THE IOWA STATE-WIDE RURAL WELL-WATER SURVEY Water-Quality Data: Initial Analysis

Technical Information Series 19

Iowa Department of Natural Resources

Larry J. Wilson, Director

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TABLE OF CONTENTS

LIST OF FIGURES

Page

LIST OF TABLES

Page

Page

Page

ABSTRACT

The State-Wide Rural Well-Water Survey (SWRL) was conducted between April 1988 and June 1989 by the Iowa Department of Natural Resources and the University of Iowa Center for Health Effects of Environmental Contamination. SWRL was designed to provide a statistically valid assessment of the proportion of private rural wells and rural Iowa residents affected by various environmental contaminants. The survey was a systematic sample, stratified by rural population density. SWRL demographic data indicate the sample is clearly representative of rural Iowans.

Primary samples were analyzed for total coliform bacteria; nitrate (+nitrite)-N, ammonium-N, and organic-N; major inorganic ions; 27 pesticides, and 5 pesticide metabolites. Existing agency and laboratory USEPA quality assurance, quality control plans were utilized and verified for SWRL. SWRL collected and analyzed 1,048 water samples from 686 sites.

SWRL was conducted during the driest consecutive two-year period on record in Iowa, with precipitation averaging 14 inches below normal. Monitoring studies indicate the drought limited the movement of contaminants to groundwater. Hence, the SWRL results may present a "best-case" water-quality situation because of the temporal coincidence with the drought.

The SWRL data provide a population-based summary of the drinking water used by rural Iowans, and a cross-section of the quality of Iowa groundwater. The variations in water quality exhibited in the SWRL data, both regionally and particularly with depth, show consistent and predictable geochemical patterns, related to contaminant sources, transport, and age effects. Iowa well waters are near neutral and dissolved ions are dominated by calcium, magnesium, bicarbonate, and in some cases, sulfate. Mean concentrations for all ions, except chloride (Cl) and nitrate (NO_3-N), increase or remain fairly constant with depth. The higher concentrations of Cl and NO_3-N at shallow depths are related to their surficial sources. State-wide, 1.3 % of private well waters exceeded the USEPA maximum contaminant level (MCL) for fluoride (F), and 2.5% exceeded the secondary standard of 2 mg/L. Private well water should be analyzed to assess the natural F content before a supplement is prescribed, to avoid problems with dental fluorosis.

About 18% of Iowa's private, rural drinking-water wells contained NO_3-N >10 mg/L, the recommended health advisory level (HAL); 37% of wells have >3 mg/L, typically considered indicative of anthropogenic pollution. Approximately 14% of wells had detections of pesticides: 16 pesticide compounds were detected, including 11 parent compounds and 5 environmental metabolites; 16 pesticides were not detected. Atrazine and its metabolites were found in 8% of wells. Multiple residues were detected in all regions of the state. The mean concentrations were generally <1 ug/L. Lifetime HALs were exceeded in 1.2 % of private, rural wells in Iowa.

Approximately 45% of sites were positive for total coliform bacteria. Total coliforms are ubiquitous constituents of soils, surface water, and shallow groundwater and cannot be equated to fecal coliforms. Only 7% of water systems were positive for fecal coliform bacteria. The only sound, general interpretation of a persistent presence of total coliforms is that the water system is allowing interaction with soil, soil-water, shallow groundwater, or possibly surface water. This can indicate that the system is prone to other forms of contamination.

Individually, or in combination, nearly 55% of rural water supplies exhibited total coliform positives, NO_3-N >10 mg/L, and/or pesticide detections. Using fecal coliforms only, this reduces to about 30% of well-water supplies. Based on 1980 Census data, about 130,000 rural Iowa residents consume drinking water from private wells with >10 mg/L, NO_3-N; 94,000 use water with one or more pesticides; 5,400 use water with a pesticide concentration above an HAL.

Statistical analyses show significant associations between many water-quality parameters but the associations are not strong predictors based on state-wide data. By far the most significant factor explaining water-quality variations is well depth. An apparent relationship among total coliforms, NO_3-N, and pesticides is primarily a function of their co-occurrence related to well depth. Total coliform bacteria are very poor predictors of these chemical contaminants. If a prediction were based on the presence of total coliform, the probability is better that they would not occur in the

water supply.

Consistent relationships among NO_3-N, dissolved oxygen, and ammonium-N with well depth suggest that nitrate reduction and/or denitrification occurs with depth in groundwater systems in Iowa. It is not clear from these data if the deeper groundwater system has the capacity to denitrify the nitrate loads currently being delivered to the system, however.

The effects of sinkholes or agricultural drainage wells are not significant in a state-wide context. Sinkholes were identified in the vicinity of only 2.1% of sites and only 0.6% of sites were near agricultural drainage wells (ADW). No sites reporting ADWs had any pesticide detections or NO_3-N >10 mg/L. Non-farm, suburban housing tracts exhibited the most significant association between landuse and water quality; proportionately, these areas show substantially *fewer* wells with >10 mg/L NO_3-N and total coliform bacteria. Wells <50 feet from septic systems, showed less nitrate and significantly fewer positives for total and fecal coliform bacteria.

Point source problems affect only a small proportion of wells state-wide. Wells located in feedlots showed significantly higher concentrations of nitrate, but not bacteria problems. Such sites comprise only about 3% of wells state-wide, and account for only about 1% of the wells with >10 mg/L, NO_3-N. Sites where herbicides were mixed within 15 feet of the well showed greater pesticide detections, but again the proportion of wells is low, about 3%, state-wide. Wells located within 15 feet of chemical storage and handling areas are uncommon, occurring at $<0.6\%$ of rural sites, and *none* of these wells contained pesticides or NO_3-N >10 mg/L.

About 5.5% of private water wells in Iowa have experienced a spill or back-siphoning accident with pesticides or fertilizers. These sites exhibit a greater proportion of pesticide detections and high nitrate concentrations than average, as expected, but at the majority of sites the pesticides detected were not those involved in the accident. The sites exceeding HALs for pesticides occurred throughout the state. The sites were dominated by shallow wells; one deep well was involved, and this was a point source case which could affect any depth of well. Two of the sites, 25%, are clearly "point source" cases, a spill and back-siphoning accident (alachlor and trifluralin); the majority, 62.5%, are probable nonpoint sources related to pesticide occurrences in shallow groundwater (alachlor and atrazine); 1 case, 12.5%, is equivocal (atrazine).

Well depth is a major variable related to well-water quality, affecting the potential for surficial contaminants to enter a well. The degree of contamination is far greater in shallow wells and significant contamination occurs in wells up to $\simeq 100$ feet deep. Wells <100 feet deep comprise 50% of wells state-wide and account for 70% of total coliform positives, 80% of fecal coliform positives, 64% of pesticide detections and total atrazine detections, and 89% of wells with NO_3-N >10 mg/L. In NE Iowa the depth of contamination is greater because of the deeper groundwater circulation. The greatest proportions of contaminated wells occur in the SC, SW, and NW regions, paralleling regional dependance on shallow wells. In these regions nearly 75% of wells are <100 feet deep (dominantly seepage wells) because alternative water sources are limited.

Certain factors of well construction or placement may afford easy entry of shallow, contaminated groundwater. But these factors are not *causes* of contamination; if the contaminants were not in the environment they would not get into the soil water and groundwater, or the well. Remediation of well construction or replacing current wells with deeper wells would undoubtedly reduce nitrate and pesticide contamination in many locations, but this would not address the cause of the contamination. The sources of contamination must be addressed because these shallow groundwaters will be the recharge for deeper groundwater with time. Sanitary and structural improvements of private water systems are also needed.

Extrapolating from temporal samples provides an upper limit estimate of wells with likely detections, at sometime over the course of a year: 1. wells >10 mg/L NO_3-N $\simeq$ 21%; 2. wells with any pesticide detections $\simeq$ 30%; 3. wells with atrazine detections $\simeq$ 15%. The systematically selected 10% repeat sites provide a consistent representation of the state-wide data, including representative detections of pesticides down to a 1% occurrence interval. These wells can provide a subset for monitoring trends over time.

INTRODUCTION

The State-Wide Rural Well-Water Survey (SWRL) was conducted as part of the implementation of the Iowa Groundwater Protection Act of 1987. The SWRL survey is a systematic, statistical sampling of the quality of private drinking-water supplies used by rural Iowans. The Iowa Department of Natural Resources and the University of Iowa Center for Health Effects of Environmental Contamination conducted the sampling between April 1988 and June 1989.

The survey addresses two questions: 1. What proportion of private rural wells in Iowa are affected by various environmental contaminants? and 2. What proportion of rural Iowa residents are utilizing well water containing these environmental contaminants? Results of the SWRL study provide, for the first time, state-wide estimates of the extent of groundwater contamination in rural private wells. The many prior studies in Iowa, while very important, were local in nature and have not afforded statistically valid state-wide estimates of groundwater quality. The details of the design and operation of the SWRL survey are reported in Hallberg et al., 1990. This report reviews pertinent elements of the design, the hydrologic conditions during the survey, and presents a summary of the basic water-quality results.

The SWRL survey was designed and conducted as a joint effort of the Iowa Department of Natural Resources (DNR) and The University of Iowa (UI) Center for Health Effects of Environmental Contamination (CHEEC). Participating units in CHEEC include the Department of Preventive Medicine and Environmental Health (PM&EH), the Department of Civil and Environmental Engineering (CEE), and the University Hygienic Laboratory (UHL). Additional support for the survey was provided by the Iowa State University Cooperative Extension Service (CES) and the Iowa Department of Agriculture and Land Stewardship (DALS). Principal funding for the SWRL study came from the Iowa Groundwater Protection Fund; related support was provided by the U.S. Environmental Protection Agency (EPA), Region 7, Kansas City.

STUDY DESIGN SUMMARY

The intent of the SWRL survey was to provide statistically valid estimates, both state-wide and within regional, multi-county areas, of the quality of private well-water supplies. Primarily a one-time sampling, the survey provides a 'snapshot' of the condition of the private water-well supplies of the state, and some insight into the condition of Iowa's groundwater resource. It may also serve as a baseline for long-term monitoring of private water supplies and for measuring future trends and changes in groundwater and/or rural private drinking-water quality.

A SYSTEMATIC, STRATIFIED SAMPLE

If there was a listing or index of private wells in Iowa, a simple random sample could be selected for a survey. However, there is no current listing of this sort. Also, the hydrogeology of the state is not a random feature; there are systematic differences in geology, soils, and landscape features across the state that affect the nature of water availability and, in turn, the nature of the wells developed. The SWRL survey includes all rural residents, not just the farm population, and the density of the rural population is not randomly distributed either. The rural population is more dense surrounding major urban population centers, which are also related, historically, to natural water availability. So, a systematic sample, stratified by rural population density, was selected to provide a statistically valid answer to the questions posed for the study.

A sampling grid was constructed from the intersection of every 5 minutes of latitude and longitude within the state. This grid provided a systematic framework across the state, unaffected by any external bias. The sample scheme was then stratified by the rural population density of each county, derived from 1980 census data and land area inventories. The counties were divided into three population strata -- high, medium, and low population density, based on statistical criteria. The number of samples to be collected were allocated among these three groups in proportion to the population represented. The sample sites were picked based on their relation to selected intersections of latitude and longitude throughout the state. The drinking-water well closest to each chosen intersection was selected as the target for sampling. In this fashion, samples were collected in every county in the state; the number of sites in a county ranged from 3 to 17 (in typical sized counties).

A target of 698 sites was defined, based on statistical considerations (to develop a sample with a margin of error of approximately 5%), available funds, and the time and logistical constraints of collecting the samples and conducting an inventory of each site. The resulting sampling framework is shown on Figure 1. Iowa State University Cooperative Extension Service (CES) staff identified eligible participants based on the study criteria. The sampling was completed with 98% success; 686 sites were sampled and field inventoried. Typical of the cooperation of Iowans, only 5 % of those people contacted were unwilling to participate; 81% agreed to participate; and 14% were unqualified (e.g., they received water from a public water supply).

Overall, 92 of 99 counties were sampled at 100% of the initial design. The lowest single county completeness rate was 67%. With repeat samples and secondary samples after water treatment, a total of 1,048 well water samples were collected and analyzed during SWRL (not counting quality control analyses).

The design of the survey dictates that the results should only be summarized at the state and regional (multi-county) level. The survey is not designed to provide county specific information.

CONSIDERATIONS FOR SEASONAL VARIABILITY

Studies in Iowa and elsewhere have shown that the quality of groundwater, particularly shallow groundwater that is affected by anthropogenic contaminants, changes over time, often seasonally. Such seasonal, or temporal variability of groundwater quality during the sampling was addressed by several elements of the survey design. The first element chose 10%, or 68, of all selected sites for a one-time repeat sampling during a different season (sites indicated with triangles in Fig. 1). A systematic counting scheme was used to identify these sites. Sixty-four of the 68 sites (94%) were successfully sampled twice. The second element divided Iowa into six general hydrogeologic regions (described below). A county or counties typifying each of these regions was selected, and 62 sites within these counties were sampled quarterly. Multiple sampling was completed at 94% of these sites. In addition, scheduling for state-wide sampling was dispersed across the state seasonally; no more than two contiguous counties were sampled within 4 weeks of each other.

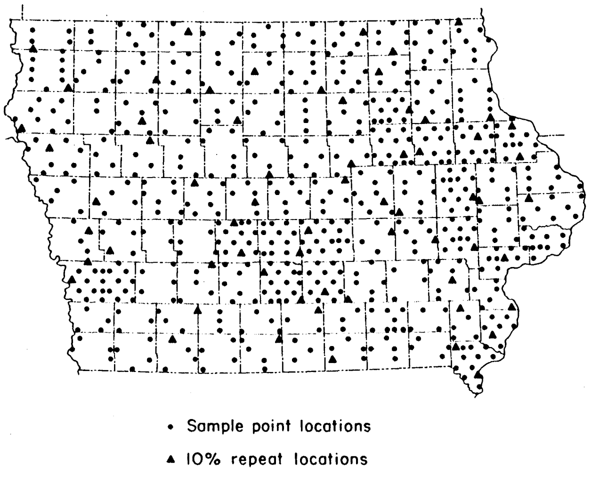


Figure 1. Location of sampling sites; 10% repeat sampling sites shown as triangles.

HYDROGEOLOGIC REGIONS

The hydrogeologic regions employed to assess temporal variability will also be used in this report to summarize regional aspects of the water-quality results. The definition of the regions is outlined below.

Iowa was subdivided into six generalized areas of broadly similar soil, landscape, and hydrogeologic characteristics. These characteristics affect the general nature of the susceptibility of aquifers to contamination, well construction practices, and water availability. Soil-landscape-hydrogeologic conditions also influence landuse and productivity. The six regions exhibit some basic differences in agricultural practices, as well. A county, or counties, judged to be representative of these areas was selected, regardless of population strata or sample numbers, and all sites within these counties were sampled on a quarterly basis. Figure 2 shows the hydrogeologic regions (delineated by county boundaries) and the counties selected for quarterly sampling. There were 62 sites included for the quarterly sampling.

The definitions of "shallow" and "deep" bedrock used in the descriptions of the regions follow those of Hallberg and Hoyer, 1982, Libra et al., 1984, and Bruner and Hallberg, 1988; these categories have been shown to be a simple, but useful method for evaluating the potential for bedrock aquifer contamination from surface activities. The definitions are: shallow depth to bedrock, $\leq$ 50 feet of cover, by glacial deposits or other aquitards over the bedrock aquifers (very shallow, $\leq$ 25 feet of cover, with areas of bedrock outcrop common); deep, > 50 feet, ranging to several hundred feet of cover; and very deep, typically > 150 feet of cover.

The six regions and some generalized characteristics are given below (counties sampled quarterly and number of sites, in parentheses):

1. Northeastern Iowa -- high relief, shallow, commonly very shallow depth to bedrock, which commonly is Paleozoic carbonate and sandstone aquifers; local karst conditions exist; 11.4% of the area of Iowa (Winneshiek County, 9 sites);

2. Eastern Iowa -- moderate relief, shallow to deep bedrock, with a nearly continuous mantle of relatively fine-textured Pre-Illinoian glacial deposits. Bedrock commonly consists of Paleozoic carbonate aquifers; 26.4% of the state's area (Bremer and Washington Counties, 19 sites);

3. South-central Iowa -- moderate relief, shallow to deep bedrock, but generally lesser thickness of glacial deposits than region 2; Pennsylvanian bedrock with highly variable lithologies and aquifer characteristics; 16.2% of the state's area (Lucas and Monroe Counties, 6 sites);

4. Southwestern Iowa -- moderate to high relief, thick loess and glacial deposits, generally deep bedrock; Pennsylvanian bedrock of variable lithologies and locally Cretaceous Dakota sandstone aquifer present, alluvial aquifers supply most community water supplies; 15.6% of the state's area (Cass and eastern Pottawattamie Counties, 8 sites);

5. Northwestern Iowa -- moderate to high relief, generally thick glacial deposits and deep (to very deep) to bedrock, Dakota sandstone aquifer and related Cretaceous bedrock units are important groundwater sources, along with alluvial aquifers; 10.1% of the state's area (Sioux County, 10 sites);

6. North-central Iowa -- low to moderate relief, high relief along major river valleys, area of youngest glacial deposits in the state (the Des Moines lobe; Clarion-Nicollet-Webster soils), shallow to deep bedrock; bedrock varies from Paleozoic carbonate aquifers to Cretaceous Dakota aquifer; 20.3% of the state's area (Hamilton and southern Kossuth Counties, 10 sites).

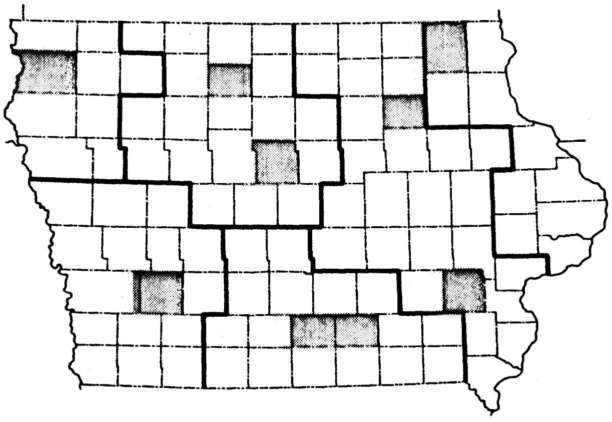


Figure 2. Generalized hydrogeologic regions (bold outlines) and counties selected for quarterly sampling (shaded) within each region.

FIELD AND LABORATORY PROCEDURES

Standardized procedures for field activities were employed during SWRL. An appointment was arranged for each site, so that a resident was available to interview. The target well was located and such items as the well construction and placement characteristics, proximity to potential point-source contamination, and surrounding landuse were inventoried by DNR and CHEEC field staff. Sampling points were chosen as close to the well as possible and before any water treatment devices. To ensure a representative sample, the wells were pumped and the water-system was purged until tracking measurements (temperature and specific conductance) stabilized. Alkalinity, pH, and dissolved oxygen concentrations were also measured in the field. Samples for laboratory analysis were collected in pre-treated containers supplied by the laboratories. Field quality assurance and quality control (QA/QC) included trip blanks, field-spikes, and blind-duplicate samples. Custody forms tracked the movement and custody of all containers, water-samples, and QA/QC samples, from each laboratory to the field and back. Hallberg et al., 1990, provide a more complete review of procedures.

All primary well-water samples were analyzed for total coliform bacteria; nitrate, ammonia, and organic-nitrogen; other major inorganic ions (such as, chloride, sulfate, calcium); 27 commonly-used pesticides; and selected pesticide metabolites (see Tables 1 and 2). Three laboratories conducted the analyses: the University Hygienic Laboratory (UHL); the Analytical Toxicology Laboratory (ATL), associated with the UI Department of Preventive Medicine and Environmental Health; and the Environmental Engineering Laboratory (EEL), associated with the UI Department of Civil and Environmental Engineering. Methods were developed and tested for additional pesticide metabolites, and for organic and toxicity screening of groundwater during the project (Table 3). Therefore, not all samples were analyzed for these additional parameters. Where water treatment systems other than water-softening were used, a secondary sample was collected after the treatment system to assess the effects of treatment.

The labs had U.S. EPA QA/QC plans in place, and the SWRL plan utilized and verified their implementation. The method detection limits (MDL) for pesticide analyses were set as the minimum practical concentration quantitation limit for each analyte in a groundwater matrix, established through QA/QC procedures. These included a minimum two-column confirmation with gas chromatography; intermittent confirmation with other columns and/or detectors, and with mass spectrometry; standards prepared with both a reagent water and a groundwater matrix; field and laboratory QA/QC samples; and co-elution and storage degradation studies. Groundwater-matrix effects typically necessitate an increase in quantitation limits, relative to a reagent water matrix. This may cause an increase in false negative detections, but should minimize false positive detections.

Different facets of the QA/QC program were examined both quarterly and following completion of specific work efforts. The only problem of concern during the course of the study was violation of maximum holding times for a few samples. The effects of the holding time variance and other QA/QC audits are discussed below.

QUESTIONNAIRES AND ON-SITE OBSERVATIONS

Personal interviews with all participants and on-site inventories were conducted by field staff who collected the samples. Information about the wells, characteristics of the sites served by these wells, agricultural practices, water treatment, past water-quality problems, waste disposal practices, and the basic health status of residents were collected using four different questionnaires designed for this study. Inventory questionnaires were completed for 99% of sites. Health assessment questionnaires were returned voluntarily from 85% of the sites, representing over 1,700 individual participants.

The site inventory and survey data will allow for additional relational analysis. For example, the following relationships will be examined: a. the relationship between well construction and placement factors and contamination of well water; b. the relationship of on-farm, or lawn and garden chemical handling practices and groundwater contamination; c. the extent of on-farm chemical spills and back siphoning accidents; d. the extent of use and type of home water treatment systems in rural Iowa; g. basic family health status of rural Iowans, and the ability to link these survey data with Iowa's existing health registries. Analysis of these data are just beginning; detailed reviews will be presented in subsequent reports.

Table 1. Summary of chemical parameters analyzed in SWRL samples, laboratory, methods, and data quality requirements. Table 2 lists references by analyte number.

No.	Analyte name	Other name	Lab	Method	MDL / method detection limit	Sample holding time	Extract holding time	Average % recovery	Relative % diff. of replicates
	Bacteria:								
1.	total coliform		UHL	Most probable number	0 to 16+ statistical function	48 hours	N/A	N/A	N/A
	Nitrogen-Series:								
2.	nitrate (+nitrite)-N		UHL	Cu-Cd reduction	0.05 mg/L	28 days	N/A	93%	7%
3.	ammonium-nitrogen		UHL	color/phenate	0.05 mg/L	"	"	98%	10%
4.	organic-nitrogen		UHL	TKN, block digest	0.10 mg/L	"	"	100%	30%
	Common Ions:								
5.	Na, Ca, Mg, K	cations	EEL	ion chromatography	0.10 mg/L	28 days	N/A	100%	10%
6.	Cl, F, sulfate	anions	EEL	" "	0.10 mg/L	"	"	"	"
	Field Measurements:				Units				
7.	Specific conductance			conductivity meter	umho/cm sq. @ 25 degrees C				
8.	Temperature			mercury thermometer	degrees C				
9.	pH			pH meter	pH units				
10.	Dissolved oxygen			D.O. probe	mg/L				
11.	Alkalinity			titration	mg/L as calcium carbonate equivalent				
	Pesticides: common chemical name	common trade name	Lab	Method	MDL / minimum quantitation limit	Sample holding time	Extract holding time	Average % recovery	Relative st. dev.
	Herbicides:								
12.	2,4,5-T	many	UHL	GC-ECD	0.10 ug/L	7 days	40 days	71% **	41% **
13.	2,4,5-TP	Silvex	UHL	GC-ECD	0.10 ug/L	"	"	" "	" "
14.	2,4-D	many	UHL	GC-ECD	0.10 ug/L	"	"	" "	" "
15.	acifluorfen	Blazer	UHL	GC-ECD	0.10 ug/L	"	"	" "	" "
16.	alachlor	Lasso	ATL	GC-ECD	0.02 ug/L	"	"	89%	15%
17.	atrazine	Atrazine	ATL	GC-ECD	0.13 ug/L	"	"	79%	20%
18.	butylate	Sutan	ATL	GC-NPD/ECD	0.10 ug/L	"	"	70%	16%
19.	chloramben	Amiben	UHL	GC-ECD	1.00 ug/L	"	"	71% **	41% **
20.	cyanazine	Bladex	ATL	GC-ECD	0.12 ug/L	"	"	42%	50%
21.	dacthal	DCPA	ATL	GC-ECD	0.01 ug/L	"	"	83%	14%
22.	dicamba	Banvel	UHL	GC-ECD	0.10 ug/L	"	"	71% **	41% **
23.	metolachlor	Dual	ATL	GC-ECD	0.04 ug/L	"	"	81%	14%
24.	metribuzin	Sencor	ATL	GC-ECD	0.01 ug/L	"	"	78%	25%
25.	pendimethalin	Prowl	ATL	GC-ECD	0.02 ug/L	"	"	73%	18%
26.	picloram	Tordon	UHL	GC-ECD	0.10 ug/L	"	"	71% **	41% **
27.	propachlor	Ramrod	ATL	GC-ECD	0.02 ug/L	"	"	80%	14%
28.	trifluralin	Treflan	ATL	GC-ECD	0.02 ug/L	"	"	75%	18%
	metabolites:								
29.	de ethyl atrazine	*a.	ATL	GC-ECD/NPD	0.10 ug/L	7 days	40 days	69%	29%
30.	de isopropyl atrazine	*b.	ATL	GC-ECD/NPD	0.10 ug/L	"	"	38%	37%
	Insecticides:								
31.	chlorpyrifos	Lorsban	UHL	GC-FP or NPD	0.10 ug/L	7 days	40 days	104% **	54% **
32.	diazinon		UHL	GC-FP or NPD	0.10 ug/L	"	"	" "	" "
33.	dimethoate	Cygon	UHL	GC-FP or NPD	0.10 ug/L	"	"	" "	" "
34.	ethoprop	Mocap	UHL	GC-FP or NPD	0.10 ug/L	"	"	" "	" "
35.	fonofos	Dyfonate	UHL	GC-FP or NPD	0.10 ug/L	"	"	" "	" "
36.	malathion		UHL	GC-FP or NPD	0.10 ug/L	"	"	" "	" "
37.	parathion		UHL	GC-FP or NPD	0.10 ug/L	"	"	" "	" "
38.	phorate	Thimet	UHL	GC-FP or NPD	0.10 ug/L	"	"	" "	" "
39.	terbufos	Counter	UHL	GC-FP or NPD	0.10 ug/L	"	"	" "	" "
40.	carbofuran (includes carbofuran and metabolites, derivatized as carbofuranphenol)	Furadan	ATL	GC-ECD	0.01 ug/L	7 days	40 days	47%	35%
41.	3-hydroxy and 3-keto carbofuran metabolites		ATL	GC-ECD	0.02 ug/L	7 days	40 days	47%	35%

*a & b. metabolites of atrazine; *b. also metabolite of cyanazine. ** Pooled data, reflects multi-residue method composite. UHL - University Hygienic Laboratory; EEL - Environmental Engineering Laboratory; ATL - Analytical Toxicology Laboratory. GC - gas chromatography; ECD - electron capture detector; NPD - nitrogen-phosphorus detector; FP - flame photometric detector.

Table 2. Summary and references for lab methods used for analyzing SWRL water-quality analytes; referenced to the analyte numbers on Table 1.

Analyte Number; Method Description and Reference

1; total coliform bacteria. Most Probable Number (MPN) method; using multiple-tube fermentation, presumptive test and confirmation test; statistical derivation of MPN of coliform bacteria in 100 ml of water sample. Based on Standard Methods for Water and Wastewater Method 908A (APHA, 1985).

2; nitrate and nitrite-nitrogen. Automated, copper-cadmium reduction and colorimetric quantitation. The method is based on U.S. EPA Method 353.2 (USEPA, 1983).

3; ammonia-nitrogen. Automated phenate reaction, and colorimetric quantitation, using Technicon auto-analyzer IM 780-86T. Based on U.S. EPA Method 350.1 and .2 (USEPA, 1983).

4; organic-nitrogen. Total Kjeldahl procedure with sulfuric acid, K_2SO_4, and $HgSO_4$ pre-treatment using Technicon IM 780-86T; semi-automated block digester, AAII, colorimetric quantitation. Organic-nitrogen is defined as the sum of free-ammonia and organic nitrogen compounds which are converted to ammonium sulfate, less the ammonia-N determined in procedure 3, above. Based on U.S. EPA Method 415.1 (USEPA, 1983).

5; cations, Na (sodium), Ca (calcium), Mg (magnesium), and K (potassium). Ion chromatography, using two columns of ion exchange resins with a filtered sample. Based on the American Society for Testing & Materials (ASTM), 1984a; O'Dell et al., 1984; Topol and Ozdemir, 1981.

6. anions, Cl (chloride), F (fluoride) and SO_4 (sulfate). Ion chromatography, using a precolumn (guard column), a separator column and a suppressor column with an anion exchange resin, and a conductivity detector, with a filtered sample. References as in 5.

7 through 11; On-site Field Measurements 7; Specific conductance; Fisher conductivity meter and temperature compensated probe, standard KCl solutions for calibration (Fisher Sci., 1987). 8; temperature, simple glass, mercury thermometer. 9 and 11; pH and alkalinity; Measured with probe and incremental titration using Beckman ph meter, Hach pH probe, Portable Water Test Kit and Digital Titrator (Beckman, 1987; Hach, 1987a, and b). 10; dissolved oxygen; YSI DO Meter and Probe with Automatic Stirrer (YSI, 1987).

12, 13, 14, 15, 19, 22, and 26; acid-based herbicides. Hydrolyze derivatives with mechanical shaking 0.1 N sodium hydroxide; extraneous organic material is removed by a solvent wash. Acidify, extract chlorinated acids with ethyl ether by mechanical shaking in a separatory funnel or mechanical tumbling in a bottle. Convert acids to methyl esters; derivatize with diazomethane. Remove excess derivatizing reagent; esters determined by GC using an electron capture detector (ECD). The method is based on U.S. EPA, National Pesticides Survey Method 3 (USEPA, 1987).

16, 17, 20, 21, 23, 24, 25, 27, and 28; common herbicides, multi-residues. Methylene chloride extraction; extract partitioned, using silica gel, into two fractions for gas chromatograph-electron capture detector (GC-ECD) analysis, employing two-column confirmation. Based on U.S. EPA methods, EPA-600/8-80-038, Section 10, A (USEPA, 1980, p. 431-456)

18; butylate. Method identical to that for analyte 16, et al., except GC-nitrogen phosphorus detector (GC-NPD) analysis is used for the first fraction. Modified method from EPA-600/ 8-80-038, Section 10, A (USEPA, 1980).

29, 30; metabolites. Method same as analyte 18, but uses GC-NPD analysis of the second fraction. Modified method from EPA-600/ 8-80-038, Section 10, A (USEPA, 1980).

32 through 39; organophosphate insecticides. Extraction with with methylene chloride as a solvent at a neutral pH, using a separatory funnel or a continuous liquid-liquid extractor. GC with a flame photometric (FP) or nitrogen-phosphorous detector (NPD) is used for this multiresidue procedure. Based on U.S. EPA, Method 81.40 (USEPA, 1986).

40; carbofuran. Similar to procedure for 16, et al.; second fraction derivated with pentafluorobenzyl bromide and partitioned, using silica gel, for GC-ECD analysis. Modified method from EPA-600/ 8-80-038, Section 10, A (USEPA, 1980, p. 431-456).

41; carbofuran metabolites. 3-hydroxy carbofuran and 3-keto-carbofuran; derivatization with pentafluorobenzyl bromide, GC-ECD analysis (after Jackson and Soileau, 1981).

Table 3. Water-quality parameters with methods under development in SWRL.

Laboratory/parameter

UHL Laboratory

Fecal Coliform Bacteria: fecal coliform tests performed on a subset of samples, using two methods; multiple-tube fermentation (MTF) and the Autoanalysis Colilert test. See discussion in text in Coliform Bacteria section.

ATL Laboratory

Pesticide metabolites:

cyanazine amide; cyanazine metabolite; modification of method for analytes 29 and 30, Table 3.

2,6 diethyl aniline (DEA); alachlor hydrolysis degradate; modification from U.S. EPA, Pesticide Analytical Manual, Vol. II, Section 120.249, p. 1-4, dated 7/1/69, provided by USEPA.

Ciba-Geigy Compound 37913 and 49751; metolachlor metabolites; extraction and derivatization followed by GC-ECD and GC-NPD analysis; modification from U.S. EPA Pesticide Analytical Manual, Vol. II, Section 180.368, p. 1-21, dated 12/82, provided by USEPA.

deamino, diketo, and deamino-diketo metribuzin; metribuzin metabolites; methylene chloride extraction; HPLC (high-pressure liquid chromatography) with UV detector, after Parker et al., 1983.

Toxicity Screening:

acute toxicity; Microtox method (Microbics Corp., Inc., 1982).

EEL Laboratory

TOX; total organic halogens; minor modification of APHA (1985) method 506.

TOC; total organic carbon; minor modification of APHA (1985) method 505B (also Dohrmann instrument reference DC-180 for TOC analysis).

TIC; total inorganic carbon; modification of APHA (1985) method 505B; ASTM (1988), D4839-88 (also Dohrmann instrument Tech. Ref. TR-022).

DROUGHT EFFECTS; PERSPECTIVES ON THE SWRL RESULTS

Ideally, this survey would have been conducted under "normal" climatic conditions, for a true reflection of the quality of the state's groundwater supplies. Unfortunately, 1988 and 1989 were the two driest consecutive years in Iowa's recorded history; the state-wide average precipitation was more than 18 inches below normal. This undoubtedly has influenced the results because recharge to the groundwater was greatly restricted. Some of the original 698 sites could not be sampled because wells had dried up in portions of south-central Iowa, for example. Never-the-less, the SWRL study provides an important overview of the condition of the state's rural water supplies.

Climatic, hydrologic, and water-quality data from long-term monitoring sites are reviewed below to provide a perspective on the SWRL results.

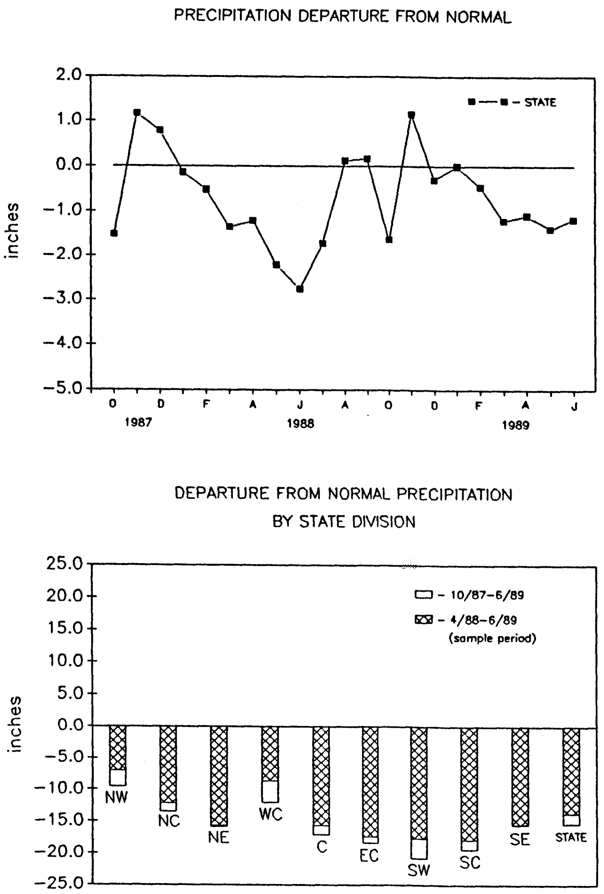


Figure 3. Departure from normal precipitation state-wide and by climatic reporting district during SWRL study.

CLIMATIC AND HYDROLOGIC OBSERVATIONS

Interpretation of the SWRL results requires an understanding of the climatic and hydrologic conditions that prevailed during the survey. The state-wide average precipitation was 18 inches below normal for 1988 and 1989, and was about 14 inches below normal during the April 1988 through June 1989 period of sample collection. All climatic data presented in this report were obtained from the Office of the State Climatologist, Iowa Department of Agriculture and Land Stewardship (DALS). The "normal" or average climatic data are based on the period 1951-1980.

Figure 3 shows the departure from normal precipitation state-wide, and by climatic reporting divisions (equivalent to DALS crop and livestock reporting districts) during SWRL. It also shows the departure from normal for the period October 1987 through the end of the SWRL sampling. During this time, precipitation was at least 10 inches below normal in all but the northwest and west central divisions, and was at least 15 inches below normal for most of the state. Figure 4 maps the departure from normal precipitation for Iowa during 1988. Some parts of northwestern Iowa received near normal precipitation; the southeastern part of the state experienced precipitation deficits of 10 inches to more than 15 inches. Figure 5 shows monthly state-wide departure from normal precipitation for the period defined by water-years 1982 through 1989 (October 1982-September 1989). Only three months of SWRL showed above normal precipitation, and only one of these was more than one inch above normal. Most months showed precipitation deficits of over one inch.

Hydrologic Conditions

Water levels in wells, and the discharge rates of rivers and springs, provide an overview of the hydrologic conditions existing during SWRL. Locations with long-term records afford further insights, as departures from long-term normal conditions may be evaluated. Water levels in wells rise when infiltrating precipitation reaches

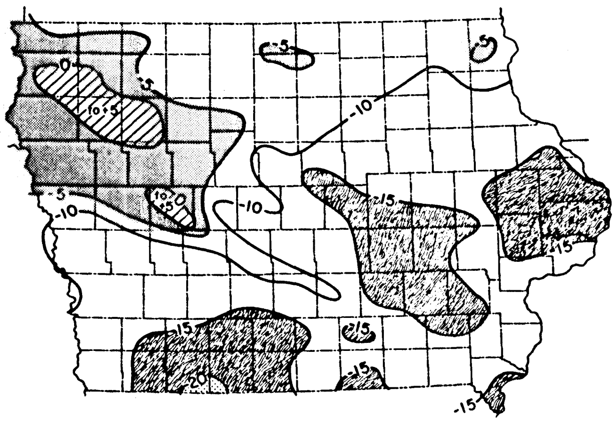


Figure 4. Map of departure from normal precipitation during 1988.

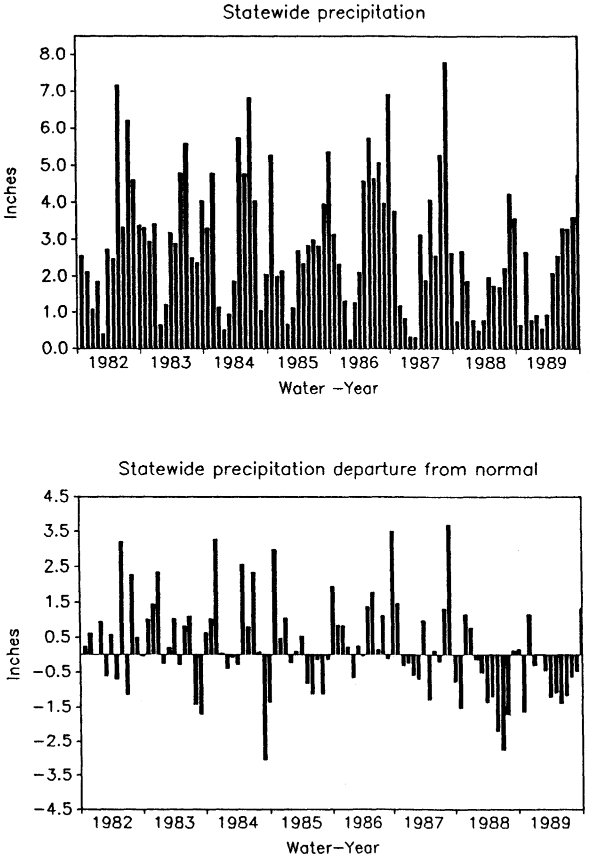


Figure 5. State-wide departure from normal monthly precipitation for water-years 1982-1989 (October 1982-September 1989).

the water table, recharging the groundwater system. They decline during periods when recharge is limited or non-existent. Perennial streams and rivers are the ultimate discharge zones for most groundwater, and the discharge of large rivers yields an overview of groundwater conditions over extensive areas. During extended dry periods river discharge is sustained by groundwaterbase-flow, and decreasing base-flow discharge reflects the same depletion of the groundwater reservoir as recorded by declining water levels in wells. Springs discharge groundwater, and declining spring discharge rates also reflect groundwater depletion in a spring's groundwater basin. Groundwater levels, and stream and spring discharge rates, from across Iowa (see Fig. 6) are discussed below. Where possible, data from the SWRL period are compared to long term normal conditions. These data are from the U.S. Geological Survey-Water Resources Division (USGS), Iowa District, and IDNR-GSB records.

Groundwater Levels in Wells

Figure 7 shows monthly water level departures from normal for four shallow (<50 feet) water-table wells completed in unconsolidated Pleistocene deposits from various parts of the state. Data are given for the period covered by water-years 1982 through 1989 (October 1, 1981 through September 30, 1989). These wells, monitored by the USGS, have data records for periods ranging from 39 to 47 years. Monthly normals are calculated by the USGS using the entire period of record. All four wells show strikingly similar trends. Water levels were generally above normal across the period until early in water-year 1988, when levels began to drop sharply to below normal. This rapid change from above to below normal conditions began in the months preceding the start of sampling for SWRL. In general, water-levels approached normal in the early part of water-year 1989. However, by late summer of

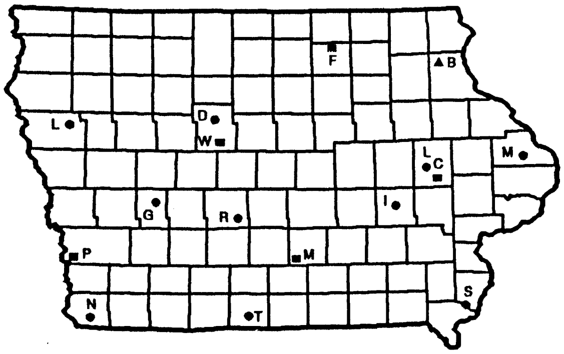


Figure 6. Locations of hydrologic monitoring sites discussed in text. Observation well records shown on Fig. 7: L=Linn County; M=Marion County; P=Pottawattamie County; W=Webster County. Stream discharge records, Table 4: C=Cedar River; D=Des Moines River; I=Iowa River; L=Little Sioux River; M=Maquoketa River; N=Nishnabotna River; R=Raccoon River; S=Skunk River; T=Thompson River. B=Big Spring basin; F=Floyd County monitoring wells; G=Upper Bluegrass basin.

