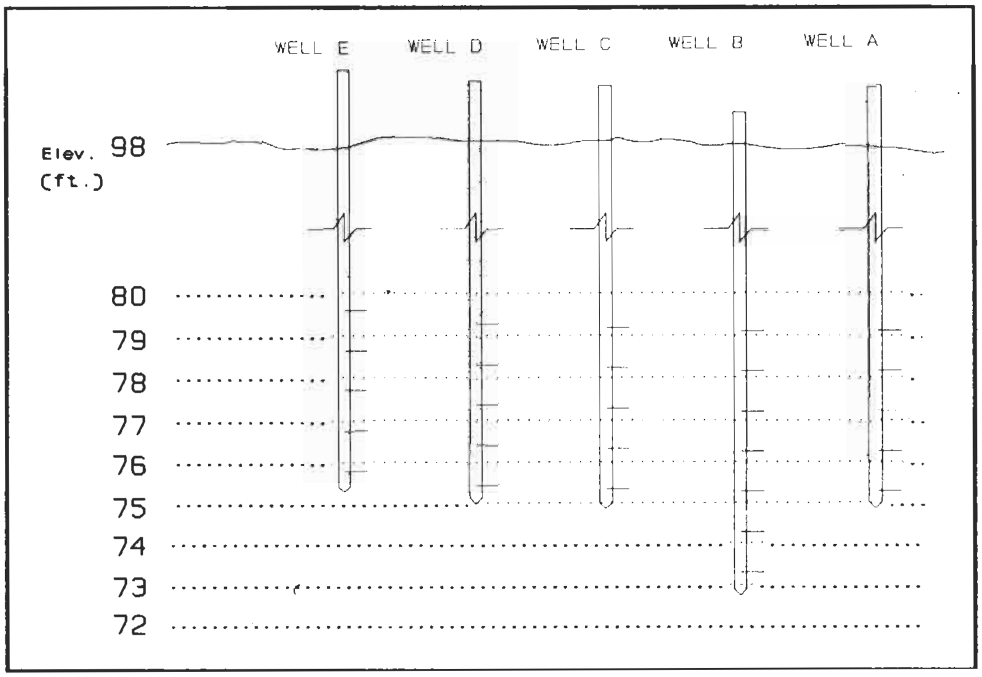

Figure 8. Cross sectional view of RSDS wells, view is from down to upgradient. Wells are located 38 meters downgradient of the drainfield at site REE. Hash marks represent the center of the 30.5 cm sampling interval.

cm as shown in Figure 9. The well was installed with a bucket auger 29 m (95 ft) downgradient of the septic drainfield vent as shown in Figure 6.

The multiport sampling wells described above required very little well development before sediment free samples were produced. Due to the small well volumes, these wells also tended to purge quite rapidly even at low pumping rates.

The PVC wells were typically developed with a large peristaltic pump or with a gasoline powered impeller-type pump. A hose attached to the pump was then surged up and down in the well in an attempt to remove or displace the finer textured formation deposits. The well was assumed to be developed when this process produced sediment-free water.

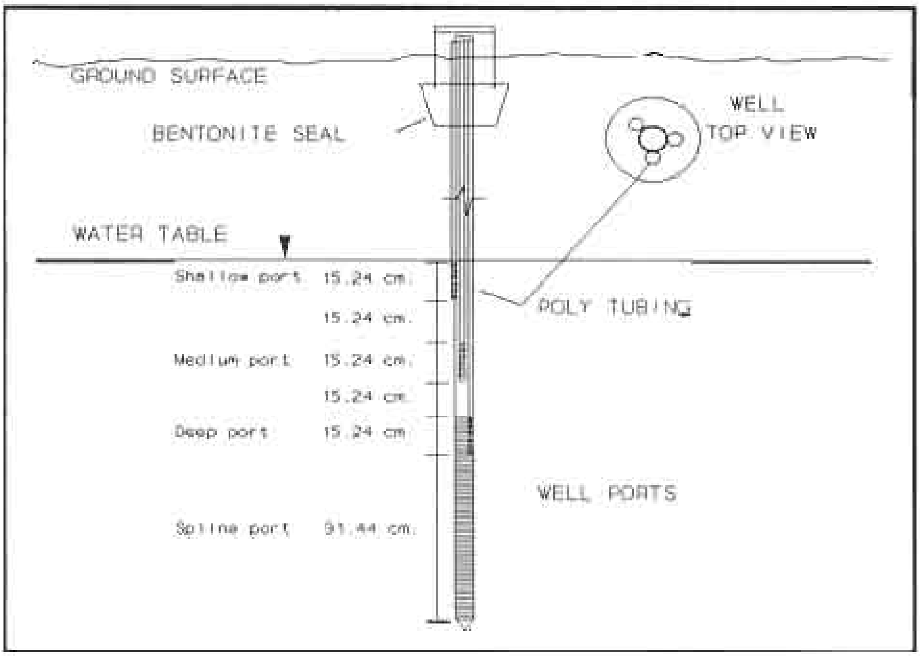

Figure 9. Design of KEP well, downgradient of site BAR in Village Green.

Groundwater Sample Acquisition

The peristaltic pump used to obtain groundwater samples was a Cole-Parmer, dual-headed, 12-volt DC electric pump. The pumping lines (the only wetted part) were silica tubing.

The multiport wells were sampled by attaching one of the pump's influent lines directly to the individual tubes, then withdrawing the water by vacuum. Because the pump had two separate pumping heads, two wells were frequently pumped at the same time. To sample the other types of wells, a length (or two) of 0.64-cm (¼-in) O.D. polypropylene tubing was lowered into the well, and the sample was withdrawn with the pump. The wells were purged prior to sampling by removing at least three times the volume of the well, or until constant temperature and conductivity readings were obtained.

Field pH and conductivity measurements were obtained by directing the pump effluent into the appropriate measurement container. The water was allowed to flow over the instrument's detector until a constant reading was obtained, at which time the value was recorded in a field notebook.

After the pH and conductivity measurements were obtained, the samples were filtered. Filtering was accomplished by using a Gelman in-line filtering cartridge and 0.45 micron filters. At least 200 ml of water was allowed to pass through the filter prior to obtaining the sample. The filtered sample was discharged directly into a 250 ml Nalgene sample bottle or other suitable sample container.

Samples for trace organic analysis were collected from monitoring wells by using a Teflon bailer after the well was purged with a peristaltic pump. The bailers were made of 1.5-m (5-foot) lengths of 2.54-cm (1-in) diameter Teflon or Schedule 40 PVC with a ball check-valve in the bottom. The bailer was lowered into the well using a length of nylon rope. Three times the well volume was purged prior to obtaining the sample. Samples from multilevel wells were collected using a peristaltic pump. All samples were kept on ice until delivery to the ETF lab.

Inorganic Chemical Analysis

Groundwater sample analyses were performed by the ETF lab at the University of Wisconsin-Stevens Point (Wisconsin lab certification #750040280).

Nitrate-N, chloride, and reactive phosphorous were analyzed using a Technicon Autoanalyzer. Nitrate-N analysis used a sulfanilamide complex read at 520 nm (Method No. 158-71W/A). Chloride analysis used a ferricyanide ion read at 480 nm

(Section 407D, APHA, 1985). Reactive phosphorous analysis used a phosphomolybdenum complex read at 880 nm (Industrial Method No. 329-74 W/B).

Sodium analyses were performed using a Varian AA475 Atomic Absorption spectrophotometer read at 589.0 nm.

Analyses for alkalinity and total hardness were performed using techniques described in Standard Methods for the Examination of Water and Wastewater (APHA et al., 1985).

Relative fluorescence was measured using a Baird-Atomic Fluoripoint. The excitation scan was set at 355 nm and the emission was set at 425 nm.

The pH and specific conductance were measured in the field using a Corning electrode meter (pH) and a YSI conductivity cell.

Organic Chemical Analysis

The groundwater samples collected from the potable, irrigation, and monitoring wells were analyzed in the University of Wisconsin-Stevens Point, ETF laboratory. The groundwater samples were analyzed for some or all of the analyte groups listed below.

Volatile organic compound (VOC) analysis was performed using EPA Methods 5030/601-602. This is a purge and trap extraction method, utilizing a photoionization detector (PID) with a 10.6 eV lamp and an Hall electrolytic conductivity detector (HECD) set in halogen mode. The detectors were set up to run in-series, with the HECD following the PID.

Polynuclear aromatic hydrocarbon (PAH) analysis was performed using the high pressure liquid chromatographic (HPLC) method in EPA Method 610. The HPLC system consisted of an automated sample injection system, a temperature controlled reverse phase column, and an ultraviolet (UV) detector and florescence detector in series.

Semi-volatile organic analyses were performed on several of the groundwater samples. An electron capture detector (ECD) was used to screen groundwater samples for semi-volatile organic compounds. A thermoionic specific detector (TSD) was used to screen groundwater samples for semi-volatile organic compounds that contain nitrogen and phosphorous. The samples for both analyses were extracted following EPA Method 608, and analyzed by gas chromatography. The sample was injected into the gas chromatograph and split between two columns, each going to a separate detector. A temperature program was used to aid in compound resolution.

D. Survey of Homeowners Chemical Use and Attitudes (Condensed from Mechenich et. al., 1991)

Introduction

Questions about the effects of unsewered residential areas on groundwater quality are being raised by groundwater planners and regulators in Wisconsin and many other states. To make good decisions about potential impacts, more information is needed about the activities of those living in these areas, such as lawn fertilization and household chemical use and disposal practices.

A number of studies documenting groundwater pollution problems from unsewered subdivisions were reviewed by Bicki and Brown (1991). Most studies they reviewed reported that a minimum lot size of 0.2 to 0.4 Ha (0.5 to 1 acre) was needed to prevent nitrate-N contamination of groundwater. However, they also noted that in some areas even larger lots were inadequate to prevent contamination. These lot sizes were based on needed separation of onsite waste disposal systems.

Nitrate-N contamination of groundwater from fertilizer was not specifically addressed. However, several authors have reported significant leaching of nitrate-N from fertilized turf grass (Morton et al., 1988; Owen and Barraclough, 1983; Rieke and Ellis, 1974). The recommended minimum lot sizes also did not account for potential effects of pesticides used on lawns and gardens, or volatile organic or other toxic compounds found in household cleaning and maintenance products. Cleaning products used in homes often contain solvents, disinfectants, and other potentially hazardous compounds. Commonly used products such as laundry detergent, toilet bowl cleaner, and tub and tile cleaners may contain a variety of chemical compounds classified by the U. S. Environmental Protection Agency as priority pollutants (Hathaway, 1980).

Many factors influence the extent to which use of these products by residents of unsewered subdivisions represent a hazard to groundwater quality. These include the chemical composition of the product, the volume used, and the method of disposal in addition to soil and aquifer attenuation potential. Volatile organic compounds disposed of in onsite sewage disposal systems have been reported to have reached groundwater by several researchers (Tomson et al., 1984; Kolega et al., 1986).

This report, part of a larger research project on the effects of unsewered subdivisions on groundwater quality, details chemical use practices and attitudes about groundwater contamination and management in two subdivisions in Central Wisconsin; Jordan Acres and Village Green Estates.

The two subdivisions are located in Portage County, in the northern portion of the Central Wisconsin sand plain (Figure 1). Jordan Acres is located about 5.2 km northeast of the city of Stevens Point, and Village Green is about 2.6 km southeast. The average age of the homes in the two subdivisions is 15 to 16 years, with the first homes being built in the 1960s. Jordan Acres had 64 developed lots, with an average lot size of 0.2 Ha (0.6 acres). The average value of homes in Jordan Acres in 1990 was $58,000, with a range of $38,000 to $86,000. Village Green had 136 developed lots with an average size of 0.16 Ha (0.4 acres). The average value of homes was $62,000, with a range of $47,000 to $123,000.

The geologic setting and groundwater pollution potential for both subdivisions is similar. A sand and gravel aquifer underlies both subdivisions to a depth of approximately 26.2 m (80 ft), with a water table depth of 6.6 to 8.2 m (20 to 25 ft). However, contaminant sources upgradient of the two subdivisions are somewhat different. A small amount of agricultural activity occurs upgradient of Jordan Acres, whereas much of the land upgradient of Village Green is intensively irrigated agricultural land, used primarily for potato production.

Within the two subdivisions, groundwater contamination problems have already occurred. The average nitrate-N concentration in private wells tested in Jordan Acres from 1976 to 1988 was 6.8 mg/l; in Village Green, .3 mg/l. Village Green had 16 samples exceeding 20 mg/l during that time period (Environmental Task Force, 1989).

Objectives

The objectives of the homeowner survey were to:

1) characterize the amounts and variety of products used for household cleaning and maintenance, and lawn and garden care, in two unsewered subdivisions;

2) evaluate the hazard to groundwater from use of these products, including their intrinsic hazards and the hazards caused by use or disposal practices, and to provide data to researchers siting monitoring wells;

3) collect nitrogen loading data for use by other researchers in a mass balance model;

4) understand how residents view the causes and severity of groundwater contamination in their county and neighborhood, and how they might respond to various solutions;

5) examine the relationships between residents' beliefs about groundwater contamination and chemical use practices; and

6) evaluate areas of greatest need for educational efforts.

Survey Results and Discussion

Chemical use data obtained from the surveys are grouped by uses; household cleaning products, maintenance products, and lawn and garden chemical use. Following discussion of each group, relationships between the groups are examined. Attitudes and opinions about groundwater protection are then discussed.

Household Cleaning Products Use

Commonly used products such as laundry detergent, toilet bowl cleaner, and tub and tile cleaners may contain a variety of chemical compounds classified by the U.S. Environmental Protection Agency as priority pollutants. One objective of this survey was to characterize the types, amounts, and variety of products used for household cleaning and maintenance in the subdivisions. Participants were asked to specify, by brand name, the products used in their household for bathroom and kitchen cleaning, laundry care, and septic system maintenance. They were also asked to specify the frequency of use. These products have a high probability of ending up in the septic tank through normal use.

Only one statistically significant difference ($p < 0.05$) was found in use rates between the two subdivisions. Bathroom rust and lime remover was used significantly more often in the Village Green subdivision than in Jordan Acres. This may be related to the differences in total hardness of water between the two subdivisions. Samples from Village Green averaged 165 mg/l total hardness, reported as $CaCO_3$, with some values as high as 250 mg/l. In Jordan Acres, total hardness averaged 108 mg/l as $CaCO_3$, with a maximum of 140 mg/l. Iron concentrations are

Product	Avg number of uses/month in one home	Range	Number of users	Percent using	Number of uses/month
Drainfield root killer	0.08	0.08	1	<1	
Laundry rust remover	0.21	0.08-0.33	2	1	0
Drain cleaner	0.54	0.03-4.0	45	34	24
Carpet cleaner	0.68	0.08-4.0	18	13	12
Septic system additives	0.98	0.08-4.0	18	13	17
Bathroom rust/lime remover	1.35	0.08-4.0	19	14	24
Bathroom floor cleaner	2.50	0.08-8.0	97	72	237
Chlorine bleach	3.28	0.17-30	72	54	265
Kitchen floor cleaner	3.51	0.25-60	92	69	316
Garbage disposal cleaner	3.57	1.0-10.0	7	5	25
Grease cutting spray	3.60	0.17-15	53	40	184
Toilet bowl cleaner	4.19	0.25-30	110	82	453
Bathroom spray cleaner	4.20	0.17-15	101	75	417
Powdered laundry sanitizer	4.61	0.17-12	6	4	28
Kitchen cleanser	4.65	0.37-30	92	69	419
Bathroom cleanser	4.99	0.5-30	96	72	469
Powdered bleach	6.83	0.33-40	35	26	232
Spot remover	7.13	0.5-20	41	30	277
Laundry detergent	15.66	2-60	114	85	1754

Table 3. Number of users and average use rates for household cleaning products in two Portage County subdivisions.

not a significant problem in either subdivision. Despite the difference in water hardness, use of other cleaning products was not significantly different, so cleaning product use data for the two subdivisions was reported together.

Some products were used frequently by those who reported using them. Laundry detergent was used an average of 15.7 times per month, followed by bathroom cleanser, used an average of 5.0 times per month (Table 3). Other products, although used slightly less frequently, were also used by a large number of participants. For example, toilet bowl cleaner was used by 110 participants (82%), and bathroom cleanser was used by 96 participants (72%). Laundry detergents, toilet bowl cleaners, and bathroom cleansers are the top three products used by homeowners.

Large use ranges were observed for many products. For example, people reported using chlorine bleach anywhere from twice a year to every day. Such wide variations make generalizations about use rates difficult.

Another way of evaluating household product use is to categorize users by quartiles as high, medium-high, medium-low, or low users. High users were those using household cleaning products 54 times per month or more; medium-high, 35-54; medium-low, 26-35; and low, less than 26 uses per month. Subtracting use of laundry detergent from the totals, high users were those using household cleaning products 34 times per month or more; medium-high, 22-34; medium-low, 14-22; and low, less than 14 uses per month.

To provide a clearer picture of household chemical use, the total number of uses per month was calculated for each product. This illustrates that some products (root killers, rust removers) are used infrequently by only a few people. Others, such as laundry detergent and bathroom cleanser, are used often by the majority of participants. However, some of the products used infrequently, such as septic system additives and wood cleaners, may be intrinsically the most hazardous.

The numbers of bathroom and kitchen cleaning products used ranged from two to nine, with most users listing four to six products as the typical number used. Cleaning frequencies for these rooms average once per week, but some reported cleaning daily.

These data were used to help design a monitoring strategy for priority pollutants in groundwater under the subdivision, both for individual homes and in the aggregate. In addition, an educational strategy presenting subdivision residents with information about the most hazardous products and safe, effective alternatives, with emphasis given to those most frequently used by large numbers of people, may reduce the risk of groundwater contamination.

Household Maintenance Products

Information on use of wood oils and cleaning products, paint thinner and strippers, car maintenance products, and "others" were also gathered. These products do not commonly enter septic systems through use, but may be improperly disposed of there. They may also be disposed of on the ground, and could contribute to groundwater contamination in that way.

Both frequency of use and number of users are lower for this category of products (Table 4) However, the method of waste disposal may be a significant concern. Paint thinner, paint stripper, and oil were all reported to have been disposed

Product	Average number of uses/month	Max	Number of users	Percent Using
Paint thinner	.87	4	25	18
Paint or varnish	.66	1	7	5
Paint	.56	2	43	31
Motor oil	.66	2	49	35
Antifreeze	.32	1	15	11
Metal cleaners	.77	1	4	3
Wood oils	2.10	4	13	9
Wood cleaners	.78	4	12	9

Table 4. Use of selected maintenance products in two Portage County subdivisions.

of in the septic system or the yard (Figure 10). However, it appears that most oil waste and at least half of paint thinner and stripper is disposed of through means not directly linked to the subdivision groundwater system

Educational efforts in this category should focus on proper disposal practices for hazardous products

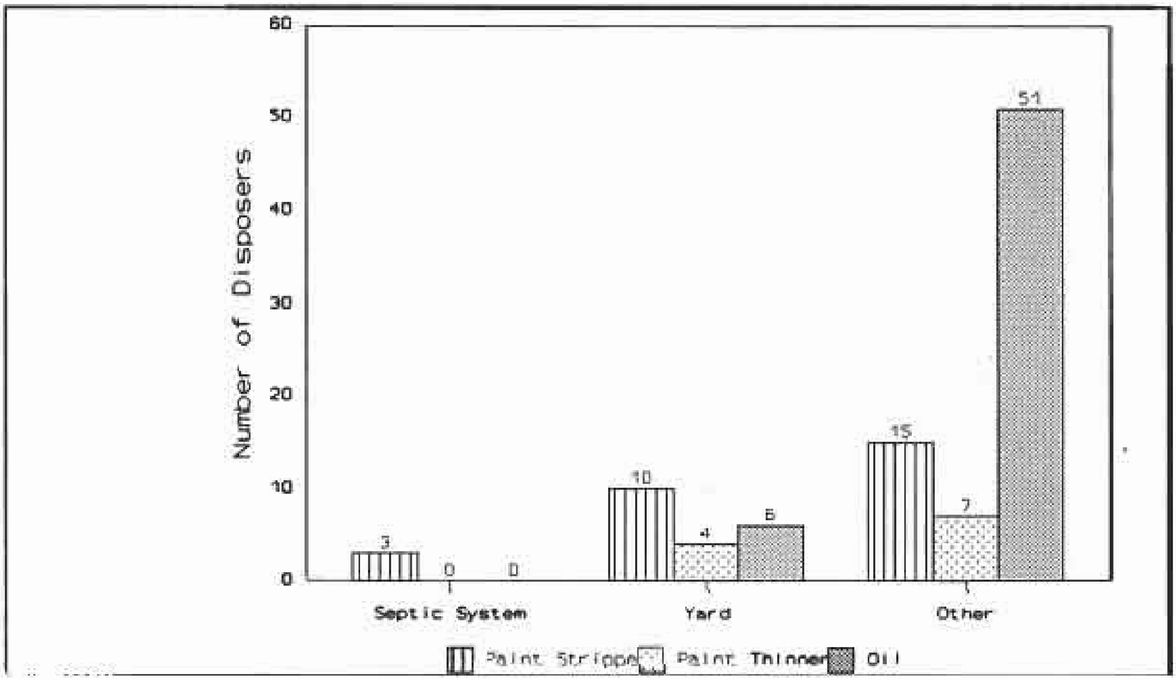

Figure 10. Number of participants reporting various disposal practices for paint thinner, paint stripper and motor oil in two Portage County subdivisions.

Lawn and Garden Chemical Use

Lawn fertilization and pesticide use on lawns and gardens are also potential threats to groundwater quality in subdivisions. Another objective of this survey was to characterize the frequency and volume of lawn and garden chemical use in these subdivisions. Questions were asked about homeowner applied and commercial applicator applied fertilizer and pesticides. In this section, comparisons are often made between overall use rates, which include all survey participants, and the use rates of "users", or those who actually reported using the product being discussed.