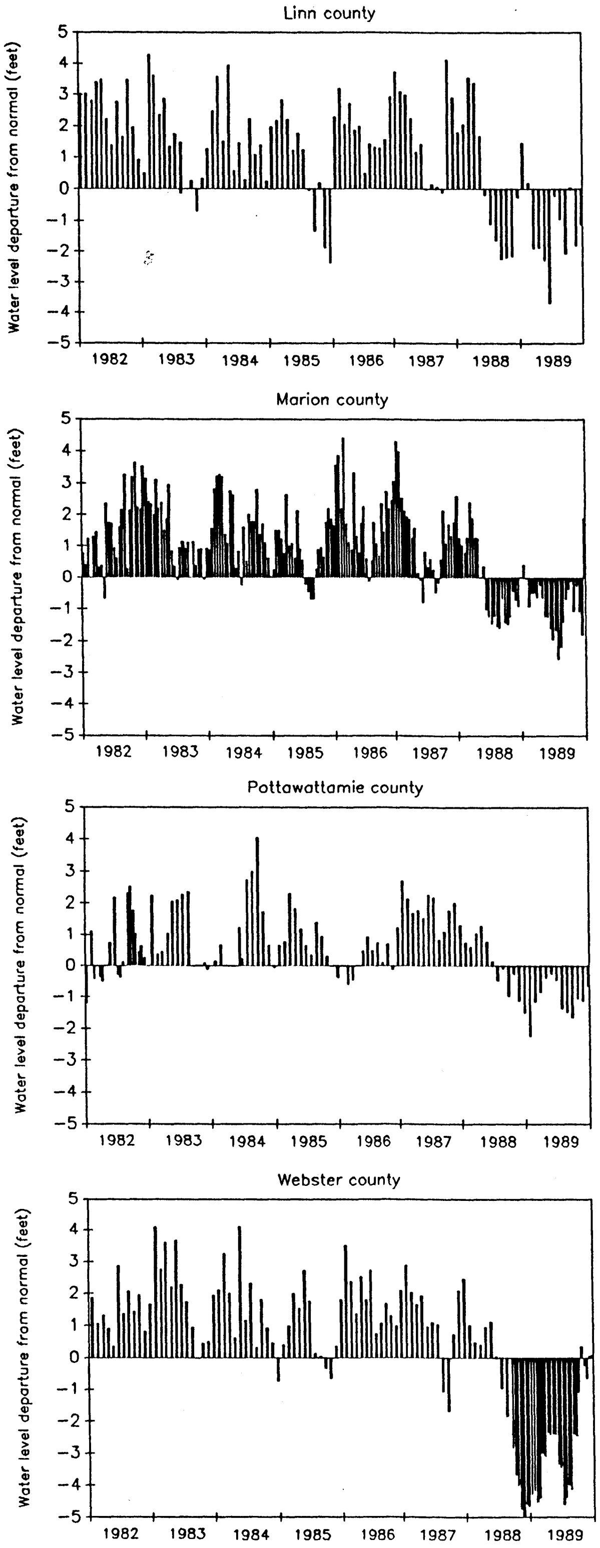


Figure 7. Water level departure from long-term normal for four water table wells across Iowa. Locations shown on Fig. 6, L=Linn County; M=Marion County; W=Webster County; and P=Pottawattamie County. Data from US Geological Survey, Water Resources Division, Iowa District.

that year levels were as far or further below normal than they were in 1988. This decline from above to below normal conditions is indicative of the lack of significant recharge across much of the period.

The lack of recharge to the water table also affects recharge to, and therefore water levels in, deeper bedrock aquifers. Figure 8 shows water-level elevations from a nest of four wells in Floyd County. Monthly precipitation and departure from normal monthly precipitation for nearby Charles City are also shown. These wells vary in depth from about 75 to 350 feet and are completed in the Devonian carbonate aquifers (see Libra and Hallberg, 1985). Water-level data are available for two periods, 1985 through 1986, and late-summer 1988, to present, coinciding with SWRL. Note that the highest water-level elevations measured during SWRL were below the lowest measured during 1985-1986, and that monthly precipitation has been below normal in over two-thirds of the months since 1986.

River Discharge Rates

River discharge rates were well below long-term normals during SWRL, also. Published discharge data are available for water-year 1988 (USGS, 1989). Table 4 compares the long-term average and water-year 1988 discharge statistics for nine Iowa streams from across the state. These streams are representative of the conditions state-wide during water-year 1988, and show discharges ranging from 40% to 80% of long term normals. The variations in discharge among rivers during water-year 1988 generally mimic the variations in precipitation across the state (Fig. 4). Some rivers draining areas that received near normal precipitation had discharges of as much as 85 per cent of the long term normals, while streams draining areas with more severe precipitation deficits discharged less than 35 per cent of normal.

Groundwater Discharge at Big Spring

Big Spring is the discharge point for a 103 square mile groundwater basin in northwestern Clayton County (Hallberg et al., 1983, 1989). Discharge from the spring has been measured since November, 1981. Figure 9 shows mean monthly discharges from Big Spring through water-year 1989. Monthly discharge during SWRL ranged from 31% to 80% of the average from the preceding period of record, with a mean of 49% of normal, again indicating severely limited groundwater recharge. Discharge from Big Spring reflects groundwater conditions in a relatively small part of the state, and the monitoring record is from a relatively short period. However, the Big Spring data allow for more quantitative estimates of groundwater recharge/discharge conditions than many other records.

WATER-QUALITY MONITORING PERSPECTIVES

SWRL was conducted during a period of pronounced drought. The drought resulted in very limited movement of water from the land surface to the water table (i.e., groundwater recharge), and therefore limited delivery of surface-related, nonpoint source contaminants to the groundwater reservoir. The drought undoubtedly affected the occurrence and/or concentrations of the contaminants analyzed for SWRL. An evaluation of these effects on SWRL results is reviewed below through comparisons with longer term water-quality monitoring data.

Water-Quality Variations

Groundwater quality, particularly as measured at individual point locations such as wells, varies in many dimensions -- spatially, with depth, and over time -- and the variation in these dimensions are inter-related (Libra et al., 1984; Hallberg, 1989a). There are many, complex factors that affect the quality of water derived from any given well. The depth of the well and the thickness of the aquifer/material tapped by the well affect the resultant sample. The chemistry of the water is affected by various environmental and landuse factors, by

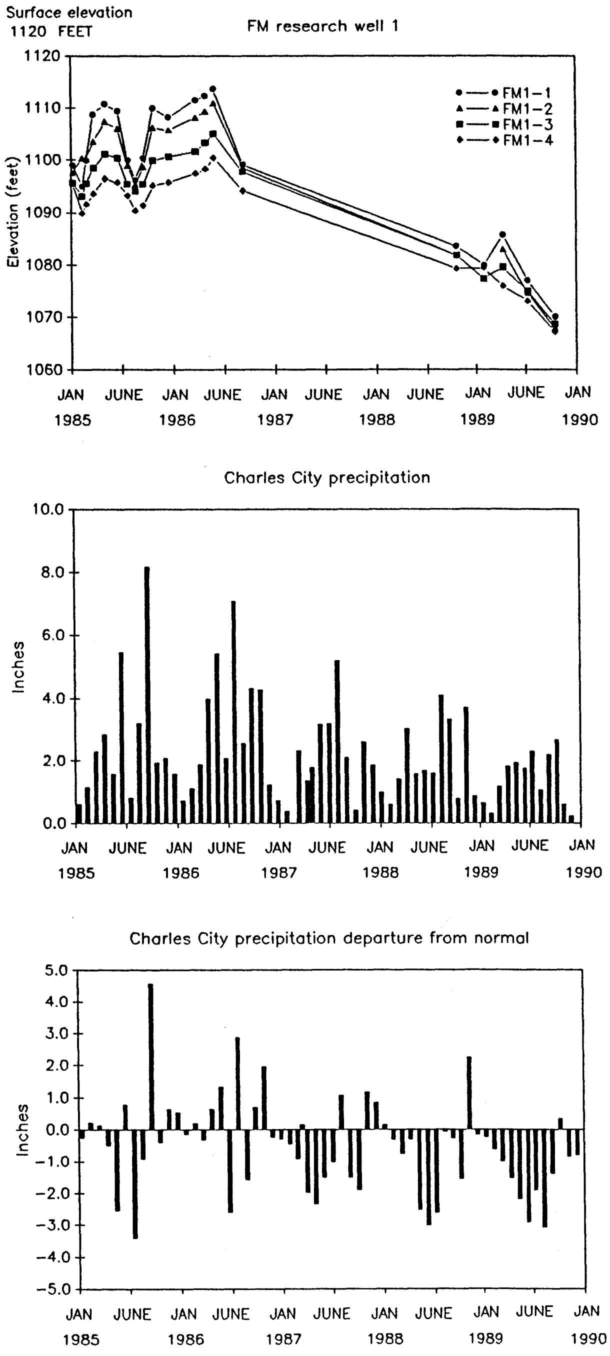


Figure 8. Water levels from nest of four wells in the Devonian aquifers in Floyd County, and departure from normal precipitation at Charles City, Floyd County. The depth of the wells is: 1-1 = 60 ft; 1-2 = 140 ft; 1-3 = 235 ft; and 1-4 = 350 ft.

Table 4. Discharge statistics for nine streams across Iowa during 1988 drought.

Stream-gaging station	Drainage Area sq miles	Normal Q inches	WY-1988 Q inches	WY-1988 as % of normal
Maquoketa River at Maquoketa	1,553	9.01	6.46	72%
Iowa River at Marengo	3,271	8.87	3.95	44%
Cedar River at Cedar Rapids	6,510	7.25	3.62	50%
Skunk River at Augusta	4,303	7.75	3.58	46%
Des Moines River at Ft. Dodge	4,190	5.08	2.27	45%
Raccoon River at Van Meter	3,441	5.62	3.62	64%
Thompson River at Davis City	701	7.34	3.14	43%
Little Sioux River at Correctionville	2,500	4.51	3.54	78%
Nishnabotna River at Hamburg	2,806	5.44	3.86	71%

hydrogeologic factors, such as the local aquifer/material properties and nature of the flow-system, and by the stability of dissolved constituents in the groundwater system. The landuse in the recharge area affects the potential sources of contaminants that can be present. The nature of the materials in the flow path affects the potential for natural substances, which may also impair the quality, to be dissolved into the water. These geologic materials also affect the potential for adsorption or degradation of contaminants. Poor well construction (or maintenance) or well placement may allow contaminants to enter the well head itself.

In some settings, even slight differences in spatial location, or in well depth (or casing depth), can result in the well tapping a substantially different portion of the groundwater flow-system. With these interactions, apparently similar wells may tap water of very different ages; the groundwater derived from many wells began as recharge before many man-made contaminants were in use (Alexander and Alexander, 1989; Hallberg, 1989a).

How these factors integrate in the subsurface can seldom be satisfactorily worked out on a point-by-point basis. As a generalization, however, "shallow" wells that tap the water table or the uppermost portions of the groundwater system exhibit different responses and water quality problems than "deep" wells that tap confined aquifers. Shallow wells are more directly responsive to annual, seasonal, or even daily changes in recharge and contaminant delivery related to land-surface activities. Deep wells may tap waters with very long flow paths and residence times and may only be subject to very long-term changes. The water quality problems they typically exhibit are related to natural factors, such as dissolved salts, iron, or radionucleides.

As previously discussed, the SWRL sampling design incorporated elements to minimize the effects of seasonal variability. Of primary interest for perspective on the SWRL results are the effects of restricted recharge related to the drought, on the water quality in the relatively

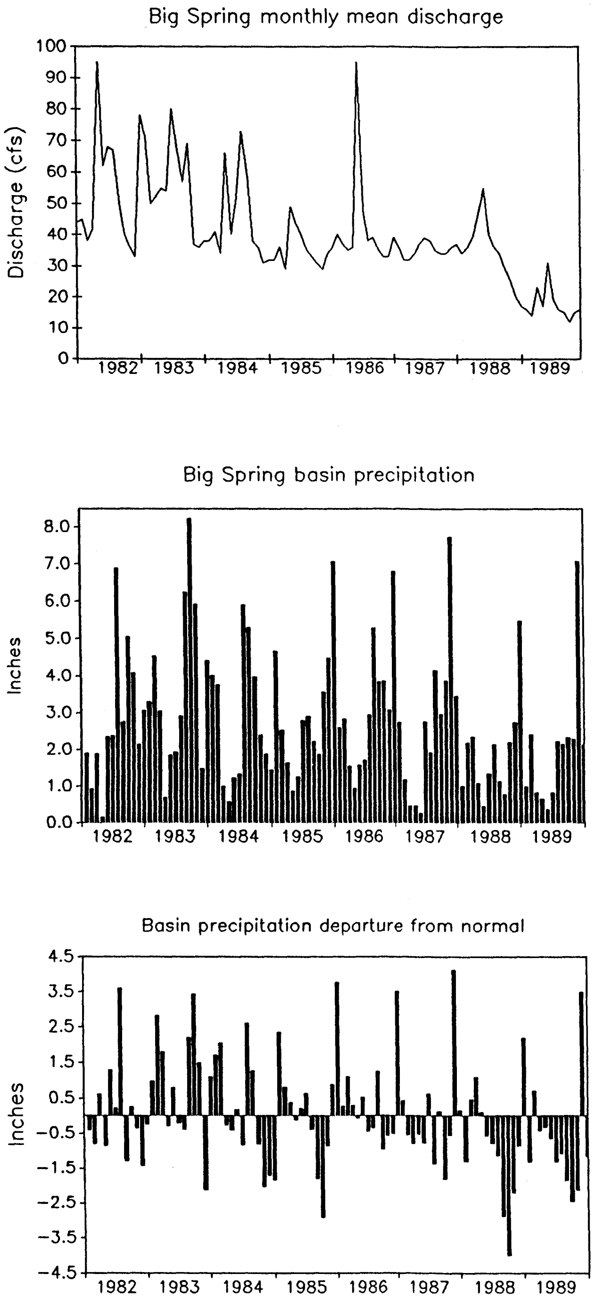


Figure 9. Groundwater discharge from Big Spring, Clayton County, and precipitation departure from normal for Big Spring basin area.

shallow wells commonly used for private water supplies.

Drought Effects

Two principle components of recharge that affect the quality of shallow groundwater are the volume of recharge water and the concentration of solutes, or chemical substances in the infiltrating water. The time of year that the recharge occurs can also affect the chemistry or quality of the recharge, in relation to the timing of application of chemicals and nutrients to the soil, or in relation to natural cycles, such as the conversion of nutrients to available and soluble forms that can move with the water. The nature of the groundwater flow system and the geologic materials in which the well is completed, also play a role. In some settings new recharge water will be more quickly dispersed laterally and to depth, which can dilute the effects of small amounts of recharge.

Limitations on the volume of water moving through the soil, i.e., the recharge to groundwater, will obviously affect the mass of substances that are leached, or moved by this water, and hence, will affect the resultant quality of the shallow groundwater. During periods of drought, when the water-table or potentiometric surface in an aquifer declines, or drops in elevation, recharge impacts can be further limited because of the greater depth needed to penetrate to the water table. However, with such dry conditions, the uptake of nutrients from the soil may be limited as may natural degradation processes. This can result in considerable carryover of compounds in the soil, affording greater than typical concentrations of chemicals once recharge does resume.

Many studies have shown that NO_3-N mass losses and, to a lesser extent, concentrations are highly dependent on the volume of flow in shallow groundwater and tile-effluent systems (e.g., Baker and Johnson, 1981; Hallberg et al., 1986). Other studies have shown that average NO_3-N concentrations typically decline during dry periods, often followed by large increases in concentration when recharge returns, mobilizing the mass of unused nitrate-N that carries over (e.g., Burt et al., 1988).

A review of temporal changes in water-quality from Iowa studies involving longer term monitoring provides a perspective on the SWRL results. Three water quality studies around the state provide some insights on the effect of the drought conditions on groundwater quality changes.

Big Spring Basin Study

The Big Spring Basin project in northeastern Iowa (Fig. 6) has the longest, continuing monitoring record for nitrates and pesticides in groundwater in Iowa (e.g., Hallberg et al., 1984, 1989). As described on Figure 9, recharge has been limited, and groundwater flow has been declining in the basin from the spring of 1986 through 1989. Over this time, with declining recharge, annual nitrate concentrations abruptly declined, dropping from about 9.0 mg/L to 5.7 mg/L NO_3-N in 1989. Annual average atrazine concentrations declined from water year 1986 to 1988, but then increased during 1989. (The details of the 1989 records are under review.) However, the percentage of detections of atrazine in the groundwater samples declined from 100% in water year 1987 to 75% and 87% in 1988 and 1989.

Floyd and Mitchell Counties

Groundwater monitoring has been underway in Floyd and Mitchell counties since 1984 (Libra et al., 1984; Hallberg, 1989a). As part of a population-based sampling program in 1986 and 1987, 184 wells were sampled in these two counties. To maintain a temporal record in relation to other studies, 10 of these wells are being sampled quarterly; the renewed sampling began in October, 1988. As described in Figure 8, water levels in these aquifers have dropped 30 feet or more from the 1986-1987 period to the 1988-1989 period. During this interval, the percentage of these wells with pesticide detections decreased from 70-90% to 70%, 30% and finally no detections in July, 1989 (Fig. 10). Similarly, atrazine detections, multiple pesticide detections, the number of compounds detected, and the concentrations all decreased. Likewise, the median nitrate concentrations and the percentage of wells exceeding the recommended drinking water standard (10 mg/L, NO_3-N) declined, from 40% to 25%.

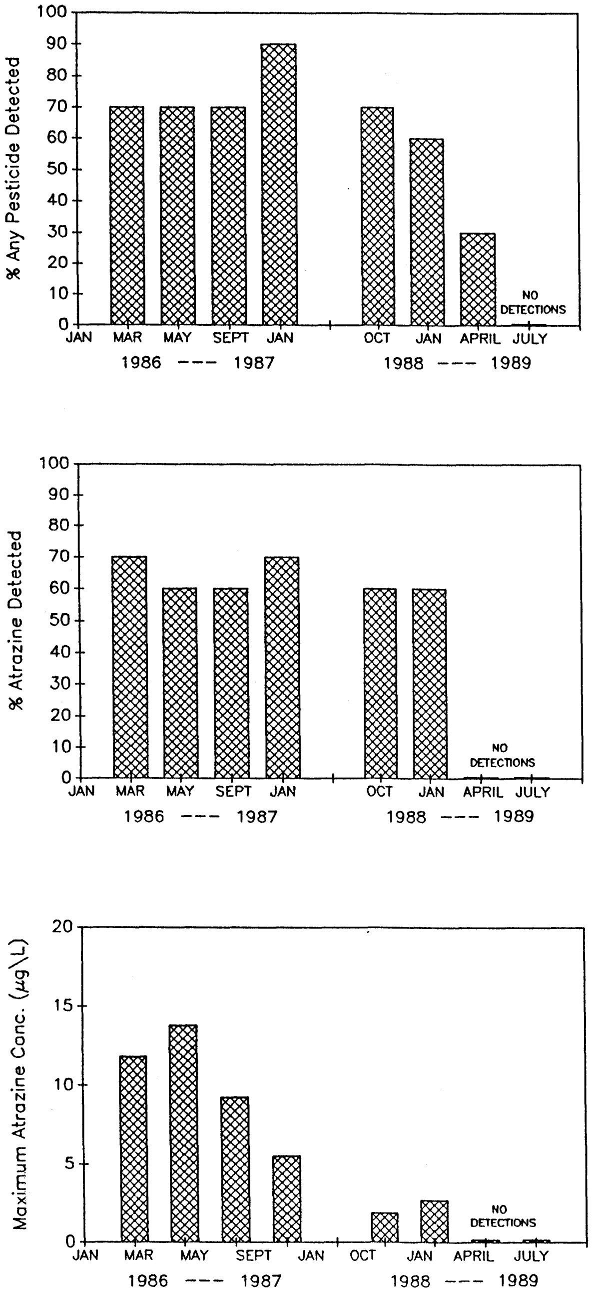


Figure 10. Summary of pesticide detection data for wells monitored in the Floyd and Mitchell counties area, 1986-1989.

Upper Bluegrass Watershed

The Upper Bluegrass Watershed Project is located in a small basin in Audubon County (Fig. 6) and involves on-farm demonstration projects with local farmers. The project is a component of the Integrated Farm Management Demonstration Project, administered by the Iowa Department of Agriculture and Land Stewardship. In addition to the on-farm demonstration projects, private wells, tile lines, surface waters, and monitoring wells in the watershed have been sampled monthly for water quality since 1987. Figure 11 illustrates the change in annual median nitrate concentrations from 1987-1989 from monitoring in the basin. The medians in all categories of sampling sites show statistically significant declines during this drought period. Figure 12 shows the departure from normal precipitation during this time, and the record of nitrate concentrations from one of these wells. As the drought progressed, recharge was limited, the water table declined, and nitrate concentrations dropped significantly. Note, with the return of more substantial precipitation in the late summer and fall of 1989, that nitrate concentrations begin to fluctuate and increase with renewed recharge.

Past research and the monitoring results from these three different locations in Iowa suggest that the SWRL results may present a "best-case" situation because of the temporal coincidence with the drought. The longer term monitoring indicates that the frequency of wells with pesticide detections or with high nitrate concentrations were clearly lower during the period of SWRL than during past periods. Hence, the SWRL sampling likely presents a conservative, or low estimate of the proportion of wells affected by various contaminants, particularly nitrate and pesticides. Never-the-less the SWRL study provides an important base-line measure of the conditions of the state's rural water supplies. Use and interpretation of these results must keep in mind the pronounced drought conditions.

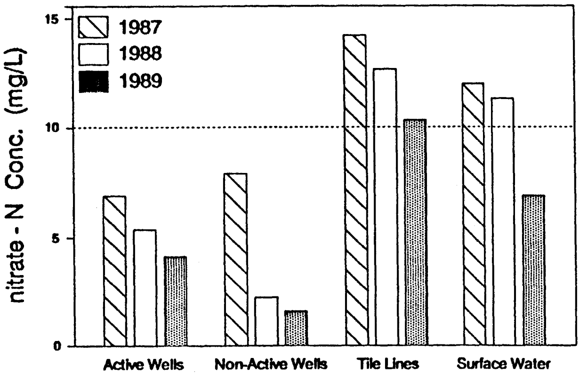


Figure 11. Annual median NO_3-N concentrations from monitoring in the Upper Bluegrass Watershed, 1987-1989.

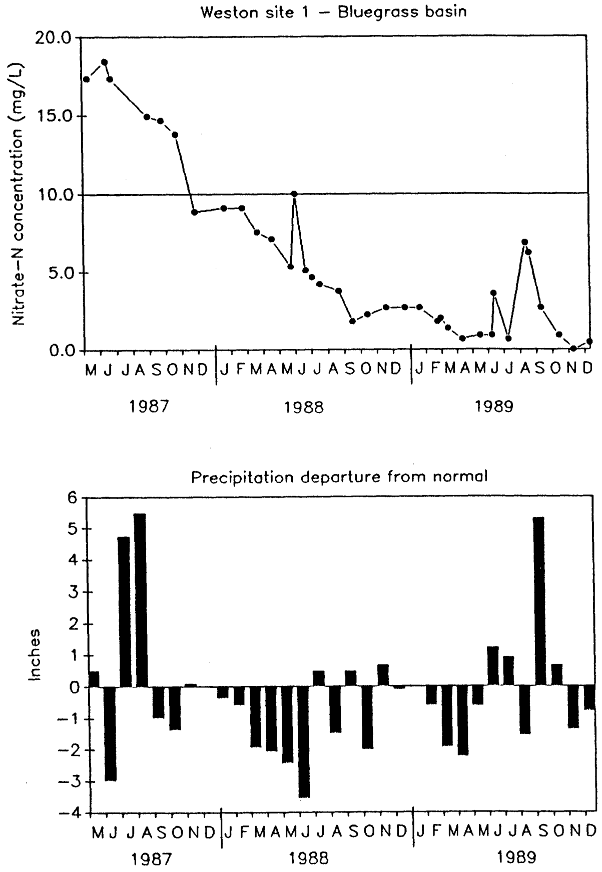


Figure 12. Monthly departure from normal precipitation (Audubon weather station) and NO_3-N concentrations from a well in the Upper Bluegrass Watershed, 1987-1989.

STATISTICAL ANALYSIS

A systematic sample design, stratified by rural population density (based on county population figures), was employed for the SWRL survey (Hallberg et al., 1990). At the end of field sampling the actual number of sites successfully sampled differed slightly from the projected design. This required that stratified estimators (weights) of the proportion of the state's rural population be recomputed to calculate population-weighted means and proportions of various parameters for the state and various subdivisions; such as hydrogeologic regions, crop reporting districts, and well depth. Estimation of means and proportions (and their standard errors) for regions which follow county boundaries is straight forward because the stratum weights can be easily calculated from the known population data.

For example, the estimation of proportions utilizes the following expression:

$$p_{str} = \sum_{h=1}^{L} W_h \, p_h \qquad (1)$$

where L is the number of strata; W_h is the stratum weight, which is the proportion of the population in stratum h; and p_h is the sample proportion in stratum h. There are 3 strata for the state, and for regions that follow county boundaries. These strata have been defined previously (Hallberg et al., 1990) and are based on the rural population density of the counties in Iowa.

The variance of the proportions is estimated as;

$$\mathrm{var}(p_{str}) = \sum_{h=1}^{L} W_h^2 \, \frac{p_h \, [\, 1 - p_n \,]}{n_h - 1} \qquad (2)$$

Similar formulas are utilized for estimation of means and the corresponding variances. The expressions are;

$$\bar{x}_{str} = \sum_{h=1}^{L} W_h \, \bar{x}_n \qquad (3)$$

and,

$$\mathrm{var}(\bar{x}_{str}) = \sum_{h=1}^{L} W_h^2 \, \frac{s_h^{\,2}}{n_h} \qquad (4)$$

The variance expressions assume that the populations within each stratum are essentially in random order. Tests of this assumption show that this holds true for the SWRL sample design (see Hallberg, et al., 1990). If this were not the case, repeated systematic sampling within each stratum would be necessary to calculate stratum-specific variances. Since there is also no reason to assume there exists a cyclical pattern within the sampling framework for each stratum, the above assumption and resulting variance formulas are appropriate.

Estimation procedures for subdivisions of the state that do not coincide with county boundaries, such as for wells <50 feet deep, are complicated because the stratum weights need to be adjusted to these secondary stratification variables.

The state-wide primary stratum weights are based on county population densities (Hallberg, et al., 1990) and are listed in Table 5. These are the weights that need to be adjusted for each secondary stratification variable. Although the population-density strata are used in this application, the proportions of each stratum in the secondary stratification variable are not known and must be estimated. Adjusting the stratum weights consists of a two-step procedure. First, the primary state-wide stratum weights (based on county population densities) are multiplied by the fraction of sites (sorted by the secondary variable) that falls in each of the primary strata. The resulting weights are then normalized across the population-density strata to sum to one for each of the secondary stratum classes.

The procedure used to estimate these new stratum weights is illustrated below using well depth (i.e., wells <50 ft deep and wells $\geq$50 ft deep) as the secondary stratification variable. In this example the number of "shallow" and "deep" wells in each primary stratum is listed in Table 6 (note; these figures do not include 100 wells with unknown depths).

The stratum weight are then apportioned by the fraction of "shallow" and "deep" wells within each stratum. This results in the estimators given in Table 7.

These estimated weights must be normalized to sum to one for the specific well depths. This results in sets of estimated weights which can then be used to estimate means and standard

Table 5. State-wide population strata and proportional weights.

Low population density	0.3331
Medium population density	0.4150
High population density	0.2519

Table 6. Frequency of wells in each primary strata sorted by well depth.

	Well Depth Class		
Stratum	<50 ft	$\geq$50 ft	Total
Low density	47	141	188
Medium density	52	190	242
High density	66	90	156
Total	165	421	586

errors for water quality parameter concentrations according to well depth (Table 8).

The estimated weights that result from this procedure, denoted as W'_h, can be used to estimate means for a specific, population-weighted, well-depth strata, as follows;

$$\bar{y}_{str} = \sum_{h=1}^{3} W'_h \bar{y}_h \qquad (5)$$

and

$$\mathrm{var}(\bar{y}_{str}) = \sum_{h=1}^{3} W'^{2}_h \frac{s_h^2}{n_h} \qquad (6).$$

The use of estimated weights complicates the calculation of secondary stratification variables. The estimator y_{str} is no longer strictly unbiased, and the variance expression is an approximation.

Table 7. Proportion of wells, by depth class, within each population stratum.

Stratum	<50 ft	$\geq$50 ft
Low density	0.0833	0.2498
Medium density	0.0892	0.3258
High density	0.1066	0.1453

[I.e., 0.0833 = 47/188 * (0.3331)]

Table 8. Normalized weights for well depth variable.

Stratum	<50 ft	$\geq$50 ft
Low density	0.2985	0.3465
Medium density	0.3196	0.4519
High density	0.3819	0.2016

[I.e., 0.2985 = 0.0833/(0.0833+0.0892+0.1066) = 0.0833/0.2791]

TEMPORAL VARIABILITY

The influence on water quality results from the temporal variability of sample collection during this project was evaluated. Repeated measures analysis of variance was used to determine the effects on water quality results among the six hydrogeologic regions related to sample collection during different seasons of the year. Seasons in this project are equivalent to quarterly sampling periods (Hallberg, et al., 1990). The General Linear Model (GLM) procedure, included in the SAS computer package, was used to complete the repeated measures analysis of variance. The format of the results are shown in Table 9.

In this project, interest centers on the possible interaction of Seasons (seasonal variation exhibited in quarterly sampling) by Regions. If this source of variation is significant, then it can be concluded that changes over the seasons have a different effect on water quality results in the six hydrogeologic regions. If so, the effects of seasons must be considered separately for each region. This analysis was completed on those analytes that are a health

Table 9. Generalized GLM output for repeated analysis of variance.

Source Variable	df
Regions	5
Sites within regions	52
Seasons (Quarters)	3
Seasons * Regions	15
Seasons * Sites within regions	153
Total	228

concern and had a sufficient number of results above their MDL. The GLM analysis indicates that this effect is not significant. While seasonal changes are evident they do not produce significantly different changes from region to region.

The GLM results for chloride, nitrate and sulfate are summarized in Tables 10 - 12.

There is not a significant interaction with season for chloride and nitrate-N. However, there is an apparent interaction between season and regions for sulfate. There are statistically significant differences by hydrogeologic regions for nitrate-N and chloride, but not by seasons. For sulfate there is an apparent quarterly effect in hydrogeologic region 5, northwestern Iowa.

REGIONAL VARIATION

Variations in water-quality results among regions were further analyzed. Each water quality parameter was evaluated statistically to determine if the variability exhibited among the six hydrogeologic regions was significant. A standard chi-square test was used to evaluate the possibility of variations among and between regions. A contingency table was constructed for each water quality parameter. The table was set up with regions as column headings and categorized water-quality data (i.e., presence or absence of atrazine, mean nitrate-N) as row headings (Rosner, 1982). Weighted proportions of detections and non-detections were multiplied by the number of water quality measurements in each region to calculate the observed values to conduct the tests. The results of these analyses are presented with the respective water-quality data.

THE SWRL SAMPLE; POPULATION DEMOGRAPHICS

Because of its statistical design, the SWRL population forms a very important sampling framework both in terms of rural wells and rural residents. The study population may be used for other appropriate investigations, with the approval of participants, to maximize the utilization of data collected during this primary study. Additional study components have been added as new research ideas and funding sources have become available, to build upon this unique sampling framework. For example, the SWRL population is participating in a systematic survey of radon in rural homes in Iowa. In addition, some of the SWRL population will be used to evaluate pesticide exposure assessment methods with the National Cancer Institute. The confidentiality of the sample population can be assured by CHEEC because of the confidentiality afforded medical and health records.

Comparison of traits (e.g., age) of the SWRL participants shows that the population and the farm sites inventoried were a typical sample of rural Iowans. Of the SWRL sites, 69% were sites where the residents operated the surrounding farm, and 31% were non-farm, "suburban" sites or sites where the residents were not farming at the immediate location. The information collected from each site will continue to be analyzed to provide further insights on the relationship between landuse, well construction, and water quality.

HOUSEHOLD CHARACTERISTICS

The majority of SWRL sites were farm households. At 69% of the sites, the residents actively farmed land at the site where the well was located; at 10% of the sites, the residents indicated that they farmed elsewhere. At the remaining 21% of the sites, the residents were not employed in farming. There were an average of 2.6 household members, or residents, at the

Table 10. GLM analysis of quarterly (seasonal) variation by hydrogeologic region for chloride.

Source Variable	df	MS	F
Region	5	6,924.74	2.97 *
Sites	52	2,333.14	
Season (Quarter)	3	206.04	0.79
Season * Region	15	281.37	1.08
Seasons * Sites	153	260.44	
Total	228		

Table 11. GLM analysis of quarterly (seasonal) variation by hydrogeologic region for nitrate.

Source Variable	df	MS	F
Region	5	987.21	3.27 *
Sites	52	301.96	
Season (Quarter)	3	7.67	0.72
Season * Region	15	11.32	1.06
Seasons * Sites	153	10.70	
Total	228		

Table 12. GLM analysis of quarterly (seasonal) variation by hydrogeologic region for sulfate.

Source Variable	df	MS	F
Region	5	740,495.26	7.87 *
Sites	52	94,125.08	
Season (Quarter)	3	4,422.85	1.10
Season * Region	15	8,450.22	2.10 *
Seasons * Sites	153	4,028.16	
Total	228		

SWRL sites.

Age Distribution

The age distribution of SWRL participants is similar to the projected 1990 state population distribution (Iowa Development Commission, 1985), differing by 4% at most across age brackets (Fig. 13). Comparison of the median age of the SWRL population (38 years) to the 1980 Census state median age (30 years), shows that the SWRL population is somewhat older than the state population. This result is not unexpected; rural Iowa tends to have a larger proportion of older residents than the urban areas.

SWRL farmer age distribution is similar to the distribution reported in the 1987 Census of Agriculture (Bureau of Census, 1989). Proportions of farmers in each age bracket are similar except for the lowest and highest age brackets (Fig. 14). The SWRL sample over represents the oldest age bracket and under represents the youngest age bracket. Correspondingly, the mean age of SWRL farmers (52) is slightly older than the mean age of farmers reported in the 1987 Census of Agriculture (49). Age differences between the Census of Agriculture and the SWRL sample are likely related to differences in the definition of farmers. For the SWRL sample, farmers were defined by their usual occupation, regardless of whether or not they were the primary operator of the farm. The SWRL sample therefore includes a group of semi-retired older farmers who would not be defined as the primary farm operators under the Census of Agriculture definition.

PROPERTY/SITE CHARACTERISTICS

Overall, the property size at SWRL sites averaged 179 acres; sites where the residents were farmers averaged 209 acres and sites where the residents were not farmers averaged 69 acres. For the farm group, 87% of the sites were between 50 and 499 acres, while 60% of the sites in the non-farm group were less than 50 acres in size. Figure 15 shows the distribution of SWRL sites by size. Rate of property ownership was similar for both farmers and non-farmers (77.5% and 78.1 % respectively).

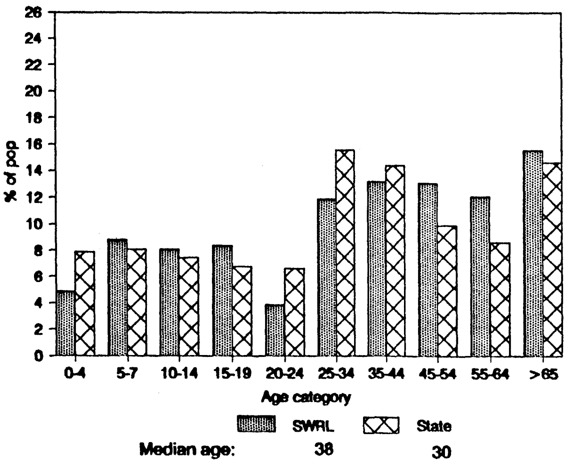


Figure 13. Age distribution of SWRL participants compared to 1990 projected state population (from Iowa Development Commission, 1985).

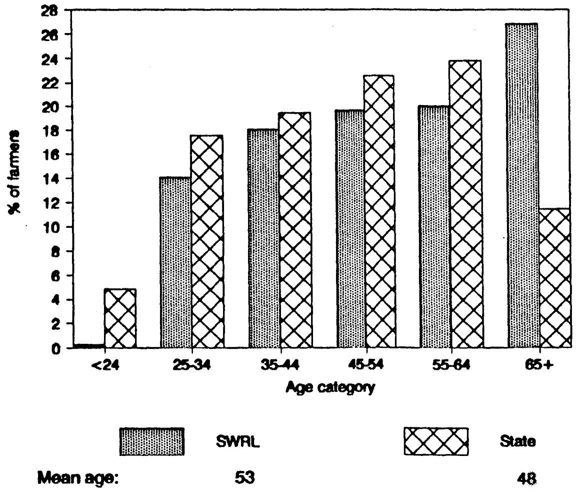


Figure 14. Age of SWRL farmers compared to the 1987 Census of Agriculture.

SWRL farmers reported an average site size smaller than the state average farm size (301 acres). This difference is an artifact of the purpose of the interview instrument used. The SWRL participants were asked how many acres comprised the property/site where the sampled well was located, but farmers were not specifically asked how many total acres they farmed (i.e., including other properties). Hence, the acreage reported by farmers, in many cases, does not include all of the land that comprises their farming operation.

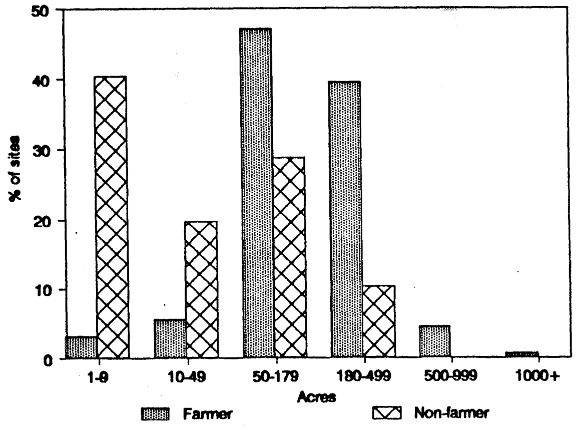


Figure 15. Percent of SWRL farmer and non-farmer sites by size category.

FARMING OPERATION

Cropping patterns reported by SWRL farmer participants are typical of Iowa, with corn (92%) and soybeans (79%) the most frequently raised crops. Livestock patterns are also typical of Iowa with swine (52%) and beef cattle (51%) being the most commonly raised livestock. Another recent statewide survey by Padgitt (1989) reported similar cropping and livestock patterns among Iowa farmers. Table 13 summarizes the cropping and livestock patterns of SWRL farmers and compares them to the results reported by Padgitt. Though both SWRL and Padgitt conducted state-wide surveys there are differences in how the studies selected and defined farmers. Padgitt's survey was for 1988; the SWRL survey information covers portions of crop-year 1987 as well as 1988. The SWRL selection process included a larger portion of smaller farms than did Padgitt's survey.

Fertilizer Usage

Farmers who actively farmed the site where the well was located were asked questions pertaining to their fertilizer use on the crops which they raised. Most SWRL farmers (74%) report applying nitrogen fertilizer to corn in early spring. A much lower proportion report late spring/early summer (13%) or fall applications (6%).

Average fertilizer application rates on corn varied by hydrogeologic region and by crop rotation (Table 14). The highest statewide average rate was, as expected, for continuous corn. The average fertilizer application rates for corn following soybeans, alfalfa, or oats were lower than for continuous corn at both state and regional levels of comparison.

The SWRL statewide average rate of 132 pounds/acre of nitrogen fertilizer applied to continuous corn is similar to the state-wide average reported by the Padgitt of 136 pounds/acre. Both values are slightly less than the 1988 average of 139 lbs/ac reported by the Agricultural Statistics Service (Skow and Holden, 1989). Given the variance in rates these differences are not significant. For added perspective, in 1987 the Agricultural Statistics Service reports an average rate of 132 lbs/ac for corn, and in 1989, an average rate of 128 lbs/ac.

At the regional level there was greater variation between the results of the Padgitt and SWRL studies. This is likely related to the relatively small sample size at the regional level in both studies, the variation in fertilizer application rates of farmers, and differences in how the samples were drawn. However, in all cases the mean values are within one standard deviation. Table 15 summarizes the average nitrogen fertilizer application rates for continuous corn by the seven ISU-CES areas.

Twenty-three percent (23%) of SWRL farmers report having reduced their use of nitrogen fertilizers since 1980, while 69% report no change and 6% report increases in use. For the most recent growing season, 44% report having soil tested. Padgitt reports from his sample that 12% of farmers note reducing N-fertilizer use since 1985; while 1% report increases. Many SWRL farmers report giving nutrient (nitrogen) credits for manure applications on corn ground or for crop rotation with alfalfa, soybeans, or other crops (Table 16).

Table 13. SWRL farmer cropping and livestock enterprises compared to those reported by Padgitt (1989). (Not all categories are comparable; nr=not reported.)

Enterprises:	% of Farmers Reporting	
Crops;	SWRL	Padgitt/CES
Corn	92	99
Soybeans	79	83
Small Grain	56	40
Wheat	6	nr
Oats	55	nr
Other	2	nr
Hay	66	58
Alfalfa	63	nr
Other	13	nr
Specialty	5	nr
Set Aside	65	85
CRP	10	11
Livestock;		
Dairy	11	13
Beef Cattle	51	nr
Stock	nr	29
Feedlot	nr	33
Sheep	11	5
Poultry	10	7
Swine	52	54
Other Animals	9	nr

Pesticide Usage

Farmers who actively farmed the site where the well was located were also asked questions relating to their herbicide and insecticide use on crops at the site. The percentages of SWRL farmers growing a particular crop and using a particular herbicide/insecticide (e.g., percent of corn growers using atrazine) are summarized. The 1985 Pesticide Survey, conducted by CES, gathered information on pesticides used on corn, soybeans, and pasture in Iowa (Wintersteen and Hartzler, 1987). Because of this study's close proximity in time to the SWRL study, and because it also used a state-wide sample frame, the 1985 Pesticide Study provides a useable comparative reference. Pesticide-use data for the 1985 Pesticide Survey was reported in percentage of acres of crops treated with pesticides (e.g., percent of corn acres treated with atrazine). The SWRL study asked farmers to report use of specific pesticides on crops rather than the number of crop acres treated with specific pesticides. Direct comparisons between the two studies are therefore not possible because of the differences in measurements of pesticide use. However, the results of the two surveys are very similar in their relative distributions.

Table 14. Average reported nitrogen (N), phosphorus (P), and pottasium (K) fertilizer rates (pounds/acre) used for corn; by crop rotation and by hydrogeologic region. s.d. = standard deviation.