Nine participants (6%) reported never fertilizing their lawns and never having them fertilized by a lawn service. Most people reported fertilizing their own lawns once or twice a year. The mean fertilization rate for the subdivisions overall was 1.6 times per year, (1.8 times for users) with a range of once every five years to four times per year (Figure 11). Seventy-four percent stated that they use the amount specified on the bag when fertilizing; 18 percent reported using more. Only two participants reported not reading the bag at all when applying fertilizer. This data is used later in the report for input to the mass balance model. Seventy-two percent of users reported using a fertilizer with a nitrogen content of 26 percent or greater. Thirty-five percent reported using a slow-release nitrogen fertilizer, but 50 percent did not know if their fertilizer was of this type. Forty-nine percent reported using a mixture of broad leaf weed killer and fertilizer (weed and feed) on their lawns. The average use rate was 0.8 times per year overall, with an average use rate of 1.2 times per year reported by users. Thirty-one participants (22%) reported never using this product, while the 68 users reported frequencies of use from once every five years to three times a year. Crabgrass killer was applied an average of once per year by 31 users (22%), with a range of once every five years to twice annually. The overall average use rate (including nonusers) was 0.3 times per year.

Application frequencies for fertilizer reported by fertilizer users were not significantly different ($p < 0.05$) between the two subdivisions (1.6 per year for Jordan Acres and 1.8 per year for Village Green). However, the overall use rate (including nonusers) for the two subdivisions was significantly different (1.3 per year for Jordan Acres, 1.7 per year for Village Green) ($p < 0.05$). This difference occurs because of the nine non-users of fertilizer, six live in Jordan Acres, accounting for twelve percent of Jordan Acres participants. In Village Green, only three percent of participants do not fertilize. The same relationship (non-significant differences for users but a significant difference overall) was observed for broad leaf weed killer (weed and feed). No significant difference was found for crabgrass control products.

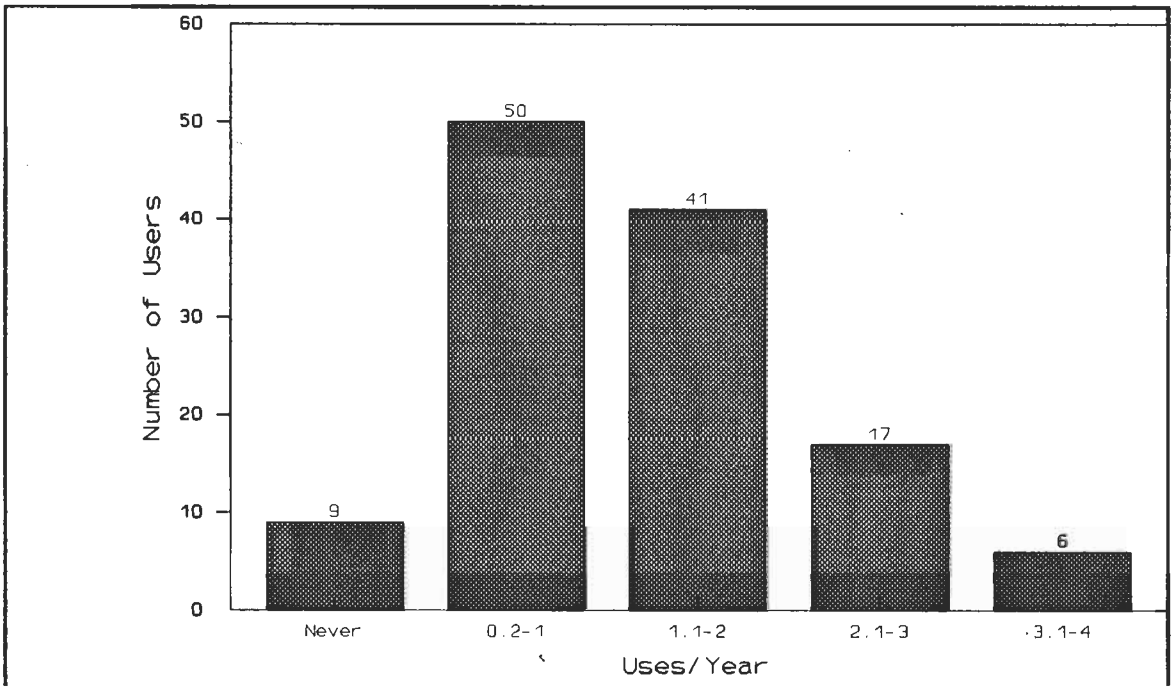

Figure 11. Frequency of lawn fertilizer use in two Portage County subdivisions.

Other common lawn care practices included mowing the lawn once per week (69%), with 14 percent mowing more frequently. Sixty-six percent removed lawn clippings after mowing. Forty percent watered their lawns an average of once a week during the growing [illegible], while 13 percent reported never watering (Table 5)

	Waters once per week or more	Mows once per week or more (%)	Removes lawn clippings (%)
Jordan Acres	59	68	48
Village Green	79	91	76
Combined	71	83	66
	Fertilizer applications/yr (%)	Uses Weed and Feed (%)	Uses Insecticides (%)
Jordan Acres	1.6	44	54
Village Green	1.8	52	47
Combined	1.8	49	50

Table 5. Lawn care practices reported in two Portage County subdivisions.

Relationships were apparent between various lawn care practices. For example, 24 percent of those who fertilize more than twice a year mowed their lawns more than once a week, while none of those who never fertilized mowed their lawns that frequently. Over 80 percent of those who fertilized more than twice a year removed their lawn clippings, compared to 44 percent of those who never fertilize. All three participants who water their lawns daily fertilize more than twice a year, while the majority of those who never fertilize, never water either. Statistically significant relationships ($p < 0.05$) were found between lawn fertilization frequency and mowing frequency, removing clippings, and watering frequency.

Cluster analysis indicates that lawn care practices can be divided into two groups. The first group, which used less fertilizer, was also likely to mow less frequently, was less likely to remove clippings, and watered their lawns less often than those in the second group.

Only ten participants reported using a commercial lawn care service. Of those, three reported that the service never applied any lawn chemicals, including fertilizer. These may have been strictly lawn mowing services. Of the remaining seven, two reported monthly fertilizer application, and two reported semi-annual application, with the other four giving no response. A total of seven reported use of herbicides, with applications of three twice a year and four once a year. Three reported application of insecticides; two twice a year and one once a year. Only one participant reported the use of fungicides.

Study participants reported to use insecticides less frequently than other lawn and garden chemicals. The most commonly used insecticides were diazinon (used by 51 participants), malathion (used by 16), and carbaryl (Sevin) (used by 17). Most reported using small amounts (less than one cup of undiluted product per year) but some used more than 10 cups per year (Figure 12).

Of the insecticides chosen by subdivision residents, diazinon is reported to have a medium potential for leaching to groundwater, and carbaryl and malathion have a low potential (Becker et al, 1990). From 1983 to 1987, the Wisconsin Department of Natural Resources pesticide monitoring report shows that five of 230 sampled wells contained detectable levels of carbaryl; none of four sampled wells contained malathion; and none of 27 wells contained diazinon (WDNR, 1987). Pesticide mixing and disposal practices in the subdivisions were not specifically surveyed, but there may be some potential for groundwater contamination from these practices as well as from routine use.

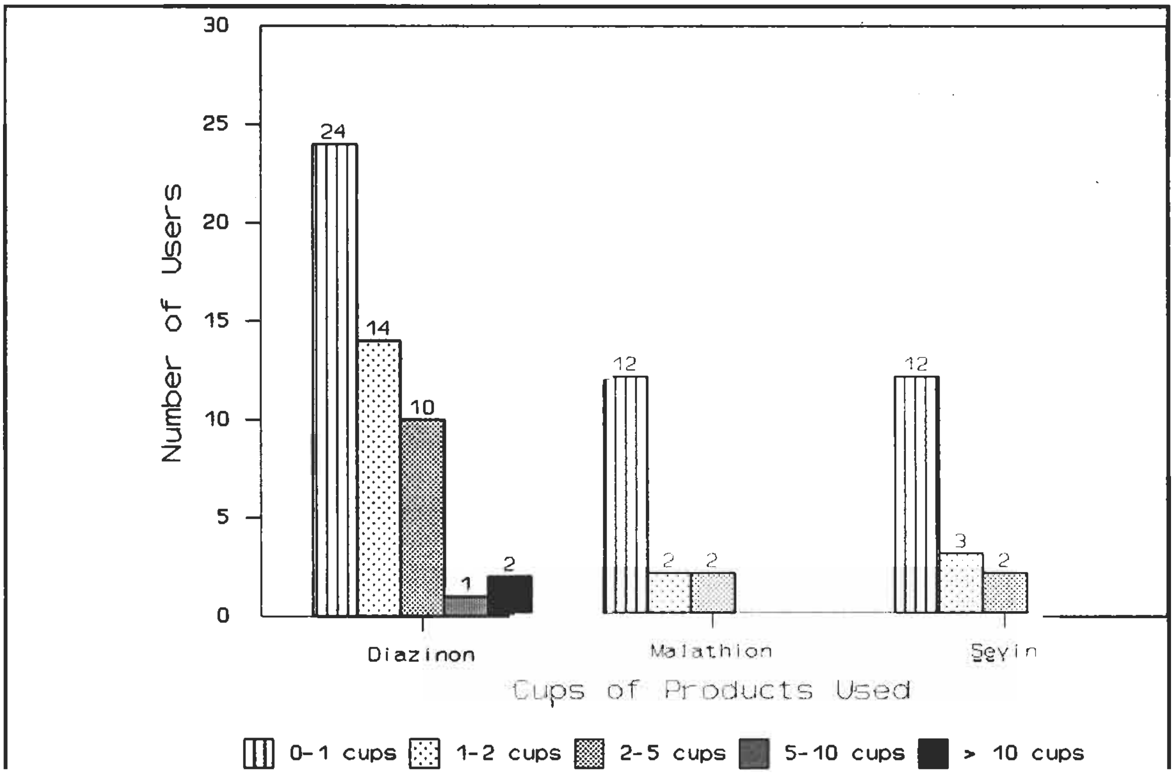

Figure 12. Type and amounts of insecticides used in two Portage County subdivisions.

To minimize fertilizer loss, educational efforts on lawn and garden practices could be focused on the benefits of modifying lawn care practices, such as leaving grass clippings on lawns and limiting irrigation. Participants might also benefit from comparison of their fertilizer application rates with the rates used by farmers to grow typical crops. Many people perceive their fertilizer application on lawns to be insignificant compared to agricultural applications, but this is not always the case. More information on the relative importance of fertilization practices compared to other sources of groundwater contamination in the subdivisions can be found in Section I. Participants may also need instruction on proper pesticide mixing, storing, and disposal practices.

Knowledge about Water Supplies and Septic Systems

Participants were asked for some basic information about their well and sewage disposal system. Wells in the two subdivisions are generally similar in construction: shallow driven-point wells with an average depth of 8.7 meters. Minimum well depths reported were 4 meters in Jordan Acres and 4.3 meters in Village Green. The deepest wells in the subdivisions were in the 13 meter range, although one person reported an estimated depth of 25 meters. The average depth to water is 5.3 meters. Only 25 participants (18%) were certain of the well depth information they reported. This probably reflects the fact that in Wisconsin, no record-keeping on driven-point wells was required at the time of the survey. Seventeen participants (12%) reported that their wells had been replaced or upgraded since original construction, 6 in Jordan Acres and 11 in Village Green.

Twenty-seven participants (19%) reported that their sewage disposal system had been replaced since original construction, 14 in Jordan Acres (28%) and 13 in Village Green (15%). Participants reported pumping their septic tanks an average of every 1.9 years. Some reported pumping as frequently as once every six months, while one participant reported an interval of 9 years. Overall, sewage disposal systems are reportedly well maintained; 119 (86%) were pumped at least once every two years, and only five (4%) were pumped at an interval exceeding once every three years.

Educational efforts about wells and septic systems should be focused on the importance of gathering and maintaining information about well depth, since depth and construction of wells is often related to the quality of the water they produce.

Well owners should also be reminded about the importance of regular water testing although this practice was not specifically surveyed. It appears, however, that survey participants have a good knowledge of proper septic system maintenance. In addition, household chemical use data and participant comments show that most have some concerns about the types of materials they dispose of as well

Attitudes and Opinions about Groundwater Issues

Participants were asked to respond verbally to questions measuring their attitudes about the severity and causes of groundwater contamination in their subdivisions and in Portage County. Overall, 63 percent of participants stated that groundwater contamination was "a serious problem" in Portage County, while 13 percent ranked it as "a very serious problem." Only 1 percent felt that groundwater contamination was "no problem at all".

When responding to an open-ended question about the causes of this problem, the words most frequently used by participants were pesticides (17%), ag fertilizer (16%), farmers (14%), potato farmers (14%), and septic systems (6%). The greatest concerns about groundwater quality were related to nitrate-N and pesticide contamination. At the time the survey was conducted, groundwater contamination with the potato insecticide aldicarb was a major issue in the county. Participants apparently followed and understood the issues in this contamination incident, and believed the information being presented. Overall, 67 percent of those who felt groundwater contamination was "serious" or "very serious" attributed the problem to agriculture. Five percent attributed it to homeowners; 24 percent said both were equally responsible (Figure 13).

Participants' assessment of the severity of water quality problems in their own subdivisions varied. In the Village Green subdivision, discussion of contamination problems had occurred in the local media, and annexation of the subdivision to the city had been discussed. In Jordan Acres, water quality problems were fewer, and there had been little public discussion about them. Accordingly, 54 percent of Village Green participants ranked groundwater contamination as a "serious" or "very serious" problem in their subdivision. On the other hand, 77 percent of Jordan Acres participants rated it a "minor problem" or "no problem at all". In comparison, our water quality survey showed that 14 percent of wells in Jordan Acres and 43 percent in Village Green exceeded the U.S. EPA maximum contaminant level for nitrate-N in that same time period, but participants did not have that information when completing

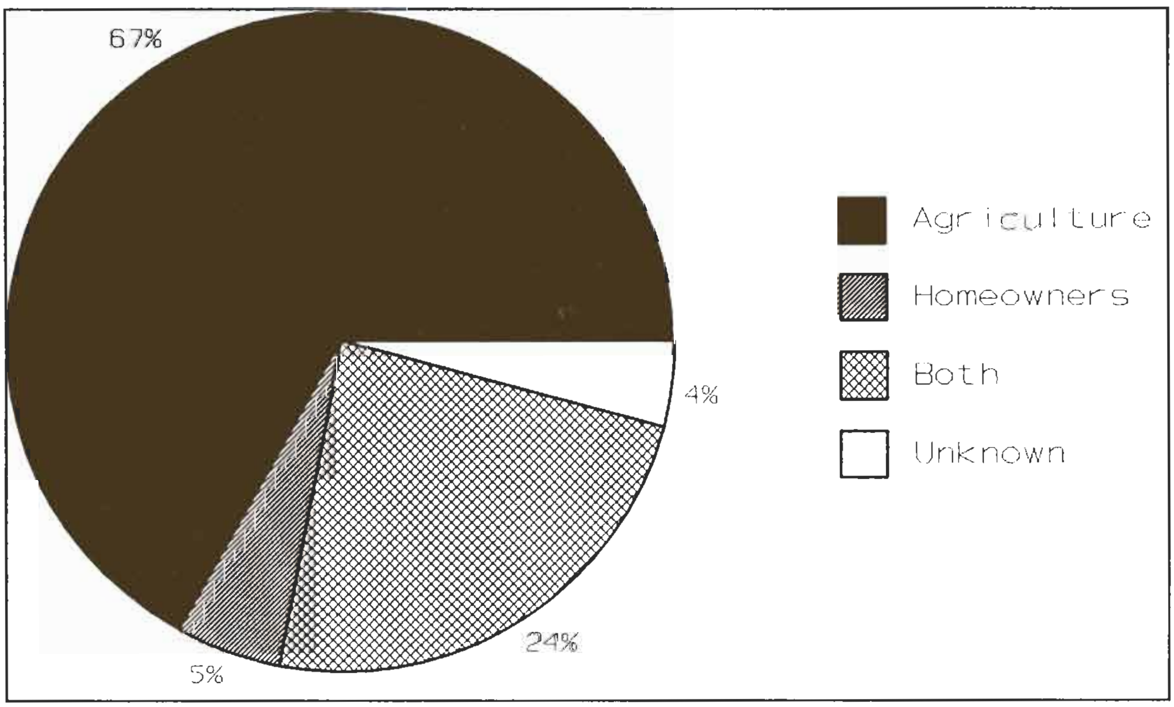

Figure 13. Participant responses about the major source of groundwater contamination problems in Portage County.

the questionnaire.

In Jordan Acres, 73 percent of those ranking it a "serious" or "very serious" problem stated that residential land use was the primary cause, using words such as homeowners (21%), lawn fertilizer (21%), septic systems (14%) and density (14%). Twenty-seven percent attributed the problem to agriculture (Figure 14). On the other hand, in Village Green subdivision, residents perceived that agricultural as well as residential activities were contributing to the problem. Thirty-nine percent of participants in Village Green attributed the problem mainly to agriculture, using words such as Blue Top (a local feedlot) (11%), potato farmers (7%), and ag fertilizer (7%). Forty-three percent named residential activities, using words such as septic systems (14%), lawn fertilizer (13%), and homeowners (10%) (Figure 14).

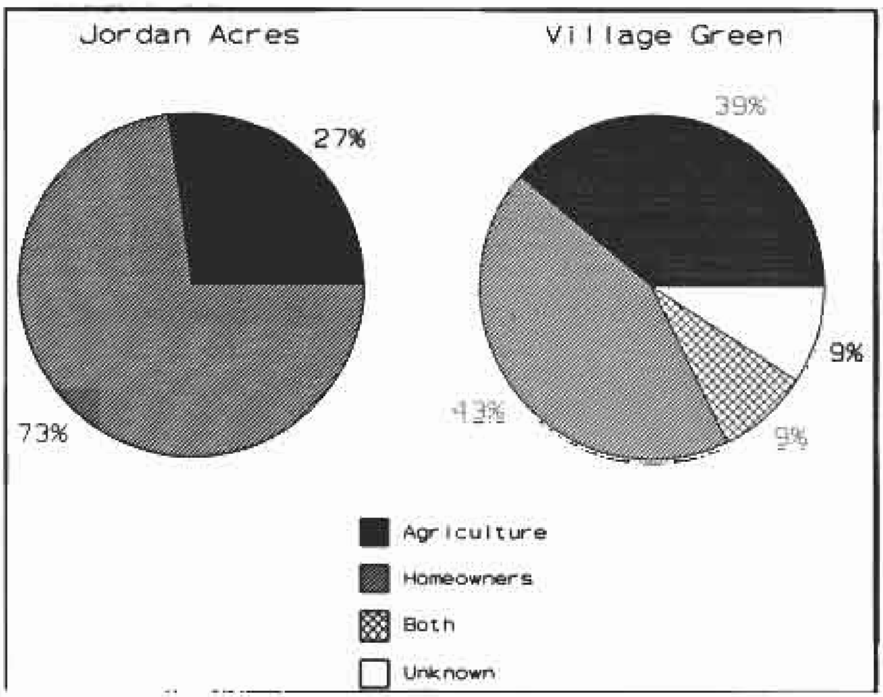

Figure 14. Reasons given by participants that groundwater contamination is a "serious" or "very serious" problem in two Portage County subdivisions.

Participants were asked to choose from a list of problems they believed had been experienced as a result of groundwater contamination in Portage County (Table 6). All the problems on the list were believed by the researchers to have actually occurred in the county. Problems ranked as the top three overall included loss of clean drinking water (102 responses), loss of property values (99 responses), and conflict between agricultural and residential land uses (97 responses). Fewer people believed the quality of life had been lowered (33), that farm animals had been affected (22), or that the area was less attractive to businesses (20).

Problems	All	Jordan Acres	Rank	Village Green	Rank
Loss of clean drinking water	102	44		58	
Loss of property values	99	36	2	63	
Conflict between ag/residential	97	36	2	61	2
Buying/hauling water	74	26	3	48	4
Human stress or illness	52	26	3	26	6
Decreased fish in streams	51	24	4	27	
Lower quality of life	33	15	5	18	7
Farm animal illness/lower productivity	22	12	6	10	8
Area less attractive to businesses	20	10	7	10	8

Table 6. Problems resulting from groundwater contamination in Portage County.

The order of responses varied between the two subdivisions, again perhaps reflecting their differing experiences with water quality problems. In Jordan Acres, where few problems had been experienced to date, "loss of clean drinking water" was identified by the greatest number of participants. On the other hand, in Village Green, "loss of property values" was chosen by the greatest number of participants. At least one participant directly stated to researchers that reports of poor water quality had prevented the sale of his home. The second highest selection was "conflict between agricultural and residential land uses", again perhaps reflecting participants' perceived problems with a nearby feedlot.

A set of twelve statements was then presented to participants with a range of answers from "strongly agree" to "strongly disagree" (Table 7). Responses to several of the statements were similar in both subdivisions. About three-quarters of participants (79 percent Jordan Acres, 71 percent Village Green) disagreed that too much emphasis is being placed on the problem of chemicals in drinking water in Wisconsin. Most participants agreed (88 percent Jordan Acres, 87 percent Village Green) that educating people on how their actions cause groundwater pollution is the most effective solution to groundwater problems. The majority of participants (85 percent Jordan Acres, 75 percent Village Green) also agreed that individual actions taken by a homeowner can make a significant difference in water quality in a subdivision, and that homeowners can pollute their own water supplies (94 percent Jordan Acres, 88 percent Village Green).

Despite the fact that 23 percent of participants in Jordan Acres felt that groundwater contamination was "a serious problem" in their subdivision, only 6 percent did not feel confident that their water was safe to drink, and 13 percent were uncertain. In Village Green, 76 percent felt confident that their water was safe to drink, although 54 percent ranked groundwater contamination as a "serious" or "very serious" problem in their subdivision.