				Hydrogeologic Regions:											
Corn Rotation:		State-wide rates		NE-1		EC-2		SC-3		SW-4		NW-5		NC-6	
		Mean	s.d.	Mean	s.d.	Mean	s.d.	Mean	s.d.	Mean	s.d.	Mean	s.d.	Mean	s.d.
								lbs-nutrient/acre							
Continuous corn:															
	N	132	43	121	48	144	42	139	36	124	36	128	57	142	15
	P	48	36	40	27	56	39	54	40	46	25	636	26	47	43
	K	53	41	55	46	60	41	52	45	39	19	38	32	54	55
Corn following soybeans:															
	N	123	38	112	42	133	36	115	44	114	35	110	24	129	35
	P	55	36	41	36	62	45	56	27	41	27	44	26	66	27
	K	64	45	64	56	76	52	64	38	36	27	42	29	78	35
Corn following alfalfa:															
	N	95	51	78	48	107	51	128	16	83	49	71	31	91	48
	P	43	39	31	34	55	49	57	1	39	30	19	24	50	53
	K	49	45	34	36	68	53	66	4	37	26	22	33	54	57
Corn following oats:															
	N	100	55	116	48	98	67	101	0	103	54	93	45	132	13
	P	42	36	62	43	41	44	70	0	36	28	41	27	53	28
	K	40	42	99	81	33	32	82	0	25	19	37	33	65	28

Table 15. Average fertilizer-nitrogen application rates for continuous corn; from SWRL survey and Padgitt state-wide survey (Padgitt, 1989). s.d. = standard deviation.

	Fertilizer-nitrogen rate (lbs-N/acre) applied to continuous corn:			
CES Areas	SWRL survey mean	s.d.	Padgitt/CES mean	s.d.
NE	138	44	127	
EC	123	52	147	
SE	141	38	141	
SW	121	35	134	
NW	128	36	118	
NC	128	57	154	
C	142	14	138	
State-wide	132	43	136	41

Herbicide Usage

A majority of farmers reported using herbicides on corn and soybeans. State-wide, 93% of SWRL farmers reported using at least one herbicide on corn, while 88% reported using at least one herbicide on soybeans. This reported herbicide usage is somewhat lower than reported rates for percent of acres treated reported in the 1985 Pesticide Survey (Table 17). In the SWRL study, herbicide usage on corn and soybeans varied among hydrogeologic regions. The reported herbicide usage on both corn and soybeans was lowest in the Northeast (region 1) and greatest in the North-Central region (6) (Fig. 16).

Corn Herbicides. State-wide, atrazine was the herbicide most farmers reported using with corn, followed by Lasso (alachlor), Bladex (cyanazine), Dual (metolachlor), Banvel (dicamba) and 2,4-D.

Figure 17 illustrates that the reported use of corn herbicides by SWRL farmers closely mirrors the usage reported in the 1985 Pesticide Use Survey. Regionally, there are variations in the reported use of specific pesticides. For example, 2,4-D/Banvel was used more commonly than atrazine in the Northwest (crop-reporting district 1), whereas reported atrazine use was more

Table 16. Percentage of all SWRL farmers applying nutrient (nitrogen) credits for manure or crop rotation.

Manure	50%
Crop rotation with:	
Soybeans	70%
Alfalfa	48%
Other Crops	41%

Table 17. State-wide percentages of reported herbicide use by SWRL farmers compared to the proportion of 1985 acres treated from Wintersteen and Hartzler (1987).

Crop	SWRL farmers using herbicides	1985 acres treated
Corn & soybeans	90%	98%
Corn	93%	98%
Soybeans	88%	98%

common in the Southeast (district 9). The regional variation seen in the SWRL data is very comparable to that reported in the 1985 Pesticide Use Survey (Fig. 18).

Soybean Herbicides. Figure 19 compares the reported soybean herbicide use in the SWRL study to the percentage of acres treated from the 1985 Pesticide Survey. The distribution patterns of the two studies are again similar, with Treflan accounting for the greatest proportion of use. In both studies, other major herbicides account for considerably less of the herbicide usage on soybeans. Figure 20 illustrates the comparable proportions of reported use by crop reporting district.

Insecticides

The proportion of SWRL farmers using insecticides on corn or soybeans is much smaller than those using herbicides on these crops. The SWRL survey shows 45% of farmers using insecticides for corn, similar to the 42% reported for corn acres treated in the 1985 Pesticide Use Survey. Fewer SWRL farmers (11%) reported using soybean insecticides. Soybean insecticide usage is not available for the 1985 Pesticide Use Survey, but insecticide use is not typically needed for soybeans in Iowa. The outbreak of spider mites during the 1988 drought prompted far greater than normal use of insecticides on Iowa soybeans. The proportion of insecticide usage on corn and soybeans varied by hydrogeologic region (see Fig. 21).

Corn Insecticides. Comparisons were made of insecticides used for control of specific pest problems (i.e., corn rootworm larval control, etc.) between the SWRL survey and the 1985 Pesticide Survey. The use distribution patterns were very similar from the SWRL survey and the 1985 Pesticide Use Survey. The most common use of insecticides is for corn rootworm larval control (Fig. 22). For European corn borer control and black cutworm control insecticide use is lower and more variable.

DEMOGRAPHIC SUMMARY

The SWRL demographic and farming practices data are very comparable to other state-wide surveys and census statistics. The comparative results for many factors, ranging from sample age distributions to farm chemical usage, indicate that the SWRL population is clearly a representative sample of rural Iowans and of Iowa farmers and their agricultural practices.

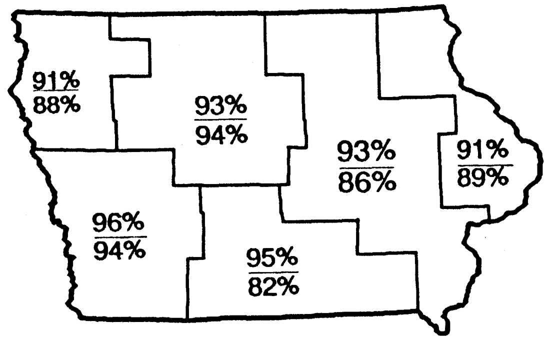


Figure 16. Percentage of SWRL farmers using herbicides on corn (upper figure) and soybeans (lower figure) by hydrogeologic region.

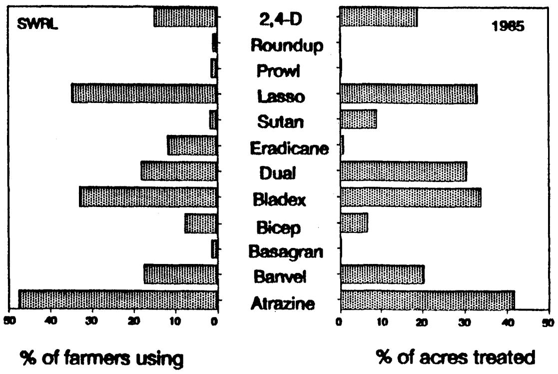


Figure 17. Major corn herbicides used by SWRL farmers compared to 1985 corn acres treated (Wintersteen and Hartzler, 1987); summarized by Crop Reporting District (Districts 1-9).

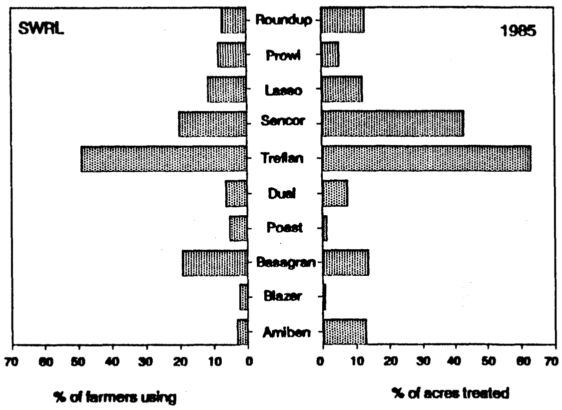


Figure 19. Major soybean herbicides used by SWRL farmers compared to 1985 survey of soybean acres treated (Wintersteen and Hartzler, 1987).

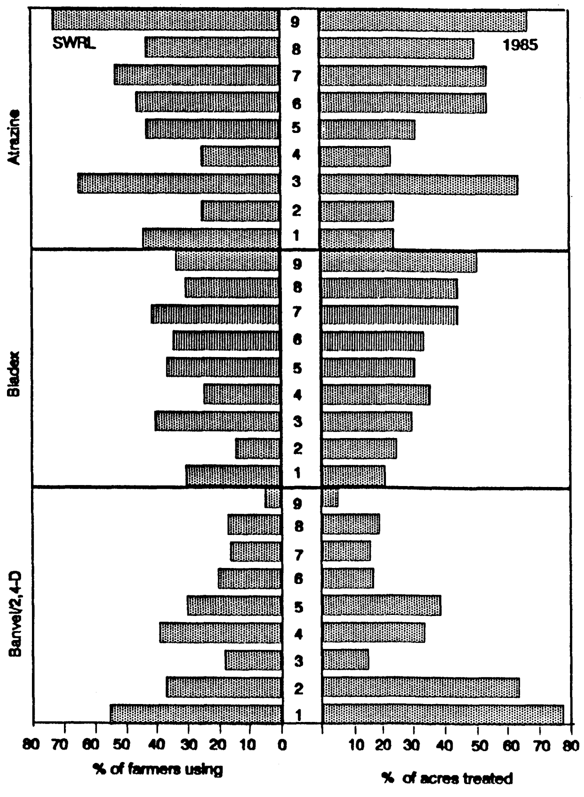


Figure 18. SWRL farmers using atrazine, Bladex, or Banvel/2,4-D on corn, compared to 1985 corn acres treated (Wintersteen and Hartzler, 1987), by Crop Reporting District (indicated by number in middle graph).

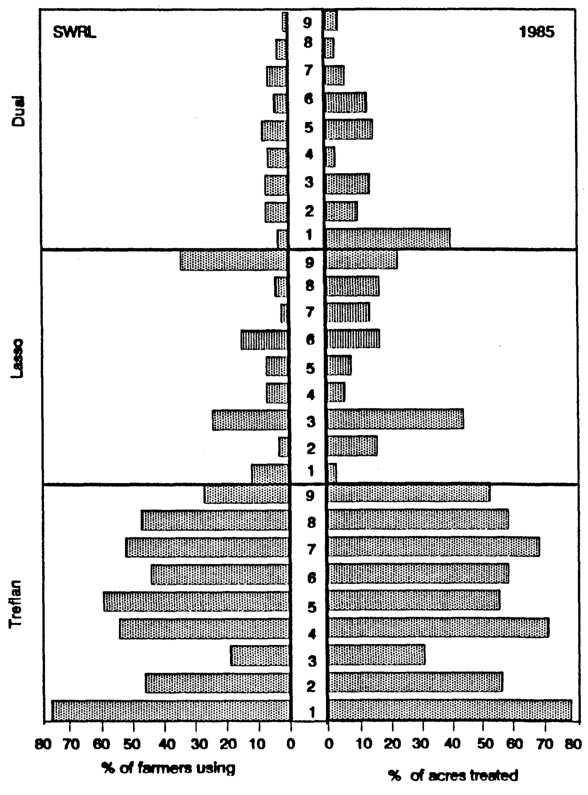


Figure 20. SWRL farmers using Treflan, Lasso, and Dual on soybeans compared to 1985 soybean acres treated (Wintersteen and Hartzler, 1987); by Crop Reporting District.

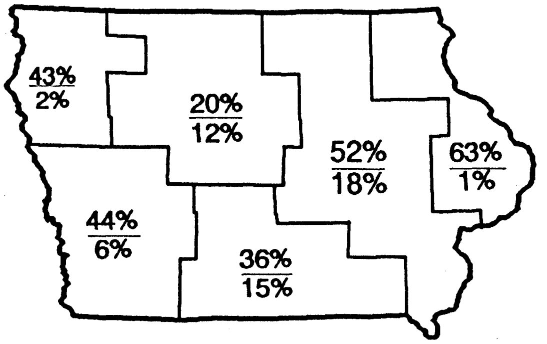


Figure 21. Percent of SWRL farmers using insecticides on corn (upper figure) and soybeans (lower figure) by hydrogeologic region.

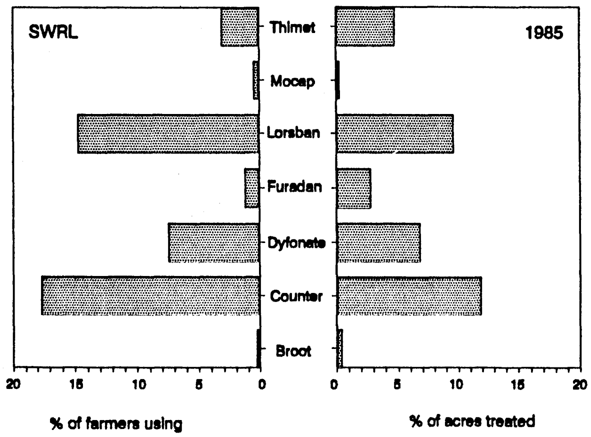


Figure 22. Reported insecticides used for corn rootworm larval control by SWRL farmers and for the 1985 Survey acres treated (Wintersteen and Hartzler, 1987).

QUALITY ASSURANCE, QUALITY CONTROL

The quarterly laboratory and field procedure quality-assurance (QA) audits (Hallberg, et al., 1990) found only minor inconsistencies in record-keeping and a minor problem that holding time requirements were exceeded for a few analytes. (The holding time is defined as the time between the sample collection and its analysis, or extraction/stabilization in the laboratory.) The record-keeping was corrected at the time of the audits. The holding time problem was related to hardware breakdown and was corrected as soon as new equipment was brought on line. The audits indicated that each lab was implementing their internal QA/QC programs and that these programs were adequate to meet SWRL project requirements.

A database for the SWRL project was implemented by PM&EH. As the different laboratories transferred analytical data to the SWRL project, the data were segregated into distinct groups for ease of processing and verification. The database (set A) contains all the well-water analyses for the primary visit to the 686 sites in the SWRL study. This is the data used in the statistical population assessment of state and regional water quality. The ten percent repeat sites, quarterly sites, and duplicate samples (Hallberg, et al., 1990) were segregated into separate data sets for faster processing. All site inventory results were set up in seperate databases, as well.

The final review and validation of the primary SWRL database, consisted of a check on data entry, data precision, and review of the holding times for both the raw sample and organic extracts. A systematic 10 per cent subset of all entries for the 686 sites in the primary database was visually reviewed to evaluate data entry. The subset selection started with a list of the site numbers in the primary database, in ascending order. This also grouped the sites by county and placed the county numbers in ascending order. The first county listed (county number 01 = Adair) had six sites. A random number between one and six was generated with a statistics software package. The number 2.31 was generated and rounded to 2. The second site in Adair county was thus the first site selected for the subset and every tenth site number there after was also chosen. This produced a list of 68 site numbers. All analytical data and site information associated with these sites were printed from the SWRL database. These printouts were visually compared to the original laboratory data sheets and site files. Any discrepancies were noted and the Quality Control officer reviewed them to verify that an error had occurred and took appropriate steps to rectify the problem. This analysis revealed that no errors occurred in the entry of the analytical results. There were eight errors in other parts of the database. These were limited to six dates that were not entered or entered incorrectly, one juxtaposition of two yes/no responses from a site description form, and one aquifer designation that had not been updated. These few errors were corrected in the database. All of these errors are minor and do not affect the integrity of the database.

The precision of the analytical data was evaluated by statistical comparison of the analyses for both the duplicate samples with the regular sample that was matched by date and time. An independent T test was calculated for all these correlated samples. For those analyses with few detections, the T test was run on only those analytes that had a detection in either sample, with the results below the detection level set to zero. There were no significant differences found for any analyte. This is one indicator of the good precision maintained by all the labs during the analysis of the SWRL samples.

A final check was made of each laboratories sample holding times during the SWRL project. EEL analyzed samples for major cations and anions. Sample holding time for major ion analysis is defined as the time between the collection of the sample and its analysis. The lab met the specified holding time requirements for cations and anions (28 days maximum) 92% and 94% of the time for samples in the primary database. The few samples that failed to meet the holding times should not adversely affect the analytical accuracy considering the nature of the analytes. The holding time for anion and cation analysis was arbitrarily set low for logistical reasons rather than just for analytical concerns. Recommended holding times for many cation and anion analyses range up to several months.

Analysis of both sample and extract holding times were made of pesticide analyses in the primary database. Sample holding time for pesticide analysis is defined as the time between the sample collection and the first organic extraction of the sample. Sample holding time was limited to seven days in the SWRL QA/QC plan. ATL analyzed samples for common herbicides and selected metabolites. Sample holding time was met over 98% of the time for all the compounds in the primary data set that was analyzed by ATL. Extract holding time is defined as the time between the first organic extraction of a sample and analysis of the final residue by gas chromatography. The extract holding time was set at a maximum of 40 days (under refrigeration). The extract holding time was not met as often, with 81% to 88% of the samples meeting the extract holding time requirement for parent compound analysis and 58% to 81% meeting the extract holding time requirement for metabolite analysis. This could increase the chance of false negatives in the primary data set for several herbicides and metabolites (i.e., compound degradation could decrease concentrations and hence may lower some true detections below the quantitation limit). A pesticide degradation analysis was attempted by using ATL's field spike data, correlating the concentration of specific compounds to the number of days since it had been extracted. There was no obvious linear correlations indicating a problem had occurred. However, there is typically considerable variance in this type of analysis.

UHL utilized the 10% subset described earlier to check compliance with specified holding times. The subset indicated that the analyses for nitrogen species and coliform bacteria all met the holding time requirements for these tests. It also indicated that the sample holding time for acid herbicide and insecticide analyses all met the QA plan requirements. All extracts for insecticide analysis and about 97% of the acid herbicide extracts were analyzed in the time frame specified for the SWRL project.

The QA/QC audits show that the primary SWRL database correctly represents the analytical results from the different laboratories. It also indicates that the precision of the analytical data is very good. The few holding times that were exceeded may increase the number of false negative results for selected herbicide and metabolite analyses. However, any increase in false negative reports makes the database somewhat more conservative when used to summarize herbicide and metabolite detections.

EQUIPMENT CHANGES

As noted above, there were some equipment problems related to the holding time problem. During the course of the SWRL study the ATL lab switched from using packed columns for the GC pesticide analysis to capillary columns to improve resolution and lab processing time. While some time of adjustment was involved, extensive replicate testing shows that the capillary columns improved the precision of the operations. The coefficient of variation with capillary columns is considerably lower than with packed columns. This change took place during the last quarter of SWRL sampling. To maintain consistency, all SWRL samples analyzed during and after the installation of the capillary columns were actually run on 4 columns; 2 with electron-capture detectors and 2 with nitrogen-phosphorus detectors. Hence, there was no adverse affect on the analyses or data from this change.

GC-MS CONFIRMATION STUDY

As part of continuing quality assurance efforts, additional confirmation of pesticide detections from gas chromatography (GC) methods was determined using gas chromotgraphy-mass spectroscopy (GC-MS). GC-MS confirmation was tested for varying concentrations of six pesticides in groundwater samples: alachlor, atrazine, cyanazine, metribuzin, metolachlor, and propachlor. These pesticides are commonly used in Iowa and are the most common pesticides detected in Iowa groundwater by the laboratories.

Four liters of groundwater from seven sites were extracted with methylene chloride, eluted through silica gel column for separation, and analyzed by ATL using gas chromatography - electron capture detectors (GC-ECD) and dual packed columns, for analyte identification and quantification. Sample extracts were also analyzed using gas chromatography-mass spectroscopy (GC-MS) for confirmation. The confirmation criteria were those specified by the EPA Contract Laboratory Program:

1. All ions present in the standard mass spectra at a relative intensity greater than 10% (most abundant ion in the spectrum equals 100%) must be present in the sample spectrum.
2. The relative intensities of ions specified in the above criterion must agree within 20% between the standard and sample spectra.
3. Ions greater than 10% in the sample spectrum but not present in the standard spectrum must be considered and accounted for by the analyst making the comparison. When GC-MS computer data processing programs are used to obtain the sample component spectrum, both the processed and the raw spectra must be evaluated.
4. If a compound cannot be verified by all of the above criteria, but in the technical judgement of the mass spectral interpretation specialist the identification is correct, then the identification shall be reported.

Table 18. Lowest pesticide concentrations confirmed by GC-MS analysis.

Analyte	GC-ECD MDL	GC-MS lowest Confirmed Concentration
	---- in ug/L	----
alachlor	0.02	0.35
atrazine	0.13	0.60
cyanazine	0.12	1.56
metolachlor	0.04	1.02
metribuzin	0.01	0.85
propachlor	0.02	0.58

The samples and standards were analyzed on the same day using the ATL GC and the GC-MS instrument located at the UHL. Water samples from wells that, in the past, had contained multiple pesticide residues, over a range of concentrations, were used for the study. In the judgement of Mr. Delon Maas, the analytical chemist performing the study, the presence of each of the six herbicides in the samples was successfully confirmed (Table 18). GC-MS detection limits are higher (i.e., not as sensitive) than for GC equipment and procedures; the confirmed GC-MS concentrations reported were for the lowest confirmations in the sample sets analyzed.

POSSIBLE ATRAZINE INTERFERANT STUDY

Tris (2-chloroethyl) phosphate, commonly called Fyrol CEF, has been reported by Monsanto Agricultural Company representatives (letters from Ronald G. Smith and Andrew Klein of June 1, 1988) as an interferant in the analysis of atrazine in some laboratories. Previous discussions with Monsanto representatives suggest that Fyrol CEF is a fire retardant used in industry. Plastic well casing material may become contaminated with Fyrol CEF, resulting in possible contamination of well water samples with traces of the compound.

Results of comparing standards of tris

(2-chloroethyl) phosphate provided by Monsanto with standards of atrazine showed no interference using the methods and two column confirmation employed in the ATL laboratory. The compound did not interfere with any of the 17 pesticides routinely measured by the laboratory. However, some possible interference was noted for a metabolite (hydroxy-alachlor).

In addition to these studies, ATL conducted an extensive holding time study for pesticide degradation prior to the start of SWRL.

WATER QUALITY RESULTS

The basic water-quality results of the SWRL survey will be summarized in the subsequent sections. Unless otherwise noted, all data reported are population-weighted values. As the first stage in making these data publicly available a series of fact sheets were prepared and released which summarized the water-quality results most important from a public health perspective. Table 19 summarizes these data, with the addition of fecal coliform data that were not available at the time of the initial summaries.

Table 19. Summary of SWRL water-quality results previously released in fact sheets.

	All wells	Wells <50 ft deep	Wells ≥50 ft deep
Nitrate-N:			
% wells >10 mg/L	18.3%	35.1%	12.8%
mean conc., mg/L	6.2	11.2	4.3
max conc., mg/L	100.	95.	100.
HAL, mg/L	10.0	10.0	10.0
Total Coliform Bacteria:			
% wells with TCB	44.6%	71.5%	36.3%
FECAL Coliform Bacteria:			
% wells with FCB	5.4%	8.4%	4.3%
Pesticides:			
% wells with any pesticide detection	13.6%	17.9%	11.9%
%wells with 2 or more pesticide detections	5.3%	8.5%	4.1%
Atrazine (total):			
% wells with detections	8.0%	9.6%	7.6%
mean conc., ug/L	1.1	1.7	0.8
max conc., ug/L	7.7	7.7	4.9
HAL, ug/L	3.0	3.0	3.0

The characteristics of Iowa groundwaters, as sampled from the population of private wells in the SWRL survey can be summarized by a few fundamental parameters. Groundwater temperatures typically approximate the mean annual air temperature, which varies from about 8°C (46°F) to 12°C (53°F), from north to south in Iowa. The mean and median temperatures recorded for water samples from the SWRL survey was approximately 12°C (53°F), with a standard deviation of about 3°C (5°F). The majority of temperatures varied from (10th percentile) 9°C to (90th percentile) 15°C. Some temperatures were affected by water storage systems, and could be expected to be slightly greater than true groundwater temperatures. In terms of acidity, Iowa groundwaters are near neutral, as reflected in the mean or median pH values of 7.1 pH units for the SWRL samples; the majority of the data only range from 6.8 (10th %) to 7.5 (90th %).

Typically, Iowa groundwaters are moderately hard with a mean hardness of 432 mg/L. With respect to alkalinity the data show a mean of 200 mg/L and a median of 190. The SWRL samples showed a mean specific conductance of 756 microsiemens/cm @ 25°C, a median of 625, ranging from 400 (10th %) to 1,280 (90th %). These values are typical of water ranging from low to moderate values of total dissolved solids. These factors will be further discussed in later sections on the ion chemistry of the waters. The degradation of many contaminants, particularly organic compounds, can be affected by temperature, alkalinity, and pH, but the narrow range of these parameters, particularly pH, does not indicate that any major differences are likely across the state.

In the subsequent sections of the report, the data will be summarized by concentrations or occurrence of particular analytes and by the proportion of the wells and the population affected. A brief discussion of the nature and source of various compounds is included. A brief review of the environmental health implications is also included where appropriate. Later sections of the report will discuss the inter-relationships among water-quality parameters, and relations to possible causal factors and environmental patterns.

WELL DEPTH

Variations with well depth will be discussed for each of the water-quality variables. As a preface to these discussions, this section will provide an overview of well depth observations from SWRL.

Well depth is clearly one of the most important variables affecting the potential for contaminants from activities at the land-surface to affect the quality of water derived from a well. In reality, it is the depth of the open or screened portion of the well from which the water is pumped that determines this. Hence, in a deeper well the depth of the casing below the land surface is the important parameter.

All regions of the state have appropriate settings for shallow wells; such as sand-point wells in alluvial aquifers along major streams. Throughout the state the water table is also quite shallow (within 5 to 20 feet), except where a combination of high relief and a substantial thickness of relatively high permeability materials are near the land surface. In these settings the groundwater surface is found at greater depths. Even in these areas, permanent zones of saturation are often perched closer to the land surface, within lower permeability glacial deposits which occur at the land surface.

We have found from many past studies, comparing actual well records with owners reports, that rural Iowa landowners recall and report well depth quite accurately. Typically, 75% are accurate (within rounding to the nearest 25-50 feet), with another 15% stating they don't know. Recollections are seldom this good for casing depths or the depth to a screened interval. SWRL participants were asked by letter and phone to review their well records before the on-site interview, which should help to maintain this accuracy.

For a majority of private wells, total well depth and casing depth are closely related, and water-quality parameters can be related to well depth (Hallberg et al., 1983; Libra et al., 1984). Some deeply drilled wells that encounter bedrock at a shallow depth, may be deeply drilled because the water table is deep or because better water producing zones are at depth. In such cases the well may only be cased into the top of the bedrock, with the well boring held open by the bedrock itself. Though the total well depth may be quite deep (e.g., 300 feet) the casing may be shallow (e.g., 30 feet), in such cases. This is commonly the case with some apparently deep wells that exhibit surficial contaminants. Because of their shallow casing, they are effectively shallow wells, allowing surficial contaminants to enter the well, at least intermittently (Libra et al., 1984; Hallberg et al., 1984).

SWRL Well Depth Variations

State-wide the SWRL survey shows that private wells exhibit a mean well depth of 136 feet, a median of 100 feet, with a maximum of 880 feet. As the maximum depth suggests, there is considerable variation. The private well data have a standard deviation of 129 feet; the 10th percentile rank of well depths is about 30 feet, and the 90th percentile is 300 feet. As discussed in the definition of the hydrogeologic regions, much of this variation in well depth occurs regionally in Iowa in relation to variations in the hydrogeologic setting. Table 20 summarizes well-depth observations from the SWRL survey. Approximately 28% of private wells state-wide are <50 feet deep; 77% are <200 feet deep; and only about 10% >300 feet in depth. Similar to other surveys, approximately 15% are reported as unknown. Water quality relations to wells <50 feet and >50 feet deep are used as a starting point for review in this report. Considerable past work has shown that a depth of 50 feet serves well as a definable approximation between "shallow" and "deep" aquifer regions and between shallow and deep wells that exhibit significant differences in water quality among surficially derived contaminants, such as nitrates and pesticides. In some hydrogeologic settings significant contamination continues to greater depth (Hallberg and Hoyer, 1982; Hallberg et al., 1983, 1984; Libra et al., 1984; Bruner and Hallberg, 1987; Hallberg, 1989a).

Figures 23 and 24 summarize these well depth variations across the hydrogeologic regions. The proportion of wells <50 feet deep varies from about 6% in Northeastern Iowa

Table 20. Proportion of wells by well depth, state-wide and by hydrogeologic regions.

Well depth range feet	State-wide %	Hydrogeological regions					
		NE (1) %	E (2) %	SC (3) %	SW (4) %	NW (5) %	NC (6) %
<25	6.5%	4.5%	4.3%	11.9%	8.9%	15.9%	1.2%
25-49	21.4%	1.2%	8.3%	43.0%	45.3%	36.9%	11.7%
50-99	20.9%	3.5%	20.7%	23.9%	25.9%	21.5%	26.0%
100-149	17.3%	28.6%	22.7%	8.2%	6.2%	6.0%	23.3%
150-199	10.5%	13.1%	17.0%	4.9%	1.5%	2.0%	13.6%
200-299	13.1%	26.5%	14.7%	2.1%	9.4%	8.0%	15.3%
≥300	10.4%	22.7%	12.2%	6.0%	2.9%	9.7%	9.0%
unknowns as % of total	14.6%	21.8%	15.7%	15.0%	14.3%	7.3%	11.2%

Well depth range feet	State-wide cumulative %	Hydrogeological regions					
		NE (1) cumulative %	E (2) cumulative %	SC (3) cumulative %	SW (4) cumulative %	NW (5) cumulative %	NC (6) cumulative %
<25	6.5%	4.5%	4.3%	11.9%	8.9%	15.9%	1.2%
25-49	27.9%	5.7%	12.5%	54.9%	54.2%	52.8%	12.9%
50-99	48.8%	9.2%	33.3%	78.8%	80.0%	74.3%	38.8%
100-149	66.1%	37.8%	56.0%	87.0%	86.3%	80.2%	62.1%
150-199	76.6%	50.8%	73.0%	91.9%	87.7%	82.3%	75.8%
200-299	89.7%	77.3%	87.8%	94.0%	97.1%	90.3%	91.0%
≥300	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%

(NE-Region 1) to over 50% in the South-Central (SC-Region 3), Southwestern (SW-Region 4), and Northwestern (NW-Region 5) regions of Iowa. In the SC, SW, and NW nearly 75% of all wells are also <100 feet deep. The median (50th percentile) well depth parallels these trends, with NE Iowa having a median of 190 feet and the southern and western areas all exhibiting medians of 40 feet. These well depths reflect the regional hydrogeology. In NE Iowa the bedrock and bedrock aquifers occur at a shallow depth, but the local relief is the greatest in the state. Hence, the potentiometric water surface can be deep, especially over the long-term, including drought periods. In this region we would expect the relationships between water quality and well depth to be much less clear, because of the greater depth of groundwater circulation and well construction considerations (e.g., deep wells, but shallow casing). The SC, SW, and NW regions are dominated by shallow wells because regional aquifers are often very deep, and hence, drilling may be prohibitively expensive. The water in deeper aquifers also can be undependable, particularly because natural water quality may be poor (e.g., high sulfates, high dissolved iron, etc.).

In the SC area, bedrock is commonly shallow, but the Pennsylvanian bedrock is highly variable in terms of lithologies and aquifer characteristics, and water is typically highly mineralized. In SW and NW Iowa, bedrock aquifers (e.g., Dakota aquifer) are typically deep to very deeply buried by glacial deposits. Hence, bedrock wells are often avoided because of their expense, and beyond certain depths they are also quite mineralized. Hence, the southern and western regions are dominated by "water-table" wells;

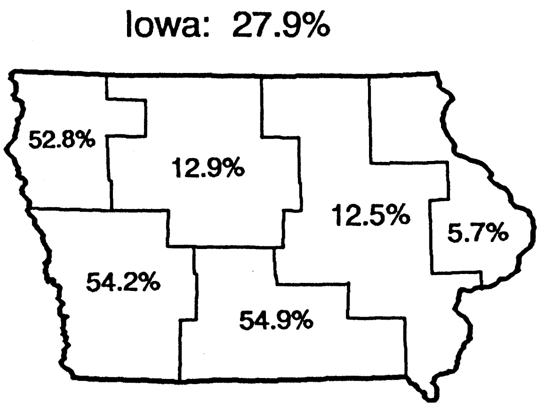


Figure 23. Percentage of wells <50 feet deep in the hydrogeologic regions.

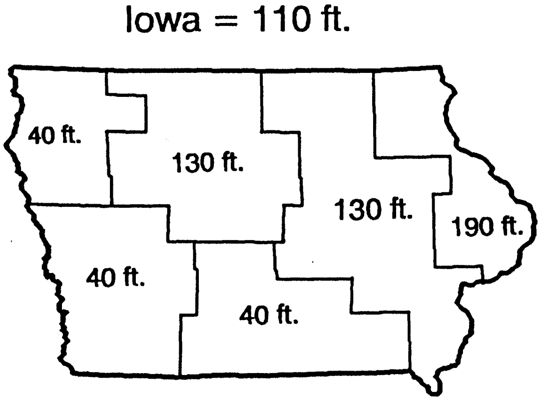


Figure 24. Median well depth in the hydrogeologic regions.

wells which by design tap the very shallowest portions of the groundwater system. Such wells often are not drilled into an aquifer, but are simply completed in the low-permeability glacial deposits, which in a strict sense, would be considered an aquitard. In these settings large diameter "seepage" wells are installed. These wells are typically about 3 feet in diameter with depths of 30 to 50 feet. Their casing, or lining, is generally comprised of stacked, 2-3 feet tall section of concrete pipe (similar to storm-sewer piping) which may or may not be perforated. For older wells the casing may be simple, stacked clay bricks. These wells provide a large perimeter area that allows water to slowly seep into them from the glacial deposits over the entire depth of the saturated thickness penetrated. The large size also provides a storage area as well. Such wells typically provide enough water for a rural household and farmstead, including small livestock operations. Often, these wells are open to the very top of the water table. Some are grouted to shallow depths to try to exclude seepage of the most shallow groundwater. Some newer seepage wells, or renovated wells, may use "buried-slab" construction, which allows use of a submersible pump and a better protected well head.

Such wells are often set in locations to enhance seepage, such as in the alluvium along small streams, in upland drainage ways or depressions. While such wells are highly vulnerable to contamination from land-surface activities, most were installed years ago when such contamination was not a consideration. They continue to be installed today as a practical matter because, as noted, there may be few economic options for the rural resident in these areas.

The Eastern (E, Region 2) and North-Central (NC, Region 6) regions exhibit similar distributions of well depths (e.g., about 13% <50 ft deep, median depth 130 ft), but their settings are distinctly different. The E region is mantled by much older glacial deposits and has greater over-all relief; the NC area is typically of lower relief, with limited development of a stream network, because it is blanketed by the youngest glacial deposits in the state. Both areas typically have dependable bedrock or Pleistocene aquifers at moderate depths, but because of the greater relief and greater age of the surficial glacial deposits in the E region, groundwater circulation occurs to a greater depth and is less restricted than in NC Iowa. This likely affects the depth distribution and regional variability in contaminants from land-surface activities.

In summary, on a state-wide basis, about 75% of all the wells in the SC, SW, and NW regions are <100 feet deep; these areas contain nearly 65% of all the wells <100 feet deep, state-wide. In contrast, over 75% of all the wells ≥200 feet deep are located in the NE, E, and NC regions of Iowa. These general observations provide a basis for evaluation and discussion of the regional and depth variations in water quality.

Table 21. Summary of nitrate-N concentration data, state-wide, by well depth, and by hydrogeologic regions.

	nitrate-N						
	% > 10 mg/L MCL	95% confidence interval	mean conc. mg/L	s.d. mg/L	median conc. mg/L	range conc. mg/L	Proportion of wells in class %
State-wide:	18.3%	(21.2-15.4)	6.2	12.2	0.55	<0.1-100	100.0%
wells < 50 ft;	35.1%	(42.8-27.4)	11.2	15.7	6.0	<0.1-95	27.9%
wells ≥ 50 ft;	12.8%	(16.1-9.6)	4.3	10.7	<0.1	<0.1-100	72.1%
Hydrogeologic Regions:							
northeastern (1)	9.2%	(15.4-3.0)	3.5	5.5	1.3	<0.1-37	13.0%
eastern (2)	11.6%	(15.9-7.3)	2.6	5.3	<0.1	<0.1-50	32.3%
south-central (3)	28.1%	(35.9-20.2)	9.5	14.0	2.1	<0.1-87	17.5%
southwestern (4)	31.4%	(40.9-22.0)	10.2	19.4	5.2	<0.1-100	14.2%
northwestern (5)	38.2%	(73.2-46.8)	14.1	19.7	5.2	<0.1-95	7.9%
north-central (6)	5.6%	(9.9-1.2)	2.4	9.0	<0.1	<0.1-79	15.1%

NITROGEN COMPOUNDS

Nitrogen is ubiquitous in the natural environment, and its movement and transformation is part of the natural functioning of any ecosystem. Nitrogen is one of the most important plant nutrients and a fundamental component of organic compounds (e.g., proteins, amino acids). The SWRL water samples were analyzed for nitrate-nitrogen (NO_3-N), ammonium-N (NH_4-N), and organic-N (O-N). These compounds form the principle end-members of the N-cycle in the soil-water system. Nitrogen is recycled through time in the soil environment by various biologically mediated processes: mineralization, including nitrification - producing NO_3-N, and ammonification - transforming other N forms to NH_4 (and to ammonia, NH_3, a gaseous form that is lost into the atmosphere); immobilization - the uptake, or conversion of N into biomass, either soil microbial forms or uptake into plants and crops, which produces O-N; and denitrification - and other processes that produce N gasses that may be lost into the air.

NH_4-N is generally a highly adsorbed ion, binding tightly to the soil and, hence, it is not common, in significant concentrations, in groundwater. It is more common in runoff and surface water, being transported along with sediment in overland flow, with some going into solution in the water phase.

O-N is the nitrogen contained in most naturally occurring complex organic compounds (i.e., in combination with carbon), such as the humus or organic matter in the soil. O-N is removed from the soil-system within harvested crops and through erosion. The plant residues and root biomass that are left to decay in the soil and soil microbes that die, for example, form O-N in the soil. This can be viewed as a storage component for nitrogen. The O-N in the soil is relatively immobile, and is waiting to be converted, by the processes noted above, into other forms. Because natural O-N is relatively immobile, it is uncommon in groundwater, except at low concentrations. It is more common in surface water, occurring as particulate matter (suspended matter) more than as dissolved O-N. Organic wastes, some

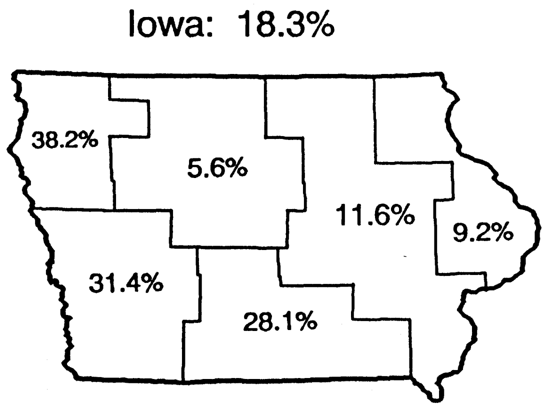


Figure 25. Proportion of private rural wells in Iowa that exceed 10 mg/L, NO_3-N, the health advisory level for nitrate.

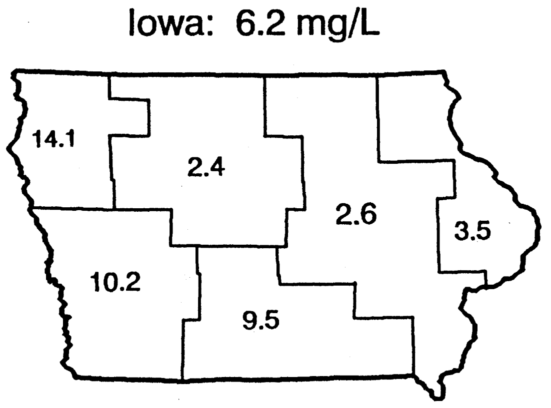


Figure 26. Summary of mean NO_3-N concentrations by hydrogeologic region.

fertilizer compounds, and many synthetic organic chemicals (SOCs) also contain organic-nitrogen. Examples include urea and many pesticides commonly used in Iowa. Hence, O-N is also a possible indicator of contaminants, as well.

Nitrate is the N-compound of greatest significance in groundwater supplies and it is the most common of all groundwater contaminants derived from society's activities. NO_3-N is formed by oxidation processes in the soil and it is highly mobile. Hence, it readily moves with water through the soil to groundwater. It is also the N-compound of greatest concern from a public health standpoint because it can be acutely toxic to infants and may be related to other chronic problems. Ammonium and ammonia, as well as nitrogen gasses, can be toxic to fish and other aquatic life in surface waters.

Nitrate Results

Low concentrations of nitrate occur naturally in some uncontaminated groundwater. Concentrations over 3 mg/L nitrate-nitrogen are typically considered indicative of anthropogenic pollution (Madison and Brunett, 1985; Hallberg, 1989a). The natural background concentration in groundwater aquifers in Iowa is less than 2 mg/L, often less than 1 mg/L nitrate-nitrogen (NO_3-N) (Libra, et al., 1984; Hallberg, 1987, 1989a). Higher concentrations in Iowa indicate a degree of pollution related to agricultural practices, fertilizer use, manure, septic tank wastes, sewage sludge, or other sources.

Nitrate concentrations are reported as either milligrams per liter of nitrate-nitrogen (mg/L of NO_3-N), or as mg/L of nitrate (NO_3). All results in this report are expressed as nitrate-nitrogen (NO_3-N). Reporting results in the nitrogen form facilitate comparison with other forms of nitrogen (e.g., ammonium-nitrogen). The drinking water standard set by the USEPA is 10 mg/L, as NO_3-N. This is equivalent to 45 mg/L, when expressed as NO_3. Nitrate concentrations reported in the SWRL study, and most other studies, are actually the total of nitrate and nitrite (NO_2) nitrogen, but the predominant species in groundwater is nitrate.

As summarized in Table 21 and Figure 25, the SWRL results clearly indicate widespread contamination of groundwater with nitrate. Approximately 18.3% of Iowa's private, rural drinking water wells contain nitrate at concentrations exceeding the recommended health advisory level of 10 mg/L NO_3-N. Statistically, the true proportion of wells contaminated above 10 mg/L lies between approximately 15% and 21% (with 95% confidence). For wells <50 feet deep, 35% are >10 mg/L, and even the mean concentration for these wells is >10 mg/L. State-wide the mean concentration for all wells exceeds half the drinking water standard.

There are statistically significant regional variations in nitrate contamination (chi-square statistic, p value <0.001). Within southern and

Table 22. Percentage of wells in the state and the hydrogeologic regions with water analyses in different ranges of nitrate concentration.

nitrate-N concentration:	< 0.1 mg/L	0.1-3.0 mg/L	3.1-10.0 mg/L	10.1-20.0 mg/L	> 20.1 mg/L	% > 3.0 mg/L	% > 10.0 mg/L
	- - - - - - - percentage in range				- - - - - -		
State-wide:	42.2%	20.4%	19.1%	10.1%	8.2%	37.4%	18.3%
wells < 50 ft;	11.7%	21.3%	31.9%	19.5%	15.6%	67.0%	35.1%
wells ≥ 50 ft;	54.1%	20.0%	13.0%	7.9%	4.9%	25.8%	12.8%
Hydrogeologic Regions:							
northeastern (1)	45.1%	20.9%	24.9%	8.3%	0.9%	34.0%	9.2%
eastern (2)	58.2%	15.0%	15.0%	7.0%	4.6%	26.6%	11.6%
south-central (3)	20.0%	33.3%	18.3%	14.1%	14.0%	46.4%	28.1%
southwestern (4)	21.4%	21.4%	25.5%	17.2%	14.2%	56.9%	31.4%
northwestern (5)	20.0%	12.7%	29.1%	18.2%	20.0%	67.3%	38.2%
north-central (6)	68.6%	17.1%	8.6%	2.8%	2.8%	14.2%	5.6%

western Iowa approximately 30% of all wells exceed the health advisory level of >10 mg/L, reaching a maximum of 38% in northwestern Iowa. In these three hydrogeologic regions the mean NO_3-N concentration is about 10 mg/L or greater (Fig. 26).

The mean concentrations for nitrate, presented in Table 21, are calculated using all samples; i.e., those samples with results that were less than the detection limits were set equal to zero. This produces a considerably lower value than a mean calculated from the quantifiable detections. It also results in a very large standard deviation, with values greater than the mean.

Tables 22 and 23 provide further detail, showing the frequency distribution of nitrate concentrations from the SWRL wells, summarized by region and by well depth. The distribution is subdivided at the 3.0 mg/L concentration, the more conservative indicator of contamination from society's activities, noted above. It is noteworthy that 40% of private rural wells exhibited no detectable nitrate (i.e., <0.1 mg/L). Yet, nitrate contamination is a widespread problem; in every region but the north-central more than one-third of the wells exhibit >3 mg/L NO_3-N, and as noted, about one third are >10 mg/L, in southern and western Iowa. As expected the degree of contamination is far greater in shallow wells; shallow groundwater is obviously the first affected by contamination related to land-surface activities, except where accidents, or poor construction or maintenance may allow the contamination of a deep well. For shallow wells, <50 feet deep, 88% have detectable NO_3 and 67% are >3 mg/L, state-wide. In the three southern and western regions over 50% of the shallow wells are >3 mg/L, ranging to 97% of the wells in northwestern Iowa. In these regions from 30-50% of the shallow wells have >10 mg/L (Fig. 27).

In all regions, except the northeast (NE), the percentage of wells <50 feet deep with >3 mg/L NO_3-N, or >10 mg/L, is much greater than for those wells >50 feet deep. In the NE region the proportion is slightly greater for the deeper wells (>50 feet deep). This is partly because there are so few shallow wells in this region; so few, that a proportion for wells <50 feet deep with >10 mg/L cannot be estimated with confidence (Table 23). This also reflects a greater depth of NO_3-N contamination in the NE, because of the

Table 23. Percentage of wells with water analyses in different ranges of nitrate concentration, with well depths by hydrogeologic region. nd = indeterminate because of small sample size.

nitrate-N concentration:	< 0.1 mg/L	0.1-3.0 mg/L	3.1-10.0 mg/L	10.1-20.0 mg/L	> 20.1 mg/L	% > 3.0 mg/L	% > 10.0 mg/L
	- - - - - - -		percentage in range		- - - - - -		
State-wide:	42.2%	20.4%	19.1%	10.1%	8.2%	37.4%	18.3%
wells < 50 ft;	11.7%	21.3%	31.9%	19.5%	15.6%	67.0%	35.1%
wells ≥ 50 ft;	54.1%	20.0%	13.0%	7.9%	4.9%	25.8%	12.8%
Hydrogeologic Regions:							
northeastern (1)	45.1%	20.9%	24.9%	8.3%	0.9%	34.0%	9.2%
wells < 50 ft:	nd	84.3%	15.7%	nd	nd	15.7%	nd
wells ≥ 50 ft:	43.8%	20.5%	26.7%	7.7%	1.3%	35.7%	9.0%
eastern (2)	58.2%	15.0%	15.0%	7.0%	4.6%	26.6%	11.6%
wells < 50 ft:	6.0%	25.3%	27.2%	33.4%	8.1%	68.7%	41.5%
wells ≥ 50 ft:	66.2%	13.5%	11.3%	4.8%	4.3%	20.4%	9.1%
south-central (3)	20.0%	33.3%	18.3%	14.1%	14.0%	46.4%	28.1%
wells < 50 ft:	13.2%	33.6%	23.1%	17.2%	13.0%	53.2%	30.1%
wells ≥ 50 ft:	34.1%	39.7%	9.0%	8.2%	9.1%	26.2%	17.3%
southwestern (4)	21.4%	21.4%	25.5%	17.2%	14.2%	56.9%	31.4%
wells < 50 ft:	14.0%	21.6%	34.3%	18.5%	11.5%	64.4%	30.0%
wells ≥ 50 ft:	28.1%	25.3%	16.4%	18.5%	11.7%	46.6%	30.1%
northwestern (5)	20.0%	12.7%	29.1%	18.2%	20.0%	67.3%	38.2%
wells < 50 ft:	nd	3.3%	45.1%	15.0%	36.6%	96.8%	51.6%
wells ≥ 50 ft:	45.3%	20.7%	8.4%	21.7%	3.9%	34.0%	25.7%
north-central (6)	68.6%	17.1%	8.6%	2.8%	2.8%	14.2%	5.6%
wells < 50 ft:	48.9%	7.4%	36.3%	nd	7.4%	43.7%	7.4%
wells ≥ 50 ft:	69.7%	18.0%	6.4%	3.4%	2.5%	12.3%	5.9%

greater depth of groundwater circulation in this hydrogeologic setting.

Environmental Health Implications of Nitrate Results

Results of the SWRL study reinforce other Iowa data showing that nitrate contamination of groundwater is a significant environmental problem, of potential significance for public health state-wide, and more pronounced within specific regions. Earlier regional studies within Iowa have identified many areas with high nitrate concentrations, particularly in shallow wells (Hallberg, 1987).

The SWRL study was designed to allow a population exposure estimate to be made. Based on 1980 Census data, about 130,000 rural Iowa residents (or about 17.9% of the rural population) are consuming drinking water with unacceptably high concentrations of nitrate (>10 mg/L, NO_3-N). The population percentage is slightly less than the percentage of contaminated wells cited earlier, because some of the participants used rural water district water supplies, and not their wells for their primary drinking water.

The drinking water standard for nitrate was set primarily to prevent infant cyanosis, or methemoglobinemia (blue-baby syndrome), a temporary blood disorder that reduces the ability of an infant's bloodstream to carry oxygen through the body (Johnson and Kross, 1990). Acute and even fatal incidents of methemoglobinemia have been reported in Iowa (Comly, 1947) and South Dakota (Johnson,

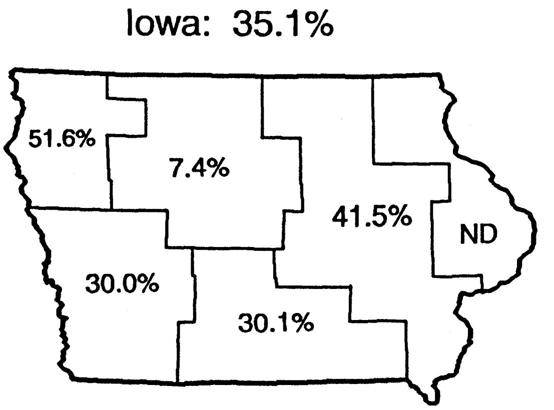


Figure 27. Percentage of wells <50 feet deep with >10 mg/L NO_3-N in the hydrogeologic regions. ND = indeterminate; the NE region has too few wells to make a reliable estimate.

1987). Water containing more than 10 mg/L of NO_3-N (45 mg/L, NO_3) should not be used in preparing infant formula or otherwise consumed by infants, particularly those less than 6 months old.

There are also growing concerns related to long-term and chronic exposure to nitrate alone and in combination with other contaminants. There are concerns that even subclinical methemoglobinemia, which still removes oxygen from the system of a developing infant, could have subtle, long-term developmental or neurological effects (Health and Environment Digest, January 1988). Some evidence exists from epidemiological studies that high nitrate ingestion is involved in the etiology of human cancer (Fraser et al., 1980). High nitrate levels in groundwater have been associated with increased rates of non-Hodgkins lymphoma (cancer of lymphoid tissues) in a Nebraska study (Weisenburger, 1985). An Australian study associated high nitrate in drinking water with increased birth defects (Dorsch et al., 1984 and Dreosti et al., 1984). A Canadian case control study (Arbuckle et al., 1988), comparing baseline nitrate levels of less than 0.1 mg/L NO_3-N to exposures of nitrate of about 6 mg/L, from private well water sources, suggested a moderate increase in risk for central nervous system birth defects (risk odds ratio = 2.30, but not significant statistically). While these and other studies have been suggestive, results have been equivocal and inconsistent among different studies.

From a chronic exposure perspective, nitrate is known to be reduced to nitrite by enzymes and bacteria in the adult human mouth and digestive systems (Forman et al., 1985). Nitrite in turn can combine with other compounds to form N-nitrosamines, known to be potent cancer-causing agent in animals. Nitrates may also interact with pesticides. Some pesticides, including atrazine, aldicarb, and carbofuran, contain secondary amine structures that may react with nitrite at low pH to form N-nitroso compounds (Health and Environment, 1988).

Such studies are complicated, and a factor that must be considered is that, at least for adults, more of the dietary nitrate intake may come from green vegetables. With children, water intake is proportionately much more important, and often the dominant input.

As indicated above, some evidence suggests that nitrate in drinking water might cause cancer, birth defects, or other developmental problems. As noted, these studies are incomplete and equivocal; the EPA has indicated that there is insufficient information currently available to determine whether or not nitrate causes cancer in humans. The current drinking water standard is based only on the non-cancer health effects related to methemoglobinemia.

Ammonium-N and Organic-N

Ammonium-nitrogen (NH_4-N) and organic-nitrogen (O-N) are the two other primary nitrogen compounds analyzed in the SWRL study. As noted concentrations of these compounds in groundwater are low relative to NO_3-N concentrations. Situations where higher NH_4-N concentrations occur in shallow groundwater can be associated with: seepage from septic systems or manure storage; point-sources, such as fertilizer storage and handling facilities, or other sites of fertilizer spills or leaks; and, for short periods in spring when residual NH_4-N, or fertilizer derived ammonium, may move quickly with preferential flow of water through the soil (Hallberg, 1989a; 1989b). Recent studies in Iowa show that ammonium is a natural constituent (in low concentrations) of many deeper, deoxygenated groundwaters, where it may be derived from degradation of natural organic material contained in the

Table 24. Summary of ammonium-N and organic-N concentration data state-wide, by well depth, and by hydrogeologic regions.

	ammonium-N				
	% detection >0.1 mg/L	mean conc. mg/L	s.d. mg/L	median conc. mg/L	range conc. mg/L
State-wide:	45.2%	0.74	1.3	<0.1	<0.1-11.0
wells < 50 ft:	13.6%	0.18	0.6	<0.1	<0.1-8.3
wells ≥ 50 ft:	57.6%	0.98	1.4	0.3	<0.1-7.1
Hydrogeologic Regions:					
northeastern (1)	33.3%	0.31	0.7	<0.1	<0.1-4.0
eastern (2)	59.5%	0.92	1.2	0.3	<0.1-11.0
south-central (3)	27.2%	0.41	1.0	<0.1	<0.1-6.4
southwestern (4)	24.8%	0.22	0.6	<0.1	<0.1-5.4
northwestern (5)	27.3%	0.41	0.9	<0.1	<0.1-4.2
north-central (6)	77.2%	1.50	1.6	1.0	<0.1-8.3

	organic-N				
	% detection >0.1 mg/L	mean conc. mg/L	s.d. mg/L	median conc. mg/L	range conc. mg/L
State-wide:	43.1%	0.27	0.8	0.1	<0.1-14.0
wells < 50 ft:	49.1%	0.27	0.7	0.1	<0.1-6.0
wells ≥ 50 ft:	41.8%	0.29	0.8	0.1	<0.1-14.0
Hydrogeologic Regions:					
northeastern (1)	25.7%	0.13	0.3	<0.1	<0.1-2.6
eastern (2)	38.5%	0.23	0.6	<0.1	<0.1-8.0
south-central (3)	52.0%	0.50	1.3	0.2	<0.1-14.0
southwestern (4)	45.9%	0.15	0.2	0.1	<0.1-1.3
northwestern (5)	60.0%	0.28	0.4	0.2	<0.1-2.6
north-central (6)	43.2%	0.25	0.6	0.1	<0.1-6.0

Table 25. Summary of historical water-quality data from non-statistical samples voluntarily submitted to the University Hygienic Laboratory, 1980-1989.

Year	Nitrate-N % >10 mg/L all data	 % >10 mg/L wells	Total coliform bacteria % unsafe all data	 % unsafe wells
1980	20%	20%	36%	38%
1981	18%	19%	35%	36%
1982	21%	23%	38%	40%
1983	23%	24%	34%	36%
1984	21%	23%	35%	37%
1985	19%	20%	32%	33%
1986	18%	19%	39%	41%
1987	18%	20%	38%	39%
1988	16%	17%	33%	34%
1989	15%	15%	33%	33%
Average	18%	19%	35%	36%

aquifer or reduction of nitrate in the groundwater.

NH_4-N is toxic to many fish in concentrations above about 1.0 mg/L (depending on water temperature). At the concentrations detected in the SWRL sampling (Table 24), NH_4-N is not thought to present a direct threat to human health.

As noted, O-N is not highly mobile and typically is not detected in well-water samples, or only occurs in low concentrations, except from very shallow groundwater. The presence of O-N in drinking water does not pose a direct health concern, but rather it may be an indicator of other potential well-water contamination. Detection of O-N in well water was evaluated as a potential indication of other contamination by pesticides, other SOCs, or their metabolites, because such compounds contain O-N. No clear relationships were discerned from the data to date. O-N above 2-3 mg/L may raise a suspicion, similar to coliform bacteria, that if degraded organic compounds can enter the well-water supply, so may other contaminants.

As summarized in Table 24, the concentrations for both NH_4-N and O-N are quite low, with 55 to 57% of the samples below the detection limits of 0.1 mg/L. There was no systematic or significant variation in O-N. Unlike NO_3-N, mean and median NH_4-N concentrations, and the percentage of wells with detections of NH_4-N are greater in the wells >50 feet deep. As noted above, this suggests that the ammonium is primarily a natural constituent and its concentrations are typically quite low. There is a tendency among the regional mean concentrations for NH_4-N, for an inverse relation with NO_3. The North-Central region has the lowest mean NO_3-N but the greatest NH_4-N; being 2-3 times higher than many of the regions. These inter-relationships will be discussed further in later sections of the report.

Discussion of Nitrogen Data

The SWRL data substantiate many earlier studies in Iowa that identified the occurrence of high nitrate concentrations in groundwater, particularly from shallow wells or areas where aquifers were shallow (e.g., Hallberg and Hoyer, 1982; McDonald and Splinter, 1982; Hallberg et al., 1983; Libra et al., 1984; Bruner and Hallberg, 1987; Hallberg, 1987). These prior studies noted that elevated nitrate concentrations were regionally ubiquitous in shallow groundwater, and particularly significant in wells < 50 ft deep. Data summaries by county (Hallberg, 1986; 1987) showed that the most pronounced problems occurred in southern and western Iowa, as also revealed in the SWRL data. While these studies were not statistically systematic in nature, they often involved much larger numbers of samples than SWRL. Future efforts will compare these data with SWRL to evaluate the potential for more refined, local assessments.

The UHL water laboratory analyzes about 10-12,000 samples/year for nitrate; the major portion of these are from private drinking water wells from throughout Iowa. Throughout the 1980s, these results have consistently shown that >15% of all samples, and 15-24% of well samples exceed the recommended HAL for nitrate (Table 25). These values bracket the SWRL findings that showed 18.3% of private wells exceed 10 mg/L NO_3-N. The aggregate UHL data show a distinct decline in the percentages of well samples >10 mg/L, NO_3-N during the 1988-89 drought period.

Well Depth and Nitrate

The SWRL data exhibit the same general relationship of declining nitrate concentrations with increasing well depth as prior studies (Fig. 28). As noted above, there is a statistically significant difference in nitrate-N between wells <50 ft deep, and those ≥50 ft deep. As shown on Figure 28, nitrate-N >3 mg/L occurs in wells considerably deeper than 50 feet. This depth distribution relationship varies regionally; these regional differences are graphically apparent in Figure 29.

Past studies by Hallberg and others in northeastern Iowa (Hallberg and Hoyer, 1982; Hallberg et al., 1983) and the Skunk River basin (Bruner and Hallberg, 1987) dealing with large data sets have provided various statistically significant insights to the regional and depth variations of nitrate in groundwater. In general, these studies have shown that: 1. there are significantly ($p < 0.01$) greater nitrate concentrations in wells <50 ft deep, compared to deeper wells, regardless of geologic setting; 2. where high landscape relief and/or karst development allows relatively deep groundwater circulation in bedrock aquifers, similarly high nitrate concentrations may extend to greater depths (approx. 100 ft); 3. in shallow bedrock aquifer areas lesser, yet significant, nitrate concentrations continue to depths of 150 ft, and in high relief or karst areas, to greater depths (see Table 26). In short, these prior studies showed the most pronounced effects of nitrate contamination, at the top of the groundwater system (wells <50 ft deep), were apparent in all settings, and in particular settings to greater depth. Somewhat muted effects are still apparent to depths of 150 ft, or 200 ft, related to the hydrogeologic conditions. These studies were the basis for the early summary of the SWRL data from wells less than and greater than 50 feet in depth. Recent work is showing, as discussed below, that the more significant break may now occur at approximately 100 feet in depth.

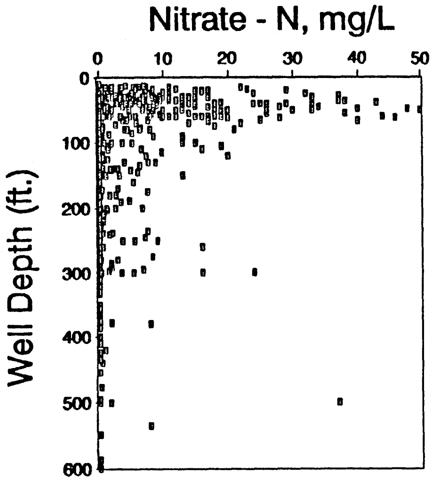


Figure 28. Plot of nitrate-N concentrations by well depth for state-wide data ; note few data points not shown because they were so far off-scale.

The SWRL data (Fig. 29) follow these patterns, related to hydrogeologic settings. The decline in nitrate contamination with depth is pronounced in the SC, SW, NW, and NC regions; it is apparent, but not as clearly defined in the E region; and no clear depth break occurs in the NE (until perhaps 300 feet). Vermace (1990) examined these records using various statistical and graphical methods. He found that nitrate concentrations were statistically greater in shallow wells, to depths of 110 to 130 feet in the

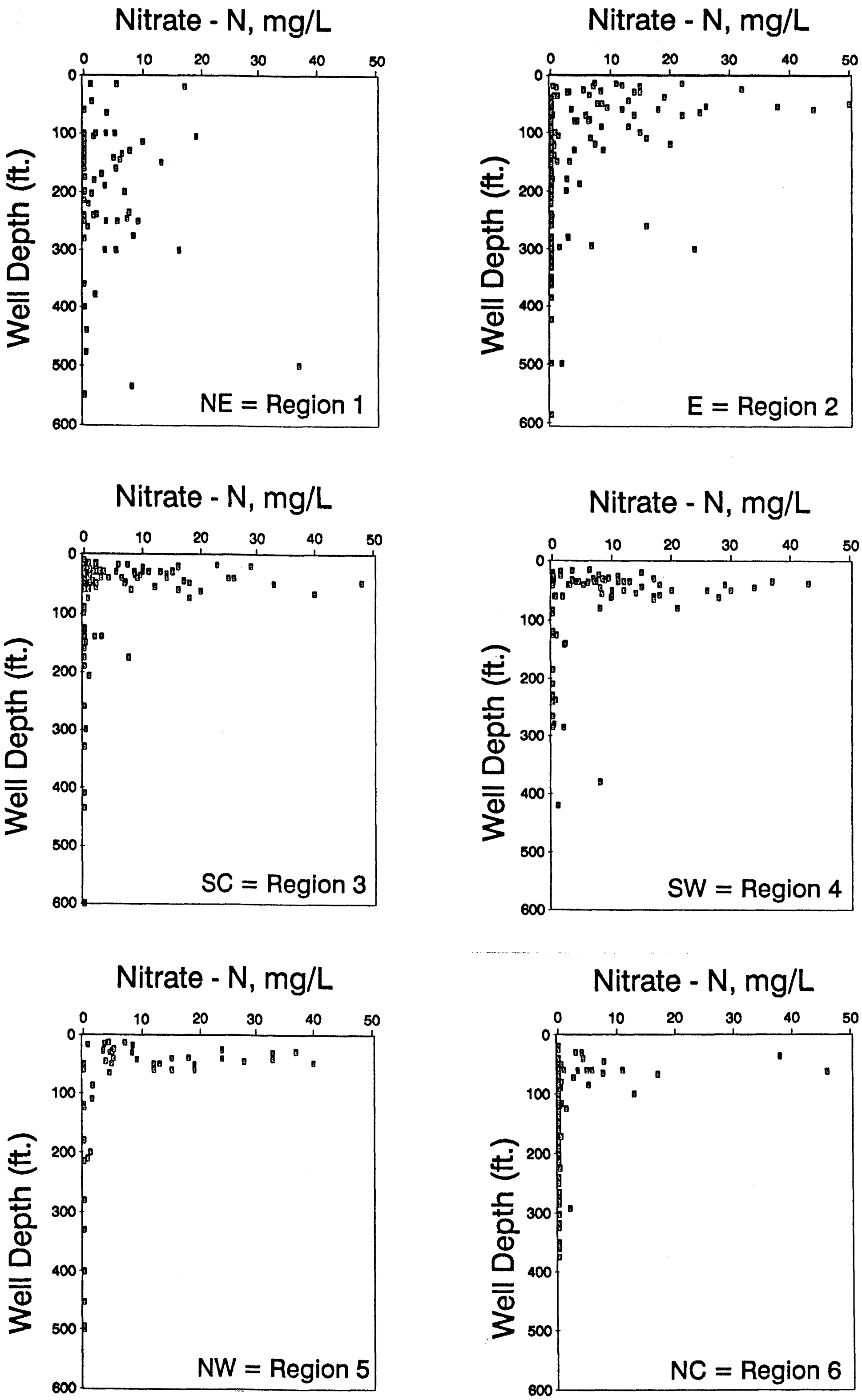


Figure 29. Relationship of nitrate concentration and well depth for each hydrogeologic region; note few data points not shown in some regions because they were so far off-scale.

Table 26. Mean nitrate-N concentration data, summarized by well depth and geologic region. From studies of Hallberg and Hoyer, 1982, for 22 counties in northeastern Iowa (including portions of the NE and E regions of this report); Bruner and Hallberg, 1987, for 21 counties in the Skunk River basin (including portions of the E, SC, and NC regions of this report). NE Iowa study includes approximately 6,000 analyses; Skunk River basin, 9,600 analyses. See discussion of Hydrogeologic Regions for definition of geologic regions. Medians followed by different letters indicate statistically significant differences (by column or row), at $p < 0.01$.

	- - - Northeastern Iowa - - -			- - Skunk R. basin - -	
Well depth	Deep bedrock aquifer	Shallow bedrock aquifer	Karst bedrock aquifer	Deep bedrock area	Shallow bedrock area
feet	- - - - - - -		median nitrate-N, mg/L		- - - - - -
< 50	7.3 a	5.8 a	6.2 a	2.1 a	3.2 b
51 - 99	1.3 b	4.2 a	7.6 c	<1.0 c	2.8 b
100 - 149	<1.0 d	3.6 e	5.1 f	<1.0 c	1.6 d
150 - 199	<1.0 d	1.3 g	2.2 g	<1.0 c	<1.0 c
200 - 249	<1.0 d	<1.0 d	<1.0 d	<1.0 c	<1.0 c
250 - 299	<1.0 d	1.0 d	<1.0 d	<1.0 c	<1.0 c
≥ 300	<1.0 d	<1.0 d	<1.0 d	<1.0 c	<1.0 c

E, SC, SW, NW, and NC regions. No significant changes could be ascertained in the NE region. The well-depth population with the greatest concentrations extend greater than 50 ft, statistically to depths of 70 to 120 feet; and as evident on Figure 29, generally declining affects are apparent from depths of 100 to 130 feet. Indeed for the state-wide SWRL data, the mean NO_3-N for wells <50 feet deep is 11.2 mg/L; for wells from 50 to 99 feet deep, 11.0 mg/L; and 1.6 mg/L for wells ≥ 100 feet deep. The respective median values are 6.0, 4.3, and <0.1 mg/L; the respective proportions of wells >10 mg/L NO_3-N is 35%, 32%, and 4%.

Because of the deep groundwater circulation that occurs in the Northeastern hydrogeologic region, there is not a pronounced depth relationship, and high nitrate concentrations extend to great depth. Similar settings provide greater depth penetration of nitrate in the Eastern region, as well.

Further relationships among depth, NO_3, NH_4, and other analytes will be discussed in subsequent sections of the report.

COLIFORM BACTERIA

Monitoring for the presence or absence of *total* coliform bacteria in drinking water has long been used as an indicator of the general bacteriological quality of public drinking water supplies. Total coliforms are common in the environment but are generally not pathogenic themselves. However, it is desirable that drinking water should be free from coliform bacteria, because their presence suggests that disease-causing bacteria or other pathogenic organisms may be present, or may be able to enter the drinking-water supply. Water analyses for total coliform have been used because total coliform are common and test procedures are relatively rapid and inexpensive, hence providing an affordable screening tool.

Some misconceptions have hampered adequate interpretation of the total coliform test with private water supplies. The presence of total coliform bacteria in private water-supply samples has often been assumed to indicate fecal contamination from human or animal wastes. Typically, faulty well construction is blamed for the presence of coliforms and, in turn, for other contaminants that may be present, particularly nitrates that could also be derived from fecal wastes. These issues require clarification.

Historically, total coliform have often been equated with fecal coliform, and even more specifically with *Escherichia coli*. These are not equivalent, however. In practice, the definition of *total* coliform bacteria is related to the test procedures; the total coliform group comprises all aerobic and facultative anaerobic, gram-negative, nonspore-forming, rod-shaped bacteria that ferment lactose with gas formation within 48 hours at 35 degrees C. (APHA, 1985).

The test defines a group of common bacteria, not a particular pathogen. The total coliform group is comprised of organisms from a fecal origin (called fecal coliforms) and those derived from non-fecal sources (non-fecal coliforms). Fecal coliform bacteria are present in human and animal wastes. Other coliforms are common constituents of soils and shallow groundwater, and they are ubiquitous in surface water. Because the total coliform group is comprised of organisms from a fecal origin and those derived from non-fecal sources, the mere presence of total coliforms in drinking water does not mean a water supply is contaminated with fecal material (Gainey and Lord, 1952; McKinney, 1962; Gaudy and Gaudy, 1980).

Historically, the soil and groundwater system were thought to be relatively sterile at depths below the root zone. The soil can act as a natural filter; microorganisms often become trapped as water percolates through unsaturated soil, and many microbes related to warm-blooded mammals are not thought to live for long in the subsoil or groundwater environment. The subsoil is not, however, a sterile environment; many important processes that take place at substantial depth (e.g., iron and sulfur reduction, natural gas and petroleum formation) are mediated or brought about by microbial activity (e.g., Beck, 1989). Recent work has shown that a variety of bacteria and viruses exist in the subsurface environment and have been extracted from groundwater samples from depths of 1,000 feet or more (e.g., Stetzenbach, et al., 1986; Ghiorse and Wobber, 1989). With preferential flow through the soil (water flow through macropores) even truly enteric organisms, such as fecal coliform or poliovirus, have been found to move rapidly through the soil to the water table and to be more mobile than previously thought (Smith et al., 1985; Stewart and Reneau, 1982).

Sources of Total Coliform

Sources of fecal coliforms are human and animal wastes, such as household sewage and animal manures. Non-fecal coliforms, as noted, are common in soils and surface water. Groundwater investigations in Iowa have found that total coliform bacteria are also common in groundwater at, and below the water table for some depth. This depth is not well defined and likely varies among areas as a function of the hydrogeologic setting. DNR-GSB research *water-table* piezometers and monitoring wells have shown high MPN ("most probable number") values for total coliform bacteria, even under controlled conditions in natural background settings, far removed from sources of contamination. Chlorination has been used to decontaminate the wells, but this has been effective only for a short period; the total coliforms always return within a few weeks after chlorination (e.g., Thompson, 1984). Further analysis has verified that these were not fecal coliforms.

Interpreting the origin of total coliform bacteria in samples from domestic water supplies may be difficult. They may enter the water system at many points. Clearly, a significant and persistent presence of total coliforms in water derived directly from a well indicates that the well is producing water that has recently interacted with the soil or soil water, shallow groundwater, or possibly surface water. In water from relatively deep wells, that are cased and grouted to some depth below the surface (typically 40 ft or more), the presence of total coliforms may indicate surface contamination (non-fecal or fecal) from shallow groundwater or even surface runoff that has entered at the well head, or through defects in the top 10-20 feet of a well, for example. Interpreting the origin of coliforms in shallow wells -- wells designed to produce water from at or just below the water table, is problematic. While well construction or maintenance problems may be involved, the source of the coliforms may simply be the shallow groundwater environment itself, as shown by recent research.

Another point of entry for total coliforms can be from the water-distribution system. Water samples are generally collected from a household water tap, or an outside hydrant, not directly from the well, and the water has passed through part of the water-distribution system. Total coliform bacteria can enter through the water distribution system, particularly when cisterns or outside storage tanks are used. Past studies have indicated that over 90% of water samples collected from taps "downstream" of such outside storage structures will exhibit total coliform ≥ 9.2 MPN (e.g., Hallberg et. al., 1983b), even though no coliforms were detected from the well. These site investigations have also shown that defects in other parts of the water system (e.g., buried water lines) may also allow total coliforms into the system. In contrast, these studies have shown no significant difference in nitrate concentrations between the well-head and tap samples.

Accidental contamination of the sterile sample container, improper sampling technique, or handling of the tap may also result in a positive total coliform result but these findings would not represent a contaminated well or water supply. Correct sampling procedures were used for SWRL samples to avoid accidental contamination (Hallberg et. al., 1990).

These complexities illustrate that it is not possible to generalize interpretations about well construction nor fecal contamination from single total coliform analyses, such as conducted during SWRL. Total coliforms can appear in a water sample because the water system uses a shallow well and this does not of necessity indicate that the well is likely to be contaminated with pathogens. The presence of fecal coliforms indicates a more acute potential for pathogenic contamination. The UHL implemented a comparative study of methods for fecal coliform analysis during the SWRL study, that were used during the later portions of the survey. Results of these tests will also be described below.

Total Coliform Bacteria Results

Methods of analysis are summarized on Tables 1 and 2. Results of the bacterial analyses give the statistically estimated number of total coliform organisms in 100 milliliters of water, rather than a direct count. The results are reported as standard MPN values ("most probable number"). There are 6 MPN numbers reported: 0, 2.2, 5.1, 9.2, 16, and 16+. The "0" level, indicating no total coliform bacteria were found, is reported as safe, while all other positive detections of total coliform (2.2-16+) are reported as "unsafe for human consumption," because their presence may suggest that disease-causing bacteria or viruses may be able to enter the drinking-water supply.

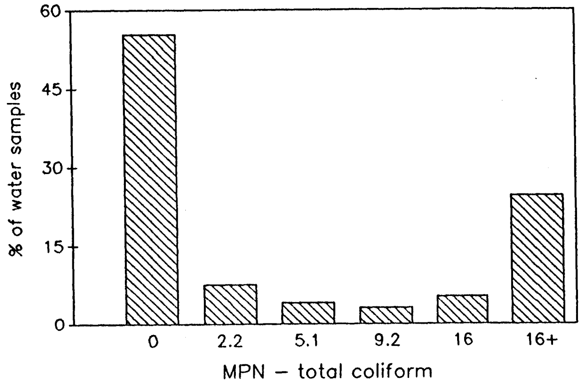


Figure 30. Histogram showing bimodal distribution of total coliform MPN results for all samples, state-wide.

The distribution of total coliform bacteria results from large populations of water samples, such as collected for SWRL, are typically bimodal. As illustrated in Figure 30, for the SWRL results, most samples are either free of total coliform bacteria (0 MPN) or the degree of contamination is great (16+ MPN). The state-wide SWRL water-system samples show that about 55%=0 MPN and 25%=16+ MPN; these 16+ values comprise 56% of the positive total coliform samples. Further comparative analysis primarily utilizes the total coliform results as positive (MPN >0) or negative (MPN=0).

The SWRL results show that 44.6% (95 % C.I. = 40.9 - 48.3%) of the private water supplies (from the wells and/or their distribution systems) in rural Iowa contained total coliform bacteria (values of 2.2 through 16+) and were reported as unsafe. The proportion of the rural, private water systems exhibiting total coliform bacteria vary significantly (p value <0.001) among the hydrogeologic regions of the state; from about 21% of wells in NE Iowa to 73% in SC Iowa (Fig. 31 and Table 27).

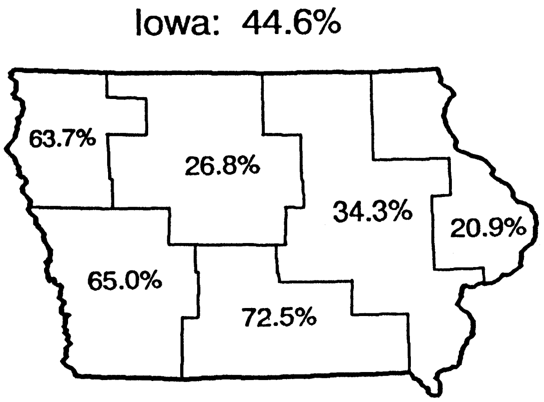


Figure 31. Map summarizing the percentage of water systems with detections of total coliform bacteria in the hydrogeologic regions.

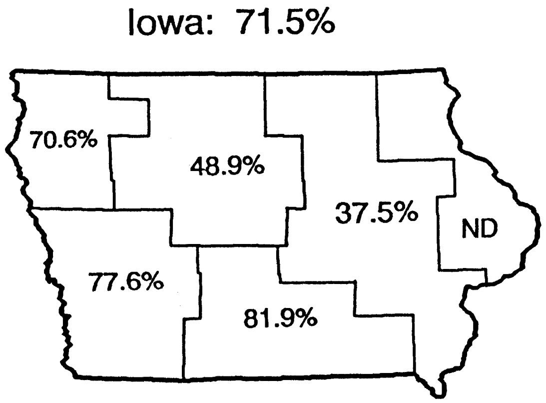


Figure 32. Percentage of water systems using wells <50 ft deep that exhibited total coliform bacteria in the hydrogeologic regions.

These significant regional differences parallel the regional differences in well depths (and typical well construction). State-wide the proportion of water systems using wells less than 50 feet deep with total coliform contamination is about 72%, compared to 36% for systems with wells $\geq$ 50 feet deep (Table 27). Table 27 shows the percentage of wells <50 ft deep, and the median well depth in each region for comparison with the total coliform results. There is a significant correspondence between the regions dominated by shallow, water-table wells and the regional proportion of water systems exhibiting total coliforms; for example, linear regression models between the regional averages for well depth and the percentage of total coliform positives, show an $r^2=0.96$, and a highly significant relationship, with $p < 0.001$. (As discussed in later sections, this relationship is not as significant for site-by-site data.)

Figure 32 shows the proportions of water systems using wells <50 ft deep that exhibited total coliform bacteria. The Southern and Western regions that are dominated by water-table wells show percentages ranging from 71% to 82%. The NE region has so few shallow wells with total coliform detections that a reliable estimate could not be made. Figure 33 shows the percentage of water systems positive for total coliforms, stratified by well depth categories, state-wide. The general decline with increasing well depth is readily apparent. While actual numbers decline, there is a slight increase in percentage of positives for the deepest well categories. The majority of these are from the NE and E regions, where local karst conditions exist and where there is a greater depth of groundwater circulation. Further analysis among other factors such as casing depth, grouting practices, and use of cisterns will be conducted.

Environmental Health Implications of Total Coliform Bacteria

The U.S. EPA is required by law, under the Safe Drinking Water Act, to regulate total coliforms in public water supplies. They have determined that the presence of total coliform in drinking water is a possible health concern. For public water supplies the EPA deems that the drinking water should be free from total coliform bacteria, even though coliform bacteria are not generally considered a health hazard themselves. This is desirable because the presence of total coliforms suggests that disease causing bacteria or other pathogenic organisms may be present, or able to enter the drinking-water supply. Drinking water that does not contain total coliform bacteria is usually not associated with a health risk from disease-causing bacteria (USEPA, Federal Register, 6-29-89, vol. 54, no. 124, p. 27545-27561).

Rural residents whose water systems tested positive for total coliforms were provided with information on possible corrective actions (e.g., chlorination, water system or well restoration) and information on suggested local contacts for

Table 27. Summary of total coliform bacteria results from private rural water systems sampled in SWRL survey.

	Total coliform bacteria results:			For comparison:	
	% "safe" MPN = 0 (no total coliforms)	% "unsafe" MPN >0	95% confidence interval	% of wells < 50 ft deep	median well depth feet
State-wide:	55.4%	44.6%	(40.9-48.3)		110
wells < 50 ft:	28.5%	71.5%	(65.4-78.5)	27.9%	
wells ≥ 50 ft:	63.7%	36.3%	(31.6-40.9)		
Hydrogeologic Regions:					
northeastern (1)	79.1%	20.9%	(12.4-29.4)	5.7%	190
eastern (2)	62.4%	37.6%	(19.4-55.8)	12.5%	130
south-central (3)	27.5%	72.5%	(64.4-80.6)	54.9%	40
southwestern (4)	30.9%	69.1%	(55.6-82.6)	54.2%	40
northwestern (5)	37.0%	63.0%	(50.6-75.4)	52.8%	40
north-central (6)	72.7%	27.3%	(18.3-36.3)	12.9%	130

assistance (e.g., county sanitarians). Resampling was also suggested, and facilitated on request, to check if high values were repeated.

It is difficult to assess the implications of the total coiform results because they are from a single sample from a single point in the water systems. The presence of total coliform bacteria could be related to fecal or natural non-fecal sources that entered the system through a shallow well, or structural defects in a deep well, or the water distribution system. The only clear implication of the total coliform positives is that shallow groundwater or surface water has entered the water system (if the sample has been properly handled).

The high proportion of sites exhibiting total coliform in the SWRL survey is a matter of concern because of their *potential* health implications. However, as noted by EPA (USEPA, Fed. Reg., op. cit.): "given that total coliforms are ubiquitous in water, EPA believes that an infrequent single coliform-positive sample does not necessarily represent a health risk." The evaluation of the *fecal* coliform results, discussed below, provides an assessment of potentially more acute situations for private water supplies in Iowa.

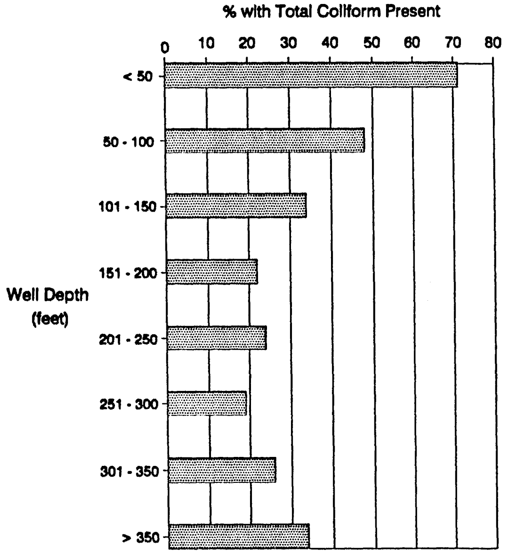


Figure 33. Percentage of water systems with total coliform bacteria by well-depth categories, for state-wide data.

Fecal Coliform Bacteria Analysis

The University Hygienic Laboratory (UHL) has recognized the need to evaluate new fecal coliform testing methods and to make them available more routinely. During the course of the SWRL survey the UHL acquired new coliform substrate for fecal coliform testing. The UHL performed two fecal coliform tests on a portion of the SWRL samples: the Autoanalysis Colilert test and the standard multiple tube fermentation method (MTF). The Colilert test had not been extensively evaluated using private well-water samples likely to contain fecal coliforms. Hall and Moyer (1990) provide a discussion of the comparative data. The MTF method was performed according to standard methods of APHA (1985) and USEPA (1982). The 5-tube MTF, using five 10 ml portions from a 100 ml sample, was performed using lauryl sulfate broth for presumptive, brilliant green lactose bile for confirmed coliforms, and EC broth for the fecal portion of the MTF test.

The results of the fecal coliform tests are also reported as MPN, with similar numbers (0-16+), as discussed in the total coliform section. In total, 465 samples were analyzed, including 256 from the primary data set used for the state-wide statistical estimates in this report. The remaining samples were from quarterly and 10% repeat sites; additional duplicates were analyzed as well. The results reported here are a composite of both methods: any sample that was positive for fecal by either method is counted as positive, and thus these data provide a maximum estimate.

Fecal Coliform Bacteria Results

As discussed in the total coliform section, fecal coliforms are a subset of the total coliform group. All sites that were positive for fecal coliform were also positive for total coliforms. Fecal coliform bacteria were found in only 5.4% of private rural water-supplies state-wide, on a population-weighted basis (Table 28). This is in sharp contrast to the 44.6% that were found to have total coliforms present and emphasizes that total coliform positives cannot be equated with the presence of fecal coliforms.

The subset of samples analyzed for fecal coliforms included sites from every hydrogeologic region. However, the western regions, which had a large percentage of sites with total coliforms, were somewhat under-represented. Table 28 shows the population-weighted estimate for total coliforms from the exact subset that were analyzed for fecal coliforms; the state-wide proportion for the subset shows 37.3% had total coliforms, slightly less than the complete SWRL data set. An estimate of the percentage of water supplies state-wide that are contaminated by fecal coliform can be made by simple proportions with the total coliform data. This suggests that 6.7% of water-supplies state-wide are contaminated by fecal coliform bacteria. The proportion of water systems with wells <50ft deep with fecal coliform contamination was 8.4%; proportioning to the state-wide estimate increases this to 8.8%, compared to the 71.5% for total coliforms. The proportion of water systems with wells >50 ft deep show 4.3% with fecal coliforms; proportioning to state-wide raises this to 5.0%, compared to 36.3% for the total coliforms.