Participants were more neutral to the idea that laws are the only way to control groundwater contamination. In Jordan Acres, 52 percent agreed with that statement,

Question	Strongly Agree (%)	Agree (%)	Uncertain (%)	Disagree (%)	Strongly Disagree (%)
Too much emphasis is being placed on the problem of chemicals in drinking water in Wisconsin.	2 (2)	18 (14)	14 (11)	74 (56)	24 (18)
I feel confident that my well water is safe to drink.	25 (19)	78 (60)	16 (12)	11 (8)	2 (2)
Educating people on how their actions cause groundwater pollution is the most effective solution to groundwater problems.	35 (27)	80 (61)	8 (6)	9 (7)	0 (0)
Laws regulating people and businesses are the only way to control groundwater contamination.	17 (13)	47 (36)	23 (18)	44 (33)	1 (1)
Individual actions taken by a homeowner can make a significant difference in groundwater in a subdivision.	28 (21)	76 (58)	17 (13)	11 (8)	0 (0)
Individual homeowners can cause the pollution of their own water supplies.	31 (24)	88 (67)	12 (9)	1 (1)	0 (0)
Property values are being affected by water quality problems in this subdivision.	22 (17)	41 (31)	24 (18)	41 (31)	4 (3)
One way to protect the groundwater in this subdivision is if all the residents work together in controlling contaminants.	22 (17)	94 (71)	10 (8)	7 (5)	0 (0)
What we do in this household has no impact on our groundwater quality.	3 (2)	22 (17)	6 (5)	69 (52)	31 (23)
Subdivisions with water quality problems should have municipal sewer and water service provided by local government.	16 (12)	55 (42)	32 (24)	24 (18)	4 (3)
Annexation to the city of Stevens Point is an acceptable option for obtaining municipal sewer and water service.	9 (7)	48 (36)	24 (18)	29 (22)	21 (16)
Having municipal sewer and water would increase the value of my home.	20 (15)	67 (51)	17 (13)	21 (16)	7 (5)

Table 7. Response to survey opinion statements.

while 31 percent disagreed In Village Green, 47 percent agreed; 36 percent disagreed.

A number of statements dealing with the acceptability of receiving municipal sewer and water service and affects on property values were also presented. Reaction to these in some cases varied significantly by subdivision. For example, in Village Green, 64 percent agreed with the statement that "property values are being affected by water quality problems in this subdivision." In Jordan Acres, only 19 percent agreed (a statistically significant difference, $p < .05$) In Jordan Acres, only 10 percent disagreed that "subdivisions with water quality problems should have municipal sewer and water service provided by local government", while in Village Green, 23 percent disagreed and 5 percent strongly disagreed (also a statistically significant difference). On the other hand, there was substantial agreement in both subdivisions (69 percent in Jordan Acres, 64 percent in Village Green) that "having municipal sewer and water would increase the value of my home." On the acceptability of annexation to the nearby city of Stevens Point, 27 percent of Jordan Acres residents were undecided, and a total of 25 percent were opposed, 13 percent strongly so. In Village Green, where annexation had been discussed as a real possibility, a total of 46 percent were opposed, 18 percent strongly so.

It is also informative to examine which opinion statements elicited the strongest agreement or disagreement from participants. In Jordan Acres, the statement most often strongly agreed with was "individual homeowners can cause the pollution of their own water supplies" (33%), followed by "one way to protect the groundwater in this subdivision is if all the residents work together in controlling contaminants" (31%). Jordan Acres participants most often strongly disagreed with "What we do in this household has no impact on our groundwater quality" (28%), followed by "Too much emphasis is being placed on the problem of chemicals in drinking water in Wisconsin" (25%).

In Village Green, participants most often strongly agreed with "Educating people on how their actions cause groundwater pollution is the most effective solution to groundwater problems" and "Property values are being affected by water quality problems in this subdivision" (each 25%). As in Jordan Acres, Village Green participants most often strongly disagreed with "What we do in this household has no impact on our groundwater quality" (21%), followed by "Annexation to the city of Stevens Point is an acceptable option for obtaining municipal sewer and water service" (18%). It appears that fewer Village Green residents were likely to feel strongly about the above issues, but that they did react strongly to some which personally affected them.

Educational efforts to increase awareness of groundwater problems in Portage County does not appear necessary at this point However, some subdivision residents need to increase their awareness of their own potential affects on their water supply and need to assume some personal responsibility for it. There appears to be a strong feeling that working together can prevent groundwater contamination Ways of encouraging that cooperation need to be explored.

Relationships of Attitudes to Age, Gender and Educational Level

Several attitude questions were significantly related to personal factors such as age, gender and education level ($p < .05$). The question "Laws regulating people and businesses are the only way to control groundwater contamination", which previously was shown to have a significant relationship to household cleaning product use, was also related to both gender and education level. Fifty-eight percent of males agreed with this statement, while only 35 percent of females agreed. Among participants with a high school education or less, 69 percent agreed, while of college educated participants, only 34 percent agreed with the statement.

In response to the statement "What we do in this household has no effect upon our groundwater quality", 31 percent of participants with a high school education or less agreed. Only 1 percent of those with some college education agreed.

Lastly, in response to the statement "Annexation to the city of Stevens Point is an acceptable option for obtaining municipal sewer and water service", a significant relationship to participants' age was found. People age 45 and over were more likely to agree with the statement (59%) than those younger than 45 (35%). Twenty-nine percent of participants younger than 45 were uncertain, as opposed to only one person in the 45 and older category.

Survey Conclusions

1. Household cleaning product use was similar between the two subdivisions. Some products, such as laundry detergent and bathroom cleanser, were used at least weekly by most participants. Some products which may be particularly hazardous to septic systems and groundwater, such as chlorine bleach, were also frequently used by participants.

2. Household maintenance products such as paint thinner and motor oil were used less frequently and by fewer participants. However, there is evidence that these materials are being improperly disposed of by some participants in ways that may adversely affect groundwater.

3. Lawn care practices were similar between the two subdivisions, with a mean fertilization rate of 1.6 times per year. Lawn fertilization frequency was related to mowing frequency, watering frequency, and tendency to remove lawn clippings.

4. Insecticides most commonly used included diazinon, malathion and carbaryl, with nearly 40 percent of participants reporting using diazinon.

5. Wells in the two subdivisions are generally shallow driven points with an average depth of 9 meters. Only 18 percent of participants were certain of the depth of their wells.

6. Participants in the two subdivisions generally reported following proper sewage disposal system maintenance, with an average pumping interval of 1.9 years.

7 A significant relationship was not found between lawn care and household cleaning product use practices.

8. Seventy-six percent of participants believed groundwater contamination was a serious or very serious problem in their county. Opinions about severity of groundwater contamination in the individual subdivisions varied by subdivision.

9. Participants were knowledgeable about groundwater contamination issues. However, some need a better understanding of how their own actions may affect groundwater quality.

10. Although some relationships were noted, in general there is not a good relationship between household chemical use practices and attitudes about groundwater contamination. A few relationships were found between attitudes and age, gender or education level.

E. Nitrogen Mass Balance Prediction using BURBS Model

One of the major objectives of the project was to determine the validity of using mass balance nitrogen models to predict subdivision impacts on groundwater quality. The BURBS model, developed at Cornell University by Hughes et. al. (1985) was selected for use in this phase of the project as it includes all the variables the authors felt were significant to predicting nitrogen impacts to groundwater. The variables used in the model are:

1) Fraction of land in turf.
2) Fraction of land which is impervious.
3) Average persons per dwelling
4) Housing density.
5) Precipitation rate.
6) Water recharged from turf.
7) Water recharged from natural land.
8) Evaporation from impervious surface.
9) Runoff from impervious surfaces that is recharged.
10) Home water use per person.
11) Nitrogen concentration in precipitation.
12) Nitrogen concentration in water used.
13) Turf fertilization rate.
14) Fraction of nitrogen leached from turf.
15) Fraction of wastewater nitrogen lost as gas.
16) Wastewater fraction removed by sewer.
17) Nitrogen per person in wastewater.
18) Nitrogen removal rate of natural land.

Each of these variables is discussed and model input values are defined.

The areas that were modelled are the sections (termed cuttings) of the subdivisions that are impacting selected downgradient multiport wells. The monitoring networks were not randomly spaced across the subdivisions; therefore, the data are more representative of a part of the subdivisions than of the entire subdivision. Because a goal of the project was to compare BURBS predictions with field monitoring values, it was necessary to define the BURBS variables in terms of the conditions impacting the monitoring network. Thus, while the demographic-type variables were defined using averages for the entire subdivision, the areal-type variables were based on specific land use within the cutting areas.

Onsite waste disposal is the primary source of nitrogen loading to groundwater from a subdivision. Once the model variables were accurately defined, simulations were run to evaluate the effect of doubling and halving the housing density (hence septic system density). Relative amounts of land use areas (i.e., turf, natural, and impervious) were adjusted to accommodate the increased (decreased) amount of impervious area associated with more (fewer) houses in a given area. For these simulations, the area of houses and driveways were doubled (halved) and the area of turf and natural land use were reduced (increased) by an amount in proportion with their baseline areas. The amount of road area was kept constant.

A number of runs were made to calibrate the model in terms of the amount of nitrogen leached from lawn fertilizers. For these simulation runs, the amount of wastewater removed by sewer was set at 1.00, to eliminate wastewater impacts from the simulation results. The leaching values ranged from 0.05 to 0.40. The leaching value considered to be most representative of observed in-field conditions was the one that yielded a nitrate-N concentration most similar to the concentrations measured in water samples of wells impacted solely by lawns (approximately 4.3 mg/l nitrate-N).

Several runs were made to demonstrate the effect of precipitation amounts on groundwater nitrate-N concentrations. Wet years and dry years were simulated.

Once the model was defined for the sandy soils, several runs were made to demonstrate how soil type and reduced groundwater recharge affect nitrate-N concentrations in groundwater.

Variable Definition

The fraction of land in turf, impervious, and natural ground covers in the cuttings were calculated by pc/ARCINFO from the subdivision maps. Maps of the cuttings are shown on Figures 15 and 16. In the simulations where the housing density was varied, the land use percentages were modified to account for the differing amount of impervious area occupied by residences. For these simulations, the fraction of impervious area was divided into roads and residences (including buildings, and driveways). The residential impervious area was modified by the changes in housing density (doubled or halved), but the road area was kept the same. The fraction of land in turf and natural were modified to account for the change in impervious area. The land use fractions used in the simulations are summarized in Table 8.

The average number of persons per dwelling was 2.97 for Jordan Acres and 3.53 for Village Green. These values were determined by surveying a portion of the subdivision occupants. Approximately 50 percent of the homes in Jordan Acres and 35 percent of the homes in Village Green were surveyed.

The housing density for each scenario was calculated using the total number of houses in each area of interest and dividing by the total area of the cutting. The value for Jordan Acres was 3.7 homes per hectare; Village Green was 2.9 homes per hectare. These values include roads, vacant lots, natural areas, and public lands.

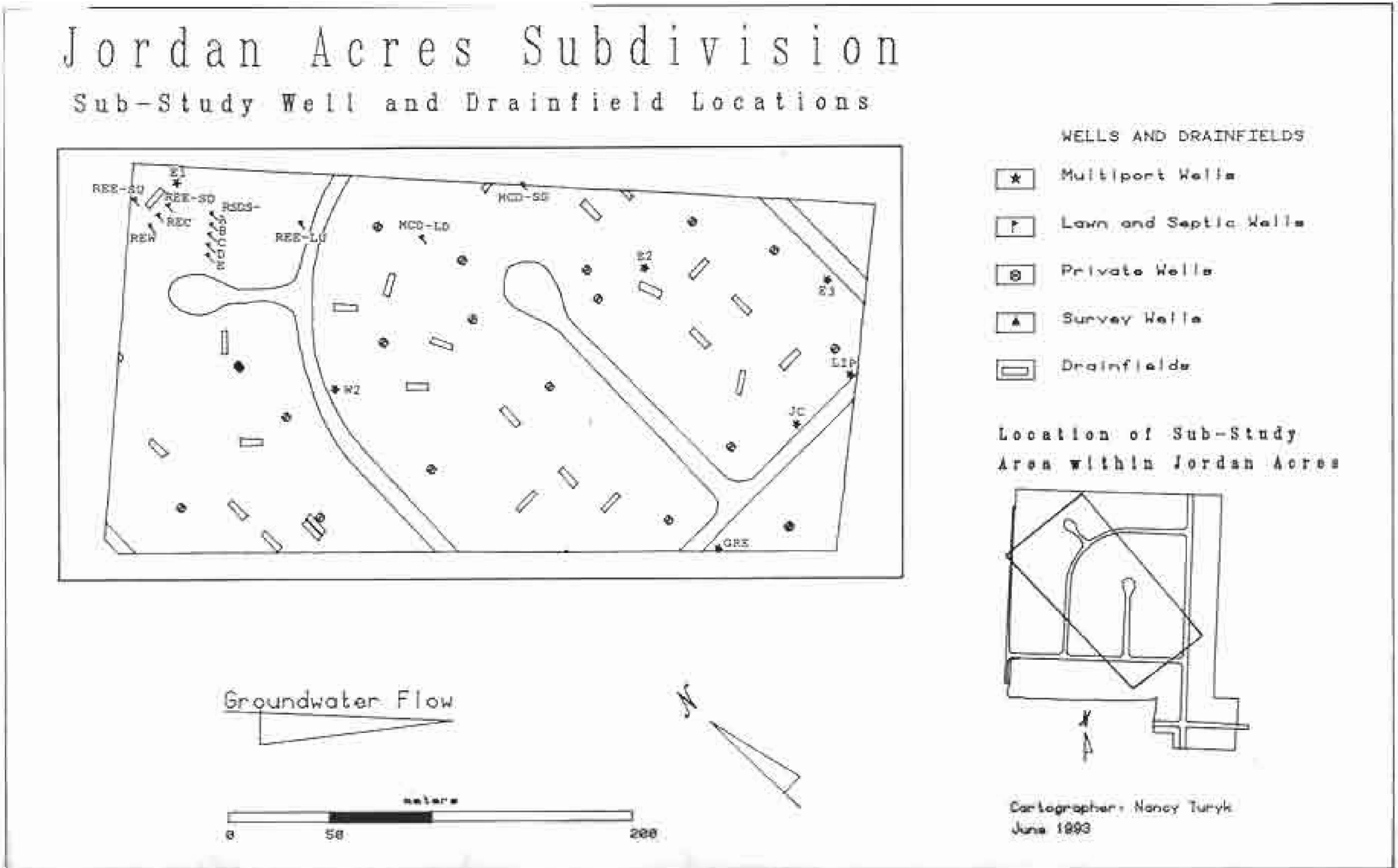

Figure 15. Map of Jordan Acres subdivision sub-study area showing well locations.

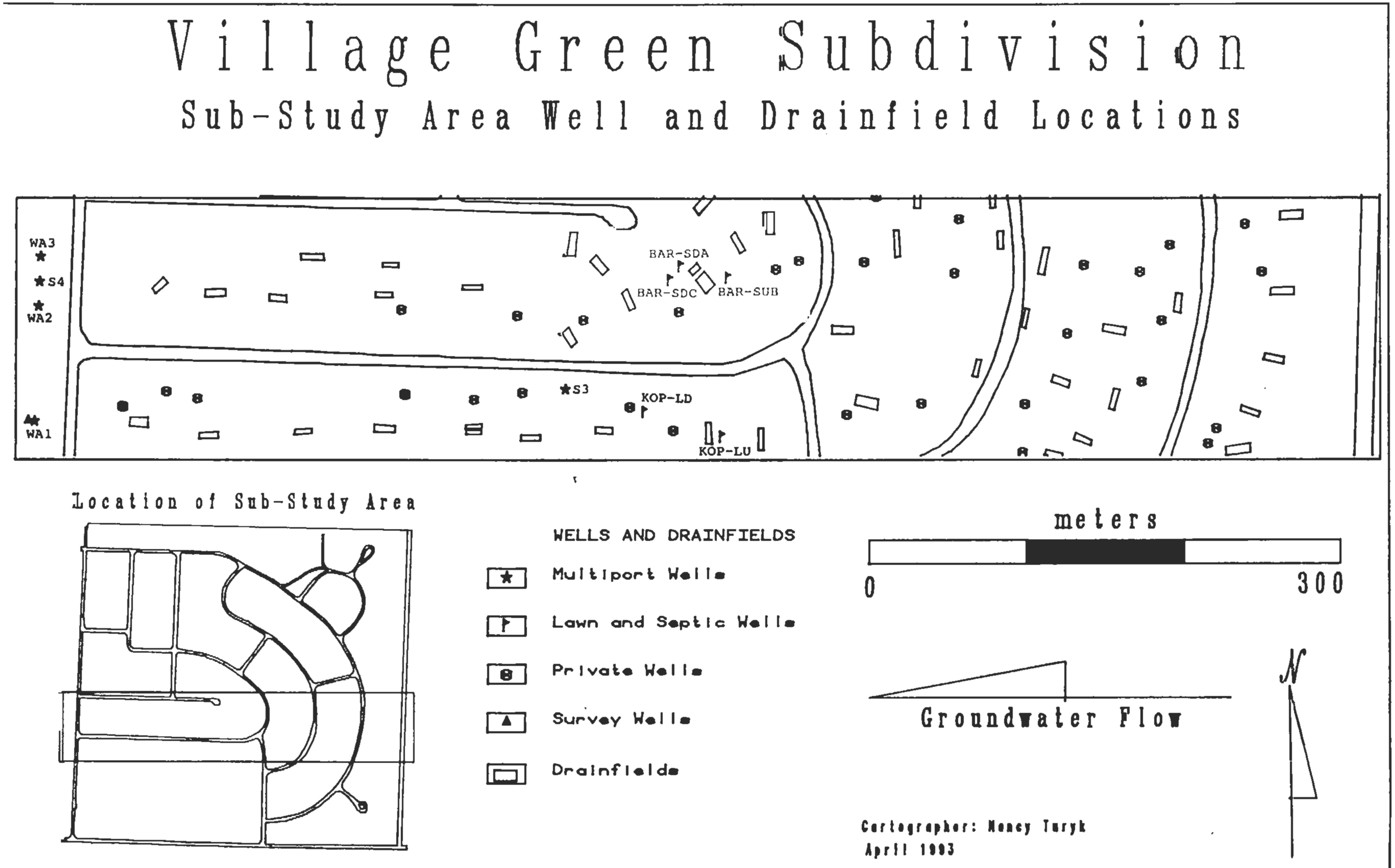

Figure 16. Map of Village Green subdivision sub-study area showing well locations.

	Jordan Acres Cutting			Village Green Cutting		
	Baseline (BL)	BL x 0.5	BL x 2	Baseline (BL)	BL x 0.5	BL x 2
Fraction of Land in Natural Conditions	0.12	0.13	0.10	0.35	0.39	0.28
Fraction of Land that is Impervious (Residential)	0.13	0.07	0.26	0.15	0.08	0.30
Fraction of Land that is Impervious (Roads)	0.09	0.09	0.09	0.09	0.09	0.09
Fraction of Land in Turf	0.66	0.72	0.55	0.41	0.45	0.33

Table 8. Relative amount of turf, natural and impervious areas in the BURBS simulation for the Jordan Acres and Village Green cuttings.

The actual lot sizes are approximately 0.17 to 0.2 hectares.

The BURBS model considers the housing density to be equivalent to septic system drainfield density. It further assumes that the drainfields are evenly distributed throughout the subdivision. Observed in-field conditions indicate that some of the wells potentially get impacted by many drainfields, while others get impacted by few or none. To simulate this variability, the drainfield density was doubled in certain scenarios and halved in other scenarios.

Precipitation data are presented in Figure .7. The estimated groundwater travel time beneath Jordan Acres ranged from 1.6 to 2.9 years; the travel time beneath Village Green ranged from 4.8 to 9.0 years. The average precipitation from the years 1985 through 1990 (78 cm) was used for BURBS simulations modeling Jordan Acres; the average precipitation from 1981 to 1991 (83 cm) was used when modelling Village Green.

In order to demonstrate how fluctuations in precipitation can affect groundwater quality, the precipitation amount from relatively wet years and dry years was used in

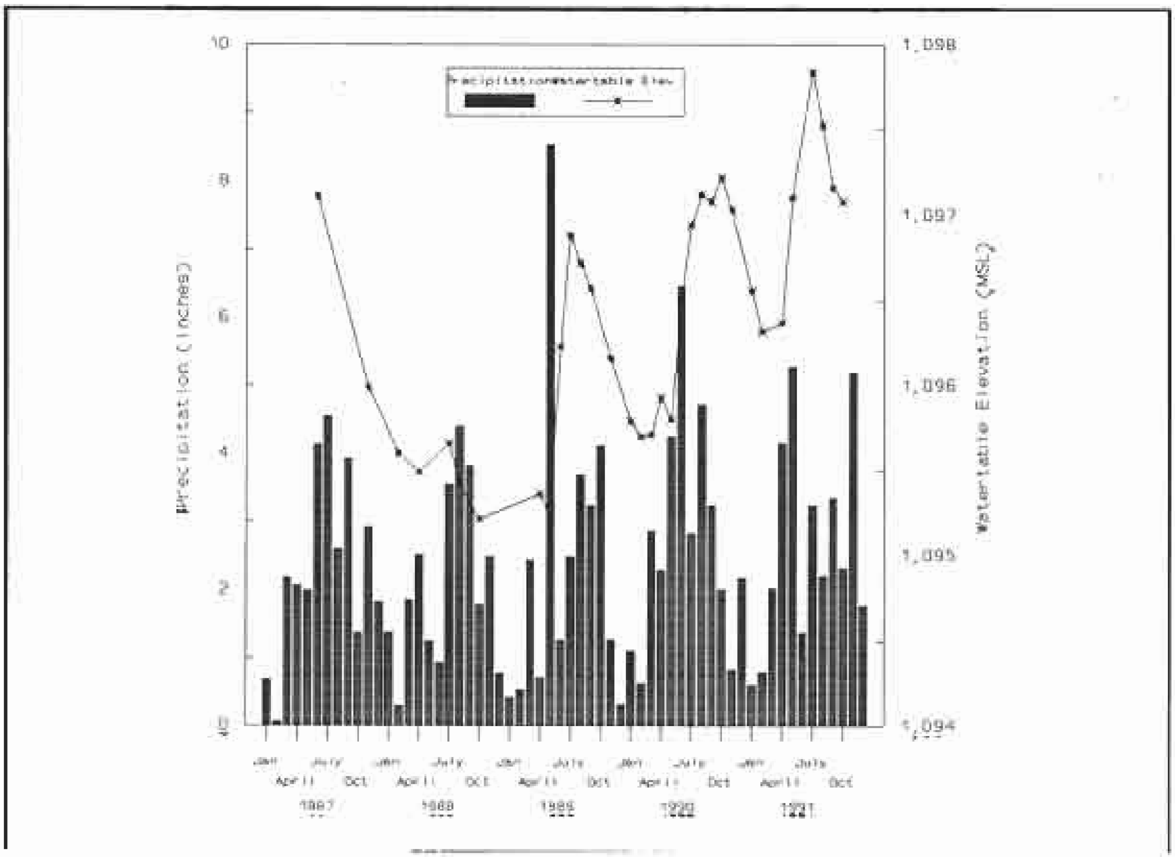

Figure 17. Water table elevation measured from the Jordan Acres northwest survey well, and precipitation measured at the Stevens Point, Wisconsin wastewater treatment plant from 1987 to 1991.

several simulations. The values that were chosen were the wettest and driest years over the time span used to determine average precipitation amounts (64 and 88 cm for Jordan Acres; 64 and 114 cm for Village Green).