Water systems with fecal coliform contamination were detected in every hydrogeologic region except the North-Central. There are too few detections to make regional estimates of occurrence. Water systems with fecal coliform contamination were detected in only 12 counties.

Environmental Health Implications of Fecal Coliform Bacteria

The U.S. EPA has determined that the presence of fecal coliforms in drinking-water supplies is a serious concern, clearly indicating a more acute hazard than total coliforms. They note (USEPA, Fed. Reg., op. cit.): "that the presence of fecal coliforms or *E. coli* in treated water is cause for grave concern and probably poses an acute risk to human health, because when fecal coliforms or *E. coli* are detected, it is likely that human pathogens are present...," because they are usually associated with sewage or animal wastes from septic tanks, or feedlot runoff, or similar occurrences. Fecal coliforms indicate that the water is being contaminated from fecal wastes that can contain organisms that cause disease and thus, represent a potential acute health hazard. Disease symptoms can include diarrhea, cramps, nausea,

Table 28. Weighted proportions of private rural water systems with fecal coliform bacteria.

Total and FECAL coliform bacteria results:	% "safe" MPN = 0 (no coliforms)	% "unsafe" MPN >0
All SWRL, State-wide, % Total Coliform:	55.4%	44.6%
SWRL subset; samples with fecal analyses, % Total Coliform:	62.7%	37.3%
% FECAL Coliform:	94.6%	5.4%
Proportioned to total state estimate, % FECAL Coliform:	93.3%	6.7%

and possible jaundice, and associated headaches and fatigue. These symptoms are obviously not unique to disease-causing organisms in drinking water, but may also be caused by a number of other factors.

The point of entry into water-supply systems are the same for fecal coliform as discussed for total coliform. Fecal coliform also clearly implicate a sewage source nearby. When fecal coliform contamination is present more immediate steps need to be taken to improve and sanitize the water supply.

Discussion of Coliform Results

The very significant difference between the total coliform results and the fecal coliform results clearly illustrate the problems in equating total coliform data with fecal contamination. State-wide, about 45% of private water supplies were indicated as unsafe by total coliform analysis; however, 85% of these sites did *not* show the presence of *fecal* coliforms, the presumed acute hazard. The total coliform test has been used as a *rapid* and *inexpensive* screening test because of the many technical difficulties in lab and field collection procedures to isolate waterborne pathogens (e.g. giardia, poliovirus).

Well Depth Distribution Related To Coliforms

Table 29 shows the relationship of total and fecal coliform positives to the depth of wells for the SWRL population water systems. The majority of positive samples come from systems using wells <100 ft deep; about 70% of all total coliform positives, and about 80% of all fecal positives. No systems using wells $\geq$200 ft deep were positive for fecal coliforms. For the comparative total coliform data, there is a significantly greater proportion of positive samples from systems using wells <100 ft deep for total coliforms; for systems with wells $\geq$200 ft deep there is a persistent 25-30% of sites with positive detections, while there were no fecal positives. For the shallow well depths at most about 22% of total coliform-positives were fecal positive, and in sum about 15% of total coliform-positives were fecal positive.

Previous studies typically shown a moderate statistical significance to greater proportions of water systems using wells <50 ft deep, and/or <100 ft deep being total-coliform positive than for deeper wells, yet there often is not an overall significant relationship between well depth and total coliforms, because the general level of occurrence is high even for systems using deeper wells (Hallberg and Hoyer, 1982; Hallberg et al., 1983a,b; Libra et al., 1984; Bruner and Hallberg, 1987). The SWRL data also show these attributes.

Total Coliform Variability

For all SWRL analytes, duplicate samples were collected in the field at specified sites and submitted to the labs as routine QA/QC checks. The results from the blind duplicates for total coliforms illustrate additional problems. Seventy-two percent of the total coliform duplicate analyses showed equivalent results; 56% had no total coliforms in either sample, and 16% had coliforms in both samples, and the same MPN value. For the remaining 28%, 7% had total coliforms in both samples but changed MPN classes, but most significantly, 21% changed from 0 MPN, or no total coliforms, in one sample, to a positive detection in another sample. In past field studies, we have also found that about 25% of duplicate samples do not repeat as positive (Hallberg et al., 1983a, b). This is in marked contrast to the repeatability expected of other analyses. For nitrate, for example, there was no significant difference between the duplicate sample results. The mean difference was 0.08 mg/L, with only 1% of samples varying from below to above the 10 mg/L mark in the paired samples, and 1% also varied from below to above detection limits (0.1 mg/L) in these replicates. Such differences in repeatability are not unexpected between biological and chemical analyses, but such variance affects the utility of single sample results.

The persistent presence of total coliforms in water-supplies is sometimes noted as an indication of a faulty water-system allowing contamination. For the SWRL quarterly repeat sites, where the same wells / water-systems were sampled (from the same point) every quarter, the proportion of water-systems exhibiting total coliforms varied from 55.5% to 27.6%; in aggregate, only 50% of sites with total coliform detections exhibit total coliforms on repeat sampling. Variability of coliform within a water system also makes single samples difficult to interpret. As noted, past field surveys have also indicated that of those sites where tap water was positive for total coliforms, at least 40 to 50% of the positives were related to the use of external storage systems, such as cisterns, or water line problems, and not well contamination (Hallberg et al., 1983a, b). Pipes and Christian (1982) describe the large variance common in the distribution of coliform densities in water systems, and EPA cites other studies, as well illustrating significant variance in total coliform values (USEPA, Fed. Reg., 1989, op. cit.).

USEPA Review of Coliform Bacteria Regulations

The EPA has recently reviewed their concerns for the microbial quality of drinking water supplies and is issuing new rules for public water supplies. This is related, in part, to the problems discussed with using total coliforms as indicators for the presence of pathogens (USEPA, Fed. Reg., 1989, op. cit.). As they note, total coliform levels have been used for decades as the primary indicator of the microbial integrity of drinking water. While coliforms are usually not pathogenic themselves, they often are present in water associated with pathogens and outbreaks of disease, particularly in surface-water systems. The Safe Drinking Water Act specifically requires EPA to regulate total coliforms. EPA states that, "conceptually, coliforms should not be present in drinking water, because they may indicate the presence of pathogenic organisms in the water." The EPA believes that treatment which provides total coliform-free water will reduce fecal pathogens to minimal levels. EPA is studying possible variances to the total coliform rule where persistent biofilm coliforms are present because of distribution system problems, but are not associated with fecal or pathogenic contamination or with waterborne disease.

EPA presents a sharp contrast for their concerns regarding the presence of total coliforms and fecal coliforms. They note that the presence of total coliforms is a possible health concern. They further state "that the presence of fecal coliforms or *E. coli* in treated water is cause for grave concern and probably poses an acute risk to human health, because when fecal coliforms or E. coli are detected, it is likely that human pathogens are present." For this reason, EPA requires more urgent public notice and action when *E. coli* or other fecal coliforms are detected, compared to when total coliforms are detected.

To tighten controls on more hazardous situations, the EPA will require that public water systems conduct additional analyses for fecal coliforms. For *any* water-supply sample that is positive for total coliforms, the system must analyze that culture to determine if fecal coliforms (or *E. coli*) are present. "If any repeat sample is fecal or *E. coli* positive, or if a fecal or *E. coli* positive original sample is followed by a total coliform positive repeat sample ... " this is considered an acute violation of the total coliform MCL, requiring immediate public notification (USEPA, Fed. Reg., op. cit.).

For many decades coliform analysis and sanitary surveys have been the foundation of programs to assure safe water supplies. Yet, as EPA notes, despite these existing drinking water regulations, waterborne disease outbreaks continue to occur. They point out that between 1971 and 1983 there were 427 reported outbreaks with over 100,000 cases of waterborne disease. However, EPA also notes that many studies suggest that the vast majority of

Table 29. Summary of total coliform and fecal coliform data related to well depth.

Well depth range in feet	% samples in well depth range with: total coliform bacteria	FECAL coliform bacteria	% all samples with: total coliform bacteria in well-depth range	FECAL coliform bacteria in well-depth range	Relative % of total coliform positives with FECAL coliforms, in well depth range
< 50 ft	69%	8%	43%	45%	12%
50-99 ft	55%	12%	26%	36%	22%
100-199 ft	29%	3%	17%	18%	10%
200-299 ft	24%	0%	7%	0%	0%
300-399 ft	29%	0%	4%	0%	0%
≥ 400 ft	32%	0%	3%	0%	0%

waterborne disease outbreaks and cases are not reported. Various studies suggest that only 1/4 to 1/25 or fewer outbreaks are actually reported, likely because the vast majority of people experiencing gastroenteritis, some of which may be waterborne in origin, do not seek medical attention, and physicians generally cannot attribute gastroenteritis to any specific source.

Keswick and Gerba (1980) have reviewed data more specifically for groundwater. They note between 1946 and 1977 there were 246 outbreaks of waterborne disease that were related to contaminated groundwater; about 50% of the recorded outbreaks. For all outbreaks of waterborne diseases over this time they conclude that about 65% can be attributed to illness of probable viral etiology (hepatitis, poliomyelitis, gastroenteritis). Their review noted many studies where no correlation could be found between total coliform indicators and the presence of viruses, and that viruses survived in water treated (chlorinated) to remove such indicator bacteria. Further studies have indicated that total coliforms, other bacteria, and viruses exhibit different mobility and survival in various environments, including the soil-groundwater system (Keswick and Gerba, 1980; Bitton et al., 1983; Pekdeger and Matthess, 1983). These studies would indicate that the absence of total coliforms may not indicate the absence of other non-coliform organisms.

Given the continued concern for waterborne disease, EPA is issuing additional requirements for public water supply systems, to add needed control. This includes: added requirements for filtration, disinfection, and maintenance of distribution systems for water supplies using surface waters; new rules for disinfection for groundwater systems; added requirements for sanitary surveys and for disinfectant residuals and residual monitoring. While these rules pertain only to public water supplies they can provide some guidance for evaluating private water supplies, as well.

Comparison With Other Data

The percentages of water systems positive for total coliforms was greater for the SWRL population than typically seen in the large number of samples analyzed by the UHL (Table 25). For the 1980s the UHL data has averaged about 35% total coliform-positive samples/year for private water-supplies, compared to the 45% recorded by SWRL. Only in 1986 did the UHL data exceed 40%; the data for the exact months coincident with SWRL have not been fully analyzed. These differences may be related to the different nature of the sample populations. (A more detailed comparison with UHL data will be addressed in subsequent reports.) The affects of the drought, and falling water tables during SWRL are also not clear.

Conclusions

The occurrence of total coliform-positives in 45% of private water supplies in Iowa is cause for concern and should not be understated. However, single positive total coliform samples from groundwater supplies do not provide unequivocal inferences, and the implications should not be overstated either. For at least two decades, the UHL data has shown 30 to 40% of samples from private wells are total coliform positive. Total coliforms, as reviewed, are ubiquitous constituents of soils, surface water, and shallow groundwater. Because the total coliform group is comprised of both fecal and non-fecal organisms, the mere presence of *total* coliforms in drinking water does not mean a water supply is contaminated with fecal material. As the comparative SWRL data show, only about 15% of total coliform positives were positive for fecal bacteria.

Well conditions should not be interpreted solely on the presence of *total* coliform bacteria in a water-supply sample, particularly a single sample. Any interpretation is confounded by the variance of the testing (about 25% do not repeat; other bacterial forms may register positive) and the high percentage of positives that typically originate from some point in the water-system (e.g., a storage tank, cistern, water line) other than the well. Hence, "poor" well construction or maintenance, or "seepage" around the well head, cannot be an automatic interpretation. The only sound, general interpretation of a persistent presence of total coliforms is that the water-system is allowing interaction with the soil or soil-water, shallow groundwater, or possibly surface water. This could indicate that the system might be prone to other forms of contamination. As discussed in later sections, however, total coliforms are not good predictors of other contaminants.

Total coliform analysis continues to be a very useful tool, particularly for public water supplies to assess the efficacy of treatment measures and to assess problems in water distribution systems. This process involves repeated sampling over time, and sample surveys throughout the distribution system. Accurate assessment of the implications of total coliforms in private water supplies should involve an analogous "sanitary survey" of the water system. Repeat samples may be needed to assess persistence. Samples may need to be collected at various isolated points in the water-distribution system to find the point of entry. Fecal coliform analysis may also be used to identify more acute problems.

When total coliform positives can be attributed to the water from the well itself (which appears to be the case in only about 1/4 of positives) it can be suggestive of well construction deficiencies, at least for deeper wells. In relatively deep wells, that are cased to some depth below the surface (typically 40 ft or more), the presence of total coliforms may indicate contamination (non-fecal or fecal) from shallow groundwater, or ponded surface water seeping into the well through defects at the well head or in the top 10 to 20 feet of the well. In many deep wells, where bedrock is shallow, the presence of total coliforms often indicate a very shallow casing, even though the well boring may be deep.

The origin of total coliforms in shallow wells is another matter. These wells were often designed to produce water from the water table and the source of the coliforms is commonly just the shallow groundwater environment itself. Construction and maintenance problems may also be involved. With today's water-quality concerns, such construction appears deficient, but these concerns were not apparent when most wells were developed. (Current standards for well construction are outlined in the Iowa Administrative Code, Environmental Protection [567], Chapter 49, *Nonpublic Water Wells*.)

The occurrence of total coliforms in shallow groundwater is reflected in the relationship between the total coliform positives and the well depth of the SWRL water systems. Approximately 72% of wells <50 feet deep were positive for total coliforms; only 9% were positive for fecal coliform bacteria. About 70% of all the total coliform positives, and about 80% of the fecal coliform positives, came from systems with wells <100 feet deep. No systems using wells ≥200 feet deep were positive for fecal coliforms. The proportions of total coliform positives sites vary regionally, paralleling the use patterns of shallow wells and related well construction practices.

Some problems can be avoided with proper development of new wells, and some can be remediated by rehabilitating older wells. Deep wells need deep casing and grouting to avoid allowing shallow water into the well. Deep wells particularly should be cased to avoid multiple aquifer completions that may transmit shallow water and possible contaminants into deeper aquifers. For shallow seepage wells improvements in the well head, such as mounding, providing a concrete slab and tight cover seal can help to prevent acute well-head problems. Casing or grouting the upper portion of the well, or conversion to buried slab design may also help to eliminate total coliform problems. However, such wells are in settings where the water table can fluctuate greatly, and during the dry conditions of the 1988 and 1989 water levels were observed to drop below the upper grouted portions of adequately constructed wells. Such conditions may still allow contaminants from the very top of the water table into the well. While such measures should be taken, and clearly can help to remediate bacterial concerns, this may not alleviate chemical contamination problems. While some acute cases are related to well-head, well construction problems, monitoring studies throughout Iowa using monitoring wells and other instrumentation clearly show that nitrates and pesticides enter groundwater as a nonpoint source problem, as well.

Results from a single total coliform test are difficult to interpret, particularly by the average private well owner. SWRL participants with positive coliforms were advised to contact local experts, such as county sanitarians, for assistance in evaluating their results and their wells, to assess the need for possible rehabilitation. For isolated testing, such as the single samples often collected from private water supplies more complete interpretive information needs to be delivered to well owners. Also, fecal coliform analysis can provide more definitive assessment of potential pathogenic contamination.

PESTICIDE RESULTS

Pesticides are organic chemicals used to kill and control insects (insecticides), weeds (herbicides), and other pests (fungicides and miticides), that can affect crop or animal production. In recent years, studies in Iowa and elsewhere have found such compounds occurring in groundwater, surface water, and drinking-water supplies. This has raised public health concerns because these are toxic and hazardous compounds. Pesticides may reach groundwater by downward movement through the soil after application, through accidental spills, by back-siphoning accidents, and through improper disposal of formulation or rinse water.

The characteristics of pesticides vary substantially. No single laboratory test is available to determine their presence. For the SWRL study, 27 pesticides commonly used in Iowa were measured using EPA approved analytical methods (Tables 1-3), summarized in a previous section (see Hallberg et al., 1990, for details of analytical methods). Pesticide concentrations are reported in units of micrograms per liter (ug/L). A microgram per liter is equivalent to one part per billion (ppb).

The pesticides whose concentrations were measured in the SWRL study are commonly used in Iowa agriculture as herbicides and insecticides. After application pesticides break down naturally in the environment through various chemical and biological processes. The resulting decomposition compounds are called environmental metabolites. Environmental metabolites of some common compounds were measured to more completely characterize the extent of pesticide contamination; the presence of metabolites serve as added "markers" of contamination by the pesticide parent compounds. Some pesticides degrade very rapidly, and parent compounds are very unlikely to be found, yet the metabolites may remain and be toxicologically active and of concern. The metabolites included in the analyses were limited by the available chemical methods which were known or could be developed or refined for use during the course of this study; also, many degradation products are unknown or poorly known. Hence, only a few compounds were focused on; some of the metabolites of atrazine, alachlor, and carbofuran. Little is known about the environmental significance, persistence, and health effects of many metabolites. Currently there are no health advisory guidelines or standards established for the herbicide metabolites included in this study.

The properties of the environmental metabolites of alachlor, and the analytical

Table 30. Pesticides included in SWRL analyses but not detected in SWRL study. (Typical trade names in parentheses.)

Non-Detected Insecticides	Non-Detected Herbicides
chlorpyrifos (Lorsban)	acifluorfen (Blazer)
ethoprop (Mocap)	chloramben (Amiben)
fonofos (Dyfonate)	dicamba (Banvel)
phorate (Thimet)	2,4,5-Trichlorophenoxy acetic acid (2,4,5-T)
terbufos (Counter)	
dimethoate (Cygon)	2,4,5-Trichlorophenoxy propionic acid (2,4,5-TP; Silvex)
diazinon (Spectracide)	
malathion (Cythion)	butylate (Sutan)
parathion	
carbofuran (Furadan)	

Table 31. Summary statistics for pesticide detections; state-wide data.

Pesticide common chemical name	Typical product name	% wells with detections	mean conc. ug/L	median conc. ug/L	max. conc. ug/L	Lifetime HAL ug/L	MDL ug/L
atrazine	atrazine	4.4%	0.90	0.41	6.61	3.0	0.13
deethyl-atrazine	AAtrex	3.5%	0.54	0.27	2.86	N/A	0.10
deisopropyl-atrazine	"	3.4%	0.68	0.29	3.54	N/A	0.10
Total atrazine	"	8.0%	1.14	0.57	7.71	N/A	N/A
metribuzin	Sencor	1.9%	0.16	0.08	0.72	200.0	0.01
pendamethalin	Prowl	1.6%	0.06	0.04	0.12	N/A	0.02
metolachlor	Dual	1.5%	0.92	0.20	9.90	100.0	0.04
cyanazine	Bladex	1.2%	0.30	0.25	0.84	10.0	0.12
alachlor	Lasso	1.2%	0.67	0.06	4.76	0.4	0.02
hydroxy-alachlor	"	0.2%	0.91	---	0.91	N/A	0.10
picloram	Tordon	0.6%	0.39	0.18	2.00	500.0	0.10
dacthal	DCPA	0.4%	0.02	0.01	0.03	4000.0	0.01
3-hydroxy-carbofuran	Furadan	0.4%	0.38	0.13	0.98	N/A	0.02
3-keto-carbofuran	"	0.4%	0.03	0.03	0.03	N/A	0.02
propachlor	Ramrod	0.4%	0.11	0.05	0.28	90.0	0.02
trifluralin	Treflan	0.4%	5.60	0.04	14.90	2.0	0.02
2,4-D	many	0.6%	0.20	0.19	0.26	70.0	0.10

Table 32. Proportion of wells with pesticide detections, state-wide and in the hydrogeologic regions; including metabolites as independent detections.

	Proportion of Wells with detections %	95% confidence interval	Wells with 1 detection %	95% c.i.	Wells with 2 detections %	95% c.i.	Wells with 3 detections %	95% c.i.	Wells with 4 detections %	95% c.i.
State-wide:	13.6%	(11.1-16.2)	8.3%	(6.2-10.3)	3.0%	(1.8-4.3)	1.6%	(0.7-2.6)	0.7%	(0.1-1.4)
wells < 50 ft:	17.9%	(11.9-23.9)	9.4%	(5.0-13.8)	3.9%	(0.9-7.0)	1.5%	(0.0-3.6)	3.1%	(0.3-5.9)
wells ≥ 50 ft:	11.7%	(8.7-14.9)	7.6%	(5.1-10.3)	2.4%	(0.9-3.9)	1.6%	(0.4-2.8)	0.0%	---
Hydrogeologic Regions:										
northeast (1)	8.3%	(2.9-13.7)	5.2%	(0.8-9.6)	1.0%	(0.0-3.1)	2.1%	(0.0-4.9)	0.0%	---
east (2)	11.1%	(7.1-15.5)	4.7%	(1.9-7.6)	3.2%	(0.9-5.5)	2.3%	(0.3-4.2)	0.9%	(0.0-2.2)
south (3)	18.4%	(11.4-25.3)	13.4%	(7.4-19.4)	3.3%	(0.2-6.5)	0.9%	(0.0-2.5)	0.8%	(0.0-2.3)
west (4)	14.0%	(7.1-21.0)	8.9%	(3.2-14.6)	4.1%	(0.1-8.0)	1.1%	(0.0-3.3)	0.0%	---
northwest (5)	23.6%	(12.8-34.5)	10.9%	(2.6-19.2)	7.3%	(0.6-13.9)	1.8%	(0.0-5.4)	3.6%	(0.0-8.7)
north-central (6)	11.3%	(5.3-17.4)	9.4%	(3.8-15.0)	0.9%	(0.0-2.7)	1.0%	(0.0-2.9)	0.0%	---

Table 33. Summary of pesticide concentration data; state-wide data.

Pesticide common chemical name	% wells with detections	95% confidence interval	mean conc. ug/L	s.d. ug/L	median conc. ug/L	range conc. ug/L
alachlor	1.2%	(0.4-2.0)	0.67	0.80	0.06	0.02 - 4.76
hydroxy-alachlor	0.2%	(0.0-0.4)	0.91	---	0.91	- 0.91
atrazine	4.4%	(2.8-5.9)	0.90	1.12	0.41	0.13 - 6.61
deethyl-atrazine	3.5%	(2.1-4.8)	0.54	0.42	0.27	0.11 - 2.86
deisopropyl-atrazine	3.4%	(2.0-4.7)	0.68	0.86	0.29	0.10 - 3.54
Total atrazine	8.0%	(6.0-10.0)	1.14	1.30	0.57	0.12 - 7.71
carbofuran	0.0%	---	---	---	---	-
3-keto-carbofuran	0.4%	(0.0-0.9)	0.03	---	0.03	- 0.03
3-hydroxy-carbofuran	0.4%	(0.0-0.9)	0.38	---	0.13	0.05 - 0.98
cyanazine	1.2%	(0.4-2.0)	0.30	0.30	0.25	0.14 - 0.84
dacthal	0.4%	(0.0-0.9)	0.02	---	0.01	0.01 - 0.03
metolachlor	1.5%	(0.6-2.4)	0.92	1.69	0.20	0.04 - 9.90
metribuzin	1.9%	(0.9-2.9)	0.16	0.17	0.08	0.02 - 0.72
pendamethalin	1.6%	(0.7-2.6)	0.06	0.04	0.04	0.02 - 0.12
picloram	0.6%	(0.01-1.1)	0.39	1.34	0.18	0.10 - 2.00
propachlor	0.4%	(0.0-0.9)	0.11	0.18	0.05	0.02 - 0.28
trifluralin	0.4%	(0.0-0.9)	5.60	---	0.04	0.04 - 14.9
2,4-D	0.6%	(0.02-1.1)	0.20	0.05	0.19	0.15 - 0.26

methods for these metabolites remain problematical. Laboratory studies, related to SWRL, assessed alachlor degradation in simulated subsurface/groundwater environments and suggested that hydroxy-alachlor and acetyl-alachlor are formed (Lynch, 1990). These metabolites are more polar than parent alachlor and appear more water soluble, as well. This may result in these compounds being incompletely retained in the extracts analyzed by SWRL methods. Refined derivatization and extraction methods will likely be necessary to accurately measure them. Hence, at best, the record of detections for hydroxy-alachlor is a minimum value.

Sixteen of the pesticides monitored during this study were not detected in any of the well-water samples (Table 30). Carbofuran was not detected during the study, but two of its environmental metabolites were observed.

The pesticide detections in the SWRL groundwater samples are summarized in Table 31. Based on the SWRL study, about 13.6 percent of private, rural drinking water wells in Iowa are estimated to be contaminated with one or more pesticides (Table 32; Fig. 34). From a statistical viewpoint, at the 95 percent confidence level, the true state-wide proportion of wells contaminated with one or more pesticides lies between 11.1 and 16.2 percent, for the time and conditions of the SWRL survey. As indicated on Table 32, 8.3% of the wells had a detection of a single pesticide compound; 5.3% had 2 or more pesticides detected (Fig. 35), with 0.7% of wells having detections of 4 compounds in a single sample. The values in Table 32 and 33 treat pesticides and their metabolites as detections of independent compounds.

Table 34 presents the same data except that detections of the environmental metabolites are included as representing their parent pesticide. For example, if de-ethyl atrazine and atrazine were both measured in a sample, Table 34 would count that observation as the detection of only one pesticide in the well. Hence, the estimated state-wide proportion of wells contaminated with *more* than one pesticide residue is slightly less. Since most previous studies in Iowa did not monitor most environmental metabolites, Table 34 provides for comparison to earlier studies.

A look at specific pesticide detections in SWRL focuses attention on atrazine (and its metabolites) as the most commonly detected pesticide, as also shown in past studies (Hallberg, 1989b). More detailed discussion about detections of atrazine and its metabolites is presented in the next section. In descending order, the most commonly detected pesticides were atrazine, de-ethyl atrazine, de-isopropyl atrazine, metribuzin, pendimethalin, metolachlor, alachlor, cyanazine, picloram, 2,4-D, propachlor, trifluralin, dacthal, 3-hydroxy carbofuran, 3-keto carbofuran, and hydroxy alachlor. In total, 16 pesticide compounds were detected in the SWRL well-water samples, including 11 parent compounds and 5 environmental metabolites. Multiple residues were detected in all regions of the state (Fig. 36).

The mean concentrations of these pesticides on a state-wide basis are generally less than one part per billion, and typically were below recommended lifetime health advisory levels (Table 31). Lifetime health advisory levels were exceeded at eight sites: five with atrazine, two with alachlor, and one with trifluralin. On a state-wide basis, 1.2 percent of the private, rural drinking-water wells in Iowa are estimated to be contaminated with a pesticide exceeding the EPA recommended lifetime health advisory levels (HAL).

Environmental Health Implications: Population Exposure

Based on the rural population in Iowa from the 1980 Census, about 94,000 rural Iowa residents (or about 13.1 percent of the rural population) are consuming drinking water from private, rural wells that contain one or more pesticides. About 5,400 rural residents (or about 0.7 percent of the rural population) are consuming drinking water from wells that contain a pesticide concentration above the HAL. These percentages are slightly less than the percentage of contaminated wells, noted above, because some rural Iowans do not use their wells as their primary source of drinking water, but rather obtain their drinking water from rural water districts. It should be reiterated that this study deals only with groundwater contamination in rural areas. While groundwater is the predominant source of drinking water in rural Iowa, in certain areas of the state, particularly portions of southern Iowa, many rural residents

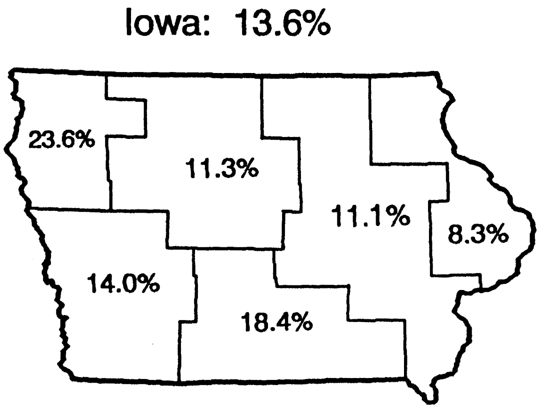


Figure 34. Proportion of wells with one or more pesticide detections state-wide, and in the hydrogeologic regions.

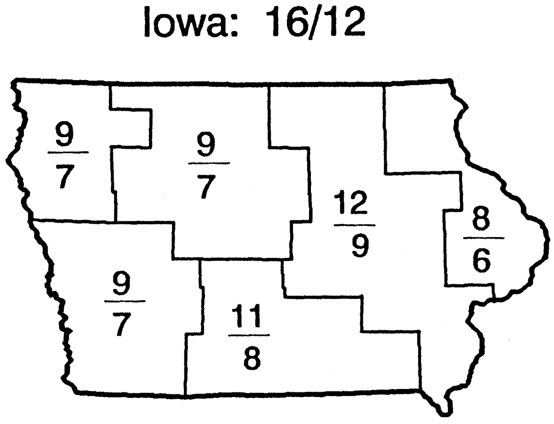


Figure 36. Number of pesticide compounds detected in well water samples from the hydrogeologic regions (total/parent compounds and carbofuran metabolites).

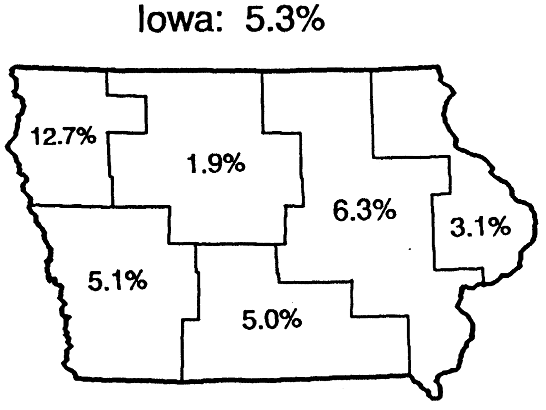


Figure 35. Proportion of wells with two or more pesticide detections state-wide, and in the hydrogeologic regions.

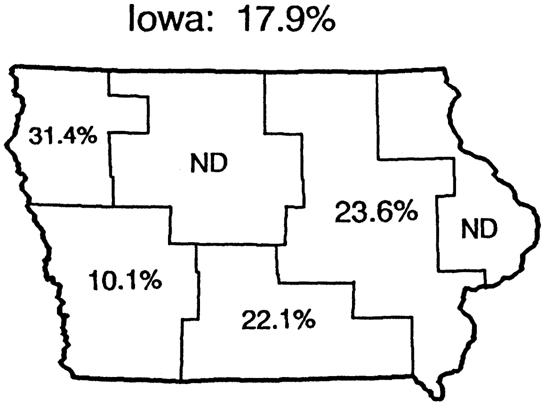


Figure 37. Proportion of wells < 50 ft deep, with one or more pesticide detections.

now receive their water from rural water systems, which utilize surface water supplies. Past surveys have indicated that these systems often have pesticide concentrations greater than those found in groundwater (IDNR Environ. Prot. Div., 1988; Hallberg, 1989b).

Distribution of Pesticide Contamination

As indicated in Table 32 and Figure 34, the extent of pesticide contamination varies regionally, from 8.3% of the wells in northeast Iowa, to nearly a quarter of the wells (23.6%) in northwest Iowa. (The regional variations are statistically significant, at the 90% confidence level; p=0.059.) Tables 35 through 40 summarize the pesticide concentration data for the individual regions.

State-wide, shallow wells (<50 ft deep) are more likely to be contaminated with a pesticide. Approximately 17.9 percent of the private, rural drinking water wells less than 50 feet deep are contaminated with one or more pesticide, while about 11.9 percent of the deeper wells are contaminated (Table 32). The difference in percent of contamination between shallow and deeper wells is statistically significant. Figure 37, summarizes the regional proportions of shallow wells contaminated with pesticides. For wells with pesticide detections, a significant decline in

Table 34. Proportion of wells with pesticide detections, state-wide and in the hydrogeologic regions; metabolites are included with parent compounds.

	Proportion of Wells with detections %	95% confidence interval	Wells with 1 detection %	95% c.i.	Wells with 2 detections %	95% c.i.	Wells with 3 detections %	95% c.i.	Wells with 4 detections %	95% c.i.
State-wide:	13.6%	(11.1-16.2)	9.7%	(7.5-11.9)	3.1%	(1.8-4.3)	0.6%	(0.0-1.2)	0.3%	(0.0-0.7)
wells < 50 ft:	17.9%	(11.9-23.9)	12.2%	(7.1-17.3)	3.0%	(0.5-5.6)	1.2%	(0.0-2.9)	1.5%	(0.0-3.6)
wells ≥ 50 ft:	11.7%	(8.7-14.9)	8.6%	(6.0-11.5)	2.6%	(1.1-4.2)	0.4%	(0.0-1.0)	0.0%	---
Hydrogeologic Regions:										
northeast (1)	8.3%	(2.9-13.7)	6.2%	(1.4-11.0)	1.0%	(0.0-3.1)	1.0%	(0.0-3.1)	0.0%	---
east (2)	11.1%	(7.1-15.5)	5.6%	(2.6-8.8)	4.1%	(1.5-6.7)	0.5%	(0.0-1.3)	0.9%	(0.0-2.2)
south (3)	18.4%	(11.4-25.3)	15.9%	(9.3-22.5)	1.7%	(0.0-4.1)	0.8%	(0.0-2.3)	0.0%	---
west (4)	14.0%	(7.1-21.0)	10.0%	(3.9-16.0)	4.1%	(0.1-8.0)	0.0%	---	0.0%	---
northwest (5)	23.6%	(12.8-34.5)	14.5%	(5.3-23.8)	9.1%	(1.5-16.7)	0.0%	---	0.0%	---
north-central (6)	11.3%	(5.3-17.4)	10.3%	(4.5-16.1)	0.0%	---	1.0%	(0.0-3.0)	0.0%	---

Table 35. Summary of pesticide concentration data for northeastern Iowa, Hydrogeologic region 1.

Pesticide common chemical name	% wells with detections	95% confidence interval	mean conc. ug/L	s.d. ug/L	median conc. ug/L	range conc. ug/L
alachlor	0.0%	---	---	---	---	-
hydroxy-alachlor	0.0%	---	---	---	---	-
atrazine	3.1%	(0.0-6.6)	0.43	0.04	0.41	0.41 - 0.48
deethyl-atrazine	2.1%	(0.0-4.9)	0.22	0.12	0.22	0.13 - 0.30
deisopropyl-atrazine	1.0%	(0.0-3.1)	0.49	---	0.49	0.49 - 0.49
Total atrazine	4.2%	(0.2-8.1)	0.60	0.53	0.44	0.15 - 1.37
carbofuran	0.0%	---	---	---	---	-
3-keto-carbofuran	0.0%	---	---	---	---	-
3-hydroxy-carbofuran	0.0%	---	---	---	---	-
cyanazine	0.0%	---	---	---	---	-
dacthal	1.0%	(0.0-3.1)	0.03	---	0.03	0.03 - 0.03
metolachlor	1.0%	(0.0-3.1)	0.04	---	0.04	0.04 - 0.04
metribuzin	2.1%	(0.0-4.9)	0.05	0.04	0.04	0.02 - 0.07
pendamethalin	0.0%	---	---	---	---	-
picloram	0.0%	---	0.10	---	0.10	0.10 - 0.10
propachlor	0.0%	---	---	---	---	-
trifluralin	2.1%	(0.0-4.9)	0.04	0	0.04	0.04 - 0.04
2,4-D	0.0%	---	---	---	---	-

Table 36. Summary of pesticide concentration data for eastern Iowa, Hydrogeologic region 2.

Pesticide common chemical name	% wells with detections	95% confidence interval	mean conc. ug/L	s.d. ug/L	median conc. ug/L	range conc. ug/L
alachlor	2.7%	(0.6-4.9)	1.03	1.27	0.14	0.02 - 4.76
hydroxy-alachlor	1.1%	(0.0-4.1)	0.91	---	0.91	0.91 - 0.91
atrazine	4.1%	(1.5-6.7)	0.49	0.44	0.38	0.13 - 1.87
deethyl-atrazine	3.6%	(1.1-6.1)	0.35	0.19	0.22	0.13 - 0.86
deisopropyl-atrazine	2.7%	(0.6-4.9)	1.77	0.17	0.23	0.11 - 3.10
Total atrazine	7.7%	(4.2-11.2)	0.89	0.96	0.45	0.13 - 4.71
carbofuran	0.0%	---	---	---	---	-
3-keto-carbofuran	0.0%	---	---	---	---	-
3-hydroxy-carbofuran	0.5%	(0.0-1.3)	0.98	---	0.98	0.98 - 0.98
cyanazine	1.8%	(0.0-3.6)	0.20	0.08	0.20	0.14 - 0.29
dacthal	0.9%	(0.0-2.2)	0.01	0	0.01	0.01 - 0.01
metolachlor	0.9%	(0.0-2.2)	0.76	0.86	0.76	0.15 - 1.37
metribuzin	1.8%	(0.1-3.6)	0.19	1.70	0.14	0.06 - 0.43
pendamethalin	1.5%	(0.1-3.4)	0.09	0.06	0.12	0.02 - 0.12
picloram	0.0%	---	---	---	---	-
propachlor	0.5%	(0.0-1.3)	0.28	---	0.28	0.28 - 0.28
trifluralin	0.0%	---	---	---	---	-
2,4-D	0.0%	---	---	---	---	-

Table 37. Summary of pesticide concentration data for south-central Iowa, Hydrogeologic region 3.

Pesticide common chemical name	% wells with detections	95% confidence interval	mean conc. ug/L	s.d. ug/L	median conc. ug/L	range conc. ug/L
alachlor	0.8%	(0.0-2.3)	0.07	---	0.07	0.07 - 0.07
hydroxy-alachlor	0.0%	---	---	---	---	-
atrazine	5.1%	(1.1-9.1)	2.80	2.02	0.46	0.14 - 6.61
deethyl-atrazine	3.3%	(0.1-6.5)	0.29	0.05	0.28	0.18 - 0.31
deisopropyl-atrazine	3.4%	(0.1-6.6)	0.80	0.22	0.79	0.63 - 1.02
Total atrazine	8.5%	(3.4-13.5)	2.53	1.25	0.92	0.14 - 6.97
carbofuran	0.0%	---	---	---	---	-
3-keto-carbofuran	1.6%	(0.0-3.8)	0.03	---	0.03	0.03 - 0.03
3-hydroxy-carbofuran	0.8%	(0.0-2.4)	0.05	---	0.05	0.05 - 0.05
cyanazine	2.5%	(0.0-5.3)	0.59	0.40	0.66	0.27 - 0.84
dacthal	0.0%	---	---	---	---	-
metolachlor	1.6%	(0.0-3.9)	0.11	---	0.16	0.04 - 0.27
metribuzin	0.9%	(0.0-2.5)	0.12	---	0.12	0.12 - 0.12
pendamethalin	3.4%	(0.1-6.6)	0.05	0.04	0.04	0.02 - 0.11
picloram	0.0%	---	0.49	1.34	0.26	0.10 - 2.00
propachlor	0.0%	---	---	---	---	-
trifluralin	0.0%	---	---	---	---	-
2,4-D	0.0%	---	---	---	---	-

Table 38. Summary of pesticide concentration data for southwestern Iowa, Hydrogeologic region 4.

Pesticide common chemical name	% wells with detections	95% confidence interval	mean conc. ug/L	s.d. ug/L	median conc. ug/L	range conc. ug/L
alachlor	0.0%	---	---	---	---	-
hydroxy-alachlor	0.0%	---	---	---	---	-
atrazine	6.3%	(1.3-11.3)	0.38	0.44	0.25	0.15 - 1.08
deethyl-atrazine	1.1%	(0.0-3.3)	0.11	---	0.11	0.11 - 0.11
deisopropyl-atrazine	2.5%	(0.0-5.5)	0.18	0.11	0.17	0.14 - 0.29
Total atrazine	7.7%	(2.4-13.1)	0.44	0.43	0.27	0.17 - 1.29
carbofuran	0.0%	---	---	---	---	-
3-keto-carbofuran	1.1%	(0.0-3.3)	0.03	---	0.03	0.03 - 0.03
3-hydroxy-carbofuran	0.0%	---	---	---	---	-
cyanazine	1.1%	(0.0-3.3)	0.21	---	0.21	0.21 - 0.21
dacthal	0.0%	---	---	---	---	-
metolachlor	0.7%	(0.0-2.1)	0.25	---	0.25	0.25 - 0.25
metribuzin	4.1%	(0.1-8.0)	0.11	0.14	0.07	0.02 - 0.30
pendamethalin	2.2%	(0.0-5.3)	0.05	0.04	0.05	0.02 - 0.07
picloram	0.0%	---	---	---	---	-
propachlor	1.1%	(0.0-3.3)	0.05	---	0.05	0.05 - 0.05
trifluralin	0.0%	---	---	---	---	-
2,4-D	0.0%	---	---	---	---	-

Table 39. Summary of pesticide concentration data for northwestern Iowa, Hydrogeologic region 5.

Pesticide common chemical name	% wells with detections	95% confidence interval	mean conc. ug/L	s.d. ug/L	median conc. ug/L	range conc. ug/L
alachlor	1.8%	(0.0-5.4)	0.05	---	0.05	0.05 - 0.05
hydroxy-alachlor	0.0%	---	---	---	---	-
atrazine	5.4%	(0.0-11.5)	1.13	2.21	0.44	0.28 - 3.41
deethyl-atrazine	14.5%	(5.5-23.5)	0.80	1.27	0.86	0.11 - 2.86
deisopropyl-atrazine	10.9%	(2.5-19.3)	0.82	0.87	0.27	0.11 - 3.54
Total atrazine	18.2%	(8.0-28.3)	1.70	1.43	1.05	0.14 - 7.71
carbofuran	0.0%	---	---	---	---	-
3-keto-carbofuran	0.0%	---	---	---	---	-
3-hydroxy-carbofuran	0.0%	---	---	---	---	-
cyanazine	0.0%	---	---	---	---	-
dacthal	0.0%	---	---	---	---	-
metolachlor	3.6%	(0.0-8.7)	5.21	---	5.11	0.31 - 9.90
metribuzin	1.8%	(0.0-5.4)	0.72	---	0.72	0.72 - 0.72
pendamethalin	1.8%	(0.0-5.4)	0.04	---	0.04	0.04 - 0.04
picloram	0.0%	---	---	---	---	-
propachlor	0.0%	---	---	---	---	-
trifluralin	1.8%	(0.0-5.4)	14.89	---	14.89	14.89 - 14.89
2,4-D	0.0%	---	0.17	0.02	0.17	0.15 - 0.18

Table 40. Summary of pesticide concentration data for north-central Iowa, Hydrogeologic region 6.