Water recharged to the groundwater from turf and natural land was assumed to be equal to the total amount of precipitation minus 53 cm of evapotranspiration. Additional recharge was calculated by the model to account for runoff from impervious areas to lawns and natural areas. The evaporation from impervious surfaces was set at 10 percent, as recommended by the BURBS documentation.

The runoff from impervious surfaces going to recharge was not defined with a great deal of certainty, due to the complexity of influencing factors. For example, rain that lands on rooftops is diverted to eaves troughs, where it is discharged to the ground in specific locations. This additional water will saturate the soil quicker than the rain would itself, thus facilitating water movement into the ground. Water that runs off from roads to ditches will behave in a similar manner. Water from impervious surfaces will be subject to some evapotranspiration (ET); however, the localized area receiving the runoff water will quickly become saturated, thus facilitating water movement through the vadose zone and into the aquifer. ET is already included in the 53 cm/year value, the additional runoff mostly goes to recharge. Because the soils have a very low water-holding capacity (which can quickly be met by the precipitation event) the additional runoff from impervious surfaces is available to recharge the groundwater. No surface runoff to storm sewers, waterways, or streams occurs in either subdivision. For modelling purposes, it was assumed that 90 percent of the water not evaporating from impervious areas goes to groundwater recharge. Because this recharge water will have low nitrate-N levels, it will tend to lower average nitrate-N concentrations (by dilution) but not significantly effect nitrogen loading. This additional recharge in areas with sandy soil helps keep nitrate-N levels in the recharge water low, compared to areas of heavier soil, where water will runoff and not aid in diluting the effects of septic systems.

The volume of water used per person in the subdivisions was estimated after considering several sources. The US Environmental Protection Agency estimates that the per capita rate of water use is 70 liters per person per day (USEPA, 1980). We conducted a survey to determine home water use in the city of Stevens Point, which indicates water consumption of 270 liters/person/day. This estimate may be high because of the uncertainty of the actual number of persons per household (assumed to be 3). Also, it has been suggested that homeowners with septic systems are more conscious of their water use than those on city water and thus tend to be more conservative in terms of water use. A water meter was installed on the well at a residence in the Jordan Acres subdivision. The two adult occupants each used approximately 190 liters of water per day over a twelve month period. Data obtained from an investigation monitoring 15 septic systems in nearby rural homes indicate that home water use is closer to 130 liters per person per day (Shaw and Turyk, 1992). A value of 150 liters/person/day was used in the simulations.

The Environmental Task Force Lab - UW Stevens Point tested for the nitrogen concentration in precipitation frequently throughout the 1980s (unpublished data). The average nitrate-N concentration determined by this study was 0.25 mg/l.

Private well data from many of the homes in the subdivision were used to calculate an average nitrate-N concentration in the water used in the subdivisions. The average for Jordan Acres was found to be 6.9 mg/l and the average for Village Green was 11.3 mg/l.

The turf fertilization rate used in the model simulation was based on data obtained by the survey of subdivision homeowners (Section D). The survey results indicated that 74 percent of all respondents used the amount specified by the manufacturer, 18 percent used more than was specified, 6 percent used no fertilizers, and 1 percent did not read the bag. The survey also revealed that the overall fertilizer application rate was 1.6 times per year (1.8 times for users). A value of 0.78 kg/100 m^2 was used in modelling both subdivisions (assuming a 0.49 kg/100 m^2 rate applied .6 times per year).

Petrovic's (1990) review of relevant research revealed that although the amount of nitrogen leached from fertilized turf grass was highly variable, it was generally less than 5 percent that was leached to groundwater. The exceptions were in areas where the fertilizers were applied in excessive amounts and/or the turf was over watered. The BURBS variable definitions cite a Long Island study that indicated up to 50 percent of lawn fertilizers used in sandy soils leached to groundwater. Because field data from lawn impacted groundwater was available, the value for this variable was defined using a range of values, then comparing the results with the field data. The leaching value that yielded the most representative results was used for the baseline value in the model. For calibrating purposes it was assumed that all of the nitrogen in wastewater was removed by sewers.

Studies have shown that in well aerated sandy soils, the amount of nitrogen in wastewater lost as a gas is negligible (Walker, et al, 1973). This conclusion was supported by studies of private waste disposal systems in a nearby subdivision (Shaw and Turyk, 1992). The value of 0 was used for this variable.

The subdivisions are unsewered, thus the wastewater fraction removed by sewer was set at 0 (except when used to calibrate the fertilizer leaching variable as discussed above).

The amount of nitrogen per person in wastewater has been fairly well documented. A value of 4.5 kg/person/year was reported by Walker et.al. (1973).

This value was also found for 15 septic systems in the Stevens Point area (Shaw and Turyk, 1992). Samples from a septic tank serving two adults in Jordan Acres contained 60 and 89 mg/l of total Kjeldahl nitrogen in the wastewater. The daily water use by this household was measured to be 190 liters/person/day, thus the annual nitrogen loading rate is estimated to be 4.4 to 6.4 kg/person/year. A value of 4.5 kg/person/year was used for modelling purposes.

The nitrogen removal rate of natural land was set at 0.9 as recommended by BURBS documentation, but it is negligible in model simulations because of the low nitrogen concentration in precipitation. Values for the variables used in the BURBS simulations are summarized in Table 9.

Variable	Jordan Acres	Village Green
Fraction of land in turf-baseline	0.66 *	0.41 *
Low upgradient drainfield density	0.72	0.45
High upgradient drainfield density	0.55	0.33
Fraction of land that is impervious-baseline	0.22 *	0.24 *
Low	0.16	0.17
High	0.35	0.39
Average persons per dwelling	2.97 *	3.53 *
Housing density (#/hectare)	3.7 *	2.9 *
Low upgradient drainfield density	1.8	1.4
High upgradient drainfield density	7.4	5.7
Precipitation rate (cm/year)	78 *	84 *
Dry year	64	64
Wet year	88	114
Water recharged from turf (cm/year)	Precipitation - 53	
Water recharged from natural land (cm/year)	Precipitation - 53	
Evaporation from impervious surface (fraction)	0.1 *	0.1 *
Runoff from impervious recharged (fraction)	0.9 *	0.9 *
Home water use per person (liters/day)	151 *	151 *
Nitrogen concentration in precipitation (mg/l)	0.25 *	0.25 *
Nitrogen concentration in water used (mg/l)	6.9 *	11.3 *
Turf fertilization rate (kg/100 m^2)	0.78 *	0.78 *
Fraction of nitrogen leached from turf (fraction)	Varied from 0.05 to 0.40 0.25 *	 0.25 *
Fraction of wastewater N lost as gas (fraction)	0 *	0 *
Wastewater fraction removed by sewer (fraction)	0 **	0 **
Nitrogen per person in wastewater (kg/year)	4.5 *	4.5 *
Nitrogen removal rate of natural land (fraction)	0.9 *	0.9 *

* Used for baseline model run
** 100 % used when calibrating fertilizer leaching

Table 9. Values for the variables used in the BURBS simulation for Jordan Acres and Village Green.

Simulation Results

The results of the various BURBS simulations are presented in Table 10 and Appendix B.

The fertilizer leaching estimates for Village Green are 30 to 35 percent lower than those for Jordan Acres. This is because Jordan Acres has a higher percentage of its land use as turf, whereas Village Green has more natural and impervious areas. The non-turf or natural areas have a diluting influence on the nitrate-N concentrations.

Results of the Jordan Acres BURBS simulations that compare fertilizer leaching rates were evaluated to determine the amount of leaching occurring within the subdivisions. Because the average nitrate-N concentration of wells monitoring lawn areas was 4.3 mg/l, the leaching rate for the baseline value used in the simulations was set at 25 percent. Jordan Acres results were used because most of the wells used to monitor lawn impacts were in that subdivision. The 4.3 mg/l nitrate-N is also close to average for the Village of Park Ridge, a sewered village adjacent to Stevens Point with all groundwater recharge originating from the urban area (ETF unpublished data).

For Jordan Acres, the 25 percent leaching rate accounts for about 21 percent of the total nitrogen budget, as compared with the results of the baseline simulation; for Village Green the 25 percent rate accounts for 18 percent. Varying the leaching rate by five or even ten percent either up or down has little significant impact on overall nitrate-N concentrations, thus the 25 percent leaching rate is considered to be appropriate.

Study area and simulation conditions	Average NO3 in Recharge	Nitrogen Leached		Water Recharged	
	mg/l	kg/Ha	lb/acre/yr	cm/year	inch/year
Jordan Acres Cutting					
Baseline Variable Values	17.2	67.3	60.1	39.1	15.4
High Upgradient Drainfield Density (7.4 dwellings/hectare)	23.7	119	106	48.3	19.8
Low Upgradient Drainfield Density (1.8 dwellings/hectare)	12.3	41.4	37.0	33.5	13.2
Wet Year (88 cm of Precipitation)	13.8	67.4	60.2	49.0	19.3
Dry Year (64 cm of Precipitation)	26.1	67.2	60.0	25.7	10.1
No Drainfield Impacts and:					
- 5% of fertilizer leaches	0.9	3.0	2.7	33.0	13.0
- 10% of fertilizer leaches	1.7	5.7	5.1	33.0	13.0
- 20% of fertilizer leaches	3.3	11.0	9.8	33.0	13.0
- 25% of fertilizer leaches	4.1	13.6	12.1	33.0	13.0
- 30% of fertilizer leaches	4.9	16.2	14.5	33.0	13.0
- 40% of fertilizer leaches	6.5	21.5	19.2	33.0	13.0
Village Green Cutting					
Baseline Variable Values	13.7	60.9	54.4	44.5	17.5
High Upgradient Drainfield Density (5.7 dwellings/hectare)	20.0	112	99.8	55.6	21.9
Low Upgradient Drainfield Density (1.4 dwellings/hectare)	9.1	35.5	31.7	38.8	15.3
Wet Year (114 cm of Precipitation)	8.3	61.2	54.6	73.9	29.1
Dry Year (64 cm of Precipitation)	23.2	60.7	54.2	26.2	10.3
No Drainfield Impacts and:					
- 5% of fertilizer leaches	0.6	2.1	1.9	38.9	15.3
- 10% of fertilizer leaches	1.0	3.8	3.4	38.9	15.3
- 20% of fertilizer leaches	1.8	7.1	6.3	38.9	15.3
- 25% of fertilizer leaches	2.2	8.7	7.8	38.9	15.3
- 30% of fertilizer leaches	2.7	10.4	9.3	38.9	15.3
- 40% of fertilizer leaches	3.5	13.7	12.2	38.9	15.3

Table 10. BURBS simulation results for the Jordan Acres and Village Green cuttings.

The results of varying the drainfield density, in addition to the results from simulations that assumed no drainfield impacts, supports the observations and conclusion of several other authors (Yates, 1985 and Perkins,1984) that septic system drainfields are the primary cause of elevated nitrate-N concentrations in the groundwater beneath unsewered subdivisions. Note that in Jordan Acres, even at a relatively low drainfield density (1.9 dwellings/hectare) BURBS predicts nitrate-N concentrations in excess of the enforcement standard for nitrate-N of 10 mg/l. In Village Green, the low drainfield density simulation yielded a result below the 10 mg/l standard. The area for this simulation was one home for every 0.7 hectares. It should be noted that the recharge rate of 29.7 cm used for Village Green is much higher than the 25.4 cm long term average for the area. Simulations were run to determine the housing density that would be needed in Village Green and Jordan Acres to achieve a 10 mg/l nitrate-N concentration in recharge. These housing densities are 1.7 dwellings/hectare in Village Green and 1.1 dwellings/hectare in Jordan Acres.

Figure 18 shows the relationship between housing density and simulated nitrate-N concentrations in groundwater recharge for Jordan Acres and Village Green subdivisions. The primary reason for the differences between the two subdivisions is that the precipitation amounts used for the two subdivisions differed by 5. cm which resulted in less recharge, and therefore less dilution in Jordan Acres simulations. The higher percent of the area in lawns in Jordan Acres resulted in more fertilizer leaching which was largely offset by a slightly higher number of people per household in Village Greens, which increases nitrate-N leaching.

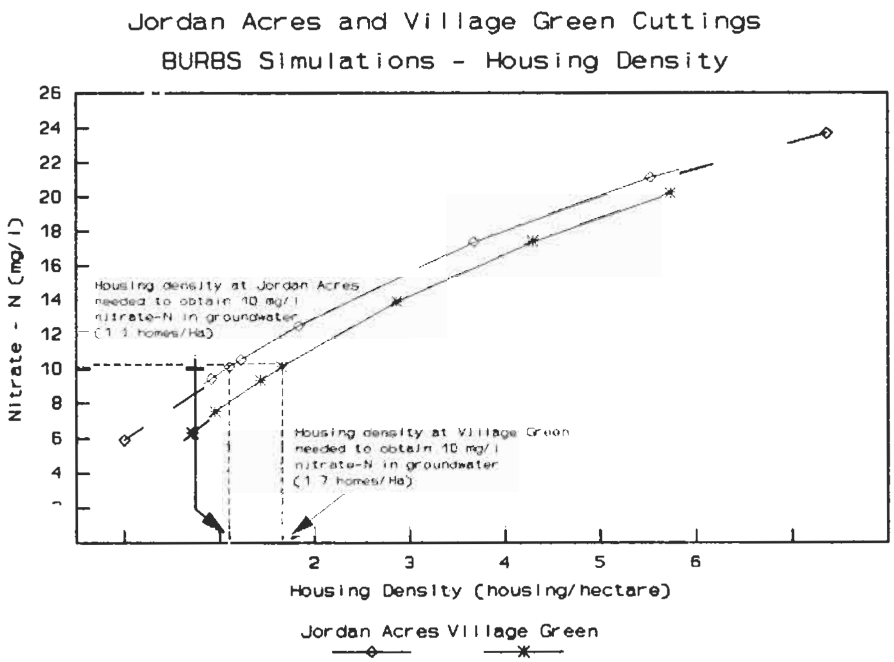

Figure 18. BURBS estimated nitrate-N concentrations related to varying housing densities at Jordan Acres and Village Green subdivisions.

Jordan Acres is the simulation that best represents the sandy soil areas of Wisconsin, as the precipitation data used is closest to the long term average for Wisconsin and the number of people per home (2.97) is close to the state per household average.

Precipitation amounts can also greatly affect groundwater nitrogen concentrations. In wet years, there will tend to be more water available to dilute the nitrogen input from septic systems; in dry years, less dilution will occur and nitrate-N concentrations will be higher. Table 10 presents results of simulations for Jordan Acres and Village Green, where precipitation extremes during the study period for each subdivision were used. The range of 64 to 114 cm used for Village Green fives simulated nitrate-N concentrations of 23.2 to 8.3 mg/l, where only this variable was changed. Precipitation extremes may have a short-term impact on groundwater quality and account for some of the variability found in shallow wells. Precipitation extremes can have a dramatic effect on groundwater quality if the conditions persist for several years.

Simulation for Heavier Textured Soils

In addition to the simulations run for Village Green and Jordan Acres, several runs were made changing the routing of runoff water and reducing groundwater recharge to 10 cm/year, which is considered to be a reasonable estimate of the statewide average for groundwater recharge. These simulations are presented in Figure 19. The simulations are considered to be indicative of what one would expect in areas of heavier textured soils and/or greater slope. The Village Green set of values were used for the variables except for the reduction of recharge from natural areas from 30.0 cm (11.8 in) to 10.2 cm (4.00 in), and recharge from runoff from 90 percent to 12 percent. The fraction of fertilizer that leaches from fine-textured soils tends to be less than in sandy soils (Petrovic, 1990), therefore, the value for this variable was reduced from 0.25 to 0.05 This resulted in nitrate-N concentrations of 34.9 mg/l, compared to 13.7 mg/l for the Village Green baseline values. Lot size to achieve a nitrate-N concentration of 10 mg/l increased from 0.6 hectares/dwelling to 2.0 hectares/dwelling.

These runs of the model indicate the importance of having good estimates of the amount of groundwater recharge that will occur from lawns, natural areas, and also that due to runoff from impervious areas. This variable is of equal importance to housing density when using a mass balance model.

Figure 19. BURBS estimated nitrate-N concentrations related to varying housing densities in heavy soils using Village Green subdivision variables.

Subdivision designs that maximize local groundwater recharge will provide maximum dilution of nitrogen inputs from septic systems. These scenarios also indicate that fertilizer leaching in sandy soil areas, while a significant part of the nitrogen budget, is effectively diluted by high recharge amounts. Decreased recharge with a similar percent of fertilizer leaching results in much higher nitrate-N concentrations reaching groundwater from lawns. More research is needed to evaluate nitrogen losses from lawns on different soil types in Wisconsin.

Overall, we believe the BURBS program provides a fairly accurate estimate of nitrogen inputs from subdivisions. Some of the variables (discussed previously) need careful evaluation for accurate application of the model. It must be recognized that the model predicts average nitrogen concentrations in the entire subdivision recharge. For these concentrations to be achieved, complete mixing of subdivision recharge would be needed, and no mixing with upgradient groundwater could occur. This is obviously not the case as demonstrated by the wide range of groundwater quality documented by this project. Careful layout of subdivisions and lots to prevent private wells from intercepting contaminant plumes is needed if current waste disposal practices are to be used.

F. Nitrogen and Water Budget Results from Field Data

The variables and results of nitrogen and water budgets using subdivision field data are presented in Table 11.

The area values used in the budget calculations (width of cross section and length of flow path) were based on data obtained from only a portion of the subdivisions (termed cuttings). The area included in the Jordan Acres cutting is shown on Figure 15; the Village Green cutting is shown on Figure 16 (pages 73 and 74).

The depth of subdivision impacted water was estimated based on the chemistry data obtained from the downgradient multiport wells that are discussed in Section H.

The average linear groundwater flow velocities were determined based on a range in hydraulic conductivity from 0.045 cm/sec to 0.085 cm/sec for both subdivisions, an effective porosity of 0.30, and hydraulic gradients of 0.0026 (Jordan Acres) and 0.0020 (Village Green). The discharge volumes were calculated based on these hydrogeologic characteristics and the cross-sectional area impacted by the subdivision.

The average nitrate-N concentrations were calculated from those ports at the downgradient multiport wells that were determined to be monitoring the groundwater recharged from subdivision sources, as discussed in Section H.

The mass of nitrogen discharged from the cuttings was calculated using the average nitrate-N concentrations and the volume of discharge (mg/l x m^3/year x 0.001 kg/year).

The groundwater flow times across the cuttings were calculated using the

Characteristic	Jordan Acres Cutting	Village Green Cutting
Width of cross section (m)	180	180
Length of flow path along cutting (m)	360	850
Depth of subdivision impacted water (m)	3.4	7.7
Area of cross section discharging groundwater from the cutting (m^2)	612	1400
Average linear groundwater flow velocity - low (m/day)	0.34	0.26
Average linear groundwater flow velocity - medium (m/day)	0.49	0.37
Average linear groundwater flow velocity - high (m/day)	0.64	0.49
Discharge of subdivision impacted groundwater from cutting - low (m^3/year)	23,000	39,000
Discharge of subdivision impacted groundwater from cutting - medium (m^3/year)	33,000	57,000
Discharge of subdivision impacted groundwater from cutting - high (m^3/year)	43,000	74,000
Average nitrate-N concentration of groundwater leaving cutting (mg/l)	9.0	13.6
Mass of nitrogen in discharge from cutting - low (kg/year)	200	530
Mass of nitrogen in discharge from cutting - medium (kg/year)	300	770
Mass of nitrogen in discharge from cutting - high (kg/year)	390	1010
Groundwater flow time across cutting - slow (years)	2.9	9.0
Groundwater flow time across cutting - medium (years)	2.0	6.3
Groundwater flow time across cutting - fast (years)	1.5	4.8
Average yearly precipitation (cm)	78	83
Volume of water recharged assuming no drainfields, no impervious areas, and recharge = annual precipitation - 53 cm (m^3/year)	16,000	46,000

Table 11. Results of nitrogen and water budget calculations based on field data obtained from Jordan Acres and Village Green subdivisions.

length of the subdivision and the range in average linear groundwater flow velocities (meters x days/meter x 1/365 = years).

The average annual precipitation was calculated based on the average precipitation that occurred over the groundwater flow time beneath the subdivision during the study period (Jordan Acres, 1986 to 1990; Village Green, 1981 to 1990).