Pesticide common chemical name	% wells with detections	95% confidence interval	mean conc. ug/L	s.d. ug/L	median conc. ug/L	range conc. ug/L
alachlor	0.0%	---	---	---	---	-
hydroxy-alachlor	0.0%	---	---	---	---	-
atrazine	2.8%	(0.0-6.0)	1.34	2.28	0.89	0.14 -3.36
deethyl-atrazine	0.9%	(0.0-2.7)	1.30	---	1.30	1.30 -1.30
deisopropyl-atrazine	2.8%	(0.0-6.0)	0.12	0.02	0.12	0.10 -0.13
Total atrazine	5.7%	(1.2-10.1)	0.97	1.54	0.16	0.13 -4.86
carbofuran	0.0%	---	---	---	---	-
3-keto-carbofuran	0.0%	---	---	---	---	-
3-hydroxy-carbofuran	0.9%	(0.0-2.7)	0.13	---	0.13	0.13 -0.13
cyanazine	0.0%	---	---	---	---	-
dacthal	0.0%	---	---	---	---	-
metolachlor	2.0%	(0.0-4.8)	0.07	0.03	0.07	0.05 -0.09
metribuzin	1.0%	(0.0-3.0)	0.04	---	0.04	0.04 -0.04
pendamethalin	0.9%	(0.0-2.7)	0.03	---	0.03	0.03 -0.03
picloram	0.0%	---	---	---	---	-
propachlor	1.0%	(0.0-3.0)	0.02	---	0.02	0.02 -0.02
trifluralin	0.0%	---	---	---	---	-
2,4-D	0.0%	---	0.23	0.05	0.23	0.19 -0.26

detections occurs at a well depth of about 100 feet (Table 41). The proportion of wells with pesticides increases again for wells >400 ft deep (20%), but this is a small proportion of the total wells with detections (6%). There is a systematic decline in the proportion of pesticide detections with well depth: approximately 40% of all the pesticide detections appeared in wells <50 ft deep, and two-thirds occurred (64%) in wells <100 ft deep. Tables 42 and 43 summarize the concentration data for wells less than 50 feet deep and greater than 50 feet deep.

For future monitoring studies of pesticides in Iowa's water supplies, environmental metabolites of pesticides, particularly those of atrazine, should be included as analytes in the study design. This will provide a more complete characterization of water-quality impacts. However, to monitor trends in contamination over time, results of subsequent studies should be reported in terms of parent pesticide detections alone, and separately with their metabolites as markers.

Atrazine and Its Metabolites

As described above, atrazine was the most commonly detected pesticide compound in the SWRL survey. Hence, this section of the report will summarize some further information and findings about atrazine. Atrazine is the common chemical name for the active ingredient in such widely-used herbicides as Atrazine and AAtrex, and in popular pre-mixed combination products, such as Bicep, Extrazine, Lariat and Marksman. It is used to control many broadleaf weeds and certain grasses in corn and sorghum. It is also used for general weed control on industrial land.

Atrazine is one of the most commonly used pesticides in Iowa. It is cited by farm operators as the most commonly applied herbicide on corn acres, with nearly 50% noting its use (e.g., Fig. 17), and regionally, up to 70% of farms noting its use (Fig. 18). It has been in use for over 30 years, longer than most other herbicides, except 2,4-D and 2,4,5-T. In 1984, over 8,000,000 lbs of atrazine (as the active ingredient) was applied to cropland in Iowa (Wintersteen and Hartzler, 1987).

The SWRL analyses show frequent detections

Table 41. Summary of pesticide detections related to well depth ranges (for wells with depths reported).

Well depth range in feet	% wells in depth range with pesticide detections	% all wells with pesticide detections
< 50 ft	18%	40%
50-99 ft	16%	24%
100-199 ft	9%	17%
200-399 ft	10%	13%
>= 400 ft	20%	6%

of low concentrations of atrazine and its environmental metabolites in Iowa groundwater. As summarized in the previous section, atrazine was the most frequently detected pesticide compound in the SWRL study, both state-wide and within each hydrogeologic region, followed in frequency by the occurrence of its metabolites. The metabolites of atrazine were commonly detected in combination with atrazine, but over half of the metabolite detections occurred when the parent atrazine was not present. Comparatively little is known about the environmental significance, persistence, or health effects of these metabolites.

Table 44 summarizes the findings for concentrations of atrazine, greater than its detection limit of 0.13 ug/L, and for two of its environmental metabolites -- deethyl-atrazine (DEA) and deisopropyl-atrazine (DIA). The mean concentration is calculated using only quantified observations; i.e., those that exceed the detection limits.

Data are also reviewed for the total atrazine detections, combining the findings of atrazine and its metabolites. Concentrations for the combined, or total atrazine compounds are first normalized to equivalent concentrations of atrazine by a factor derived from the molecular weight ratios of the compounds (factor = 1.16 for DEA and 1.25 for DIA). The combined concentration is the sum of atrazine and equivalent metabolite concentrations for each site.

Distribution of Atrazine Contamination

Based on the SWRL findings, approximately 8% of rural, private, drinking-water wells in Iowa are contaminated with atrazine or its metabolites. From a statistical viewpoint, with 95% confidence, the proportion of wells contaminated with atrazine, state-wide, lies between 6% and 10%. Atrazine occurrence varies regionally from 4% of wells in northeastern Iowa, to over 18% in northwest Iowa (Fig. 38). Statistically, these regional variations are significant (at the 90% confidence level; p=0.064).

Other detailed studies in Iowa have shown much greater proportions of wells containing atrazine for more localized areas. In nearly every study in Iowa and the Midwest, atrazine has been the most common pesticide shown to contaminate groundwater (Hallberg, 1989b). All these studies have indicated that shallow wells show a greater degree of contamination by man-made contaminants. The SWRL findings indicate that approximately 10% of wells less than 50 feet deep, across the state, show atrazine contamination (Table 44; and Fig. 39).

As noted, atrazine is also one of the most widely used herbicides. Considering how long it has been used, and that it has been one of the most widely used herbicides for 2 to 3 decades, there has probably been more atrazine applied to the land in Iowa than any other pesticide.

Atrazine Metabolites

One of the few other monitoring studies to include atrazine metabolites was done in Ontario, Canada (Ontario Ministry of Environment, 1987 a,b). In this study, deethyl atrazine occurred in about equal concentrations with the parent compound, at about the same frequency, and often occurred without the parent atrazine. In plot studies Muir and Baker (1976) observed deethyl and deisopropyl atrazine leaching through the soil and into tile waters draining the test plots. Deisopropyl atrazine was also determined to be a degradation product of cyanazine. They observed various triazine metabolites occurring in the shallow groundwater after the parent compounds were no longer detected.

As noted, in the SWRL samples metabolites of atrazine were commonly detected in

Table 42. Summary of pesticide concentration data for wells <50 ft deep, state-wide.

Pesticide common chemical name	% wells with detections	95% confidence interval	mean conc. ug/L	s.d. ug/L	median conc. ug/L	range conc. ug/L
alachlor	3.0%	(0.3-5.7)	0.39	0.73	0.05	0.02 - 2.34
hydroxy-alachlor	0.0%	---	---	---	---	-
atrazine	5.2%	(0.7-8.8)	1.08	1.49	0.42	0.17 - 6.61
deethyl-atrazine	5.8%	(2.1-9.5)	0.81	0.80	0.24	0.11 - 2.86
deisopropyl-atrazine	4.0%	(0.7-7.2)	1.29	1.41	0.53	0.11 - 3.54
Total atrazine	9.6%	(4.9-14.3)	1.73	1.62	0.85	0.14 - 7.71
carbofuran	0.0%	---	---	---	---	-
3-keto-carbofuran	1.2%	(0.0-2.9)	0.03	---	0.03	0.03 - 0.03
3-hydroxy-carbofuran	1.5%	(0.0-3.6)	0.05	---	0.52	0.05 - 0.98
cyanazine	2.3%	(0.0-4.6)	0.47	0.32	0.44	0.17 - 0.84
dacthal	0.0%	---	---	---	---	-
metolachlor	3.6%	(0.4-6.7)	1.45	4.73	0.27	0.04 - 9.90
metribuzin	3.0%	(0.3-5.7)	0.25	0.14	0.20	0.02 - 0.43
pendamethalin	1.9%	(0.0-4.0)	0.07	0.01	0.03	0.02 - 0.12
picloram	0.0%	---	0.70	1.34	0.26	0.10 - 2.00
propachlor	0.0%	---	---	---	---	-
trifluralin	0.4%	(0.0-1.1)	14.89	---	14.89	14.89 - 14.89
2,4-D	0.0%	---	0.17	0.02	0.17	0.15 - 0.18

Table 43. Summary of pesticide concentration data for wells ≥50 ft deep, state-wide.

Pesticide common chemical name	% wells with detections	95% confidence interval	mean conc. ug/L	s.d. ug/L	median conc. ug/L	range conc. ug/L	MDL ug/L
alachlor	0.7%	(0.0-1.5)	1.17	3.32	0.22	0.06 - 4.76	0.02
hydroxy-alachlor	0.0%	---	0.91	---	0.91	0.91 - 0.91	0.10
atrazine	3.8%	(2.0-5.6)	0.76	0.54	0.34	0.13 - 3.41	0.13
deethyl-atrazine	2.5%	(0.9-4.0)	0.43	0.24	0.35	0.13 - 1.30	0.10
deisopropyl-atrazine	3.5%	(1.8-5.3)	0.53	0.56	0.29	0.10 - 3.10	0.10
Total atrazine	7.6%	(5.1-10.2)	0.83	1.06	0.38	0.12 - 4.86	
carbofuran	0.0%	---	---	---	---	-	0.01
3-keto-carbofuran	0.3%	(0.0-0.8)	0.03	---	0.03	0.03 - 0.03	0.02
3-hydroxy-carbofuran	0.3%	(0.0-0.8)	0.13	---	0.13	0.13 - 0.13	0.02
cyanazine	0.9%	(0.0-1.9)	0.23	0.07	0.25	0.14 - 0.29	0.12
dacthal	0.2%	(0.0-0.6)	0.03	---	0.03	0.03 - 0.03	0.01
metolachlor	1.2%	(0.1-2.2)	0.14	0.08	0.09	0.04 - 0.31	0.04
metribuzin	1.4%	(0.3-2.5)	0.18	0.03	0.05	0.02 - 0.72	0.01
pendamethalin	1.1%	(0.1-2.0)	0.03	0.02	0.03	0.02 - 0.06	0.02
picloram	0.0%	---	0.10	---	---	-	0.10
propachlor	0.4%	(0.0-1.0)	0.05	0.18	0.15	0.02 - 0.28	0.02
trifluralin	0.4%	(0.0-1.0)	0.04	---	0.04	0.04 - 0.04	0.02
2,4-D	0.0%	---	0.19	---	---	-	0.10

Table 44. Summary of detections of atrazine and its metabolites.

	atrazine	DEA	DIA	Total/ combined atrazine
Detection limits, ug/L:	0.13	0.10	0.10	---
State-wide data:				
% all wells with detections:	4.4%	3.5%	3.4%	8.0%
mean concentration, ug/L:	0.90	0.54	0.68	1.14
maximum concentration, ug/L:	6.61	2.86	3.54	7.71
% wells < 50 ft:	5.3%	5.8%	4.0%	9.6%
mean concentration, ug/L:	1.08	0.81	1.29	1.73
maximum concentration, ug/L:	6.61	2.86	3.54	7.71
% wells ≥ 50 ft:	3.8%	2.5%	3.5%	7.6%
mean concentration, ug/L:	0.76	0.43	0.53	0.83
maximum concentration, ug/L:	3.41	1.30	3.10	4.86

combination with atrazine, but over 50% of the metabolite detections were observed when the parent atrazine was not present. Future monitoring studies should include atrazine metabolites, where possible, to gain a more complete understanding of the distribution and persistence of atrazine in groundwater.

Health Significance of Atrazine Detections

The U.S. Environmental Protection Agency has set a Lifetime Health Advisory level (HAL) for atrazine in drinking water at 3 micrograms per liter (3 ug/L). This infers that consuming water containing atrazine at or below 3 ug/L every day, over the course of one's lifetime, should not pose any adverse, non-cancer health risk. This level includes a margin of safety to protect human health and should be regarded as a guideline. The combined effects with other chemicals in water or food are not considered. This advisory considers only concentrations of the parent-compound atrazine, not atrazine combined with its metabolites.

Laboratory animal studies have shown that consuming atrazine at levels well above the Lifetime Health Advisory, over a long period of time, can result in adverse health effects, including tremors, changes in organ weights, and damage to the liver and heart.

The EPA also considers atrazine to be a possible human carcinogen (cancer causing agent). There is limited but equivocal information indicating that atrazine causes cancer in animals.

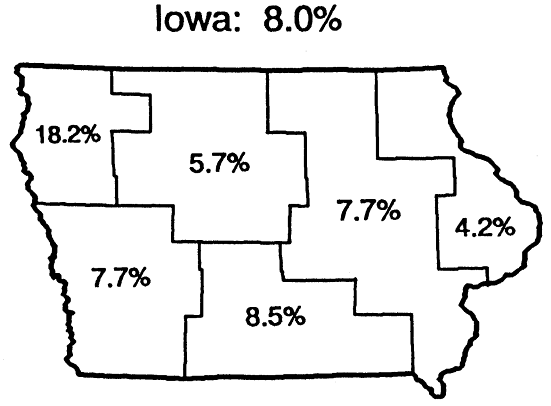


Figure 38. Percentage of wells with detections of atrazine and/or its metabolites.

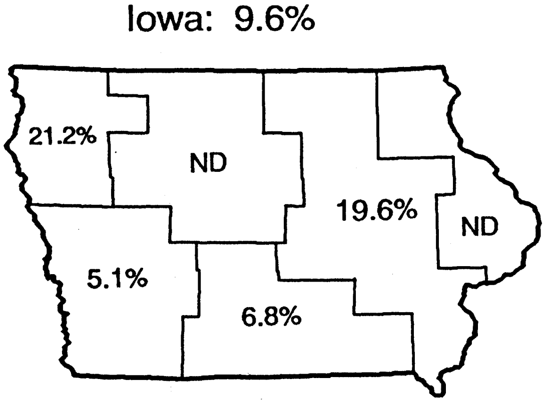


Figure 39. Percentage of wells <50 feet deep with detections of atrazine and/or its metabolites.

Because atrazine in drinking water may possibly increase the risk of cancer in humans, the Lifetime Health Advisory includes an additional margin of safety. Health advisories are not available for the metabolites.

The concentrations of atrazine observed in water supplies in this study were generally less than 1 ug/L. Atrazine was detected in 0.6% of the wells state-wide at concentrations exceeding the Lifetime Health Advisory Level. (These sites will be further discussed in the next sections of the report.) The highest concentration measured in this one-time sampling was 6.6 ug/L for atrazine, and 7.7 ug/L when combined with metabolites.

Discussion of Pesticide Detections

Given the general concerns with pesticide occurrences in groundwater and drinking-water supplies, and because use limitations have been placed on atrazine in geographically restricted portions of the state, the distribution of detections will be summarized by county occurrences. As repeatedly noted, neither SWRL proportions or concentration data provide valid summaries at the county level. However, a simple depiction of which counties had sites where atrazine and/or any pesticide was detected is a valid representation. These findings are summarized on Figures 40 and 41. Atrazine was detected throughout Iowa; it was detected at sites in 43 counties, covering every hydrogeologic region. In total, sites where any pesticide was detected in a water supply included locations from 59 counties throughout the state.

Atrazine Concentrations

Other water quality studies may have different detection or quantitation limits for atrazine. Table 45 presents a frequency distribution data for the atrazine and atrazine metabolite concentrations measured in the SWRL study.

Health Advisories and Pesticide Results

The Health Advisory Program of the U.S. EPA Office of Drinking Water provides information on the health effects of many drinking water contaminants. Health Advisory reports serve as informal, nonregulatory technical guides to assist public officials and the general public regarding the health implications of specific drinking water contaminants. Health Advisories and the Health Advisory Levels (HAL) are NOT legally enforceable Federal standards; they are simply advisory. Health Advisories are updated periodically, and HALs may change to reflect new information about the health consequences of specific contaminants.

A typical Health Advisory contains 20-30 pages of technical data (with references) about noncarcinogenic end points of toxicity. Data considered in the Health Advisory are based on both human and animal studies. Non-cancer toxicological data are usually the basis for the

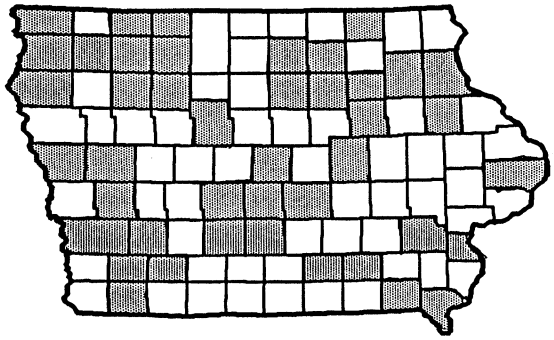


Figure 40. Counties (shaded) that included sites where atrazine and/or its metabolites were detected in one or more private well-water supplies.

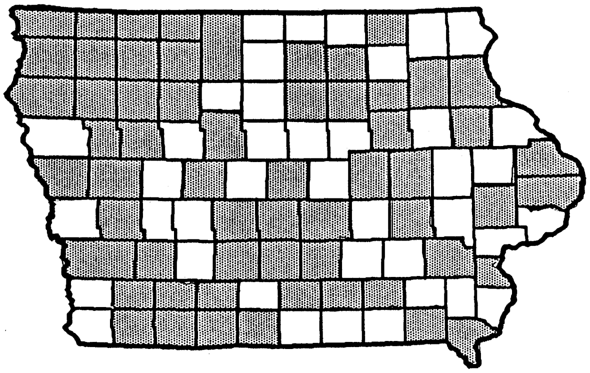


Figure 41. Counties (shaded) that included sites where any pesticide and/or metabolites were detected in one or more private well-water supplies.

Table 45. Proportional frequency of atrazine concentrations from SWRL well-water sampling.

Compound and concentration range, ug/L	% of sites/analyses in conc. range	Relative % of detections
atrazine		
< 0.13	95.6%	
0.13 - 0.19	0.9%	20%
0.20 - 1.00	2.6%	59%
1.01 - 3.00	0.3%	7%
> 3.00	0.6%	14%
deethyl atrazine		
< 0.10	96.5%	
0.10 - 0.19	1.2%	34%
0.20 - 1.00	1.6%	46%
1.01 - 3.00	0.7%	20%
deisopropyl atrazine		
< 0.10	96.5%	
0.10 - 0.19	1.6%	47%
0.20 - 1.00	1.3%	38%
1.01 - 3.00	0.1%	3%
> 3.00	0.4%	12%

risk assessments, which establish the Health Advisory Levels (HAL) for acute and chronic (lifetime) exposure to a contaminant. The HALs are the concentration limit estimate, at or below which no observable health effects would be expected; they contain a margin of safety to protect sensitive members of the populations.

Health Advisories do not quantitatively incorporate any potential carcinogenic risk from exposure into the recommended advisory levels. For those substances that are "known" or "probable" human carcinogens (Group A or B in the U.S. EPA Classification scheme), Lifetime Health Advisories are generally not determined. A separate carcinogenic risk estimate method is utilized involving a cancer potency factor, assumptions for lifetime exposure, and low-dose extrapolation models. Since current understanding of the biological mechanisms involved in cancer is limited, the estimates of cancer risk for the same chemical derived from the above method can differ by several orders of magnitude, depending upon the specific assumptions and formula used in the method.

New pesticides are subjected to rigorous testing for health related effects before EPA allows consideration for registration. However, complete data bases on many of the older pesticides are lacking. In addition to concerns with the consumption of pesticide residues in water, added data are also needed about exposure through inhalation and skin adsorption from water use in cooking and bathing. While cancer is a toxicological endpoint often discussed for chronic exposure, other subtle health affects related to neurological problems, reproductive effects, and problems related to fetal and infant development are also poorly known, and a matter of growing concern.

As part of the U.S. EPA National Pesticide Survey, one-page Health Advisory summaries were developed. A Lifetime HAL is available for many of the pesticides included in the SWRL study. Consuming water containing the chemical at or below the Lifetime Health Advisory Level is not expected to result in adverse non-cancer health effects. This level includes a margin of safety, but combined effects of several chemicals present in drinking water are not considered. For a few pesticides where cancer risk is of greater concern than non-cancer health effects, a cancer risk estimate is stated in lieu of a Lifetime Health Advisory.

A total of 12 pesticides and/or four of their environmental metabolites were detected in Iowa groundwater during the SWRL study. Health Advisory Summaries exist for 11 of the parent pesticides but none of the environmental metabolites. A brief review of the EPA summaries is provided below, for each of the pesticides detected in the SWRL study.

Alachlor

Alachlor is the common name for the active ingredient of the widely used herbicide Lasso. It is also used in various pre-mixed combination products such as Lariat and Cannon. Alachlor is a herbicide used to control annual grasses and broadleaf weeds in field corn, soybeans, and peanuts.

Non-Cancer Effects. Consuming alachlor has been shown to result in damage to the liver, kidneys, spleen, and eyes, in animal studies.

Cancer Risk. Alachlor also causes cancer in laboratory animals that are given high doses of the pesticide over the course of their lifetimes. Therefore, EPA considers alachlor to be a probable human carcinogen (cancer causing agent). Because alachlor is considered a probable human carcinogen the health advisory level is based on an estimate of the one-in-one million (10^{-6}) cancer risk; for alachlor this concentration is set at 0.4 ug/L. EPA estimates that if an individual consumes water containing alachlor at **0.4 micrograms per liter** over his or her entire lifetime, that person would theoretically have no more than a one-in-one-million chance of developing cancer as a direct result of drinking water containing this pesticide.

The proposed Maximum Contaminant Level (MCL), which would become the legally enforceable standard for public water supplies is **2.0 micrograms per liter**; for any possible carcinogen the recommended MCL goal is zero.

Atrazine

Atrazine is the common name for the active ingredient of such widely-used herbicides as AAtrex and Atrazine, and in pre-mixed combination products such as Bicep, Extrazine, Lariat, and Marksman. It is used to control certain broadleaf weeds and grasses in corn and sorghum. It is also used for general weed control on industrial and non-cropped land.

Non-Cancer Effects. EPA has set a Lifetime Health Advisory level for atrazine in drinking water at **3 micrograms per liter**. This level includes a margin of safety to protect human health and should be regarded as a guideline. EPA believes that water containing atrazine at or below this level is acceptable for drinking every day over the course of one's lifetime, and does not pose any health concerns.

However, consuming atrazine at high levels well above the Lifetime Health Advisory level over a long period of time has shown to result in adverse health effects in animal studies, including tremors, changes in organ weights, and damage to the liver and heart.

Cancer Risk. Atrazine is considered by EPA to be a possible human carcinogen (cancer causing agent). There is limited or uncertain information indicating that atrazine causes cancer in animals receiving high doses of the chemical over the course of their lifetimes. Because atrazine in drinking water may possibly increase the risk of cancer in humans, the Lifetime Health Advisory includes an additional margin of safety.

Carbofuran

Carbofuran is the common name for the active ingredient in the insecticides marketed as Furadan or Curaterr. It is a pesticide used to control insects, mites, and nematodes on corn and a variety of other crops.

Non-Cancer Effects. EPA has set a Lifetime Health Advisory level for carbofuran in drinking water at **40 micrograms per liter**. However, consuming carbofuran at high levels well above the Lifetime Health Advisory level over a long period of time has been shown to result in damage to the testes and uterus in animal studies, as well as inhibition of an enzyme called cholinesterase, in both humans and animals. Symptoms associated with inhibition of this enzyme in humans include nausea, vomiting, blurred vision, stomach cramps, excessive sweating, muscle weakness, headaches, and rapid heart rate.

Cancer Risk. Sufficient data are available from animal studies to indicate that carbofuran does not increase the risk of cancer in humans.

Cyanazine

Cyanazine is the common name for the active ingredient in the herbicide Bladex, and in combination products such as Extrazine and Conquest. It is used as a herbicide for the control of annual grasses and broadleaf weeds.

Non-Cancer Effects. EPA has set a Lifetime Health Advisory level for cyanazine in drinking water at **10 micrograms per liter**. However, consuming cyanazine at high levels well above the Lifetime Health Advisory level over a long period of time has been shown to result in adverse health effects in animal studies, including damage to the liver, changes in organ weights, changes in the blood, birth defects, and convulsions.

Cancer Risk. Data from laboratory studies are *inadequate* for EPA to determine if cyanazine can increase the risk of cancer in humans.

Dacthal

Dacthal, also known by trade names such as DCPA, Chlorothal, or Dacthalor, is a selective pre-emergence herbicide used to control various annual grasses in turf, ornamentals, strawberries, certain vegetables, and soybeans.

Non-Cancer Effects. EPA has set a Lifetime Health Advisory level for dacthal in drinking water to **4000 micrograms per liter**. However, consuming dacthal at high levels well above the Lifetime Health Advisory level over a long period of time has been shown to result in damage to the thyroid, histological changes in the liver, and increased kidney weights, in animal studies.

Cancer Risk. Data from the laboratory studies are *inadequate* for EPA to determine if dacthal can increase the risk of cancer in humans.

2,4-Dichlorophenoxyacetic Acid

2,4-dichlorophenoxyacetic acid, also known as 2,4-D, is the common chemical name for the widely used herbicide active ingredient, often marketed directly as 2,4-D, or by trade names such as Crossbow, Aqua Kleen, or Weed-B-Gon. It is marketed under many names for broadleaf weed control on lawns. It is a herbicide used on wheat, corn, and barley, on rangeland and pasture, and on lawns for broadleaf weeds.

Non-Cancer Effects. EPA has set a Lifetime Health Advisory level for 2,4-D in drinking water at **70 micrograms per liter**. However, consuming 2,4-D at high levels well above the Lifetime Health Advisory level over a long period of time has been shown to result in damage to the liver and kidneys, gastrointestinal irritation, changes in the blood, and decreased fetal weight, in animal studies.

Cancer Risk. Data from laboratory studies are *inadequate* for EPA to determine if 2,4-D can increase the risk of cancer in humans. However, recent studies in Nebraska (Weisenburger, 1985) and Kansas (Hoar, et al., 1986) that link 2,4-D exposure to increased relative risk in humans for non-Hodgkin's lymphoma are currently under review.

Metolachlor

Metolachlor is the common chemical name for the active ingredient in the herbicide Dual, and is also used in Bicep, Turbo, and Primextra. Metolachlor is a herbicide used for weed control in woody ornamentals, sunflowers, corn, and soybeans.

Non-Cancer Effects. EPA has set a Lifetime Health Advisory level for metolachlor in drinking water at **100 micrograms per liter**. However, consuming metolachlor at high levels well above the Lifetime Health Advisory level over a long period of time has been shown to result in adverse health effects in animal studies, including damage to the testes and a blood condition known as methemoglobinemia. Methemoglobinemia, also known as blue baby syndrome, is characterized by a reduced ability of the blood to carry oxygen.

Cancer Risk. Metolachlor is considered by EPA to be a possible human carcinogen (cancer causing agent). There is limited or uncertain information that metolachlor causes cancer in animals receiving high doses of the chemical over the course of their lifetimes. Because metolachlor in drinking water may possibly increase the risk of cancer in humans, the Lifetime Health Advisory includes an additional margin of safety.

Metribuzin

Metribuzin is the common chemical name for the active ingredient in the herbicides Lexone and Sencor, and is also contained in Turbo, Salute, and Preview. It is a herbicide used to control a large number of grass and broadleaf weeds infesting agricultural crops.

Non-Cancer Effects. EPA has set a Lifetime Health Advisory level for metribuzin in drinking water at **200 micrograms per liter**. However, consuming metribuzin at high levels well above the Lifetime Health Advisory level over a long period of time has been shown to result in kidney damage in animal studies.

Cancer Risk. Data from laboratory studies are *inadequate* for EPA to determine if metribuzin can increase the risk of cancer in humans.

Picloram

Picloram, also known under the trade name Tordon, is a broad-spectrum herbicide used to control broadleaf and woody plants in rangelands, pastures, and rights-of-way for powerlines and highways.

Non-Cancer Effects. EPA has set a Lifetime Health Advisory level for picloram in drinking water at **500 micrograms per liter**. However, consuming picloram at high levels well above the Lifetime Health Advisory level over a long period of time has been shown to result in damage to the liver, thyroid, testes, and arteries, and possibly reduced fertility, in animal studies.

Cancer Risk. Data from laboratory studies are *inadequate* for EPA to determine if picloram can increase the risk of cancer in humans.

Propachlor

Propachlor is the common chemical name for the active ingredient in the herbicides Bexton and Ramrod. It is a herbicide used to control many grasses and certain broadleaf weeds.

Non-Cancer Effects. EPA has set a Lifetime Health Advisory level for propachlor in drinking water at **90 micrograms per liter**. However, consuming propachlor at high levels well above the Lifetime Health Advisory level over a long period of time has been shown to result in damage to the liver, kidneys, and blood in animal studies.

Cancer Risk. Data from laboratory studies are *inadequate* for EPA to determine if propachlor can increase the risk of cancer in humans.

Trifluralin

Trifluralin is the common chemical name for the active ingredient in the widely used herbicide Treflan, and is also used in combination products such as Cannon, Commence, and TriScept. It is a herbicide used for control of annual grasses and broadleaf weeds in soybean, cotton and vegetable crops, fruit and nut trees, shrubs, and flowers. It is also used on golf courses, rights-of-way, and domestic and industrial sites.

Non-Cancer Effects. EPA has set a Lifetime Health Advisory level for trifluralin in drinking water at **2 micrograms per liter**. However, consuming trifluralin at high levels well above the Lifetime Health Advisory level over a long period of time has been shown to result in liver and kidney damage, decreased fetal weight and size, and increased numbers of miscarriages, in animal studies.

Cancer Risk. Trifluralin is considered by EPA to be a possible human carcinogen (cancer causing agent). There is limited or uncertain information indicating that trifluralin may cause cancer in animals receiving high doses of the chemical over the course of their lifetimes. Because trifluralin in drinking water may possibly increase the risk of cancer in humans, the Lifetime Health Advisory includes an additional margin of safety.

Sites Exceeding Pesticide Health Advisory Levels.

A total of eight sites in the SWRL study recorded pesticide concentrations in groundwater that exceeded health advisory levels. On a statewide basis, 1.2% of private, rural drinking water wells are estimated to be contaminated with one or more pesticides in excess of recommended health advisory levels.

Further analyses of field observations and questionnaire results from these eight sites provide some insight into the causes of these relatively high pesticide contaminations in groundwater. For reference and discussion, the sites will simply be referred to by letter.

Site A. Alachlor was detected at a concentration of 2.3 ug/L in January, 1989, in a sample collected from the kitchen tap at SWRL site A. This site was not a farm site; it was on an acreage and the house was built on land previously farmed. The property is currently surrounded by farm land, with corn on three sides of it during the period of record. Other than using an occasional portion of Sevin on the garden, agricultural chemicals are not stored or used at this non-farm site.

The property lies on an alluvial plain, and a shallow sand-point well (20 feet) is used for the water supply. However, the well head is well protected; the well is located in the basement of the house. No coliform bacteria were present in the sample; it did contain a moderate NO_3-N concentration, of 7.2 mg/L. The well-water also contained multiple pesticide residues, including atrazine at 1.9 ug/L, metribuzin at 0.2 ug/L, and a carbofuran metabolite, 3-hydroxy at 1.0 ug/L, in addition to the alachlor (at 2.3 ug/L).

Soils in the immediate area range from units that have >60 inches of silt loam to silty clay loam textures, to higher bottom areas (low terraces) that have 36 to 60 inches of loam or silt loam textures over sandy materials.

There are no apparent on-site chemical handling factors that contribute to these pesticide detections. Leaching of pesticides, through the soil and into the alluvial aquifer, from normal agricultural applications to the surrounding cropped fields is the likely cause of this contamination.

Site B. Alachlor was detected at a concentration of 4.8 ug/L in April, 1989, from a sample collected at a hydrant near the well at site B. The sample was positive for coliform bacteria (2.2 MPN); nitrate-N levels were relatively low, 1.8 mg/L. Cyanazine was also detected, at a concentration of 0.23 ug/L.

The water is supplied by a relatively new deep well (500 feet in depth), that was constructed in 1978. It has a six-inch steel casing, but the depth of casing is not known. The well head is sealed and a submersible pump is used. Hence, the results could seem enigmatic.

No back-siphoning accidents were reported, but in the personal interviews, a spill of an undetermined amount of Lasso (alachlor) reportedly occurred about ten feet from the well head in 1986. The on-site inspections of the site also indicate the presence of an abandoned well (80 feet deep, static water level about 50 feet) within ten feet of the present well. It seems likely that the spill and the nearby abandoned well may have caused the contamination of this deep well. Because of the great difference in depth of the wells, this is still problematic. However, in this area, the regional bedrock aquifer is quite shallow (ranging from about 25 to 70 feet in depth) and would be tapped both by the abandoned well and the deep well. In such a setting it is quite possible that the current water-supply well, while drilled deeply, may not be cased deeply; it may only be cased into the top of the bedrock.

Site C. Atrazine and an environmental metabolite, deethyl-atrazine were detected at concentrations of 3.4 and 1.2 ug/L, respectively. The combined equivalent concentration expressed as atrazine is 4.8 ug/L. The sample was collected in August, 1988, from a hydrant near the well at SWRL site C.

The 60 foot deep well was constructed in 1918. The 18-inch diameter well is reportedly cased to a depth of about 60 feet also. The well has an unsealed frost pit, but the well head is reportedly sealed. Field observations suggest it was possible for shallow groundwater seepage to enter the well.

The well is no longer used as the primary drinking water source since the farm is served by a rural water supply. The well-water also contained high levels of total coliform bacteria (16+ MPN), 19 mg/L NO_3-N and an additional pesticide, metolachlor, was detected at 0.3 ug/L.

Good chemical management practices are used on the farm. Pesticides are mixed and equipment is rinsed at a hydrant 300 feet from the well. No spills or back-siphoning accidents were reported. Excess pesticide formulations are sprayed on the field with another pass and/or sprayed on the road while returning home. All empty pesticide containers are rinsed and then burned in the field. Banvel (dicamba), Dual (metolachlor), Roundup (glyphosate), and Salute (trifluralin and metribuzin) are used on the farm.

There are no apparent on-site factors to cause these pesticide detections. Leaching of pesticides from normal agricultural applications appears the likely source of the residues. The shallow depth of the well, age of casing, and likelihood of the well taking water-table seepage contribute to the well's susceptibility.

Site D. Atrazine metabolites were detected at concentrations of 3.5 ug/L (deisopropyl atrazine) and 2.8 ug/L (deethyl atrazine). The combined atrazine equivalent concentration is 7.6 ug/L. The detections occurred in March, 1989, in a sample from a hydrant at the water-system storage tank at SWRL site D.

This farm site does not have a drilled well; instead, the flow a natural spring was piped or tiled to a 2,000 gallon, concrete storage tank. From the storage tank the water is pumped into the distribution system. The contributing basin for the spring is not known, but is likely part of a relatively shallow flow system in this area. The water showed no total coliform, but the nitrate-N concentration was high at 12 mg/L.

Several agricultural chemicals, including Conquest (atrazine and cyanazine), Treflan (trifluralin), Banvel (dicamba), 2,4-D, and Preview (chlorimuron ethyl and metribuzin) are used and stored on this property. No spills or back-siphoning accidents were reported; all pesticides are mixed, and application equipment is rinsed at a hydrant approximately 150 feet from the spring and storage tank. Excess formulation is discharged in the field with an extra pass.

There are no apparent on-site causes. Leaching of pesticides from normal agricultural applications in the spring's contributing watershed appears the likely source of the residues. The likely shallow nature of the spring water supply contributes to the system's susceptibility.

Site E. Trifluralin was detected at a concentration of 15 ug/L in June, 1988, from an outside hydrant at this site. The well is a shallow (40 feet deep) seepage well, constructed in 1920. It has a brick or tile lining, with wood framing around the ground level portion of the well. It has been modified from its original condition and uses a submersible pump with pitless adapter, and the well head has been grouted and sealed.

The farm owner/operator reported in the interview during the site investigation that a back-siphoning accident involving 40 gallons of a trifluralin mixture occurred on June 1, 1980. Following the back-siphoning accident, the well was pumped continuously for several days and then temporarily abandoned for household use. Livestock watering continued from this well. In 1985, the well was again put into use for drinking water, with a cartridge carbon filter installed on the kitchen tap.

The well-water was resampled in July, 1988. The second sample showed trifluralin at a concentration of 5.6 ug/L; a sample from the kitchen tap, following the carbon filter, contained no trifluralin. The initial sample was positive (16+ MPN) for coliform bacteria, but the resample was negative. The nitrate-N concentration was 5.4 mg/L.

Chemicals are stored about 200 feet from the well. Treflan (trifluralin), Sencor (metribuzin), AAtrex (atrazine), Lasso (alachlor), Basagran (bentazon), and Banvel (dicamba) are all used on the farm. All pesticides are mixed in the field from a hydrant near a field entrance over 200 feet from the well; this is where equipment rinsing is also done. Extra formulation is field applied or sprayed on the gravel road bed when returning home. Sometimes rinse water is drained into the road ditch. No spills have occurred near the well.

Even though the back-siphoning accident occurred 8 years prior to the SWRL sampling, it would still appear to be the most likely cause of the high trifluralin concentrations at this site. Trifluralin is considered to have a low potential for groundwater contamination because of its strong adsorption to soil and low solubility. However, these same properties might allow trifluralin to be very persistent following a back-siphoning incident.

Site F. Atrazine and an environmental metabolite, deethyl-atrazine, were detected at concentrations of 3.4 and 1.3 ug/L. The combined equivalent concentration expressed as atrazine is 4.9 ug/L. The sample was collected in January, 1989, from an outside tap on the house at this farm site. The nitrate-N concentration was high, 13 mg/L, and the water was positive for total coliform bacteria (16+ MPN).

The well is a large diameter bored well, but it is relatively deep, at 100 feet; it was built in 1953. It is lined with 36-inch diameter concrete pipe to a depth of 70 feet, and 30-inch pipe from 70-100 feet. It uses a pump jack (piston-pump) system and the frost pit around the well head is not sealed, though the upper blocks of the concrete pipe are reportedly grouted.

The farmer/operator uses a large variety of chemicals for the farm operation: Basagran (bentazon), Bladex (cyanazine), Dual (metolachlor), Banvel (dicamba), Eradicane (EPTC+R-25788), Lasso (alachlor), Poast (sethoxydim), Scepter (imazaquin), Treflan (trifluralin), 2,4-D, Commence (trifluralin & clomazone), and Broot (trimethecarb). All pesticides are mixed, and equipment rinsed in the field where they are applied; extra formulation and rinse are also field applied. The owner reported a spill of one gallon of dicamba on May 1, 1986, approximately 30 feet from the well head. No other accidents were reported.

There are no apparent on-site factors or accidents to cause these pesticide detections. The well is designed to tap shallow groundwater because there are few other options in this area. Even though it exhibits some better construction features than older seepage wells, it is still by design a seepage well and susceptible to contamination. Leaching of pesticides from normal agricultural applications appears the likely source of the residues.

Site G. Atrazine and deethyl-atrazine were detected at concentrations of 6.6 and 0.3 ug/L. The combined equivalent concentration expressed as atrazine is 7.0 ug/L. Sampling occurred in May, 1988, from a household faucet at this SWRL site. Two additional pesticides were detected, metolachlor at 0.3 ug/L and cyanazine at 0.7 ug/L. The nitrate-N concentration was low at 0.3 mg/L and no coliform bacteria were present.

The shallow (35 feet deep) well is an older large diameter seepage well, apparently lined with rock. The well-water is used for operating toilets and bathing, but it is not currently used for drinking water. Rural water district water is used for drinking and cooking.

Atrazine, Lasso (alachlor), and Lorox (linuron) are used on the farm, but all chemicals are custom applied. No chemicals are stored on the property. All remaining chemicals after application are taken off the property by the custom applicator. No spills or back-siphoning accidents were reported at this site.

There are no apparent on-site factors or accidents to cause these pesticide detections. The well is a shallow seepage well and is designed to tap shallow groundwater; it does appear to be poorly maintained. The seepage of shallow groundwater to the well appears the likely source of the residues.

Site H. Atrazine was detected at a concentration of 4.8 ug/L in October 1988, in a sample from a hydrant near the well at site H. A duplicate sample taken at this site, as part of the normal quality assurance program, showed an atrazine concentration of 4.7 ug/L. Nitrate-N concentration was moderate at 8.0 mg/L; the sample was positive for total coliform bacteria (16+ MPN).

This well is not currently used for drinking water; a rural water supply hookup is used. However, analysis of the rural water supply taken from a kitchen tap detected the following: 0.52 ug/L atrazine, 0.13 ug/L metolachlor, 0.34 ug/L cyanazine, 0.11 ug/L deethyl-atrazine, 0.2 mg/L NO_3-N, and no total coliform bacteria.

The well is a large diameter "seepage well" and its depth was not known; it is undoubtedly relative shallow. The well head itself was not sealed. While no pesticides are stored on the property, the well is used to mix or formulate farm chemicals from a hydrant within 15 feet of the well head. A spill of farm "chemicals" (unknown product) was reported to have occurred on June 1, 1981, about 20 feet from the well head.

The well is a shallow seepage well and is designed to tap shallow groundwater. While few details of the chemical spill are available, this is a possible contributor to the problems at this site. There are no unequivocal answers at this site.

Discussion

These sites exceeding HALs for pesticides occur throughout the state, with at least one occurrence in 5 of the 6 hydrogeologic regions. Only the Southwestern region did not have a site. These sites constitute only 1.2% of sites state-wide. The sites are, not unexpectedly, dominated by shallow wells or water sources (88% of sites); only one deep well was involved, and this was a point source case which could affect any depth of well. Two of the sites, 25%, are clearly point source cases, a spill or back-siphoning accident; the majority, 62.5%, appear to be nonpoint sources related to routine pesticide occurrences in shallow groundwater; 1 case, 12.5%, is equivocal.

Various detailed information was collected at each site, through observation of staff collecting the samples and through questionnaires and interviews with the occupants and the farm operators. Based on these site inventories, a few other observations can be afforded. Well owners were asked during on-site interviews if there had been any agricultural chemical spills within 15 feet of their wells; 5% of the private, rural drinking water wells in Iowa have experienced one or more agricultural chemical spills near the well. Other direct point sources of contamination include back siphoning accidents and tile line discharges into the wells; these occurred at <1% of the sites. These percentages are essentially identical to those reported from other detailed inventories in local areas of Iowa (Hallberg et al, 1984; Libra et al., 1984; Hallberg, 1989a,b). At the sites where pesticide spills within 15 feet of the well were reported, seven (7) of the wells contained one or more pesticides. When a specific product name was provided, only two of the seven wells containing pesticides actually contained the named product involved in the spill. At two of the three sites reporting a fertilizer spill, the concentration of nitrate observed in the wells exceeded health advisory levels. The correlation between reported back-siphoning accidents and observed pesticide contamination in the well during this study was better than this. Half of the sites' wells contained the product reportedly involved in the incident, and at relatively high concentrations. Two incidents occurred in 1975, with no traces of the named products (atrazine or alachlor) detected during the SWRL study.