The estimated volume of water that would recharge the aquifer under natural conditions (i.e., if there were no human impacts) is estimated by using precipitation minus 53.3 cm evapotranspiration. This volume is included to demonstrate the increase in recharge that occurs in subdivisions on sandy soils. The volume of recharge from a subdivision is expected to be greater than the amount from an equal area of natural land because more of the water that falls on impervious surfaces (90 percent of precipitation) will recharge to the groundwater, as compared with about 25 percent from vegetated areas. This is discussed in greater detail in Section E.

G. Comparison of the Results of the Nitrogen and Water Budgets Determined by Two Separate Methods

The nitrogen and water budget results determined using the BURBS computer program and the results based on field data are presented in Table 12. Three field data scenarios are presented for comparison purposes with the BURBS baseline results.

There is very good agreement between the two methods for the Village Green subdivision, both nitrogen loss and water budget calculations are in agreement for the medium to high groundwater flow velocity values. We feel this validates the results of the BURBS model. Results from the Jordan Acres subdivision do not agree as

Budget Results	Average NO_3 in Recharge (mg/l)	Nitrogen Leached (kg/yr)	Water Recharges (m3/yr)
Jordan Acres			
BURBS: Baseline values.	17.2	440	25,000
Field data: Low hydraulic conductivity	9.0	210	23,000
Field data: Medium hydraulic conductivity	9.0	300	33,000
Field data: High hydraulic conductivity	9.0	390	43,000
Water recharged assuming no impervious areas			16,000
Village Green			
BURBS: Baseline values	13.7	930	68,000
Field data: Low hydraulic conductivity	13	530	39,000
Field data: Medium hydraulic conductivity	13	770	57,000
Field data: High hydraulic conductivity	13	1000	74,000
Water recharged assuming no impervious areas			46,000

Table 12. Nitrogen and water budget results for Jordan Acres and Village Green cuttings. Results were calculated using both the BURBS computer program and actual field data.

well. If we use the low hydraulic conductivity value, the water budget for the methods generally agree (Table 12), however, the estimated nitrogen loss (210 kg) would only be about half of that predicted by the BURBS model (440 kg). The primary reason for the discrepancy in this subdivision is the high nitrate-N concentrations predicted by BURBS (17.2 mg/l), compared to that observed from the six downgradient multilevel wells (9 mg/l).

We have no reason to suspect any nitrogen loss by denitrification in the Jordan Acres soils or aquifer and believe the nitrate-N discrepancy between predicted values and multilevel wells is due to the groundwater chemistry data obtained from the monitoring network not being truly representative of overall subdivision impacts. The extreme variability of nitrate-N in monitoring wells downgradient of this subdivision, ranging from 1 to 50 mg/l, (Figure 36, page 114) clearly indicates a wide range of water quality values downgradient of this subdivision as compared to Village Green. At Village Green the well placement was much easier due to the accessibility of a vacant field downgradient of the subdivision and because the groundwater flow is generally parallel to the subdivision layout. At Jordan Acres the monitoring wells were placed where homeowners would permit their installation. Therefore we are not confident that even this large number of multilevel wells is providing a representative sample of groundwater at the Jordan Acres site. We feel that the results of the BURBS simulations are more representative of actual recharge characteristics than the data obtained from the monitoring wells.

H. Groundwater Quality Downgradient of Subdivisions

A number of multilevel wells were installed downgradient of each subdivision to determine subdivision impact on groundwater quality, determine the variability of water chemistry horizontally and vertically downgradient of the subdivision, and to determine changes in water chemistry over time.

Figures 3 and 4 (pages 33 and 34) show the location of downgradient wells used for this part of the project. Initially (in 1987), there were only two multilevel wells installed downgradient of each subdivision; E4 and W4 in Jordan Acres, and S4 and N4 in Village Green. Data from these wells was not considered to be sufficient to evaluate subdivision impact on groundwater. Additional multilevel wells were installed in 1989 to determine the variability of groundwater chemistry downgradient of the subdivisions, to provide better quantitative estimates of water chemistry leaving the subdivisions, and to aid in making recommendations on future well designs for subdivision evaluations.

Comparing upgradient water chemistry with downgradient concentrations can be very misleading. The shallowest downgradient well ports are sampling water recharged only from the subdivision. Mid-depth wells are believed to be sampling a mixture of water recharged from upgradient of the subdivision and that originating from within the subdivision, while deeper well ports are sampling water originating only in upgradient areas. Changes in water chemistry with depth were very useful in identifying the parts of the aquifer impacted by recharge from different land uses. The monitoring well system installed in Village Green turned out to be easier to quantitatively evaluate than that for Jordan Acres, however, both show good relationships between water quality, well depth, and land use.

The depth of groundwater impacted by the subdivision is important in calculating the extent of subdivision impact and also to validate the nitrogen mass balance model. This depth can be estimated using water chemistry graphs of multilevel well data. For the Village Green subdivision (which had a salted four lane highway separating the subdivision from an intensively managed agricultural field upgradient of the subdivision) the relative amounts of chloride and sodium proved to be most useful. Figure 20 presents the chloride to sodium ratio and Figures 21 and 22 show the chloride and sodium graphs for the same wells. In general, the upgradient water chemistry in Village Green has very high chloride to sodium ratios due to large chloride impacts from agricultural fertilization with low inputs of sodium. Recharge from the highway and from septic systems will increase the concentrations of both chloride and sodium, thereby reducing the chloride to sodium ratio.

Figures 23 and 24 show fairly high concentrations of relative fluorescence and phosphorous in the shallower depths of the aquifer from Village Green subdivision. These chemicals, however, do not move through the aquifer as easily as nitrate-N or chloride, and are used primarily to verify the presence of subdivision impacts.

From these graphs, we estimate the upper 4.7 meters of the aquifer are composed of subdivision originated water, with the 4.7 to 12 meter depth being a mixture of subdivision recharge and that from upgradient of the subdivision. If we assumed this was a 40:60 mixture of the two, the amount of subdivision recharge would be 4.7 meters plus 40 percent of the 4.7 to 12 meter depth, for a total of 7.6

meters of subdivision originated water. The volume of water represented by this effective aquifer thickness compares favorably with the estimate of subdivision recharge from the BURBS model, discussed earlier in this report (Section E).

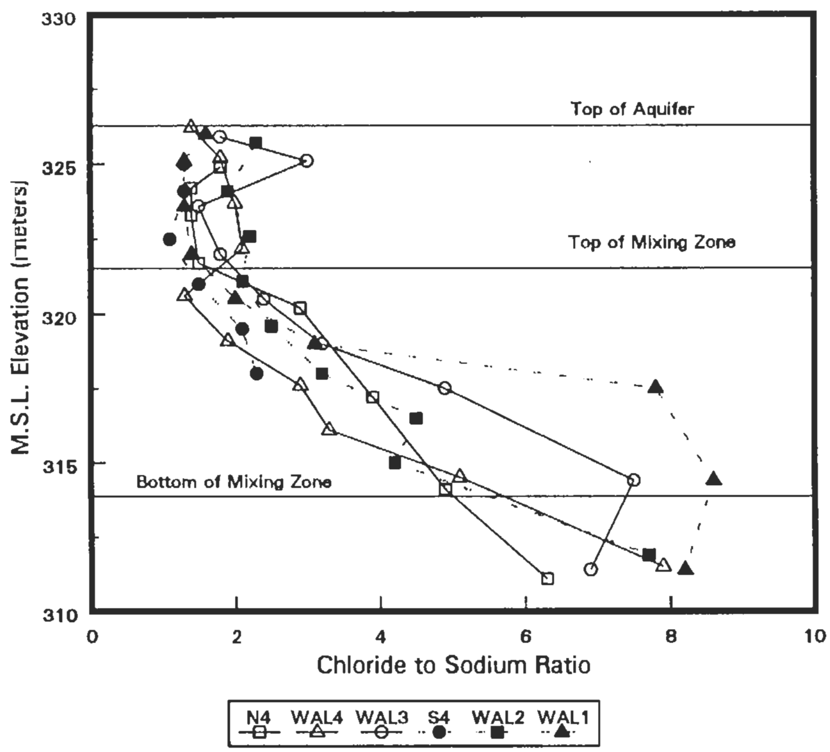

Figure 20. Average chloride to sodium ratios with depth in groundwater from downgradient wells at Village Green.

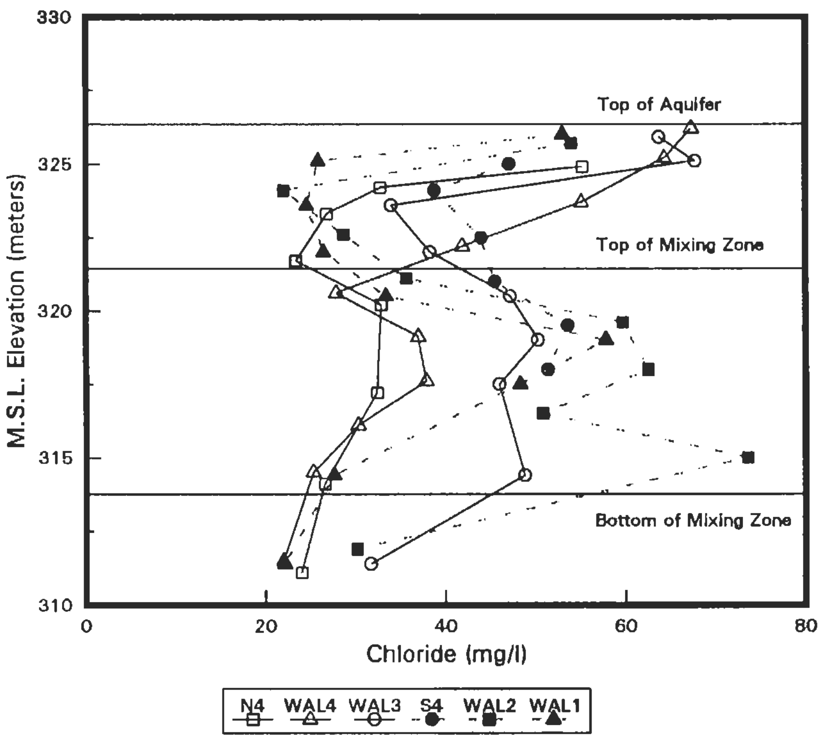

Figure 21. Average chloride concentrations with depth in groundwater from downgradient wells at Village Green.

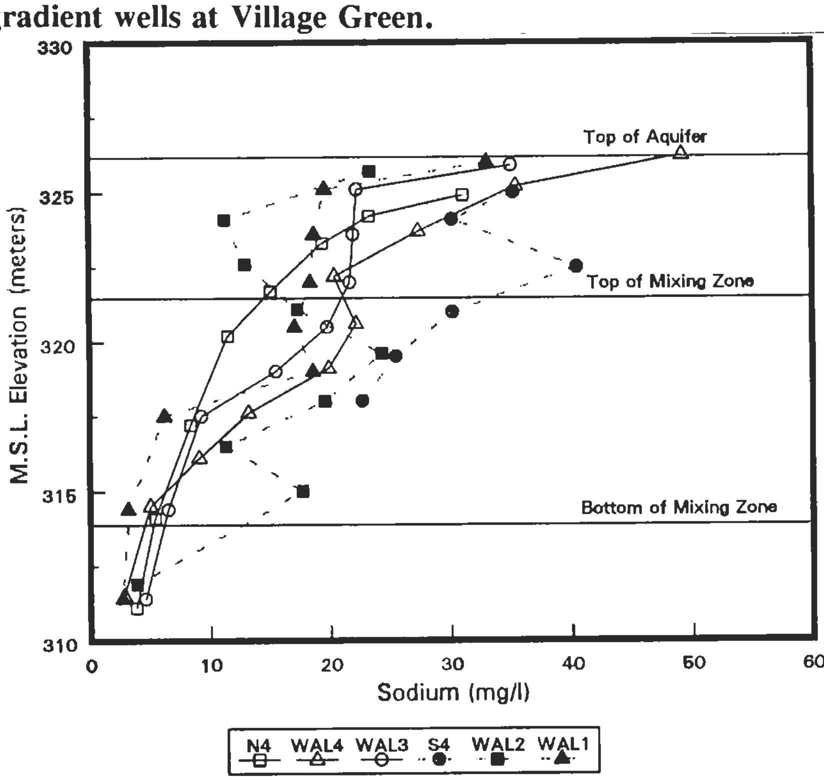

Figure 22. Average sodium concentrations with depth in groundwater from downgradient wells at Village Green.

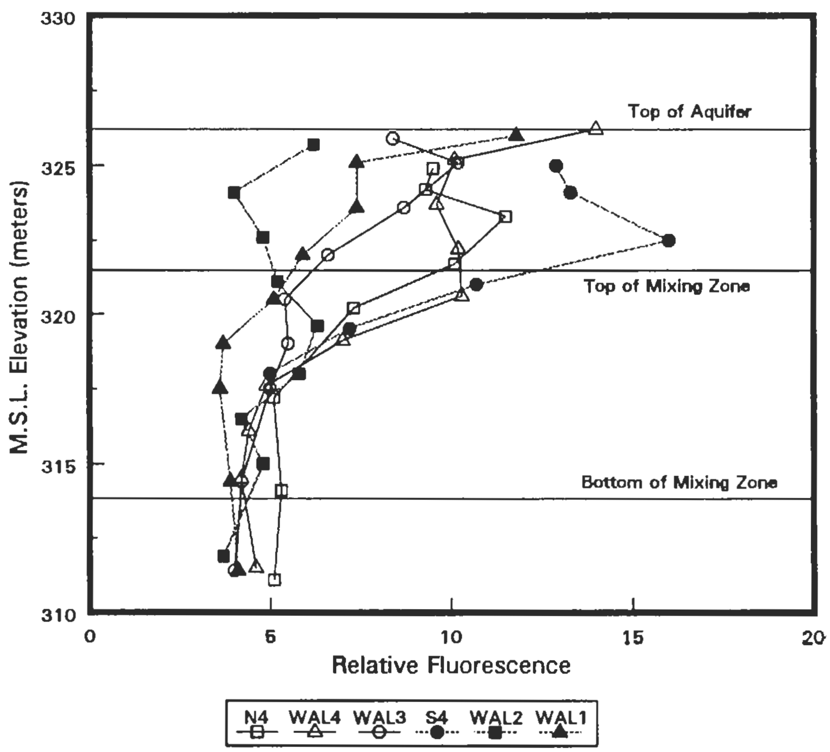

Figure 23. Average relative fluorescence with depth in downgradient wells at Village Green.

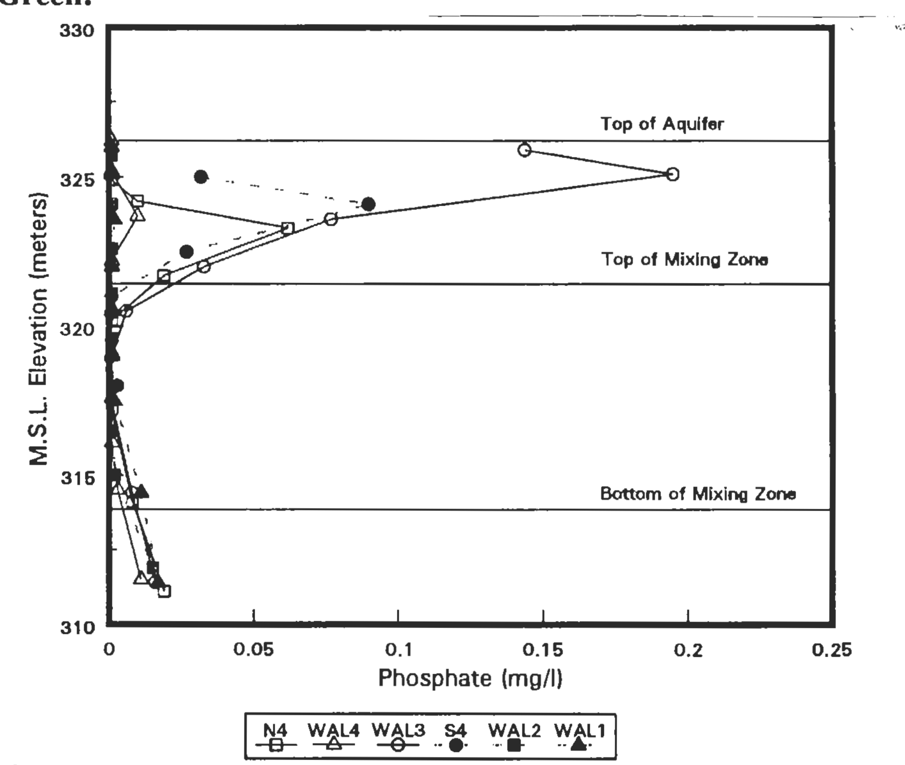

Figure 24. Average phosphate concentrations with depth in downgradient wells at Village Green.

Due to different upgradient land uses at Jordan Acres, sodium to chloride ratios were not as useful. Nitrate-N, chloride, fluorescence, and phosphorous data are presented as Figures 25, 26, 27, and 28. All show similar results and indicate depths of impact of 1.3 meters primarily from subdivision recharge; 1.3 meters to 9.5 meters into the aquifer for the mixed zone, and water below 9.5 meters predominantly from upgradient recharge. Apparently, there is some localized mixing down to 9.5 meters into the aquifer under this subdivision, while other areas show minimal mixing as evidenced by shallow concentrated plumes over 30 meters downgradient of drainfields. To estimate the volume of subdivision impacted water, we used the upper 1.3 meter depths plus one quarter of the next 8.2 meters for a total of 3.4 meters. We used the average nitrate-N concentrations of the upper 1.3 meters to estimate the amount of nitrogen leaving the subdivision as discussed in the previous section. Comparisons of these values to BURBS predictions are discussed in section G.

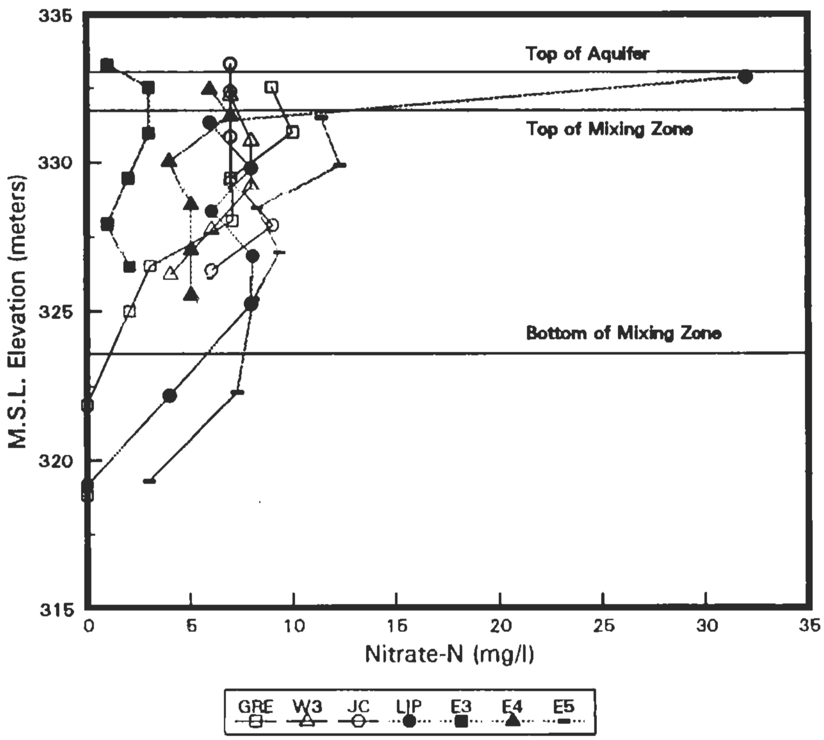

Figure 25. Average nitrate-N concentrations with depth in downgradient wells at Jordan Acres.

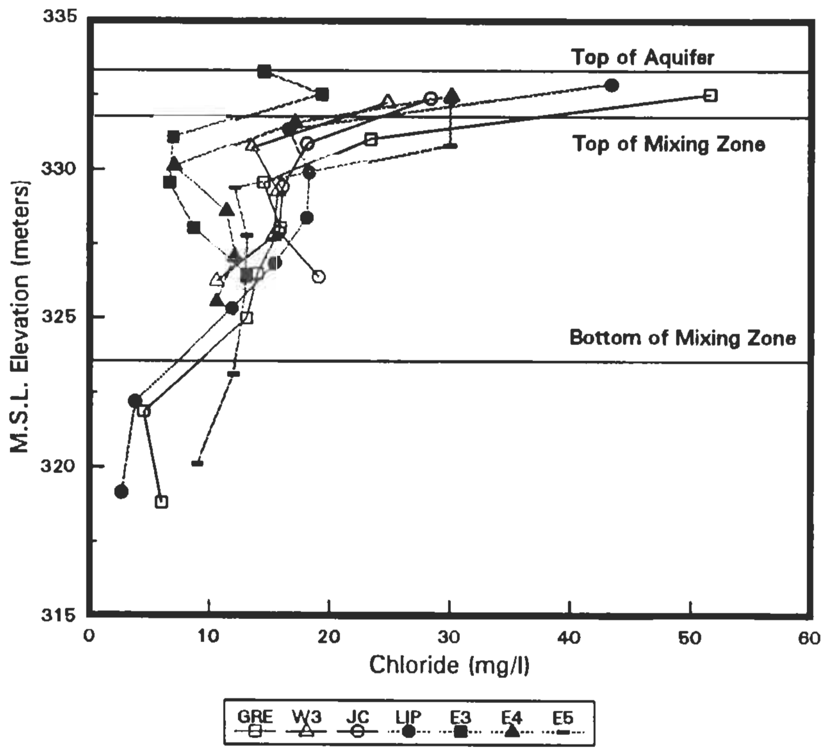

Figure 26. Average chloride concentrations with depth in downgradient wells at Jordan Acres.

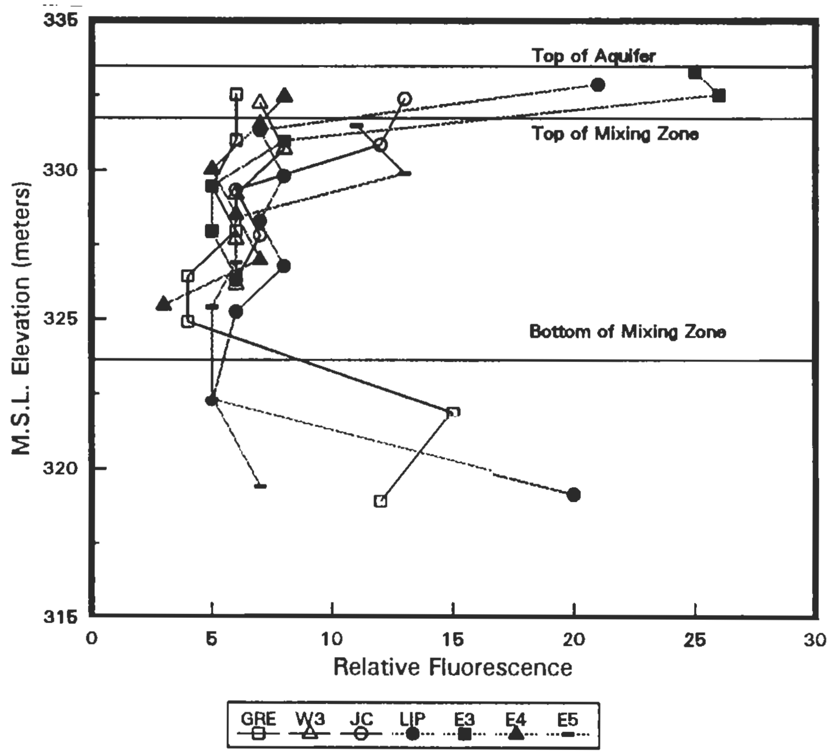

Figure 27. Average relative Fluorescence with depth in downgradient wells at Jordan Acres.

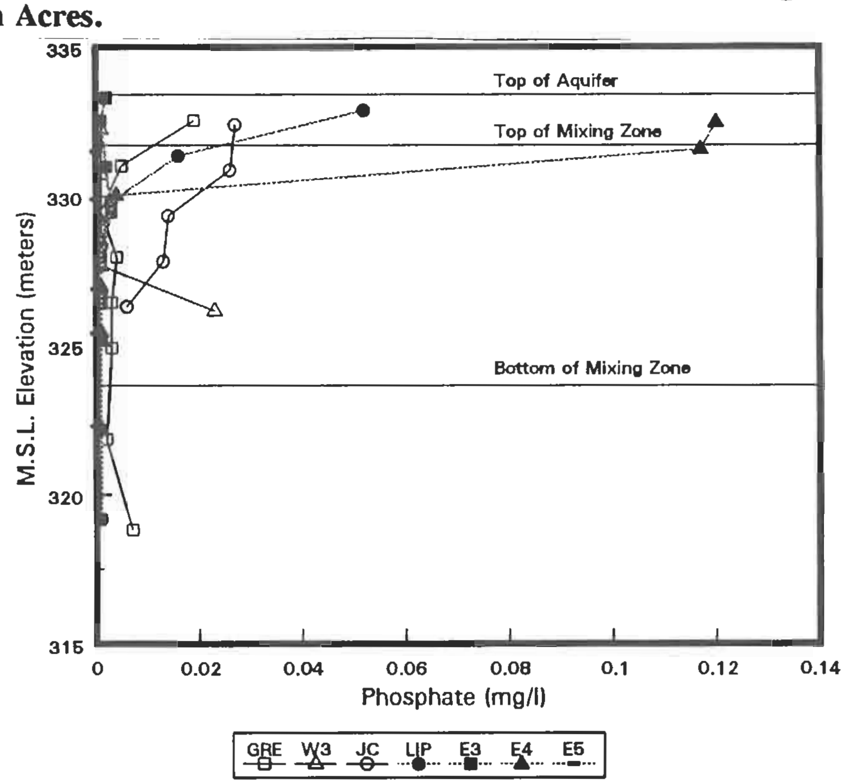

Figure 28. Average phosphate concentrations with depth in downgradient wells at Jordan Acres.

I. Impact of Lawns on Groundwater Quality

Several of the monitoring wells installed throughout the subdivisions were designed to monitor groundwater recharged from lawn areas and were not significantly impacted by septic systems. Data for five of these wells are presented in Table 13. The well that showed the greatest groundwater impact (MCD LD) was downgradient of a lawn that received four fertilizer applications per year. Chemistry data for all sampling dates from the upgradient and downgradient wells monitoring this lawn are presented in Table 14 and Figure 29.

Well Location	Well Point	# of Samples	Monitoring Period	NO_3 (mg/l)	CL (mg/l)	NA (mg/l)	PO_4 (mg/l)
FIR	SD	11	July '88 - Jan '90	4.0	13.3	5.1	<0.002
MCD	LD	12	June '88 - Aug '90	7.8	14.3	5.1	<0.002
E2	22	8	Sep '87 - Aug '89	2.9	4.8	3.9	0.011
E3	25	14	July '87 - May '90	2.7	19.3	12.1	<0.002
S3	22	12	Sep '87 - Mar '89	5.3	37.8	14.7	<0.002
Average				4.5	17.9	8.2	0.001

Table 13. Average groundwater chemistry data from wells primarily impacted by lawns.

The upgradient well was consistently low in nitrate-N (less than 1 mg/l), while the downgradient well fluctuated from 1 to 14 mg/l, with an average of 7.8

There appears to be a seasonality to this data, with the highest concentrations found in summer and fall, and lowest concentrations in winter and early spring. This pattern would be consistent with the time of year the fertilizer is applied. Winter sampling occurred when no recent recharge had occurred from the lawn area, and samples would represent chemistry more characteristic of upgradient land use. Early spring samples correspond to spring recharge events, when a lack of large amounts of residual nitrate-N combined with larger volumes of recharge produced reduced nitrate-N concentrations. It is widely believed that little residual nitrate-N remains in sandy soils over winter due to removal of most of the nitrate-N during fall leaching.

Sample	REE-LU			MCD-LD		
Date	NO3	Cl	Na	NO3	Cl	Na
06/30/88	0.5	7	2.0	1.0	6	2.0
08/05/88	0.5	6	0.8	3.5	9	2.3
10/20/88	0.8	7	1.5	9.8	13	3.0
01/18/89	0.8	5	2.5	7.8	16	5.5
03/31/89	1.0	7	1.	5.8	9	3.0
05/26/89	1.2	6	1.6	2.2	10	4.1
08/08/89	1.5	3	1.0	9.8	23	2.0
09/08/89	0.8	5	1.5	14.4	30	11.5
10/26/89	1.2	<1	1.5	13.0	27	7.0
01/08/90	1.0	3	1.5	14.2	11	8.5
02/14/90	0.5	3	1.4	4.8	8	6.9
05/17/90	<0.2	4	1.6	7.0	9	4.8
Count	12	12	12	12	12	12
Average	0.83	5	1.5	7.8	14	5.1
Std.Dev.	0.354	2.2	0.43	4.37	7.7	2.84

Table 14. Chemistry data from two wells monitoring the groundwater upgradient (REE-LU) and downgradient (MCD-LD) of an intensively managed lawn in Jordan Acres.

These patterns indicate that fall fertilization on sandy soils should not be done with fertilizers containing nitrate-N. If fall fertilization is practiced the residents should use slow release fertilizer applied late in the year to prevent its convergence to nitrate-N. The mean value of 4.5 mg/l nitrate-N for the five sites (Table 13) is consistent with private well data from the Village of Park Ridge, a nearby municipality; which is on public sewer and has private wells. These data were used in the mass balance calculations for the subdivisions, as discussed earlier.

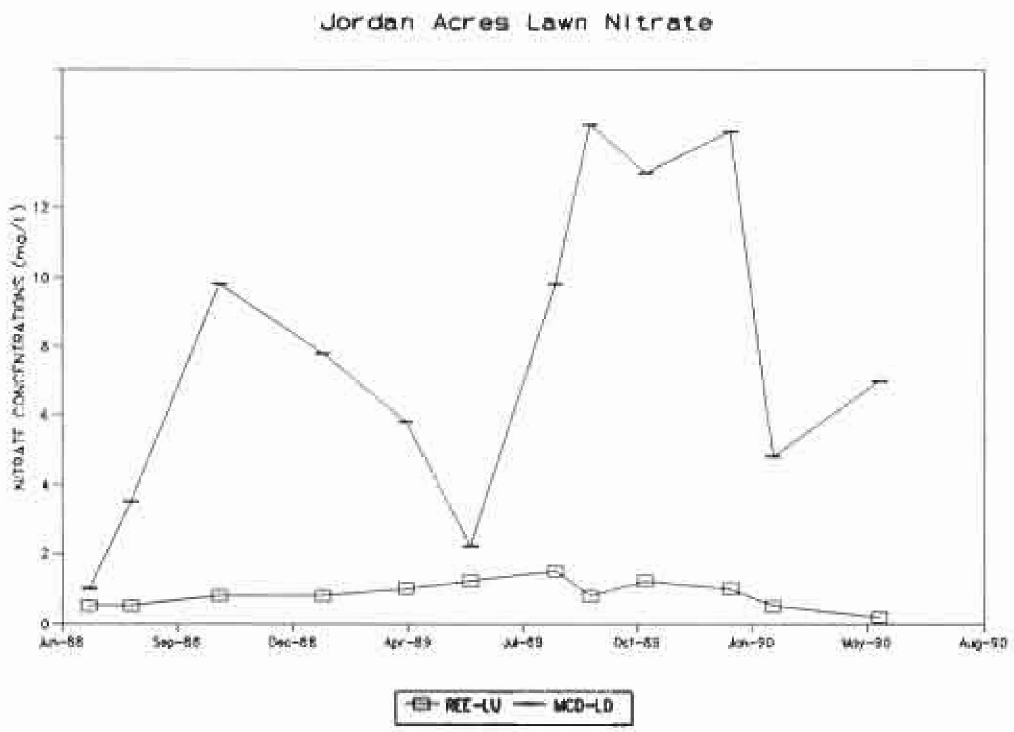

Figure 29. Plot of groundwater nitrate-N concentrations vs time for wells REE-LU and MCD-LD in Jordan Acres. REE-LU well is upgradient of a lawn, MCD-LD is downgradient of the lawn.

J. Septic System Research Site Results

Several sites were instrumented with monitoring wells designed to determine the impact of septic systems on groundwater quality. Many of these were also used for evaluation of the trace organic chemical impacts (described in section O).

Of the ten sites chosen for the septic system monitoring, only five turned out to generate useful data. The monitoring wells at the other sites apparently missed the contaminant plume or only sampled it seasonally. One of the sites (REE) was further investigated to determine the location and size of the plume, and its dispersal with distance downgradient of the drainfield.

Average water chemistry data for the wells that were found to be in at least part of the contaminant plume are presented in Table 15 A and B. Results from corresponding upgradient wells are also shown in these tables.

At sites MCD, ZAK, and ENG, the downgradient wells were apparently near the edge of plume, as indicated by the variable results (Appendix A). The two wells at site KOP were originally intended to sample lawn impacts, however, they appear to be impacted by an upgradient drainfield. Wells at sites FIR and LOD miss the plume entirely. Site locations are shown in Figures 3 and 4 (pages 33 and 34)

The original wells at site REE also appeared to totally miss the plume, however, additional wells installed at this site located and tracked the contaminant plume from this septic system. The results from sites where the contaminant plume was intersected and monitored are presented in Table 15 A. Some of the wells appeared to be located near the edge of the plume, as evidenced by the wide fluctuations in chemistry results (Figure 15 B). As shown on Tables 15 A and B, the distance between the drainfield and the monitoring wells also varied, which may account for some of the observed variability.

Results from wells at Sites REE, AMD, BAR, MOR, and S1 are believed to be most representative of shallow groundwater within 6.6 meters downgradient of the

15 A. Wells consistently monitoring the contaminant plume.

Well Location	Well I.D.	# of Samples	NO3-N (mg/l)	Cl- (mg/l)	Na (mg/l)	PO4 (mg/l)	Fluorescence	Distance from Drainfield (m)
REE	SU	12	0.7	3	1.3	<0.002	5	Upgradient
REC	SDS	11	48.4	36.2	19.8	<0.002	30	1.5
AMD	SU	9	2.9	44	12	0.004	7	Upgradient
AMD	SD	9	33.7	133	108	7.0	35	3
BAR	SDA	9	30.9	69	69	6.5	125	8
KEP	MED	8	40.9	85	13	5.03	74	29
MOR	SU	7	5.6	22	16.5	<0.002	16	Upgradient
MOR	SD	7	19.2	51	41	3.5	31	3.2
ENG	SUA	9	8.5	80	48.6	<0.002	11	Upgradient
LIP	25	6	32	43	54	0.052	21	25
VS1	22	15	24	78	54	0.452	36.4	16

15 B. Wells occasionally monitoring the contaminant plume.

Well Location	Well I.D.	# of Samples	NO3-N (mg/l)	Cl- (mg/l)	Na (mg/l)	PO4 (mg/l)	Fluorescence	Distance from Drainfield (m)
ENG	SUA	9	8.5	80	48.6	<0.002	11	Upgradient
S2	22	13	16.4	22	13	0.004	4.9	20
ENG	SDC	10	10.9	65	42	<0.002	12	16
FIR	SU	10	3.7	23	7.6	0.001	7	Upgradient
MCD	SD	11	14.1	32	196	<0.002	18	13.3
ZAK	SD	9	11.8	20	14	<0.002	14	7.5

Table 15. Average groundwater chemistry of wells in contaminant plumes originating from nearby septic systems.

drainfield. Wells at sites REE, AMD, BAR, and MOR were installed specifically to monitor the drainfield and were apparently located near the middle of the contaminant plume. The 7.2 meter well at S1 also did a good job of sampling the contaminant plume from an upgradient septic system, as did the LIP 8.2 meter well in Jordan Acres Subdivision.

Some wells showed considerable variability between sample dates, indicating they were near the edge of a contaminant plume. This suggests that the plumes apparently move horizontally and vertically on a seasonal basis as shown in the plot of groundwater nitrate-N concentrations over time for several of the wells (Figure 30).

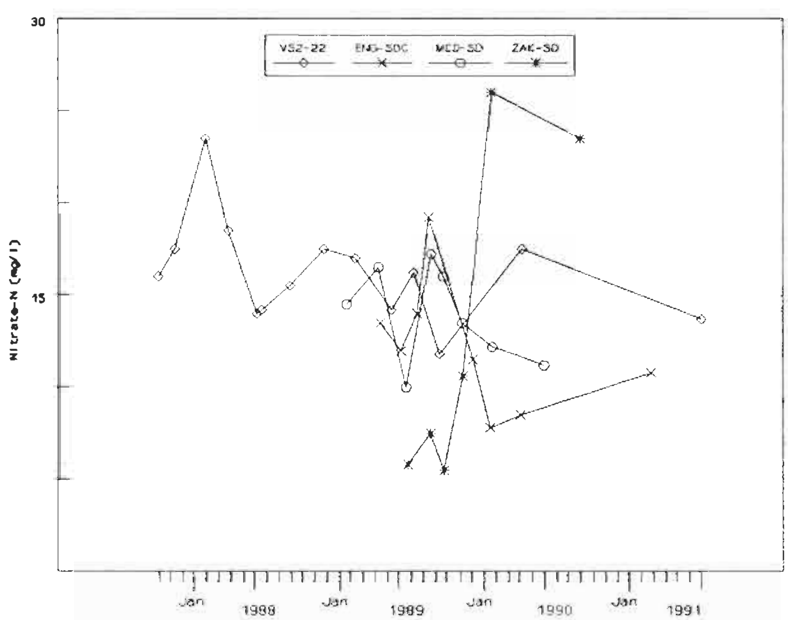

Figure 30. Nitrate-N concentrations (mg/l) in wells S2-22, ENG-SDC, MCD-SD, and ZAK-SD, located downgradient of septic system drainfields.

Detailed Septic Systems Investigation

Detailed site evaluation was conducted at Site REE in an attempt to better identify groundwater impact from individual septic systems. Details of this investigation can be found in Master of Science Thesis of William VanRyswyk, 1993. Figure 31 shows the monitoring well network installed at the REE site, along with average nitrate-N concentrations for each well.

From this figure alone it is obvious the initial wells (REE-SD and E1) were not in the contaminant plume, even though they both contain shallow well ports

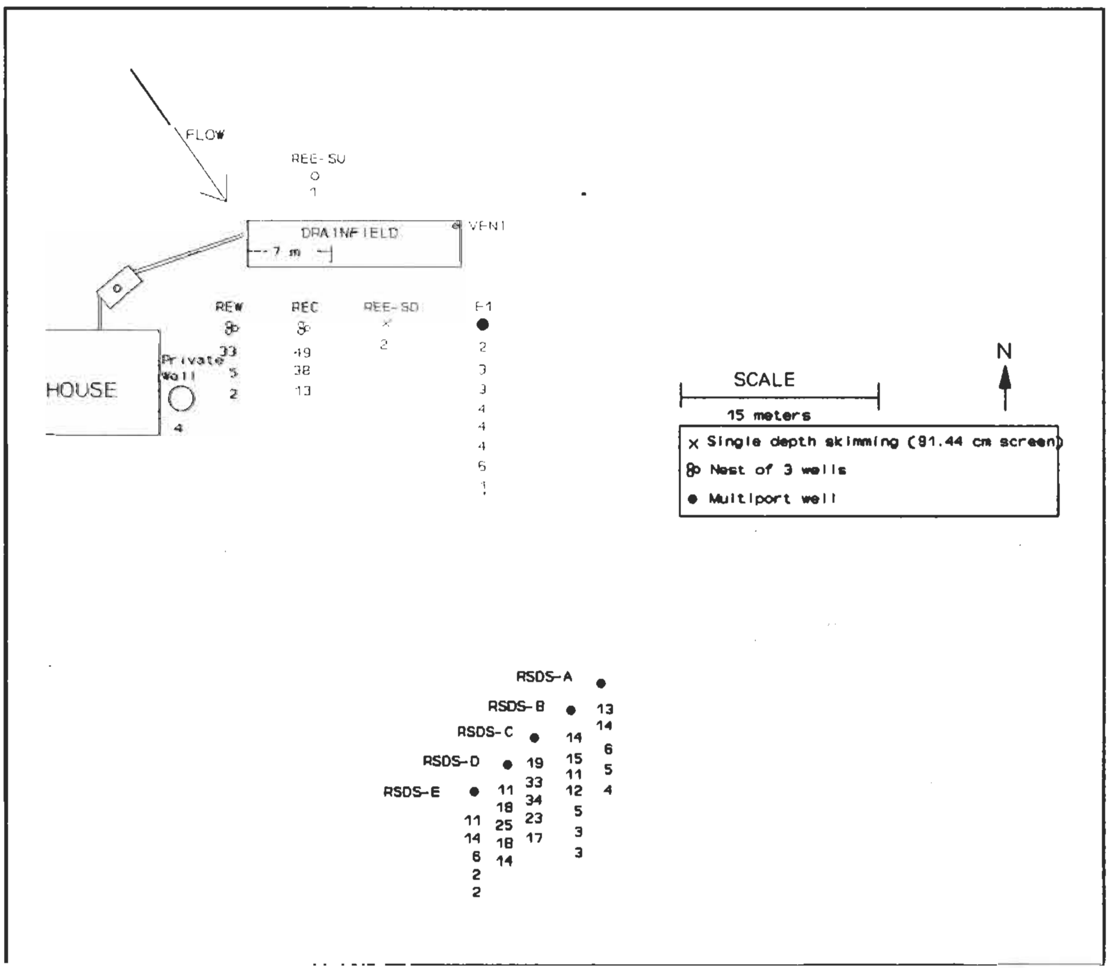

Figure 31. Location of wells at Jordan Acres septic site REE, with average nitrate-N concentrations (mg/l).

downgradient of the drainfield. The contaminant plume was found to be primarily located near well nest REC, with most of the effluent entering the soil and aquifer at the west end of the drainfield. This phenomenon is not uncommon in highly permeable soils as reported by Reneau et al (1989).

To determine the dispersion of the plume with distance, a series of five multiport wells were installed 38 meters downgradient from the drainfield in the summer of 1989. Data from these wells are presented in Figure 32, which shows the configuration of the contaminant plume 38 meters from the drainfield.

Figure 33 shows the water table fluctuation at Site REE, and Figure 34 shows the nitrate-N concentrations in the shallow, medium, and deep wells located 6 meters downgradient of the drainfield. The wells have 30 cm screens spaced 45 cm apart.

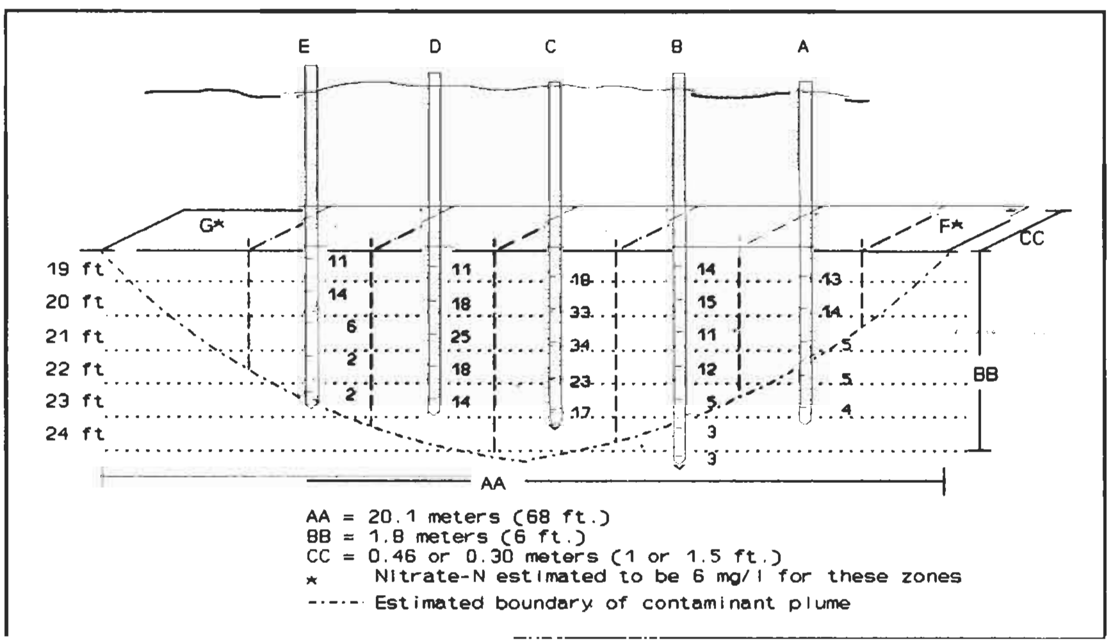

Figure 32. Average nitrate-N concentrations in RSDS multilevel wells 38 meters downgradient of the REE drainfield.