Less than 1% of wells were located within 15 feet of chemical storage areas; none contained pesticide residues in their water. Similarly, none of these wells contained unacceptable concentrations of nitrate.

Further implications derived from the site inventories will be discussed in later sections of this report.

MAJOR ION CHEMISTRY

Major ion analyses of the SWRL groundwater samples were included to provide an understanding of the basic water chemistry (i.e., the "mineral" content) at each site. Tables 1 and 2 contained a listing of the ions measured and the methods used; further detail of methods are given in Hallberg, et al. (1990). Sodium (Na^+), potassium (K^+), magnesium (Mg^{++}), calcium (Ca^{++}), fluoride (F^-), chloride (Cl^-), and sulfate (SO_4^{--}) analyses were performed at EEL; nitrate-N (NO_3^-) analysis was done by UHL. Bicarbonate (HCO_3^-) was calculated from Total Inorganic Carbon (TIC) and pH (Johnson et al., in SWRL work plan; see Hallberg et al., 1990); pH was measured in the field at the time of sampling.

These ions occur as natural constituents of groundwater, however, as discussed for nitrate, society's activities on the land surface may contribute contamination that significantly increases concentrations above their natural background. Detectable contributions of certain ions (e.g., nitrate, chloride, and sulfate) have been related to fertilization (Hill, 1982), septic tank effluents (Daly and Daly, 1982) and urbanization (Eisen and Anderson, 1979). Hence, major ion analyses may also aid in understanding the source and transport of various contaminants. In addition, the relative concentrations of ions may provide indications that the groundwater is relatively old and still unaffected by anthropogenic contaminants.

Results of Major Ion Analyses

The presence of soluble carbonate minerals in virtually all of Iowa's near-surface geologic deposits and aquifers results in groundwater chemistry dominated by calcium and magnesium, bicarbonate, and in some cases, sulfate. This is reflected in the SWRL study data. Tables 46 and 47 contain descriptive statistics for each ion; Tables 47 provides descriptive statistics for the ions by hydrogeologic region.

Examination of the ion data shows that some SWRL samples were collected, by necessity or inadvertently, after water-softening treatment. Minimum calcium and magnesium, and therefore hardness, concentrations for several hydrogeologic regions are less than 0.1 mg/L, a virtual impossibility in Iowa's carbonate dominated terrain. Water-softeners remove calcium and magnesium via ion-exchange processes, replacing these ions with sodium.

Table 46. Ion concentration data for SWRL groundwater samples; state-wide and by well depth.

Ions	Mean	s.d.	Min.	Max.
	- - - -	as mg/L	- - - -	
State-wide:				
Na, sodium	33.8	43.2	<0.1	514
K, potassium	3.7	5.0	<0.1	71.0
Mg, magnesium	43.0	24.5	<0.1	177
Ca, calcium	102	59.6	<0.1	511
Hardness	432	239	<0.1	1789
F, fluoride	0.3	0.7	<0.1	7.0
Cl, chloride	19.1	30.9	<0.1	269
NO3-N, nitrate-N	6.3	12.6	<0.1	100
SO4, sulfate	132	245	<0.1	1938
HCO3, bicarbonate	371	119	7.9	823
Wells < 50 ft deep:				
Na, sodium	24.1	23.3	3.2	153
K, potassium	2.6	3.5	<0.1	31.8
Mg, magnesium	41.8	23.4	8.7	161
Ca, calcium	104	50.6	21.4	325
Hardness	431	211	96.3	1226
F, fluoride	0.2	0.6	<0.1	5.2
Cl, chloride	26.2	33.5	1.4	269
NO3-N, nitrate-N	11.7	16.3	<0.1	95.0
SO4, sulfate	88.2	120	2.7	732
HCO3, bicarbonate	326	124	7.9	650
Wells 50-99 ft deep:				
Na, sodium	29.3	29.6	<0.1	126
K, potassium	3.6	4.5	<0.1	32.7
Mg, magnesium	43.1	24.4	<0.1	176
Ca, calcium	105	56.8	<0.1	357
Hardness	440	236	<0.1	1615
F, fluoride	0.2	0.5	<0.1	4.0
Cl, chloride	22.8	35.0	<0.1	255
NO3-N, nitrate-N	8.9	16.3	<0.1	100
SO4, sulfate	88.9	147	<0.1	804
HCO3, bicarbonate	393	120	153	823
Wells ≥ 100 ft deep:				
Na, sodium	44.3	58.0	<0.1	514
K, potassium	4.6	6.2	<0.1	71.0
Mg, magnesium	44.3	26.7	<0.1	177
Ca, calcium	99.8	69.5	<0.1	511
Hardness	432	273	<0.1	1789
F, fluoride	0.4	0.9	<0.1	7.0
Cl, chloride	11.8	24.4	<0.1	171
NO3-N, nitrate-N	1.5	4.1	<0.1	37.0
SO4, sulfate	193	377	<0.1	1938
HCO3, bicarbonate	393	111	134.0	735

About 2% of the samples were clearly softened and potentially 4% were affected to some degree. Therefore, the mean calcium, magnesium, and hardness concentrations given are slightly lower than what is truly present in well-water supplies; the mean sodium concentrations are slightly higher. These affects will be further reviewed in subsequent reports when site inventory data are further linked with the water-quality database.

Regional Variations

Differences in mean concentrations of some ions between regions are apparent. Region 1 (NE) has the lowest mean concentrations for most ions while the highest means are generally in regions 3 (SC), 5 (NW), or 6 (NC). Ions with the greatest regional variation are sulfate and sodium. Mean sulfate concentrations vary from 32.1 mg/L in region 1 (NE) to 230 mg/L in region 5 (NW). Mean sodium concentrations range from 11.5 mg/L in region 1 (NE) to 49.5 mg/L in region 3 (SC). Moderate variation in hardness concentration is also apparent. In region 5 (NW), the mean hardness is 646 mg/L as $CaCO_3$, much higher than the other regions; mean hardness is lowest in region 1 (NE), at 346 mg/L.

Variability in mean sulfate concentration is largely related to regional differences in the mineral/chemical composition of geologic materials. For instance, gypsum and anhydrite (the mineral forms of calcium sulfate-$CaSO_4$) are present in variable amounts in Iowa bedrock and unlithified Quaternary deposits. These minerals are readily soluble and hence may strongly affect the chemistry of groundwater that comes in contact. Another geologic source of sulfate may be the oxidation of secondary sulfide minerals (i.e., pyrite - FeS_2), which are locally abundant in some Iowa rock units.

Like sulfate, the variability in sodium concentrations may be related to regional differences in the chemical composition of aquifer materials. Marine shales, for instance, may contain relatively higher amounts of sodium. However, mean concentrations may also be influenced by relatively high concentrations at just a few sample sites. As noted, some of these high results are caused by samples collected after ion exchange softening systems and thus do not accurately reflect the chemistry of the

Table 47. Ion concentration data for SWRL groundwater samples; by hydrogeologic regions.

Ions	Mean	s.d.	Min.	Max.	Ions	Mean	s.d.	Min.	Max.
	- - - - as mg/L - - - -					- - - - as mg/L - - - -			
Northeastern (1):					Eastern (2):				
Na, sodium	11.5	12.7	2.1	93.6	Na, sodium	34.6	42.5	<0.1	304
K, potassium	1.9	2.4	<0.1	18.9	K, potassium	3.3	4.0	<0.1	32.7
Mg, magnesium	41.8	14.5	<0.1	105	Mg, magnesium	39.9	25.9	<0.1	177
Ca, calcium	77.1	25.0	<0.1	182	Ca, calcium	89.6	53.8	<0.1	343
Hardness	346	113	<0.1	885	Hardness	388	234	<0.1	1455
F, fluoride	0.2	0.7	<0.1	5.7	F, fluoride	0.3	0.6	<0.1	4.6
Cl, chloride	13.4	20.6	<0.1	162	Cl, chloride	14.4	27.9	<0.1	246
NO3-N, nitrate-N	3.6	5.7	<0.1	37.0	NO3-N, nitrate-N	3.6	7.7	<0.1	50.0
SO4, sulfate	32.1	37.5	<0.1	284	SO4, sulfate	134	272	<0.1	1686
HCO3, bicarbonate	368	73.3	223	556	HCO3, bicarbonate	349	111	46.7	735
South-Central (3):					Southwestern (4)				
Na, sodium	49.9	65.6	<0.1	514	Na, sodium	25.1	28.8	6.3	134
K, potassium	3.5	7.6	<0.1	71.0	K, potassium	3.4	3.5	<0.1	15.3
Mg, magnesium	36.9	19.9	<0.1	132	Mg, magnesium	36.9	18.2	8.4	99.5
Ca, calcium	96.0	56.2	<0.1	456	Ca, calcium	108	56.9	5.3	415
Hardness	392	215	<0.1	1683	Hardness	421	211	47.8	1447
F, fluoride	4.0	0.9	<0.1	7.0	F, fluoride	0.2	0.5	<0.1	4.8
Cl, chloride	30.5	34.5	0.5	168	Cl, chloride	25.4	38.1	<0.1	269
NO3-N, nitrate-N	9.6	15.5	<0.1	87.0	NO3-N, nitrate-N	10.0	15.0	<0.1	100
SO4, sulfate	160	283	<0.1	1938	SO4, sulfate	74.3	163	3.8	1257
HCO3, bicarbonate	348	137	96.2	717	HCO3, bicarbonate	338	143	7.9	811
Northwestern (5):					North-Central (6):				
Na, sodium	28.2	27.0	6.9	121	Na, sodium	44.1	36.3	<0.1	159
K, potassium	4.9	5.8	<0.1	31.8	K, potassium	5.9	4.3	<0.1	27.0
Mg, magnesium	67.6	32.6	29.1	176	Mg, magnesium	50.7	23.2	1.1	135
Ca, calcium	147	66.7	59.8	357	Ca, calcium	127	71.6	<0.1	511
Hardness	646	280	290	1615	Hardness	527	267	4.5	1789
F, fluoride	0.3	1.0	<0.1	5.2	F, fluoride	0.3	0.9	<0.1	5.4
Cl, chloride	24.0	37.3	1.2	255	Cl, chloride	12.0	21.9	<0.1	182
NO3-N, nitrate-N	13.9	20.1	<0.1	95.0	NO3-N, nitrate-N	2.6	9.8	<0.1	79.0
SO4, sulfate	230	274	10.7	1240	SO4, sulfate	188	244	0.2	1510
HCO3, bicarbonate	418	65.4	307	616	HCO3, bicarbonate	454	88.7	301	823

groundwater.

Depth Variations

Included in Table 46 are descriptive statistics for three well depth categories; 0-50 ft, 50-99 ft, and ≥100 feet. For the SWRL data, mean concentrations for all ions except chloride and nitrate, increase or remain fairly constant with depth. In general, deeper groundwater has spent more time in contact with various geologic materials, relative to shallower groundwater. The deeper groundwater has therefore had more time to dissolve soluble minerals and increase its ion concentrations. The decrease with depth in mean concentration of chloride and nitrate is suggestive of surficial sources. Agricultural applications are probably the largest contributor of nitrate and even chloride (from potassium chloride, KCl, used for potassium, K, fertilizer), though road-salt use and geologic materials augment some of the chloride input.

Sulfate and sodium show relatively large increases with depth. Mean sulfate concentrations increase from 88.2 mg/L to 193 mg/L. Mean sodium concentrations increase from 24.1 mg/L to 44.3 mg/L. While the mean sodium values may be influenced by results from softened samples, this trend is apparent in other water-quality data.

Ion Balances; Piper Diagrams

Figures 42 through 47 are Piper diagrams for each of the six hydrogeologic regions. These diagrams provide a graphic summary of the cation and anion composition and balance (Freeze and Cherry, 1979). The tables associated with each figure (Tables 48-53) summarize the ion data, calculated as their percentage of total cations or anions, based on concentrations in meq/L (milliequivalents/Liter). The diagrams illustrate that Iowa groundwaters are predominantly Ca, Mg, and HCO_3 type waters. In all regions except Northeastern Iowa (region 1), however, sulfate results were found to be appreciable in some samples. Samples plotting near the Na+K end-members are samples likely affected by softening.

The mean calcium (Ca) contribution remains fairly constant between regions, at approximately 50% of total cations. Mean sodium and magnesium percentages vary. Sodium (Na) ranges from 7% in region 1 (NE) to 22% in region 3 (SC). Magnesium (Mg) ranges from 31% in region 3 (SC) to 44% in region 1 (NE).

As noted, the anions are dominated by bicarbonate (HCO_3); mean values for bicarbonate range from 59% in region 5 (NW) to 84% in region 1 (NE). In region 1, 95% of the results were >60% HCO_3. Regions 3, 4, and 5 had the highest $Cl+NO_3$ results; 16% in regions 3 (SC) and 4 (SW), and 13% in region 5 (NW). Sulfate compromises 2% and 25% of the anions in regions 5 (NW) and 6 (NC).

Environmental Health Implications of Major Ion Concentrations

Fluoride (F), sulfate (SO_4), chloride (Cl), and nitrate (NO_3-N) are the ions for which drinking water standards have been set. These standards have been established because of concerns for either these ions effects on human health and/or the aesthetic quality of water. The nitrate results were summarized in a previous section. The implications of SWRL results for fluoride, sulfate, and chloride are discussed below.

Fluoride

The presence of fluoride in drinking water at about 1-2 mg/L has repeatedly been shown to be beneficial. At these concentrations, fluoride decreases the incidence rate of dental cavities in children. To date there is no evidence that fluoride in drinking water is associated with cancer or birth defects (Corbin, 1989).

However, the United States Environmental Protection Agency (EPA) has established a National Secondary Drinking Water Regulation for fluoride at 2 mg/L to protect against objectionable dental fluorosis (tooth mottling and staining) and a Maximum Contaminant Level (MCL) of 4 mg/L to protect against skeletal, as well as dental fluorosis (National Research Council, 1980).

The proportion of wells with elevated fluoride concentrations is low; about 2.5% of rural wells exceed 2.0 mg/L, and 1.3% exceed 4.0 mg/L. Table 54 summarizes the fluoride data in three concentration ranges because of the general uncertainty regarding health consequence

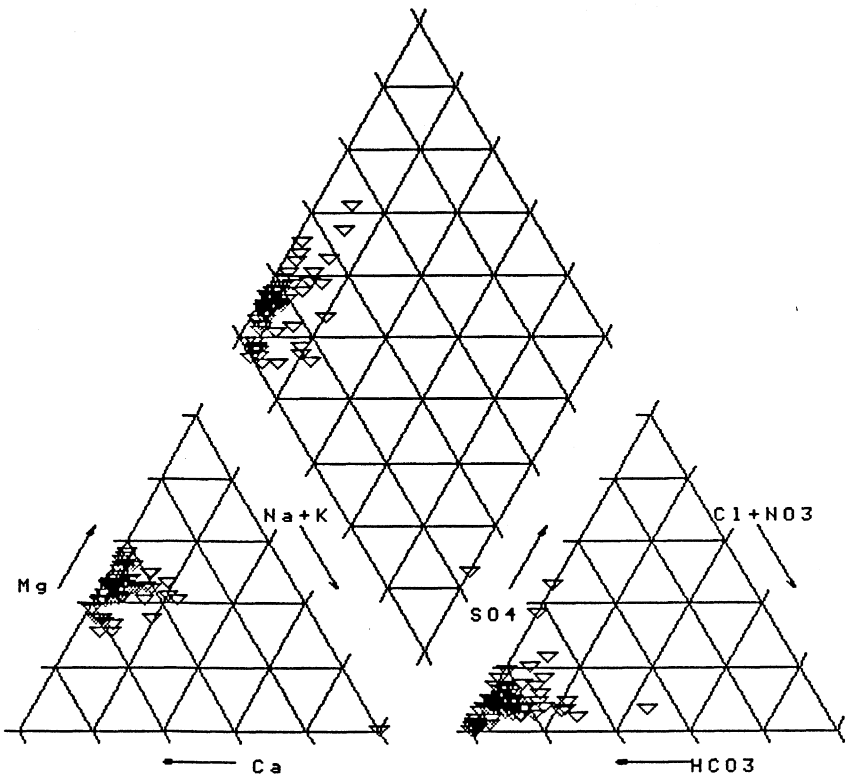


Figure 42. Piper diagram illustrating the groundwater-chemistry, ion-balance for the Northeastern Hydrogeologic Region (Region 1). (Axes of the diagram are scaled from 0 to 100%; with 100% of the ion located at its labeled apex.)

Table 48. Summary of ion concentrations in groundwater from Hydrogeologic Region 1, as meq/L.

Ions	Mean	s.d.	Min.	Max.	Mean +2 s.d.	Mean -2 s.d.
			%, Meq/L			
Na + K sodium plus potassium	7%	11%	1%	100%	0%	29%
Mg magnesium	44%	7%	0%	56%	29%	58%
Ca calcium	49%	8%	0%	63%	32%	66%
Cl + NO3-N chloride plus nitrate-N	7%	8%	0%	45%	0%	23%
SO4 sulfate	9%	8%	0%	46%	0%	24%
HCO3 bicarbonate	84%	11%	48%	100%	63%	100%

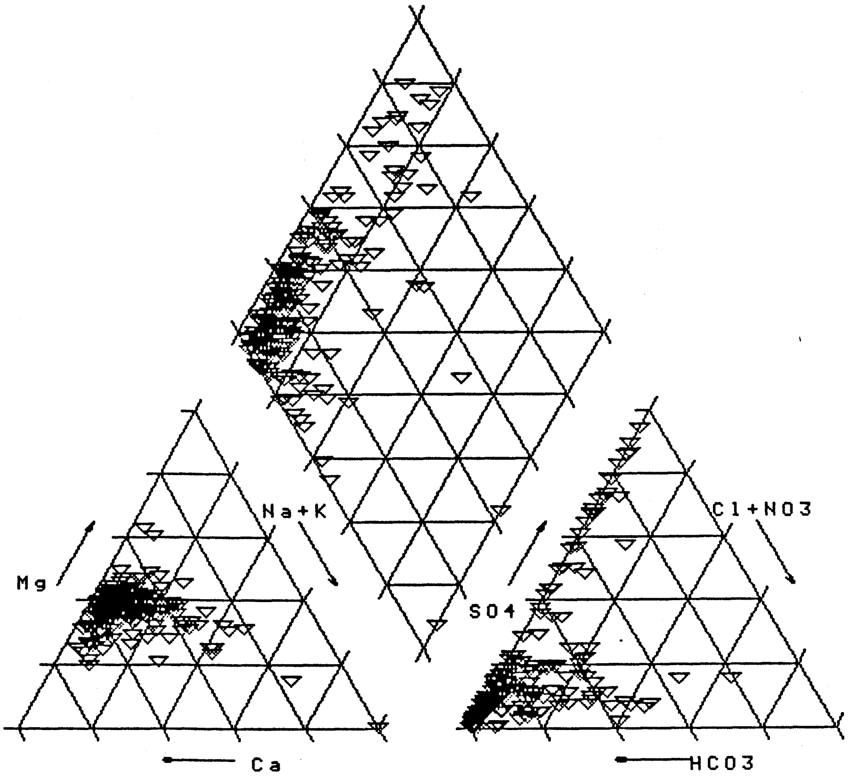


Figure 43. Piper diagram illustrating the groundwater-chemistry, ion-balance for the Eastern Hydrogeologic Region (Region 2). (Axes of the diagram are scaled from 0 to 100%; with 100% of the ion located at its labeled apex.)

Table 49. Summary of ion concentrations in groundwater from Hydrogeologic Region 2, as meq/L.

Ions	Mean	s.d.	Min.	Max.	Mean +2 s.d.	Mean -2 s.d.
			%, Meq/L			
Na + K sodium plus potassium	16%	17%	2%	100%	0%	50%
Mg magnesium	35%	8%	0%	63%	18%	52%
Ca calcium	49%	13%	0%	71%	24%	75%
Cl + NO3-N chloride plus nitrate-N	8%	11%	0%	64%	0%	30%
SO4 sulfate	19%	23%	0%	94%	0%	64%
HCO3 bicarbonate	73%	24%	6%	100%	26%	100%

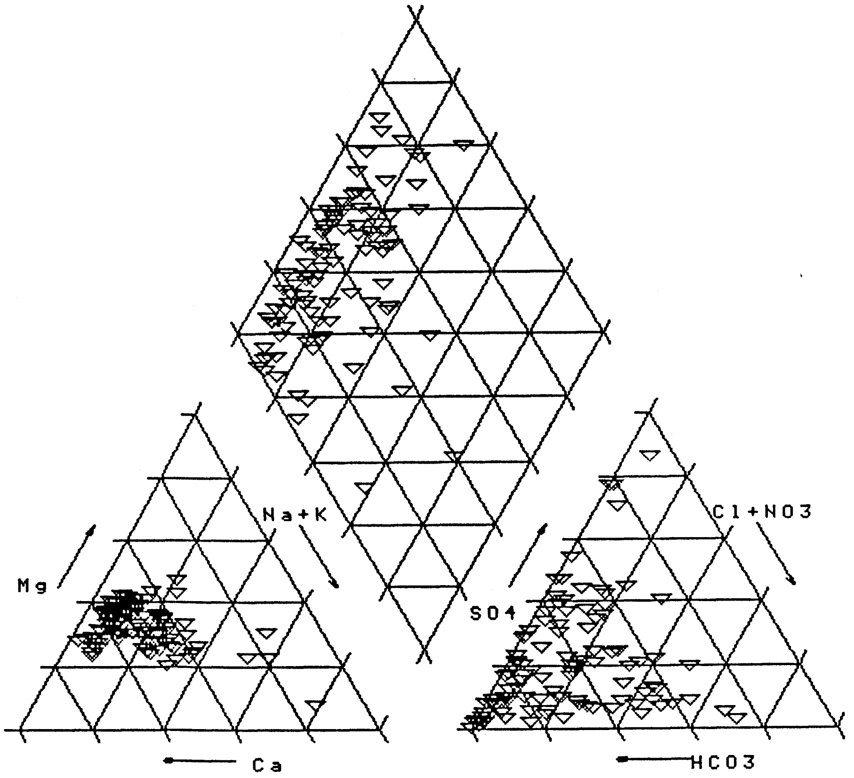


Figure 44. Piper diagram illustrating the groundwater-chemistry, ion-balance for the South-Central Hydrogeologic Region (Region 3). (Axes of the diagram are scaled from 0 to 100%; with 100% of the ion located at its labeled apex.)

Table 50. Summary of ion concentrations in groundwater from Hydrogeologic Region 3, as meq/L.

Ions	Mean	s.d.	Min.	Max.	Mean +2 s.d.	Mean -2 s.d.
			%, Meq/L			
Na + K sodium plus potassium	22%	20%	3%	100%	0%	62%
Mg magnesium	31%	9%	0%	48%	13%	48%
Ca calcium	47%	14%	0%	74%	19%	76%
Cl + NO3-N chloride plus nitrate-N	16%	16%	0%	72%	0%	48%
SO4 sulfate	22%	20%	0%	86%	0%	62%
HCO3 bicarbonate	62%	22%	7%	100%	18%	100%

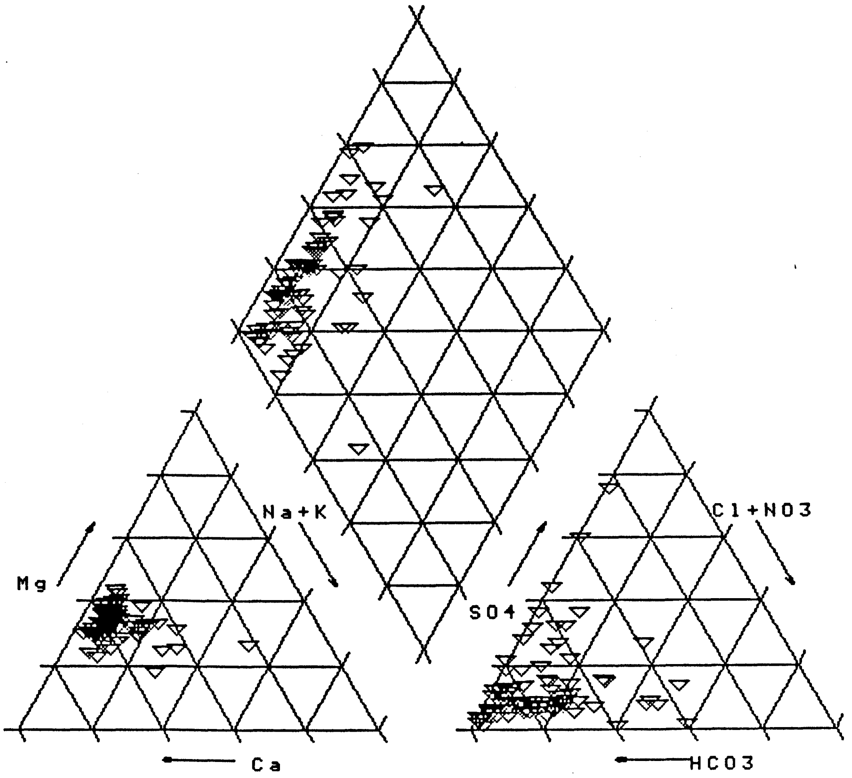


Figure 45. Piper diagram illustrating the groundwater-chemistry, ion-balance for the Southwestern Hydrogeologic Region (Region 4). (Axes of the diagram are scaled from 0 to 100%; with 100% of the ion located at its labeled apex.)

Table 51. Summary of ion concentrations in groundwater from Hydrogeologic Region 4, as meq/L.

Ions	Mean	s.d.	Min.	Max.	Mean +2 s.d.	Mean -2 s.d.
			%, Meq/L			
Na + K sodium plus potassium	12%	11%	3%	85%	0%	34%
Mg magnesium	32%	5%	11%	43%	22%	42%
Ca calcium	56%	9%	4%	69%	39%	74%
Cl + NO3-N chloride plus nitrate-N	16%	16%	0%	71%	0%	47%
SO4 sulfate	13%	13%	1%	75%	0%	40%
HCO3 bicarbonate	71%	19%	33%	97%	33%	100%

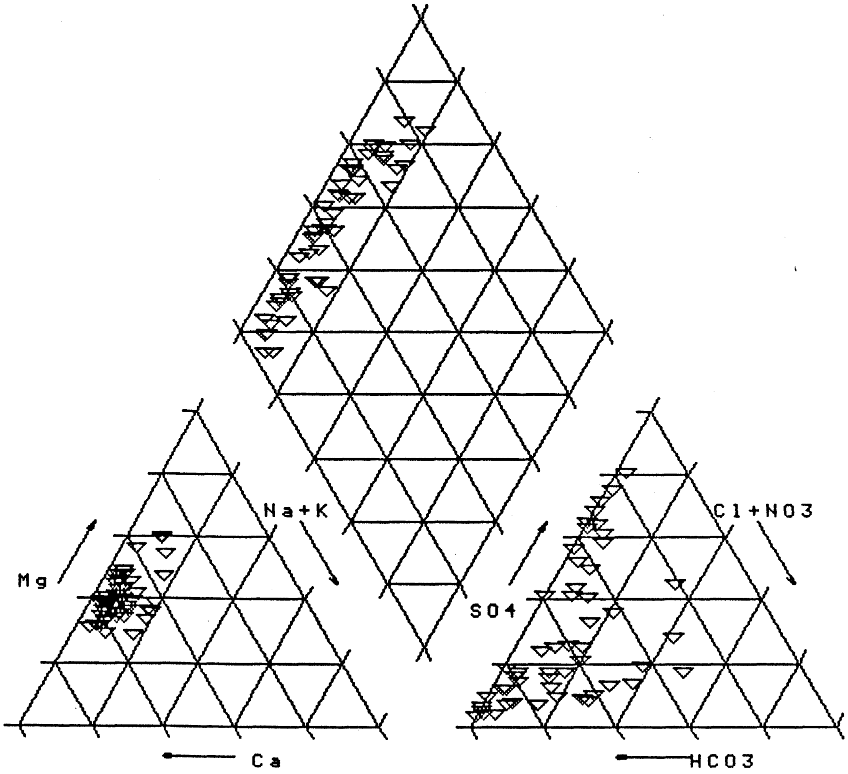


Figure 46. Piper diagram illustrating the groundwater-chemistry, ion-balance for the Northwestern Hydrogeologic Region (Region 5). (Axes of the diagram are scaled from 0 to 100%; with 100% of the ion located at its labeled apex.)

Table 52. Summary of ion concentrations in groundwater from Hydrogeologic Region 5, as meq/L.

Ions	Mean	s.d.	Min.	Max.	Mean +2 s.d.	Mean -2 s.d.
			%, Meq/L			
Na + K sodium plus potassium	8%	4%	3%	19%	0%	17%
Mg magnesium	40%	7%	29%	60%	25%	54%
Ca calcium	52%	8%	30%	65%	36%	68%
Cl + NO3-N chloride plus nitrate-N	13%	12%	0%	50%	0%	37%
SO4 sulfate	28%	22%	3%	80%	0%	73%
HCO3 bicarbonate	59%	22%	15%	97%	15%	100%

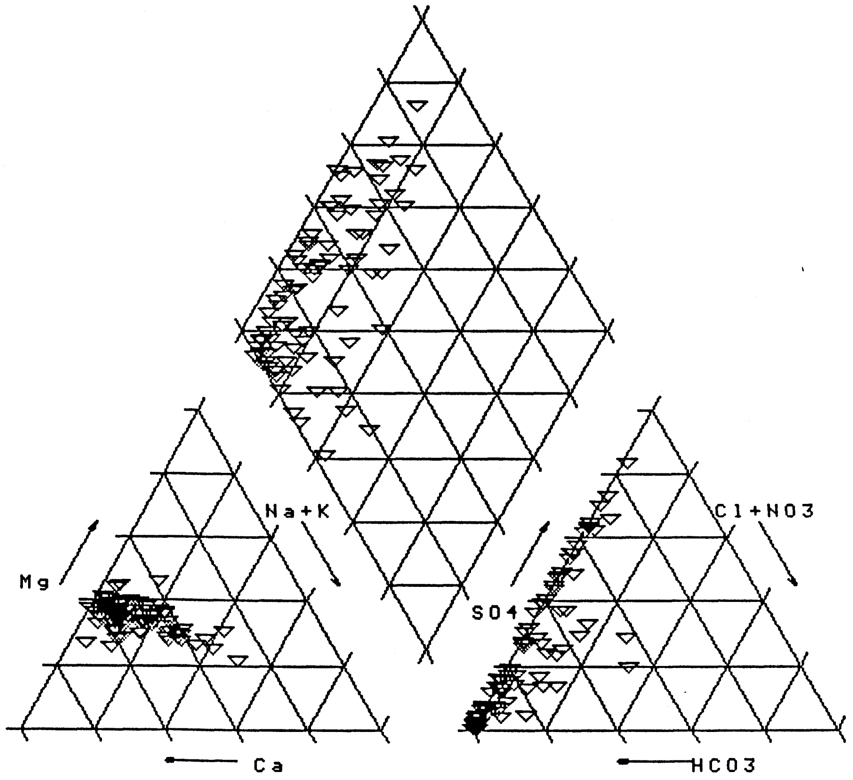


Figure 47. Piper diagram illustrating the groundwater-chemistry, ion-balance for the North-Central Hydrogeologic Region (Region 6). (Axes of the diagram are scaled from 0 to 100%; with 100% of the ion located at its labeled apex.)

Table 53. Summary of ion concentrations in groundwater from Hydrogeologic Region 6, as meq/L.

Ions	Mean	s.d.	Min.	Max.	Mean +2 s.d.	Mean -2 s.d.
			%, Meq/L			
Na + K sodium plus potassium	17%	16%	1%	98%	0%	48%
Mg magnesium	33%	7%	2%	49%	20%	47%
Ca calcium	50%	11%	0%	69%	28%	71%
Cl + NO3-N chloride plus nitrate-N	4%	6%	0%	33%	0%	15%
SO4 sulfate	25%	22%	0%	83%	0%	70%
HCO3 bicarbonate	71%	23%	16%	100%	26%	100%

interpretation significance for fluoride concentrations in the 2.1 to 4.0 mg/L range.

In rural areas, a fluoride supplement is often recommended for young children, because it is assumed that because they are not consuming fluoridated water and that they may need this for protection for their developing teeth. However, the drinking water should be analyzed before a supplement is used to avoid dental fluorosis. If the water already contains some natural fluoride any supplement that may be needed should be adjusted to protect against fluorosis.

Sulfate and Chloride

For sulfate and chloride the U.S. Public Health Service has established secondary drinking water standards of 250 mg/L. The taste threshold for sulfate in water is between 300-400 mg/L. Sulfate concentrations above this range can have a purgative, or laxative effect on humans (especially when occurring with Mg). Results of the SWRL study indicate that about 15.3% (at the 95% confidence level = 12.6-18.0%) of the rural private wells in Iowa exceed this advisory level for sulfate. For wells greater than 50 feet in depth about 18.8% percent of the wells have sulfate concentrations exceeding 250 mg/L (Table 55). Higher sulfate concentrations are typically found in deeper wells, because the sulfate occurs as a natural constituent. At a given depth, sulfate concentrations vary on a regional basis. The NW (5) and NC (6) regions have the greatest proportions of wells with >250 mg/L sulfate.

The chloride standard is based on the taste threshold for this ion, which varies between 210 and 310 mg/L. The proportion of the private rural wells that produce water with 250 mg/L or greater of chloride is only 0.3% (0 to 7% at 95% confidence interval).

Table 54. Summary of fluoride analyses from SWRL survey.

Concentration range, mg/L	Percentage of of wells in range	Cumulative %
	State-Wide	
< - 2.0	97.5%	97.5%
2.1 - 4.0	1.2	98.7
> 4.0	1.3	100.0
	Wells < 50 feet deep	
< - 2.0	98.8%	98.8%
2.1 - 4.0	0.0	98.8
> 4.0	1.2	100.0
	Wells ≥ 50 feet deep	
< - 2.0	96.9%	96.9
2.1 - 4.0	1.7	98.6
> 4.0	1.4	100.0
	Northeastern, Region 1	
< - 2.0	98.9%	98.9%
2.1 - 4.0	0.0	98.9
> 4.0	1.1	100.0
	Eastern, Region 2	
< - 2.0	97.2%	97.2
2.1 - 4.0	2.3	99.5
> 4.0	0.5	100.0
	South-Central, Region 3	
< - 2.0	97.5%	97.5
2.1 - 4.0	0.8	98.3
> 4.0	1.7	100.0
	Southwestern, Region 4	
< - 2.0	99.0%	99.0
2.1 - 4.0	0.0	99.0
> 4.0	1.0	100.0
	Northwestern, Region 5	
< - 2.0	96.4%	96.4
2.1 - 4.0	0.0	96.4
> 4.0	3.6	100.0
	North-central, Region 6	
< - 2.0	96.0%	96.0
2.1 - 4.0	2.0	98.0
> 4.0	2.0	100.0

ORGANIC SCREENING TESTS

A secondary objective of the SWRL survey was to evaluate the potential use of total organic carbon (TOC) and total organic halide (TOX) analyses as screening tools for a variety of groundwater contaminants. The measurement of TOC is a useful tool for many applications. Most groundwater contains low concentrations (about 1 mg/L TOC) of a diversity of natural organic compounds (EPA, 1985). To quantify these compounds by chromatographic or mass spectrometric techniques isn't typically feasible because of the number of compounds that are present in groundwater and the diversity of their physical properties. In most cases, the concentration of TOC in groundwater is far below

Table 55. Summary of sulfate analyses from SWRL survey.

Concentration range, mg/L	Percentage of of wells in range	Cumulative %
	State-Wide	
≤ 250	84.7%	84.7%
> 250	15.3	100.0
	Wells < 50 feet deep	
≤ 250	92.7%	92.7%
> 250	7.3	100.0
	Wells ≥ 50 feet deep	
≤ 250	81.2%	81.2
> 250	18.8	100.0
	Northeastern, Region 1	
≤ 250	98.9%	98.9%
> 250	1.1	100.0
	Eastern, Region 2	
≤ 250	84.9%	84.9
> 250	15.1	100.0
	South-Central, Region 3	
≤ 250	84.8%	84.8
> 250	15.2	100.0
	Southwestern, Region 4	
≤ 250	93.9%	93.9
> 250	6.1	100.0
	Northwestern, Region 5	
≤ 250	65.5%	65.5
> 250	34.5	100.0
	North-central, Region 6	
≤ 250	72.7%	72.7
> 250	27.3	100.0

those that can be quantified accurately by the more traditional Biological Oxygen Demand (BOD) and Chemical Oxygen Demand (COD) tests. The TOC test determines the total mass of dissolved organic carbon that is present in aqueous samples by oxidizing the compounds to CO_2.

TOC analysis of groundwater has been used in determining the fate and transport of organic leachates from solid and hazardous waste landfills (Levine and Kroemer, 1988). Also, organic macromolecules or colloids have been shown to increase the mobility of some organic contaminants (Enfield, 1985; Enfield and Bengtsson, 1988). Additionally, TOC is a source of carbon that bacteria require for denitrification, sulfate reduction, and other biologically mediated processes that can occur in the groundwater environment. Most pesticides are synthetic organic compounds, as are many other anthropogenic contaminants (i.e., some solvents and petroleum products). However, the nature of TOC, its minimum detection level and natural background concentrations, verses typical pesticide concentrations, prohibit its singular use as a pesticide screen.

Measurement of total organic halides (TOX) has become a widely used parameter in environmental analysis. TOX is designed to provide screening information on a wide variety of sample types, especially water, waste oils, solvents, and sediment samples. TOX measures the total organic "halides" -- organic complexes with chlorine, bromine, and iodine, but does not detect organics containing fluoride. Concentrations are reported in ug/L as Cl, with method detection limits of 2 ug/L as Cl (Hughes, 1989). The combination of TOC and TOX analyses could possibly provide a screening tool for a variety of contaminants that have been detected in other studies in Iowa such as pesticides, petroleum compounds, and solvents.

All pesticides analyzed in the SWRL study are organic compounds, however, not all of them contain halides. Table 56 lists the weight fraction of carbon and halide, for the pesticide analytes included in the SWRL study. If the weight fraction of halide of an individual pesticide equals 0.30, and if that pesticide were present at 10 ug/L, the theoretical TOX concentration (assuming no other organics were present) would equal 3 ug/L.

A recent study in Oregon produced mixed results in an attempt to use TOX as a screening tool for pesticides (Pettit, 1988). However, the limited scope of the study and analytical difficulties may have contributed to its inability to find a significant correlation between TOX and pesticide detections.

TOC and TOX Results

TOC values from SWRL samples ranged from below the detection limit of 0.1 mg/L to 33.0 mg/L (see Table 57). Approximately 89% of the 686 observations were below 4 mg/L, and nearly 99% were below 10 mg/L. The state-wide mean

Table 56. Weight fraction of carbon and halides in pesticides analyzed in the SWRL study samples.

Pesticide	Carbon	Halide
alachlor	0.66	0.14
hydroxy alachlor	0.71	0
atrazine	0.45	0.16
de-ethyl atrazine	0.54	0.26
de-isopropyl atrazine	0.51	0.30
carbofuran	0.65	0
3-hydroxy carbofuran	0.67	0
3-keto carbofuran	0.67	0
cyanazine	0.45	0.15
dacthal	0.36	0.43
2,4-D	0.43	0.32
metolachlor	0.64	0.13
metribuzin	0.38	0
pendamethalin	0.56	0
picloram	0.30	0.44
propachlor	0.63	0.17
trifluralin	0.47	0

Table 57. Summary of detections of indicator organic analyses, TOC and TOX.

Analyte	Mean	s.d.	Median	Max
	----	concentrations		----
	State-wide			
TOC, mg/L MDL=0.1 mg/L	2.3	2.3	1.9	33.0
TOX, ug/L MDL=2.0 ug/L	17.4	65.6	6.5	1136.5
	Wells < 50 feet deep			
TOC, mg/L	2.6	2.9	2.0	31.4
TOX, ug/L	30.1	93.1	6.9	1136.5
	Wells ≥ 50 feet deep			
TOC, mg/L	2.2	2.2	1.8	33.0
TOX, ug/L	12.4	24.2	6.7	219.4

TOC concentration was 2.3 mg/L.

Mean TOC concentrations were calculated for each hydrogeologic region. TOC concentrations below detection limits were set equal to zero for calculating the regional means. The Eastern region (Region 2) had the lowest average TOC value of 2.0 mg/L, while North-Central Iowa (Region 6) had the highest at 3.0 mg/L. The variability of TOC between hydrogeologic regions, was tested with a one-way analysis of variance (ANOVA); there were no significant differences among the six regions.

TOX analyses were performed on samples from 339 of the 686 primary SWRL sites. TOX concentrations ranged from less than the detection limit to 1,136 ug/L as Cl. The state-wide mean TOX concentration was 17.4 ug/L as Cl; 63% of the analyses were below 10 ug/L and 94% were below 30 ug/L as Cl.

TOX displays regional trends similar to TOC. The Eastern Region (Region 2) had the lowest average TOX concentration at 5.6 ug/L as Cl. The North-Central Region (Region 6) would have had the highest mean TOX (19.5 ug/L Cl) except for the impact of one very high value (1,136 ug/L) in South-Central Iowa (Region 3). Initially this was considered a possible error, but in comparing it to other data, this site also produced an above average TOC concentration and had two relatively high concentration pesticide detections which supported the high TOX concentration as being correct. Regional TOX averages were also analyzed using the ANOVA test. A log transformation of the data was used to achieve a normal distribution and test statistics showed that the regional differences in TOX concentrations were statistically significant, though largely because of the very high value noted above.

Discussion of TOC and TOX Results

The contribution of detected pesticides to TOC and TOX were calculated and compared to the measured concentrations. The contribution of organic carbon from pesticides to TOC measurements was quite low. Average TOC results were, in general, three to four orders of magnitude above "typical" pesticide concentrations. The greatest portion of a sample's TOC that could be attributed to detected pesticides was 0.3%.

The same determinations using calculated TOX and observed TOX produced more interesting results. In 15% of the cases, the percent of TOX calculated from halogen containing pesticide concentrations exceeded 10%. In one case the percentage of halide attributed to the occurrence of detected pesticides reached as high as 35% of the observed TOX concentration. Of the 34 samples that contained halogenated pesticides and detectable TOX concentrations, 19 (56%) of the measured TOX values could be partially explained by the presence of a pesticide (Hughes, 1989). However, there was not a significant correlation between TOC, TOX and pesticide detections in the SWRL study.

This study only related the contribution of the pesticide analytes measured in SWRL. There may be various other pesticides, their metabolites, or other organic compounds (i.e., pesticide inert ingredients, solvents or carriers, industrial or household compounds, or other petroleum distillates) that could be contributing to both the TOC and TOX values. Also, because of limited use of the tests, the natural background concentrations are not yet well known.