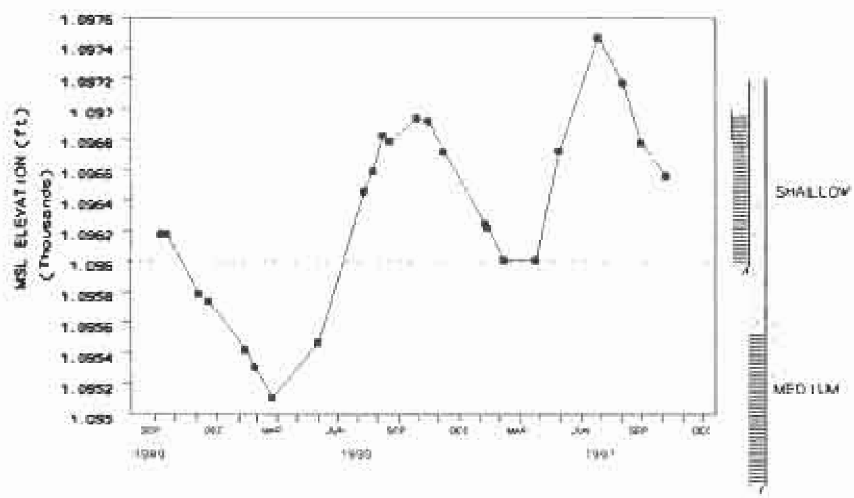

Figure 33. Watertable elevations as measured in well REC-Medium. Dashed line represents screen bottom elevation of well REC-Shallow.

The fact that the well closest to the water table generally had the highest nitrate-N concentration demonstrates that the contaminant plume is quite thin within 6.6 m of its source. A temporary drop in nitrate-N concentrations followed the pumping of the septic tank and a one week vacation by the residents in Sept. 1990, illustrated by a rapid change in groundwater quality following this reduced loading (Figure 34).

Seasonal movement of the contaminant plume toward the west was documented by increased concentrations of nitrate-N in monitoring well REW. The movement is attributed to heavy use of the private well, which only occurred during periods of irrigation (Figure 35). This well is located between the private well and well REC (which was consistently in the plume). Similar shifts in plume location likely occur throughout the subdivision and may account for some of the variability found at other monitoring sites.

The data from the wells near the drainfield and 38 meters downgradient of the drainfield clearly show the contaminant plume remaining very narrow as it moves

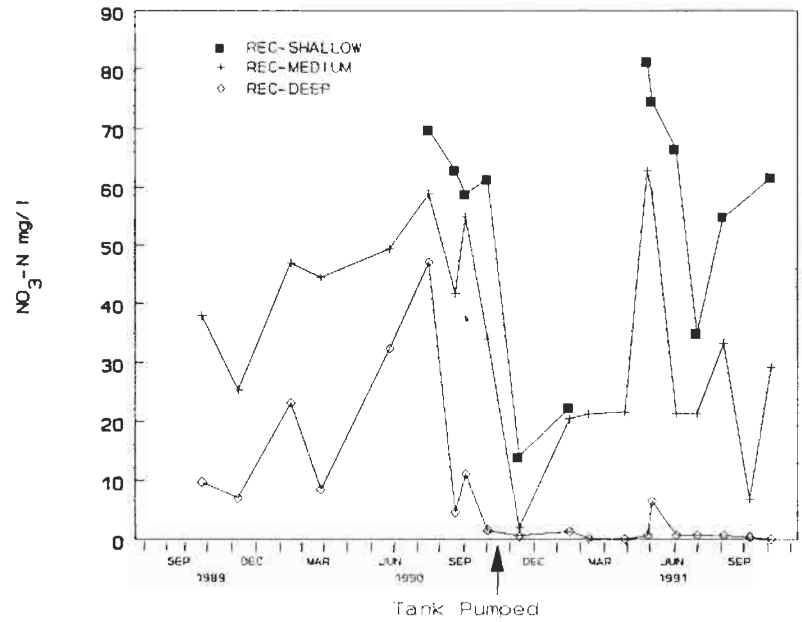

Figure 34. Nitrate-N concentrations for REC-Shallow, Medium and Deep ports. The effects of the septic tank pumping are apparent for several weeks after the pumping event.

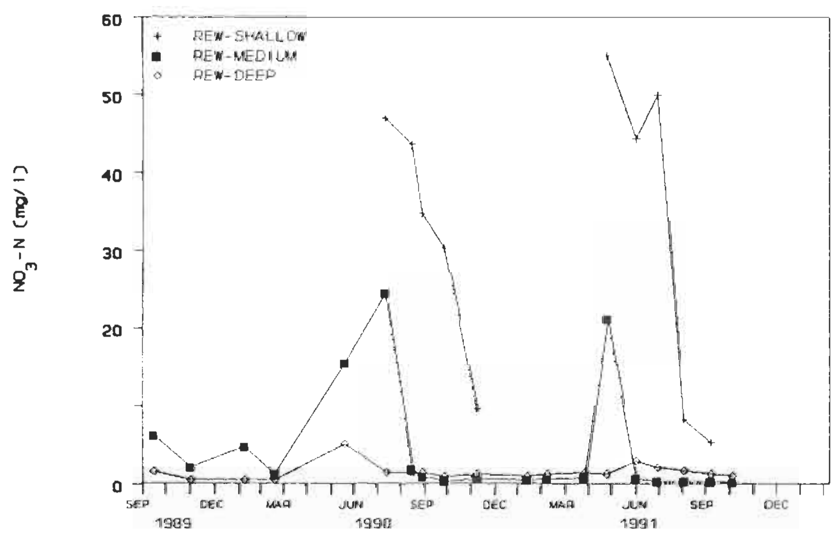

Figure 35. Well REW located 4.9 meters downgradient of a drainfield. Nitrate-N concentrations increase in May, June, July and August corresponding to irrigation well pumping resulting in changes in the plume configuration.

away from its source. A mean nitrate-N concentration of 48.4 mg/l at well REC-Shallow, compared to 24.9 mg/l for the most impacted well 38 meters downgradient of the drainfield, shows a 50 percent reduction due to dispersion and mixing with low concentrations of nitrate-N in upgradient recharge water. Maximum nitrate-N concentrations of 70 mg/l in well REC compared to an average total nitrogen concentration of 79 mg/l in the septic tank suggests minimal nitrogen is removed by the drainfield and shallow aquifer. Comparing total nitrogen to chloride ratios in the septic tank to those in the contaminant plume also indicates little if any nitrogen removal. Concentrations of seven samples (collected from the septic tank in 1991 and the first half of 1992) averaged 79.3 mg/l total nitrogen and 53.3 mg/l chloride, with a mean ratio of 1.5. This is very similar to the 1.4 nitrogen:chloride ratio found for well at REC-Shallow, indicating little if any denitrification at this site. A slight lowering of the ratio to 1.2 at RSDS-C was not found to be a statistically significant change with the Kouskan-Wakis Test, and may be due to mixing in the aquifer rather than chemical or biological changes.

Estimates were made of the total mass of nitrogen entering the drainfield and present in the downgradient network of monitoring wells (Figure 32). Details of the analysis are presented in VanRyswyk (1993). These calculations estimated the total annual per capita nitrogen loading to the drainfield to be 5.5 kg/capita/year. The weighted average nitrate-N in the plume at the RSDS wells times a flow rate of 0.3 and 0.5 meters/day, gives a range of 9.6 to 14.4 kg nitrate-N flowing in this plume 38 meters downgradient of the drainfield. This results in a range of 4.8 to 7.2 kg nitrate-N/capita/year. Similarly, multiplying the maximum concentrations of nitrate-N in groundwater adjacent to the drainfield by the per capita water use gives a value of 5.5 kg/capita/year. All the values are in good agreement and within the range of 3.2 to 8.0 kg/capita/yr reported by Gold et al (1990) and Walker et al (1973 II). They are also all in the range of values found by Shaw and Turyk (1992).

K. Phosphorous Impacts on Groundwater from Septic Systems

Data presented in Table 15 (page 106) show an increase in chemicals other than nitrate-N downgradient of septic systems. The presence of these chemicals is useful in tracking septic system impacts, and was particularly useful in this study in determining the part of the aquifer downgradient of the septic systems that was impacted by septic systems rather than lawns or other sources.

Phosphorous data presented in Table 15 and Figures 24 and 28 (pages 98 and 101) showing elevated concentrations in downgradient multiport monitoring wells in Village Green and Jordan Acres indicate a significant impact of phosphorous on groundwater quality. These data clearly indicate that the sandy soil present in these subdivisions can become saturated with phosphorous within 20 years of septic system use, thereby allowing high concentrations to reach the aquifer five meters below drainfields.

Site BAR had an additional multilevel well (KEP) installed 29 meters downgradient of the septic system. The well KEP-MED averages 5.0 mg/l phosphorous and 41 mg/l nitrate-N. Phosphorous values in the wells downgradient of the subdivision, as shown in Figures 24 and 28, show elevated concentrations in the shallower well ports, as compared to deeper wells (which sample groundwater that originates upgradient of the subdivision). These data indicate that phosphorus can be transported a fairly long distance, which would be important if these subdivisions were located near lakes. Under these conditions lakes could be subjected to the eutrophying effects of high phosphorus loading from groundwater.

L. Variability of Groundwater Chemistry

Downgradient of Subdivisions Relative to Land Use

The variability of groundwater chemistry both vertically and horizontally perpendicular to the groundwater flow was documented by use of a number of multilevel monitoring wells located downgradient of Village Green. Average groundwater data for the downgradient multiport wells are presented in Figure 36. Figure 4 (page 34) shows the location of the Village Green wells relative to groundwater flow, and the land use in that subdivision. The water chemistry of groundwater downgradient of the subdivision is obviously quite variable both horizontally and vertically. The upper three sample ports at well WA2 are particularly low in nitrate-N, compared to other wells only 17 meters away. We believe the concentrations of nitrate-N in these ports are lower because the recharge that occurs over half the flow distance of the subdivision upgradient of this well is from yards and road ditches, and few septic system drainfields. Wells WA1, S4 and WA3 are believed to be sampling groundwater that has recharged in backyard areas where most of the drainfields are located.

A similar wide range of nitrate-N concentrations were found downgradient of the Jordan Acres subdivision, as shown in Figure 36. For interpretive purposes, the locations of septic systems and roads are not as conveniently situated as in Village Green (Figures 3 and 36, pages 33 and 118). It is obvious that some wells intercept contaminant plumes while others miss the impact of drainfields almost entirely.

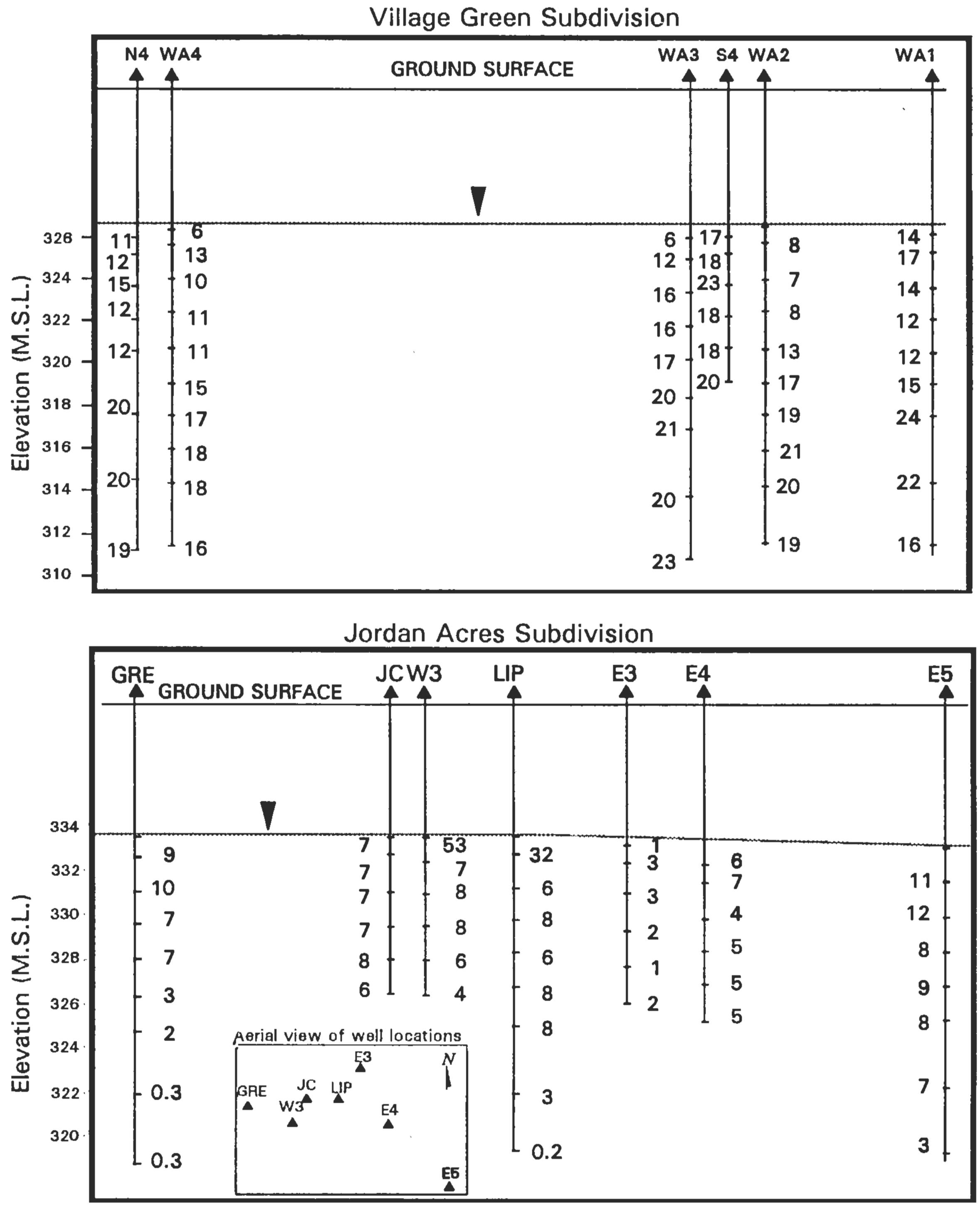

Figure 36. Profiles of average nitrate-N concentrations (mg/l) in wells located downgradient of the subdivisions. View is generally perpendicular to groundwater flow.

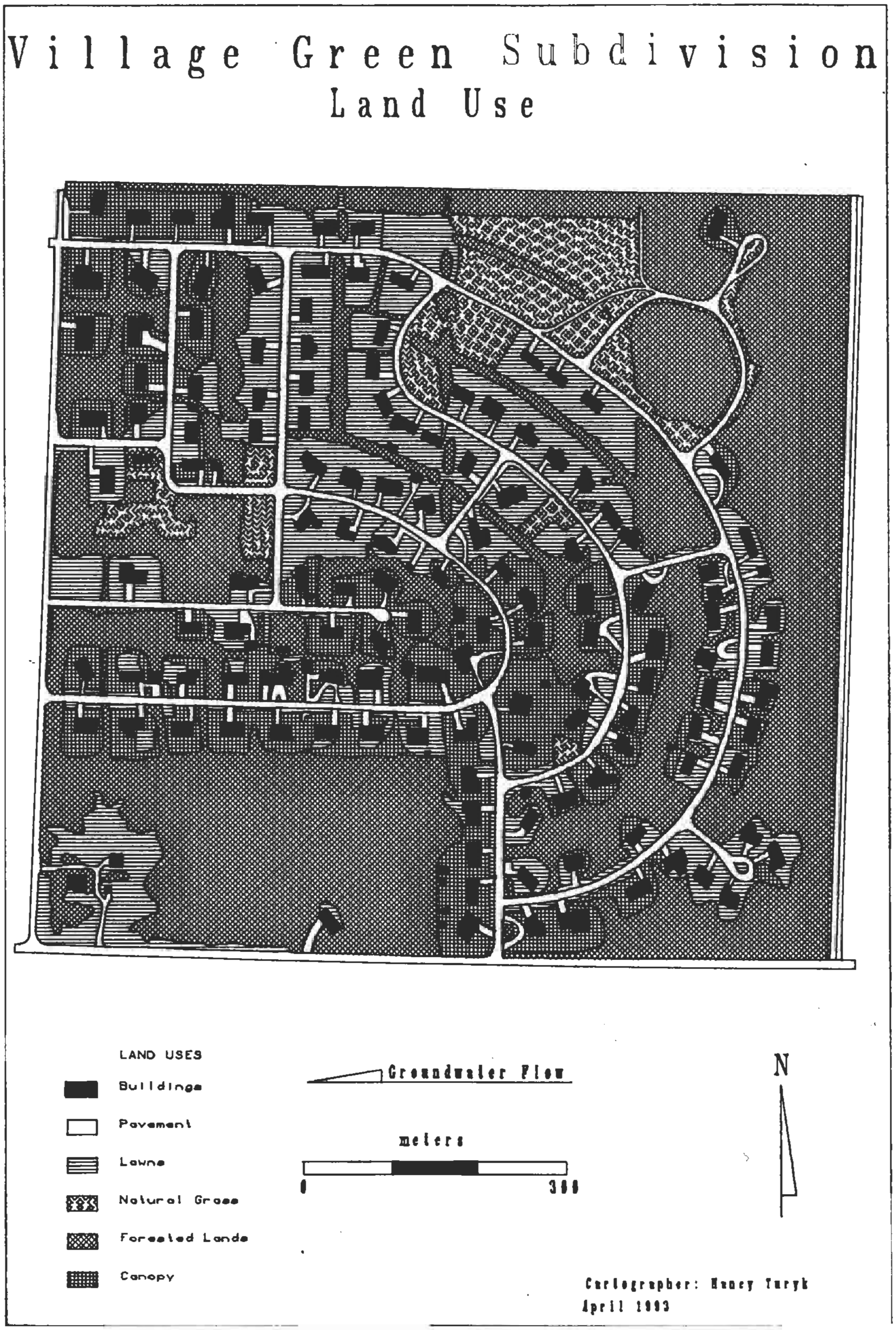

Figure 37. Map showing land uses of Village Green subdivision.

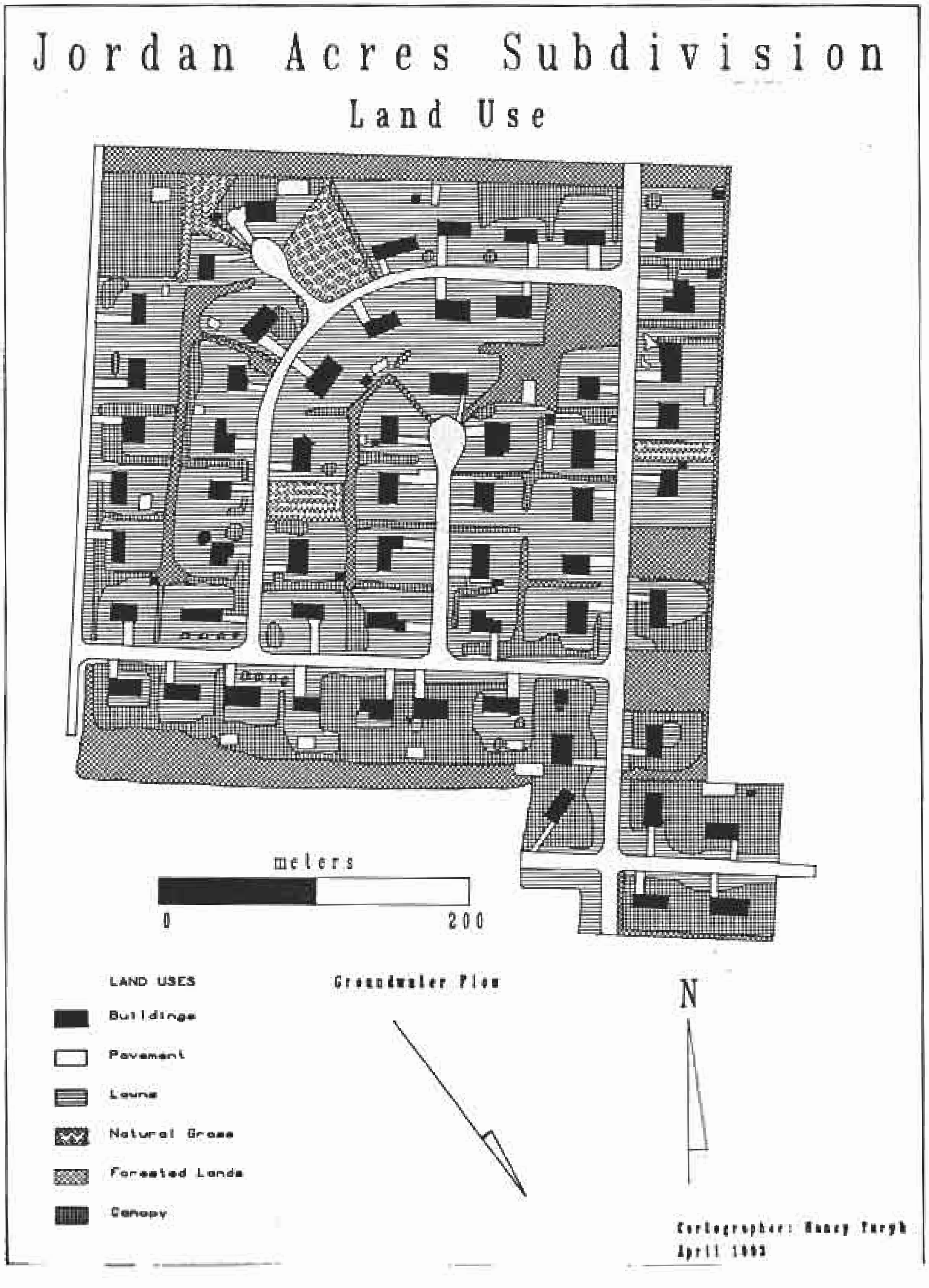

Figure 38. Map showing land uses of Jordan Acres subdivision.

These results lead to several conclusions relative to groundwater impacts from these subdivisions:

1 Water flow and contaminant transport processes occur with minimal mixing, allowing for a high degree of chemical variability in groundwater.

2. Septic systems contribute larger amounts of nitrate-N to groundwater than do lawns.

3. From a practical standpoint, it is not feasible to install a statistically valid, randomly placed monitoring network in a developed subdivision; therefore, the well locations for monitoring subdivision impacts need to be carefully chosen to avoid overestimating or underestimating specific impacts.

4. The use of shallow private wells in subdivisions with onsite waste disposal requires careful consideration of drainfield location and groundwater flow direction to prevent private wells from intercepting the contaminant plumes.

M. Water Chemistry Changes Over Time Downgradient of Subdivisions

Some of the multilevel wells used in this study were sampled and analyzed over a four year time period. During this time period, population density and amounts of groundwater recharge varied considerably. As discussed in Section E, the amount of groundwater recharge can have a large effect on nitrate-N concentrations in groundwater when septic systems are the major nitrate-N source. This is logical because nitrogen inputs to septic systems remain relatively constant year round and from year to year, whereas the amount of recharge (which varies) acts as the primary dilution mechanism. Changes in land use over time can also lead to changes in downgradient water quality. Increases or decreases in population density, and therefore waste generation are the major subdivision land use practices that can cause changes in downgradient water chemistry. The BURBS model projection for low, medium, and high septic system density clearly show these results (Table 10 page

Figures 39 and 40 show the nitrate-N concentrations from each well port of monitoring well nests N4 and S4 for the period of September 1987 to April 1991. Figure 17 (page 76) shows the precipitation amounts and the water table elevation for well PAR during this time period. These figures illustrate the degree of chemical variability that occurs in the shallower ports of the aquifer that sample groundwater originating from subdivision recharge. Well ports 22 through 40, sampling the upper seven meters of the aquifer, show considerable variability over time, compared to well ports 45 to 70, which sample down to 23 meters below the land surface.

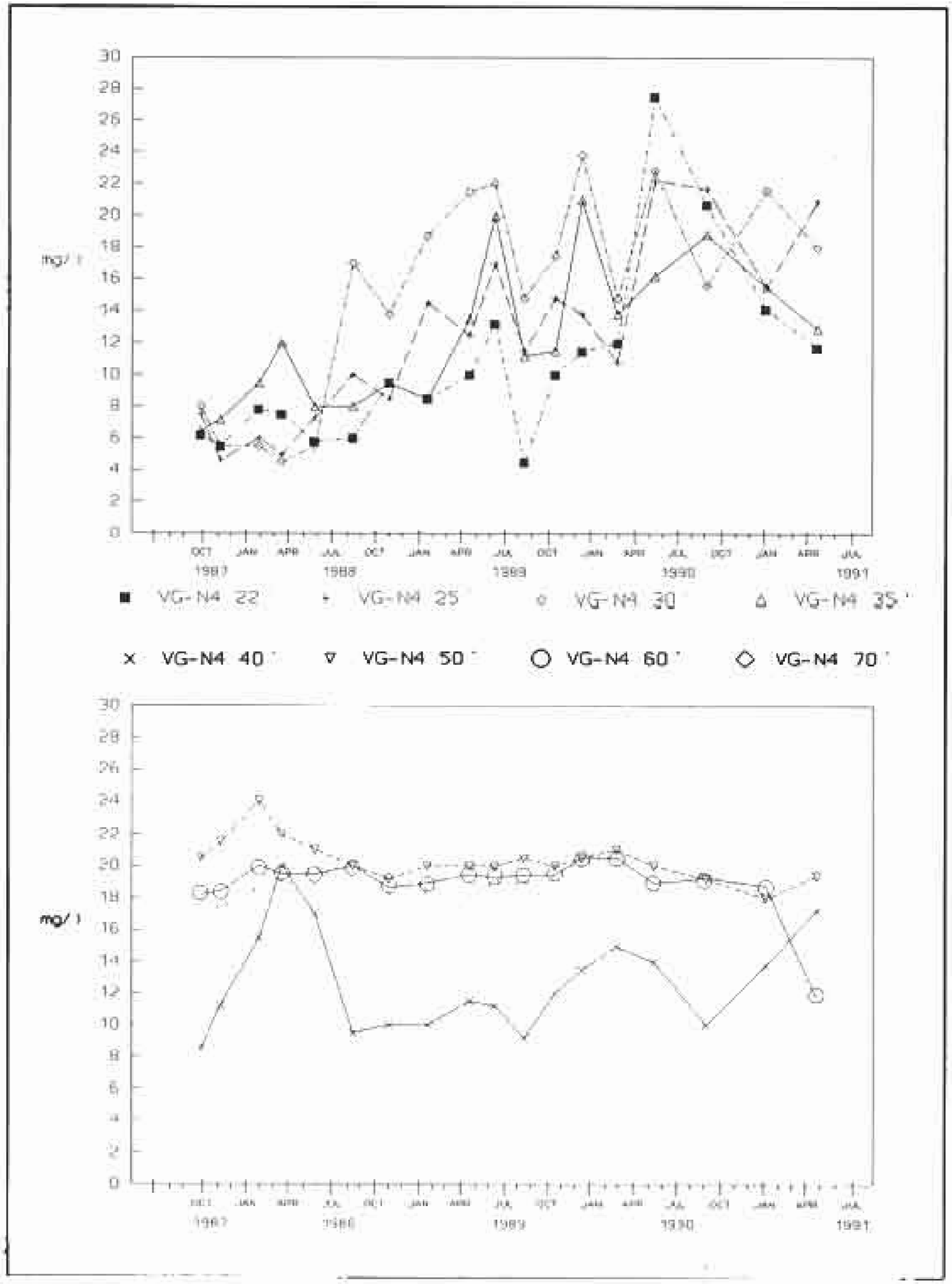

Figure 39. Nitrate-N concentrations (mg/l) of well N4 in Village Green, 1987 to 1991.

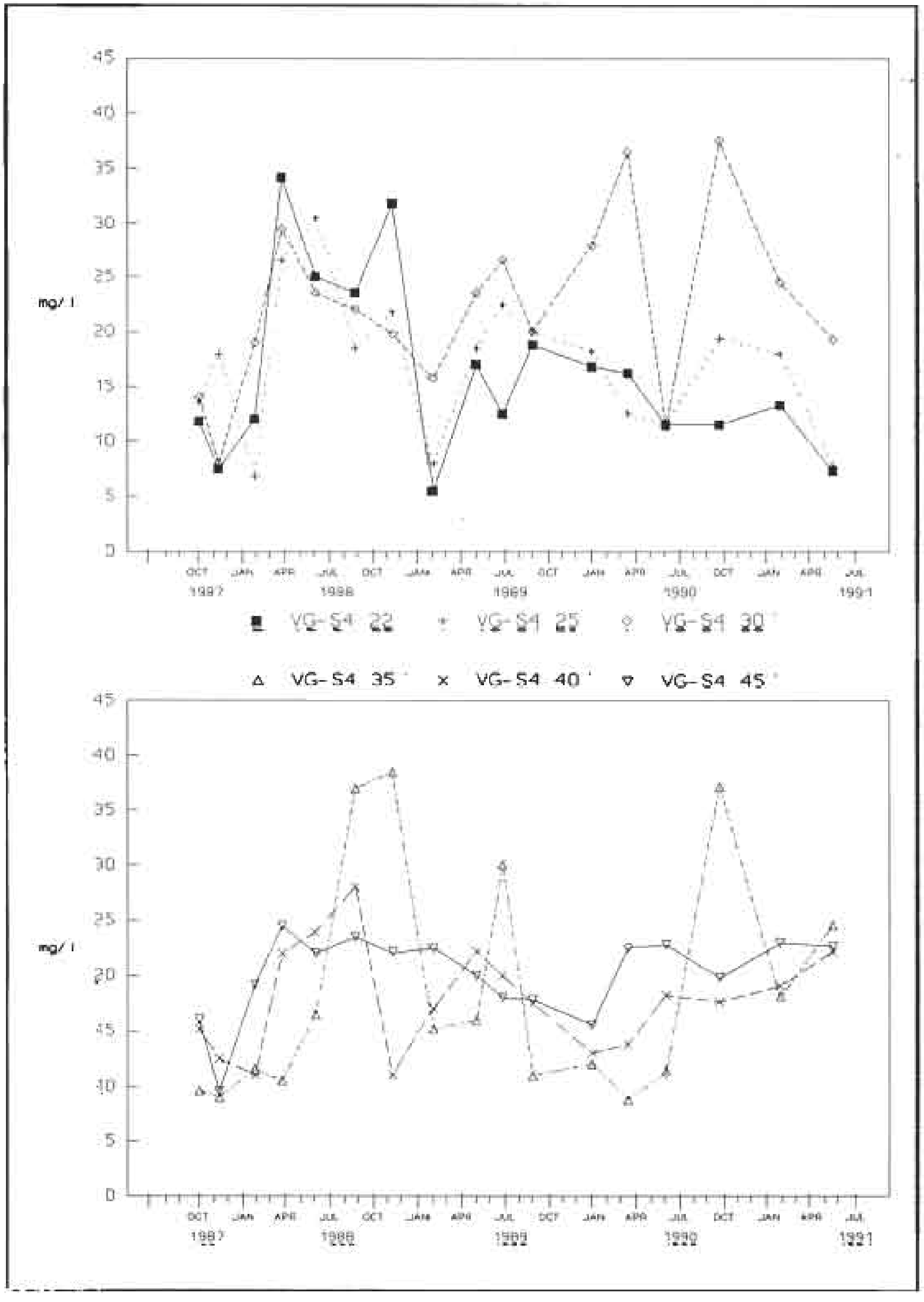

Figure 40. Nitrate-N concentrations (mg/l) of well S4 in Village Green, 1987 to 1991.

Nitrate-N values for the upper seven meters of the aquifer at S4, while showing considerable variability over time, do not show a long term trend of changing water chemistry. Wells sampling the same water depths at N4 do show a definite increase in nitrate-N concentrations over this four year study period. The primary factor contributing to this difference is the increasing amount of development upgradient of N4 during and in the seven years preceding the study. Most of the lots upgradient of S4 were developed at least ten years before the start of the study and their impact would have reached the downgradient well nests before the study began.

Shallow well ports in both well nests show steep increases of nitrate-N from 1987 to 1988. Most of this increase is attributed to the fact that for several years preceding the study there was above normal precipitation and groundwater recharge, which caused the groundwater nitrate-N to be relatively low. In 1988 there were drought conditions and significantly reduced recharge, which resulted in increased concentrations of nitrate-N. Dropping water levels during this time period shown in Figure 17 (page 76), illustrate this effect. Increased recharge in 1989-91 (indicated by a rise in the water table) is considered to have caused greater dilution of septic effluent, thereby reducing nitrate-N concentrations.

Well S4-30 does not show the same decrease in nitrate-N in 1989-91 as the shallower wells. This is attributed to a longer travel time for this water, which still shows increases from the drought years, while shallower wells show the dilution of more recent recharge events.

These data illustrate the wide variety of nitrate-N concentrations that can occur vertically, horizontally, and over time downgradient from land uses such as subdivisions. Drawing conclusions from a single sample from a single depth at one point in time is virtually impossible. Use of carefully located multilevel wells, and sampling for a number of years is essential for any sound conclusions to be drawn relative to subdivision impacts on groundwater quality.

N. Geophysical Techniques

Electrical Resistivity (ER) and Ground Penetrating Radar (GPR) were evaluated for their potential value in locating plumes from septic systems. The consistent geologic nature of the alluvial outwash sand, the relatively shallow watertable, and an extensive monitoring well network provided ideal conditions for such an evaluation. If a geophysical technique was determined to be effective at locating septic plumes in the sand plain, it would be a useful tool for siting local private water supply wells, and prove very useful for future hydrogeological investigations in the area.

ER was evaluated at two different septic systems where sufficient downgradient space was available for the proper electrode spacing. The space required by the electrode configuration proved very limiting in the subdivision environment. Although increases in electrical conductivity approaching five times that of background were measured in groundwater at one of the sites, the narrowness of the septic plume as compared to the required electrode spacing likely resulted in non-impacted areas masking the affect of the impacted zone. Interferences from underground and overhead utilities (prevalent throughout the subdivision) also made interpretation of the measured resistance readings very difficult. It was concluded that ER was of limited value for locating septic plumes in the subdivision environment.

GPR was evaluated at the Jordan Acres septic study site where the downgradient septic plume was well defined, where the technique proved ineffective at locating the edge of the septic plume. GPR was also evaluated at several nearby mound systems where monitoring well networks were also installed. The depth to groundwater in this region was generally much shallower (<3.3 m) and the radar seemed to respond with a reduced signal over the plume at one or two of the sites, but results later proved inconclusive. The groundwater contaminant plumes from septic systems are apparently not of sufficient strength for this technique to be effective, particularly in areas where five to eight meters of unsaturated sands exist above the plume.

O. Trace Organic Chemical Investigation

A number of private wells, multiport monitoring wells and septic system monitoring wells were sampled and analyzed for volatile organic compounds (VOCs) and polynuclear aromatic hydrocarbons (PAHs). Detailed results of these studies are reported in Henkel (1992).

Occurrence of trace organic compounds from septic system disposal was investigated by installing monitoring wells to sample the upper 0.9 m (3 ft) of groundwater downgradient of, and within 9 m (30 ft) of drainfields. A total of five systems were evaluated where the downgradient monitoring wells intercepted the contaminant plume from the septic systems (Table 16).

Samples from the five systems on three dates were analyzed for VOCs. Four of the five systems had detectable VOCs present on at least one sample date. No sites had VOCs present on all three sample dates, illustrating the ephemeral nature of VOC contamination of groundwater from household practices. The chemicals found, and the measured concentrations are presented in Table 16.

Benzene, toluene, dichloroethane (DCA), trichloroethane (TCA), and tetrachloroethylene (PCE) were identified in the groundwater samples. Additional peaks were occasionally present, but not in the group of chemicals identifiable by EPA Methods 5030/601-602. Some detects of VOCs were found in private wells and downgradient monitoring wells, however the concentrations were low and not reproducible on subsequent sampling dates. Occurrences appeared to be localized and in low concentrations.

These data clearly indicate VOCs can reach groundwater from household chemical use and disposal into septic systems. The ephemeral nature of occurrence, and relatively low concentrations help minimize the health impact, however, more significant concentrations are possible if larger quantities were disposed of by homeowners.

WELL	VOC Analysis			PAH Analysis		ECD Analysis	TSD Analysis
	Oct 1988	Jan 1989	April 1989	Jan 1989	April 1989	April 1989	April 1989
MCD-SUA	nd	-	-	-	-	-	-
MCD-SDB	nd	nd	2.53 BEN	nd	nd	+	+
MOR-SUB	2.1 TOL	-	-	-	-	-	-
MOR-SDA	8.8 DCA 21.6 TCA	2.4 DCA 5.1 TCA	nd	nd	nd	*	*
BAR-SUB	1.9 PCE	-	-	-	-	-	-
BAR-SDA	nd	nd	nd	nd	*	*	+
BAR-SDC	nd	nd	nd	nd	bdl	*	*
KOP-SUB	nd	-	-	-	-	-	-
KOP-SDA	nd	-	2.17 BEN	-	bdl	*	+
AMD-SUA	2.4 TOL	-	-	-	-	-	-
AMD-SDB	nd	nd	nd	nd	nd	*	*

Results are $\mu g/l$ (parts/billion)

nd	chemical was not detected in the sample
-	sample was not analyzed for that analyte
*	sample contained numerous and/or off scale unidentified peaks
+	sample contained detectable concentrations of that chemical group
bdl	sample contained identifiable PAH compounds below the method detection limit
BEN	Benzene
TOL	Toluene
DCA	1,1-Dichloroethane
TCA	1,1,1-trichloroethane
PCE	1,1,2,2-tetrachloroethene

Table 16. Summary of organic chemicals detected in groundwater monitoring well samples between October 1988 and April 1989.

Limited sampling and analysis for PAHs did show some movement of these chemicals to groundwater. PAHs with concentrations less than one ppb were detected in two wells downgradient of drainfields. Chemicals identified in these two wells were benzo(ghi)perylene, phenanthrene, gluoranthene, pyrene, and benzo(c)pyrene. Many of these chemicals are found in household products. The lack of groundwater standards for these chemicals makes it difficult to address the significance of these findings. Further research on the presence of these chemicals in groundwater and concentrations downgradient of septic systems should be conducted.

Other Organic Chemicals

A series of samples were collected from monitoring wells up and downgradient of septic systems and analyzed using a methylene chloride extraction and gas chromatography analysis using electron capture (ECD) and thermoionic specific detectors (TSD). This was done to determine the relative abundance of semi-volatile chemicals in groundwater downgradient of septic systems compared to groundwater upgradient of the same drainfields. Results from all five sites indicated the occurrence of a large number of unidentified organic chemicals in groundwater downgradient of drainfields, but few upgradient. Further research using GCMS is needed to identify these chemicals and their concentrations.

The above series of analyses demonstrated a wide range of organic chemicals moving from the septic tank through conventional drainfields to the groundwater underlying these systems (at a depth of approximately 7 meters and 7 meters downgradient of the drainfield).

Potential Sources of the Detected Organic Compounds

The following information was compiled by Hathaway (1980) from the list of the USEPA Priority Pollutants and the household products they are commonly associated

VOCs are generally a component of metal degreasers, solvents, and detergents. Benzene is found in adhesives, deodorants, paint solvents and thinners, and dandruff shampoo. Toluene is commonly found in solvents, cleaning products, and cosmetics. The compound ,1-dichloroethane (11DCA) is a solvent found in degreasers; 1,1, trichloroethane (111TCA) is a solvent found in drain and pipe cleaners, oven cleaners, degreasers, deodorizers, and photographic supplies; and 1,1,2,2-tetrachloroethene (PCE) is a solvent found in upholstery and rug cleaners, contact cement, degreasers, wax removers, and is a component of pesticides used in garden sprays.

PAHs are common ingredients of dandruff shampoos, eczema and psoriasis remedies, antibiotic creams, athletes foot remedies, deodorants, insect repellents, some detergents, and are commonly used in the manufacture of dyes. These compounds would likely be present in drainfield effluent, but are generally immobilized by particulate absorption, and (except for naphthalene) are relatively insoluble in groundwater (Verschueren, 1983).

The ECD is sensitive to a wide range of semi-volatile halogenated organic compounds such as pesticides and PCBs. The TSD is selectively sensitive to the nitrogen and phosphorous containing semi-volatile organic compounds such as those found in many herbicides. Many chemicals found in household products such as chlorophenols, phthalates, and nitrobenzenes can also be detected by these instruments.

The above lists are not complete, but rather, are a sampling of many commonly used products that contain these organic compounds. These data imply that the wells with VOC, PAH, ECD, and/or TSD detects are being impacted by residential land use and/or septic system discharges.

P. Conclusions

1. Lawns and septic systems contribute nitrate-N to groundwater, with septic systems having a greater impact than lawns.

2. The BURBS mass balance computer model does a good job of estimating subdivision water and nitrogen mass balances as long as the variables are well defined for the subdivision.

3. Housing densities of less than 1.1 to 1.7 dwellings per hectare were found to be needed to maintain nitrate-N concentrations below the 10 mg/l standard in the subdivisions studied.

4. In the sandy soil area of Central Wisconsin, using average groundwater recharge of 24.6 cm per year and three people per household would require housing densities less than 1.1 dwellings per hectare. Lower housing densities would be needed in areas with less groundwater recharge.

5. Mixing of subdivision-originated groundwater with that from upgradient sources is minimal and occurs in some areas more than others. This is apparently due to effects of private wells and differential recharge which cause local advectual processes.

6. Due to the recharge of most of the runoff water from roads and roofs, groundwater recharge from within a subdivision on sandy soils is considerably greater than from adjacent fields and woods. This results in greater dilution of septic system contaminants. The opposite would be true in areas where most road and roof runoff went to surface runoff rather than to groundwater recharge.

7. The amount of fluorescence in groundwater was generally a good indicator of septic effluent and was useful in identifying water originating from within the subdivision.

8. The ratio of sodium to chloride was useful in determining whether groundwater originated from agricultural sources, the subdivision, or a highway right-of-way.

9. Plumes from single or even a row of septic systems show minimal horizontal or vertical mixing with groundwater from other sources. Average reduction in nitrogen content from septic tank to groundwater adjacent to drainfields is only on the order of a two-fold dilution.

10. Phosphorus concentrations found in groundwater downgradient of four septic systems and two entire subdivisions indicate that sandy soil can become saturated with phosphorus within less than 20 years, and results in significant leaching of even this generally immobile chemical. Concentrations ranging from 1 to 11 mg/l were found downgradient of four septic systems.

1. A limited number and relatively low concentrations of VOCs were found in the groundwater associated with subdivision and septic system monitoring wells. These chemicals can and do get to groundwater from homeowner use, but current levels of use and disposal of VOCs were low enough to prevent any high concentrations from reaching groundwater under the studied subdivisions.

12. Well placement and depth of wells for homeowners in subdivisions needs to be carefully considered relative to septic system location and groundwater flow to prevent unwanted recycling of wastewater.

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