The higher values of TOX that should be indicative of contamination were associated with measurable pesticide residues. However, the overall TOC and TOX analyses did not provide a satisfactory indicator of pesticide contamination.

Specific analysis for other synthetic and volatile organic compounds was beyond the feasibility of SWRL, but the TOX and TOC tests could have provided some insight to well contamination by many of these petroleum based compounds. However, the human senses are very sensitive to the volatile organic compounds that occur in products such as gasoline, diesel fuel, heating oils, and many solvents. These compounds are often detected by taste and/or smell at concentrations lower than lab analyses can confirm. There were no reports of well contamination from fuel compounds in the SWRL survey. Though such cases occur on private property (DNR routinely handles a small number of cases per year in rural areas) they were evidently below the threshold of sampling used in SWRL. Most petroleum leak and spill problems occur in more urbanized areas. There were no obvious incidents of water contamination related to landfills or other waste sites detected in the SWRL review either.

ADDITIVE WATER-QUALITY EFFECTS

The preceding sections presented the impacts of various water quality parameters individually. At some sites particular contaminants are present alone but at other sites they occur in combination. To understand the total impact on rural well-water supplies the additive impacts of the major contaminants must also be assessed. Table 58 summarizes the occurrence of three contaminants of primary concern, nitrate-N >10 mg/L, pesticides, and total coliform bacteria, individually and in various combinations.

Individually, NO_3-N >10 mg/L occurred alone in about 4% of the water supplies state-wide; pesticides were present alone in about 5%. Total coliform positives occurred alone at 27% of the sites, which is over 60% of the total coliform positives. In a cumulative sense, these three contaminants, individually or in combination, were detected in nearly 55% of rural private water supplies. If only fecal coliform occurrences are used, this reduces to about 30% of private well-water supplies state-wide. (No sites where fecal coliforms were present had pesticide detections, and only about 2.5% were positive for both fecal coliforms and had NO_3-N >10 mg/L.)

The combination data also illustrate that total coliforms are a poor predictor of these other forms of contamination. As noted above, 60% of all sites with total coliforms did not exhibit pesticide detections and/or NO_3-N >10 mg/L. This illustrates that if a prediction where to be made based on the presence of total coliform, the probability is better that these other contaminants would not occur in the water supply, than that they would occur. There is, however, an association among these contaminants. For water supplies that have NO_3-N >10 mg/L and pesticides, about 63% also had total coliforms. Similarly, for any site with NO_3-N >10 mg/L, about 70% also had total coliforms. Conversely, however, 70% of supplies that were total coliform positive did *not* exhibit NO_3-N >10 mg/L. For pesticides, this association is more pronounced, with sites having pesticide detections nearly split at 50-50 between those with and without total coliforms. Conversely, 84% of total coliform positive supplies did not have any pesticides detected. In short, high NO_3-N concentrations are a moderately good predictor of the presence of total coliforms, but total coliforms are a very poor predictor of NO_3-N or pesticides.

Table 58. Proportions of wells affected by combinations of nitrate-N >10 mg/L, any pesticide detections, and/or total coliform bacteria positives.

Water-quality parameters	State-wide % of SWRL %	Cumulative nitrate-N >10 mg/L %	Cumulative any pesticide detection %	Cumulative tot. coliform positive %
nitrate-N >10 mg/L, ALONE;	3.7%	3.7%		
pesticide(s) detection, ALONE;	5.2%		5.2%	
total coliform bacteria, ALONE;	26.8%			26.8%
nitrate-N >10 AND any pesticide; (but no total coliforms)	1.4%	5.1%	6.5%	
nitrate-N >10 AND tot coliform; (but no pesticide detections)	10.7%	15.8%		37.5%
pesticide det. AND tot coliform; (but no nitrate >10 mg/L)	4.6%		11.1%	42.1%
N >10 & any pest. & tot coliform;	2.5%	18.3%	13.6%	44.6%
Cumulative total; % wells with nitrate-N >10 mg/L, &/or any pesticide detected, &/or total coliform positive;	54.8%			
Cumulative total; as above, BUT with fecal coliform instead of total coliforms;	30.5%			

	State-wide % of SWRL	Relative %	
nitrate-N >10 &/or any pesticide; (but no total coliforms)	10.3%	36.6%	
nitrate-N >10 &/or any pest. & total coliform;	17.8%	63.4%	39.9%
total coliform bacteria, ALONE;	26.8%		60.1%
Any nitrate-N >10 mg/L W/OUT total coliform;	5.1%	27.8%	
Any nitrate-N >10 mg/L with total coliform;	13.2%	72.2%	29.7%
Other total coliform positives;	31.3%		70.3%
Any pesticide W/OUT total coliform;	6.5%	48.1%	
Any pesticide with total coliform;	7.1%	51.9%	15.8%
Other total coliform positives;	37.5%		84.2%

RURAL WATER SYSTEMS

Ten of the SWRL sites sampled relied on rural water systems (RWS). One of these sites was a 10% repeat site, and one was located in a quarterly repeat county. These sites were included in SWRL because all the sites contacted, within a reasonable distance of the primary target site, also used the RWS for their primary drinking water source. RWS sites were located in Cherokee, Dallas, Jasper (10% repeat site), Lyon, O'Brien, Polk, Sioux (quarterly site), and Wapello counties. For these sites, water samples were collected from both the most often used well on site, and the RWS supply.

A one-time sampling of the 10 RWS sites (i.e., repeat samples are not included) detected herbicides in four of the supplies. Two contained pendamethalin at 0.05 ug/L (Sioux and Dallas counties); one contained atrazine at 0.14 ug/L (Polk County); and the fourth contained atrazine at 0.52 ug/L, deethyl-atrazine at 0.11 ug/L, cyanazine at 0.34 ug/L, and metolachlor at 0.13 ug/L (Wapello County). Repeat sampling verified the detection of pendamethalin at 0.05 ug/L. A maximum nitrate-N concentration of 5 mg/L was measured in the RWS samples; the median nitrate-N concentration was less than 1 mg/L. Coliform bacteria were not detected in these samples.

TEMPORAL SAMPLING; POTENTIAL FOR MONITORING

As described in the review of the study design, several steps were taken to assess potential significant temporal changes in hydrologic conditions and water quality during the course of the SWRL survey. Results from two of these components will be reviewed here: the systematically selected 10% repeat sites; and the hydrogeologic regions subset that was sampled quarterly. These will be referred to as the 10% and the quarterly sites. There are three primary issues of interest in the evaluation of these sites: 1. the nature of any seasonal, temporal changes, in relation to the entire sampling frame; 2. potential implications for the survey as a whole, related to such changes, and related to repeat sampling of sites; and 3. the potential of these subsets to be used as a manageable network for long-term monitoring.

As was briefly discussed in the statistical methods section, any temporal variations that occurred during the SWRL survey did not vary significantly in response among the regions. This was of primary concern because if some regions had a very different response to seasonal changes it could upset the intent of the survey to provide a state-wide baseline assessment. As noted, while the state was influenced by climatic and hydrologic aberrations, related to the drought, the entire state was affected.

TEMPORAL VARIATIONS

Some temporal variations are evident in the SWRL data (Fig. 48). In the midst of the drought, the fall and winter of 1988 and 1989 was a period of near normal precipitation, with some months slightly above normal. Other monitoring programs showed some hydrologic responses during this period. Figure 48 summarizes the monthly mean nitrate-N concentration and total coliform values for all the sample data (the primary set plus the quarterly and 10% repeat sites). While the means are variable from month to month some trends are present. As apparent, the mean of total coliform positives rises from the beginning of the survey, peaking during the late fall and winter, and then declines to the end of the survey period. Nitrate-N is more variable, but shows its greatest mean values in the late fall and winter as well, and declines rather sharply at the end of the survey period. Pesticide detections show a similar trend, with peak proportions of wells with detections in December and January, and the lowest period of detections in April through June of 1989.

As described, the SWRL sampling frame was scheduled for spatial coverage by quarter, not by month. Table 59 summarizes pertinent SWRL data by quarter. These general trends are still apparent, with peak occurrences in the third or fourth quarter, and the lowest occurrences in the last quarter. This may be related to well depth, although differences in well depth do not adequately explain all the variance. These trends are somewhat unusual; most temporal monitoring would show peaks in the spring and early summer months, and lower or average conditions in the winter (e.g., Hallberg and Hoyer, 1982; Hallberg et al., 1984; Hallberg,

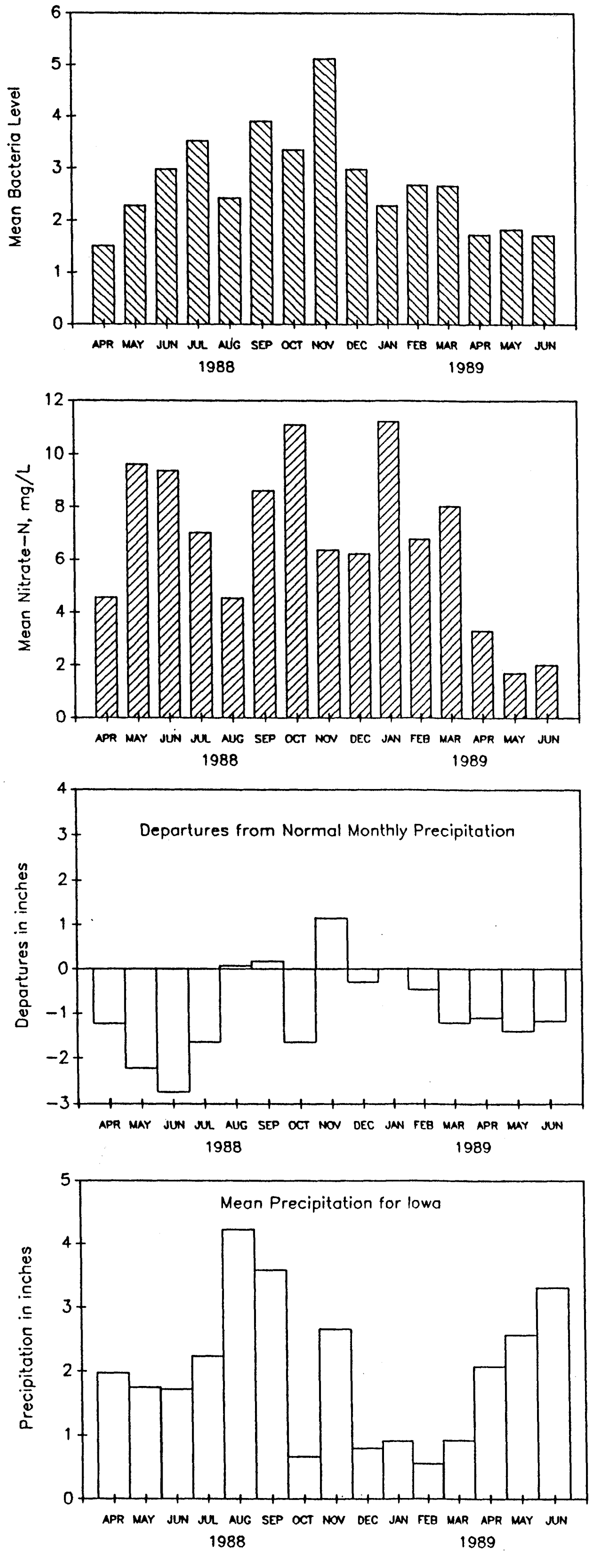


Figure 48. Bar graph summary of monthly precipitation, mean nitrate-N concentration (mg/L), and total coliform MPN (1=0 MPN; 2=2.2 MPN; 3=5.1 MPN; 4=9.2 MPN; 5=16 MPN; and 6=16+ MPN; 0=missing sample) for all SWRL data.

1989a; Libra et al., 1984; 1987). These trends partially coincide with the hydrologic changes; with somewhat greater precipitation in the late fall and winter, there is an increase, and the peak, of nitrate-N, pesticide detections, and total coliform positives. The most pronounced trend that is apparent in the data, however, is the very low value of all parameters in the April-June, 1989 period, normally a time of peak values. This seems to parallel the return of drought conditions in the late winter and spring of 1989.

Repeat Sampling Subsets

Table 60 summarizes selected water-quality data for the primary state-wide data set, the 10% repeat sites and the quarterly repeat sites. As noted, by design, the 10% repeat sites, sample-1, and the quarterly repeat sites, period-1, are included in the primary state-wide data set. When compared with the primary data set (which incorporates conditions over the entire time of sampling), the repeat sampling shows: 1. a decline in total coliform positives (no repeat sample group or period had a greater proportion of total coliform positives than the primary, state-wide data set); 2. NO_3-N shows relatively little variation; 3. pesticide detections, and atrazine detections, were significantly greater in some periods, but lower in others. Similar to the aggregate survey data (Table 58) the peak of pesticide occurrences was during the January to March, 1989 period, with the lowest detections occurring in the last quarter, April through June, 1989.

Implications From Temporal Sampling

The observations from the temporal sampling can also be used to assess an upper bound on the estimates of private-well contamination. Simply adding in the actual observations from the 10% repeats and the quarterly repeat samples to the primary data set, and then estimating state-wide proportions, provides a

Table 59. Selected SWRL water-quality parameters summarized from state-wide data, by quarterly sampling period. Total coliform MPN converted to numerical values for averaging: 1=0 MPN; 2=2.2 MPN; 3=5.1 MPN; 4=9.2 MPN; 5=16 MPN; and 6=16+ MPN; 0=missing sample.

Quarter	nitrate-N mean mg/L	nitrate-N % >10 mg/L	atrazine detections %	total coliform mean MPN	wells <50 ft %
1. 4-6/88	8	18%	5%	2.1	31%
2. 7-9/88	6	19%	8%	3.2	25%
3. 10-12/88	9	18%	13%	3.4	31%
4. 1-3/89	10	26%	14%	2.4	33%
5. 4-6/89	3	8%	2%	1.8	15%

measure of the proportion of wells state-wide that have particular detections over the course of one year. These data would increase the state-wide proportions to: 1. the % private wells >10 mg/L NO_3-N = 19% (compared to 18.3%); 2. the % wells with any pesticide detections = 18% (compared to 13.6%); and 3. the % wells with total atrazine detections (atrazine and/or metabolites) = 9% (compared to 8%).

Extrapolating from the temporal samples to the complete study data set provides an estimate of the upper limit of wells likely to exhibit detections, at sometime over the course of 1 year (but not necessarily continuously), based on the climatic conditions during SWRL. This approach estimates these proportions: 1. the % wells >10 mg/L NO_3-N = approximately 21%; 2. the % wells with any pesticide detections = approximately 30%; 3. the % wells with atrazine detections = approximately 15%.

POTENTIAL MONITORING NETWORK

The data in Table 60 also provide a comparison of the subsets to assess their potential for use as a manageable monitoring network. Both the 10% repeats, sample-1, and the quarterly repeats, period-1 subsets compare very favorably with the state-wide SWRL sample. Both subsets generally exhibit mean values close to the state-wide mean. As would be expected, because it was systematically selected, the 10% repeat subset is closest to the state-wide data; the quarterly sites do have a few mean values outside of the 95% confidence interval of the state-wide data. For either subset, because they are smaller data sets, the standard deviations are smaller than the state-wide data, but the 95% confidence intervals are much greater. In that context, the best subset may be a combination of the 10% sites and the quarterly sites; the combined data on Table 60 generally exhibit means or percentages very close to the state-wide data, and because of the larger sample size, the confidence intervals are reduced.

As Table 60 illustrates, the 10% sites provide a very consistent representation of the state-wide data, including proportionately representative detections of pesticides down to about a 1% occurrence interval. Table 61 provides further insight into the structure of these subsets, reviewing the proportions of wells within depth classes and related water-quality parameters. Overall, the 10% subset again appears more suitable, more closely reflecting the structure of the entire state-wide data.

The systematically selected 10% sites can provide a very good monitoring base for updating the SWRL survey and assessing trends over time. Using both the 10% sites and the quarterly sites would improve the confidence interval of the state-wide estimates. In either case, such a monitoring subset would only be appropriate for state-wide summaries and for tracking trends. However, with the inclusion of the quarterly sites, focused on the hydrogeological regions, better regional tracking of temporal changes might be afforded.

Table 60. Comparison of selected water-quality parameters among SWRL sample subsets.

Analyte	units	State-wide SWRL Sample	10% repeats sample-1	Quarterly repeats period-1	Qrtrly-1 & 10%-1 (combined)	10% repeats sample-2	Quarterly repeats range
nitrate-N							
mean	mg/L	6.2	5.4	5.8	5.8	7.0	5.0 - 5.8
median	mg/L	0.6	0.0	1.7	0.5	0.0	1.0 - 1.7
maximum	mg/L	100.0	31.0	61.0	61.0	140.0	45.0 - 61.0
>10mg/L	%	18.3%	20.0%	14.0%	18.6%	20.2%	11.6% - 17.8%
95% ci	%	15.5-21.3%	9.6-30.3%	5.9-22.2%	11.3-25.9%	9.8-30.6%	
ammonia-N							
mean	mg/L	0.8	0.9	0.6	0.7	1.0	0.6 - 0.6
median	mg/L	0.1	0.2	0.0	0.0	0.3	0.0 - 0.1
maximum	mg/L	11.0	6.7	3.6	6.7	13.0	3.4 - 3.8
> mdl	%	45.2%	51.6%	45.9%	47.7%	54.9%	42.6% - 45.9%
95% ci	%	41.5-49.0%	38.7-64.6%	32.3-59.5%	38.3-57.0%	42.0-67.8%	
organic-N							
mean	mg/L	0.3	0.2	0.2	0.2	0.8	
median	mg/L	0.1	0.1	0.0	0.0	0.1	
> mdl	%	43.1%	43.8%	33.5%	40.2%	43.1%	33.5% - 38.1%
95% ci	%	39.3-46.8%	31.5-56.1%	21.0-46.0%	31.2-49.2%	30.3-55.9%	
Total coliform bacteria							
MPN>0	%	44.6%	45.4%	55.5%	51.8%	42.0%	27.6% - 55.5%
95% ci	%	40.9-48.3%	33.0-57.8%	43.2-67.8%	42.5-61.1%	29.4-54.6%	
calcium							
mean	mg/L	101.7	102.8	96.7	99.9	91.7	77.1 - 112.3
median	mg/L	87.1	91.3	82.6	87.1	77.7	63.3 - 100.7
magnesium							
mean	mg/L	43.0	43.1	43.7	43.6	42.9	40.9 - 43.7
median	mg/L	37.2	38.7	41.7	39.6	40.5	39.4 - 41.7
sulfate							
mean	mg/L	131.5	122.8	113.2	120.5	127.6	109.7 - 123.1
median	mg/L	38.5	41.2	38.9	38.7	43.3	38.0 - 50.8
maximum	mg/L	1,938.4	771.2	1,240.0	1,240.0	932.6	1,168 - 1,242
Any pesticide							
detection	%	13.6%	14.5%	11.4%	13.7%	19.3%	7.4% - 40.0%
95% ci	%	11.1-16.2%	5.5-23.5%	2.8-19.9%	7.3-20.1%	9.3-29.3%	
atrazine							
detection	%	8.0%	8.2%	6.1%	7.6%	3.2%	0.0% - 14.1%
95% ci	%	6.0-10.0%	1.3-15.1%	0.1-13.2%	2.5-12.6%	0.0-7.5%	
mean	ug/L	1.1	0.4	0.4	0.4	1.0	0.4 - 0.0
median	ug/L	0.6	0.4	0.3	0.3	1.0	0.3 - 0.0
maximum	ug/L	7.7	0.8	0.8	0.8	1.7	0.8 - 0.0
metribuzin							
detection	%	1.9%	1.6%	0.0%	0.7%	1.6%	0.0% - 2.9%
95% ci	%	0.9-2.9%	0.0-4.7%	---	0.0-2.2%	0.0-4.7%	--- ---
mean	ug/L	0.2	0.2	---	0.2	0.1	--- ---
median	ug/L	0.1	0.2	---	0.2	0.1	--- ---
cyanazine							
detection	%	1.2%	1.6%	1.7%	1.6%	6.5%	0.0% - 2.8%
95% ci	%	0.4-2.0%	0.0-4.7%	0.0-4.9%	0.0-3.8%	0.3-12.8%	
mean	ug/L	0.3	0.2	0.2	0.2	0.2	0.2 - 0.0
median	ug/L	0.3	0.2	0.2	0.2	0.2	0.2 - 0.0
propachlor							
detection	%	0.4%	0.0%	0.0%	0.0%	0.0%	0.0% - 1.7%
95% ci	%	0.0-0.9%	---	---	---	---	--- ---

Table 61. Comparison of relative proportions of wells and detections of selected analytes among SWRL subsets.

Well depth class	SWRL state-wide %	10% subset %	Quarterly subset %
	Relative % of wells in depth class		
< 50	28%	31%	27%
50 - 99	21%	19%	22%
≥ 100	51%	50%	51%
(<25)	7%	6%	9%
(100-149)	17%	19%	19%
	Relative % of wells with nitrate-N > 10 mg/L		
< 50	52%	56%	68%
50 - 99	39%	39%	24%
≥ 100	9%	6%	9%
	Relative % of wells with any pesticide detection		
< 50	40%	48%	44%
50 - 99	24%	15%	18%
≥ 100	36%	37%	38%
	Relative % of wells with total coliform positives		
< 50	43%	38%	52%
50 - 99	26%	28%	20%
≥ 100	31%	34%	28%

DISCUSSION: RELATIONSHIPS AMONG WATER-QUALITY PARAMETERS AND SITE CHARACTERISTICS

This section of the report will present more interpretive discussion and data analysis of the SWRL survey results. In particular, relationships among water-quality parameters will be described to assess possible predictive capabilities, contamination and transport mechanisms, and chemical changes occurring within the groundwater system. A first-stage analysis of the relationships among site inventory characteristics and water quality findings will be presented. Much further analysis will be performed for future reports, but the first-stage analysis will review basic relationships among water quality, land-use, local chemical use, well siting and construction features.

In the following discussion the data sets used to calculate the summary values presented are typically somewhat smaller than the complete number (686) of SWRL sites. Only data from sites that have analytical results for all the water-quality parameters of interest will be used to discuss their inter-relationships; i.e., if discussing the relationship among NO_3-N, NH_4-N, and DO, sites without a DO measurement are not used in the calculation of summary data even for NH_4-N. This avoids any bias or aberrations related to missing data. As a consequence, however, summary values presented may differ slightly from those presented in other sections of the report.

STATISTICAL ASSOCIATIONS

Tables 62 and 63 present nonparametric Spearman rank correlation coefficients (rho or r_s) between pairs of various data. Spearman rank correlation coefficients vary between 1 (complete concordance) and -1 (complete discordance). These correlation coefficients assess the correlation among the ranks of the paired variables, not their interval or numerical value. Correlation matrices were developed for all paired data, but only the more significant relationships between variables of interest are summarized. None of the correlations are particularly strong even though the probability values are highly significant (the magnitude of the p-values is, in part, related to the large number of data). In essence, while there are significant associations between some of the parameters, the associations are not strong predictors. Even the strongest association, the inverse relation between nitrate-N and ammonium-N (-0.69), is analogous to an r^2 that only explains approximately 47% of the variance.

The strongest suite of associations are the inverse relations between nitrate-N and ammonium-N, well depth and nitrate-N, and ammonium-N and DO; the strongest positive relations between nitrate-N and chloride (Cl), nitrate-N and DO, and well depth and

Table 62. Spearman rank correlation coefficients (rho or r_S) and probabilities (from t statistics) between paired variables; well depth and various water-quality parameters.

Water-quality parameters: (independent)	(dependent)	Spearman correlation coefficient	Probability (p-value)
well depth	nitrate-N	-0.50	0.0001
"	ammonium-N	0.46	0.0001
"	tot. coliform	-0.38	0.0001
"	Cl	-0.36	0.0001
"	pH	0.27	0.0001
"	DO	-0.25	0.0001
"	alkalinity	0.19	0.0001
"	# pesticides detected	-0.17	0.0001
"	Any pesticide detection	-0.11	0.01
"	metolachlor	-0.10	0.05
"	atrazine	-0.10	0.05
"	fluoride	0.09	0.1
"	organic-N	-0.03	0.5
"	sulfate	0.01	0.8

Table 63. Spearman rank correlation coefficients and probabilities between paired water-quality parameters.

Water-quality parameters: (independent)	(dependent)	Spearman correlation coefficient	Probability (p-value)
total coliform bacteria	well depth	-0.38	0.0001
	nitrate-N	0.37	0.0001
"	ammonium-N	-0.35	0.0001
"	Cl	0.30	0.0001
"	DO	0.25	0.0001
"	Any pesticide detection	0.06	
"	# pesticides detected	0.08	
"	atrazine	0.09	
nitrate-N	ammonium-N	-0.69	0.0001
"	Cl	0.65	0.0001
"	DO	0.57	0.0001
"	well depth	-0.50	0.0001
"	# pesticides detected	0.32	0.0001
"	atrazine	0.18	0.0001
"	Any pesticide detection	0.17	0.0001

ammonium-N. The interaction among nitrate-N, ammonium-N, DO, and well depth provides some indications of changing chemical conditions and processes in the groundwater system. This will be discussed further in subsequent sections.

The general relationship between NO_3-N and Cl was discussed earlier. As noted in other studies, some of the Cl delivered from groundwater is related to the use of KCl for potash fertilizers. The Cl ion is highly mobile, similar to NO_3-N. The association between well depth and Cl is not as strong as with NO_3-N because Cl first declines with depth, related to surficial sources, but then in deeper wells Cl increases again, related to the natural increase in dissolved solids in older groundwaters.

Table 62 summarizes a range of other associations, for illustration. There are significant, but weak associations, of increasing pH, decreasing DO, and decreasing pesticide detections with increasing well depth. For any individual pesticide the associations become very weak, and less significant. There is no apparent association, on a state-wide basis, between well depth and parameters such as fluoride, organic-N, or sulfate, although in particular regions and aquifers there are known relationships.

There is no apparent relationship between pesticide detections and pH. While groundwater pH can be a factor in the degradation of pesticides, the pH of Iowa groundwater varies over such a narrow range, being essentially neutral to slightly basic, that there is no discernible association with pesticide detections or concentrations. Nitrate-N is significantly, but weakly associated to pesticide detections. As pointed out in many previous reports (e.g., Hallberg, 1989b), there is an association between pesticide detections and nitrate-N concentrations, but the relationship is a poor predictor and best assessed on a probability basis.

Table 63 reiterates the prior discussion, noting that total coliform bacteria is not a good indicator of other forms of contamination. While there are significant, but weak associations between total coliform and nitrate-N, there is no relationship between total coliform and pesticides; the best association is with well depth. This helps to illustrate that the apparent relationship between total coliforms and other contaminants, such as nitrate-N, is primarily a function of their co-occurrence (autocorrelation) related to well depth. As discussed, any shallow well, particularly if it is open to the water-table, will likely have total coliforms present, at least intermittently. This is simply from the influx of naturally occurring coliforms in the soil environment. Such shallow wells are also those most prone to contamination by activities and compounds used at the land surface.

Several other statistical methods were used to assess the possible predictive relationships among the water-quality parameters; linear and multiple linear regression, serial correlations, and logistic regression-probability analyses. There was no significance to any predictive models between total coliform bacteria (as the independent variable) and nitrate-N or pesticide detections. In contrast, quite significant models ($p=0.001$) were developed using well depth as an independent variable to predict the presence of total coliform bacteria. Logistic models predicting pesticide or atrazine detections from nitrate-N concentrations were also statistically significant, but not very accurate.

TOTAL COLIFORM BACTERIA

Tables 64 and 65 review relationships between coliform bacteria and nitrate-N occurrence. Reviewing these data: there is a trend that as NO_3-N increases the proportion of water systems positive for coliforms increases; for those sites with >10 mg/L NO_3-N 72% were positive for total coliforms. Conversely, however, considering all sites positive for total coliforms, less than one-third exhibit >10 mg/L NO_3-N. Reviewing the data by nitrate concentrations ranges (Table 65) shows that, for both total and fecal coliforms, the modal group, the median, or the majority of positives coincide with relatively low nitrate concentrations. About 70% of the total coliform positives show <10 mg/L NO_3-N; about 35-45% are <3 mg/L NO_3-N (Table 65). Table 64 also reiterates the general relationship between well depth and coliform positives. Even within the total coliform classes there is a consistent trend, with mean/median well depths decreasing with greater MPN values. As noted in Table 29, about 70% of total coliform positives,

Table 64. Summary of coliform data in relation to samples with nitrate-N >10 mg/L and well depth.

Coliform bacteria class	% all samples with total coliform that had nitrate-N >10 mg/L	% all samples with fecal coliform that had nitrate-N >10 mg/L	% all samples with nitrate-N >10 mg/L in total coliform class	% all samples with nitrate-N >10 mg/L in fecal coliform class	well depth for coliform class mean feet	well depth for coliform class median feet
No bacteria			28%	86%	165	130
Positive	31%	38%	72%	14%	98	55
Relative % of those total coliform positives with nitrate-N >10 mg/L:						
2.2-9.1 MPN	12%	--	10%	--	110	68
16-16+ MPN	36%	--	62%	--	84	50

Table 65. Summary of coliform data in relation to nitrate concentrations.

Nitrate-N conc. range in mg/L	% samples in nitrate-N range positive for: total coliform bacteria	% samples in nitrate-N range positive for: FECAL coliform bacteria	% all samples with: total coliform bacteria in nitrate-N range	% all samples with: FECAL coliform bacteria in nitrate-N range
< 0.1	24%	1%	22%	6%
0.1-2.9	58%	12%	24%	31%
3.0-9.9	52%	8%	23%	25%
10.0-19.9	68%	16%	16%	25%
> =20.0	78%	11%	15%	13%

Table 66. Summary of total coliform data related to pesticide detections.

Total coliform bacteria class	% samples in total coliform class that had: any pesticide detected	% samples in total coliform class that had: atrazine detected	% all samples with: any pesticide detected in total coliform class	% all samples with: atrazine detected in total coliform class
No bacteria			48%	52%
Positive	16%	8%	52%	48%
Relative % of those total coliform positives with pesticide detections:				
2.2-9.1 MPN	16%	6%	17%	11%
16-16+ MPN	16%	10%	36%	37%

Table 67. Summary of water-quality parameters and well depth.

% of all wells with water-quality parameter in the well depth range:	Wells < 50 ft deep	Wells 50-99 ft deep	Wells ≥ 100 ft deep
wells in depth range;	28%	21%	51%
total coliform positives;	43%	26%	31%
fecal coliform positives;	45%	36%	18%
pesticide detections;	40%	24%	36%
atrazine detections;	33%	31%	35%
nitrate-N >10 mg/L;	52%	37%	11%
concentrations, mg/L:			
mean nitrate-n;	11.2	11.0	1.6
median nitrate-N;	6.0	4.3	<0.1

and 80% of fecal coliform positives, are associated with systems using wells <100 feet deep.

Table 66 reiterates the lack of any clear association or predictive value between total coliform positives and pesticide detections. As discussed, *no* fecal coliform positive sites had any pesticides detected.

RELATIONSHIPS TO WELL DEPTH

Table 67 summarizes some of the inter-relationships among the occurrence of contaminants and well depth. Earlier reports (e.g., Hallberg and Hoyer, 1982; Hallberg, 1986), and the prior discussion in this report, had indicated significantly greater contamination of wells <50 feet deep; and also that contamination was still evident, at least in local settings to 100 feet and in some settings even deeper. The SWRL data shows that the most pronounced, significant difference occurs at a depth of about 100 feet. As summarized on Table 67, wells <100 feet deep comprise about 50% of wells state-wide, but these wells account for about 70% of total coliform positives, 80% of fecal coliform positives, 64% of pesticide detections and total atrazine detections, and 89% of wells with NO_3-N >10 mg/L.

Table 68 provides added details for nitrate-N and the hydrogeologic regions. This emphasizes the pronounced break in the data between wells <100 feet deep and those >100 feet deep. This significant change in the aggregate data is evident in every region, except the Northeast. In the Northeastern region wells tend to be deeper than in other regions, but because of the hydrogeology many wells are still commonly drawing water from the top of the groundwater system and, as noted, the depth of groundwater circulation is much greater in northeastern and eastern Iowa. Even the average data clearly indicate NO_3 contamination to depths >100 feet. As discussed, 67% of the pesticide detections from sites with wells >300 feet deep come from the Eastern and Northeastern hydrogeologic regions.

These data also point out differences in the North-Central region. The median nitrate-N concentrations, in particular, illustrate the markedly lower nitrate-N present.

NITRATE-N

There is a tendency for wells with greater nitrate-N concentrations to more commonly exhibit pesticide detections (Table 69). As noted, while this trend afforded a statistically significant model, it was not an accurate predictor. Table 69 illustrates why; 22% of all wells with pesticide detections had no detectable nitrate. Yet nearly 60% of all wells with pesticide detections had >3.0 mg/L NO_3-N, with 27% exhibiting >10 mg/L.

Simple linear regression models, using nitrate-N concentrations to predict pesticide

Table 68. Summary of nitrate-N results related to well depth and hydrogeologic region. ** = too few sites in range to calculate %.

Hydrogeologic Region	Wells < 50 ft deep	Wells 50-99 ft deep	Wells $\geq$100 ft deep	Unknown	All Wells
	- - - - -	mean nitrate-N, mg/L			- - - - -
Northeastern (1):	2.6	1.9	3.9	2.9	3.6
Eastern (2):	9.0	8.7	1.3	2.4	3.6
South-Central (3):	10.1	12.6	0.7	14.3	9.6
Southwestern (4):	9.7	16.0	0.9	13.1	10.0
Northwestern (5):	21.5	13.6	0.3	7.9	13.9
North-Central (6):	4.8	7.7	0.3	0.5	2.6
State-wide:	11.2	11.0	1.6	6.1	6.3
	- - - - -	median nitrate-N, mg/L			- - - - -
Northeastern (1):	1.3	1.9	1.5	1.1	1.3
Eastern (2):	7.3	3.8	< 0.1	< 0.1	< 0.1
South-Central (3):	4.9	2.0	< 0.1	11.5	2.1
Southwestern (4):	6.3	11.0	0.1	9.9	5.6
Northwestern (5):	11.0	12.0	< 0.1	8.1	5.2
North-Central (6):	0.1	0.6	< 0.1	< 0.1	< 0.1
State-wide:	6.0	4.3	< 0.1	0.7	0.6
	- - - - -	% nitrate-N >10 mg/L (% in depth class)			- - - - -
Northeastern (1):	**	**	9%	11%	9%
Eastern (2):	42%	26%	4%	3%	12%
South-Central (3):	30%	32%	0%	50%	28%
Southwestern (4):	30%	50%	0%	25%	31%
Northwestern (5):	52%	55%	0%	46%	38%
North-Central (6):	7%	17%	2%	0%	6%
State-wide:	35%	32%	4%	19%	18%

concentrations (e.g., atrazine) did not provide a significant predictive model. Logistic regression did provide a statistically significant model relating NO_3-N concentration to the presence or absence of pesticide detections. The results of such a logistic probability analysis from data from another study in Floyd and Mitchell counties (Libra et al., 1987; Hallberg et al., 1987; Hallberg 1989a and b) is shown in Figure 49. This method of analysis provides a probability estimate of the occurrence of the detection of any pesticide based on the NO_3-N concentration from the well. The state-wide SWRL data set did not, unfortunately, provide such a well-defined relationship; while the model was significant statistically, the confidence intervals, and the errors in prediction were too great to be useful.

Table 70 summarizes the inter-relationships among nitrate-N (NO_3-N), well depth, and ammonium-N (NH_4-N), organic-N (O-N), and dissolved oxygen (DO). Comparing either means or median concentration data, as well depth increases NO_3-N and DO decrease but NH_4-N increases, and there is no apparent trend for O-N. These trends suggest that some level of denitrification is taking place, with increasing depth in groundwater flow-systems in Iowa. This will be further explored in the next sections.

AMMONIUM-N

In contrast to NO_3-N, the proportion of wells with detectable NH_4-N increases with depth; 74% of all wells with detectable ammonium are >100 feet deep (Table 71). The average

Table 69. Summary of pesticide detections in relation to nitrate-N concentrations.

Nitrate-N conc. range in mg/L	% wells in nitrate range with pesticide detections	% all wells with pesticide detections in nitrate range
< 0.1	7%	22%
0.1-2.9	12%	20%
3.0-9.9	20%	31%
10.0-19.9	16%	14%
> =20.0	22%	13%

concentrations of NH_4-N significantly increase with depth also, though the maximum values are similar across depth ranges. The maximum NH_4-N observed was 11.0 mg/L. (Note: in relation to Tables 71 and 72, the maxima of 11 mg/L comes from a well of unknown depth.)

NH_4-N concentrations are typically much lower than NO_3-N and NH_4-N occurrence is inverse to NO_3-N (Table 72). As the concentrations of nitrate-N increase, the occurrence and average concentrations of ammonium-N consistently decrease; 90% of all wells with detectable NH_4-N have <3.0 mg/L NO_3-N. The mean and median NH_4-N for wells with >5.0 mg/L NO_3-N is <0.1 mg/L NH_4-N; even the maximum concentrations recorded show an inverse relation to NO_3 concentrations.

Table 73 summarizes the occurrence of pesticide detections in relation to NH_4-N concentrations. Pesticide detections increase with decreasing ammonium concentrations; 65% of pesticide detections coincide with <0.1 mg/L NH_4-N and 87% with <0.9 mg/L. This relationship would be expected because of their respective general trends with well depth. While there is no direct causative relation, the association is very consistent and may be useful in further predictive models.

NITROGEN, DO, WELL DEPTH, AND REDOX REACTIONS

Reinforcing the implications between nitrate and ammonium the inter-relationships among NO_3-N, NH_4-N, O-N, DO, and well depth are summarized in Table 74. The relationships among these parameters are very consistent. Dissolved oxygen concentrations decline with well depth, as would be expected. As DO declines, so do NO_3-N concentrations, but NH_4-N increases. Organic-N does not show any consistent trend, and as indicated in prior discussion, has rather uniformly low concentrations in relation to other conditions.

Redox Reactions

These relationships provide a consistent indication of changing oxidation conditions with depth in the groundwater system that in turn afford changes in the nitrogen compounds that may be stable. Reduction-oxidation, or "redox" reactions involve changing the oxygen carrying capacity of ions, or more generally, changing the charge that an ion carries. In the groundwater system, at more shallow depths, oxidizing conditions often occur, that promote the weathering of sulfide minerals. Iron sulfides, FeS_x compounds, are oxidized to form SO_4, and a variety of iron compounds. With the movement to depth in the groundwater system, DO is consumed by chemical reactions, microbial consumption, and related degradation of organic materials, producing reducing conditions. Under reducing conditions reactions are driven a different direction; sulfate may be reduced to

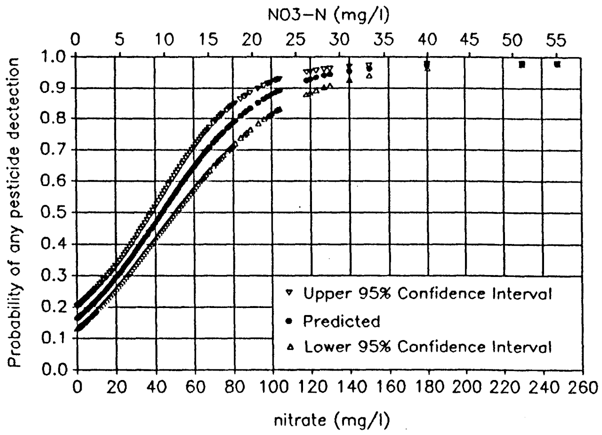


Figure 49. Probability relation between nitrate concentration and the detection of any pesticide in well-water samples from Floyd and Mitchell counties, from data collected in 1986 and 1987.

Table 70. Mean and median nitrate-N, ammonium-N, organic-N, and dissolved oxygen (DO) concentrations by well depth ranges.

Well depth range feet	nitrate-N mean	nitrate-N median	ammonium-N mean	ammonium-N median	organic-N mean	organic-N median	DO mean	DO median
				in mg/L				
<50 ft	11.2	6.0	0.2	<0.1	0.3	0.1	5.2	5.0
50-99 ft	11.0	4.3	0.5	<0.1	0.2	0.1	4.4	3.1
100-199 ft	1.5	<0.1	1.1	0.6	0.3	0.1	3.4	2.6
200-399 ft	1.4	<0.1	1.4	0.9	0.3	0.1	4.0	2.3
>=400 ft	2.5	<0.1	1.2	0.8	0.3	<0.1	2.8	2.5

hydrogen sulfide, H_2S, a stable form under such conditions. H_2S gives the "rotten egg" smell that may occur when opening a water tap from a deep well.

Many such reactions are biologically mediated, i.e., they occur because of bacteria that promote, or actually conduct the reaction as part of their metabolism (e.g., Freeze and Cherry, 1979; Matthes, 1982). Such microbes are adapted to environments where there is no free oxygen (DO) for their metabolism, and they obtain their oxygen supply by "reducing" compounds such as sulfate or nitrate. Most reactions in the nitrogen series are biologically mediated by "nitrifiers" and "denitrifiers." Such bacteria are present in the soil-groundwater system but population densities vary in relation to the suitability of the micro-environment. Their populations may be limited by the absence of sufficient organic carbon as a "substrate," needed as an energy-food source for the bacteria.

Free oxygen does not have to be present in an oxidizing environment, but in the soil-groundwater system dissolved oxygen (DO) is typically present. The DO is derived from the atmosphere by precipitation that becomes recharge water and from interaction in the soil as microbes release oxygen during respiration. Nitrate is the stable form of N in an oxidizing environment (e.g., Lowrance and Pionke, 1989). It is in this oxidizing environment in the soil that NO_3 is formed by the mineralization and oxidation of O-N and NH_4.

DO in groundwater is depleted through chemical reactions and/or biological processing as the water passes through the system to depth, and, hence, reducing conditions evolve. Under reducing conditions ammonium is stable and the decomposition of organic materials may produce NH_4. Under particular conditions, even nitrite, NO_2, can be stable. Under reducing conditions, denitrification can occur. Denitrification is a biologically mediated reaction and NO_3 can be reduced to NO_2 or NH_4, but typically in the shallow-soil environment (and surface water environment, of streams, lakes, and wetlands) it is reduced to NH_4 and then to N_2O or N_2 gas. Chemical denitrification, involving oxidation of ferrous iron, can also take place (Lowrance and Pionke, 1989; Strebel et al., 1989).

Oxidizing and reducing conditions can coexist in close proximity, and can change temporally in the same setting. In the small pores within aggregates in the soil profile, or in the matrix of an aquifer, permeability is limited, and DO may not readily diffuse into these domains. This may allow local reducing conditions to dominate. In close proximity to these sites, water and air (DO) may be recharged more readily through more open portions of the soil or an aquifer (such as around macropores of various origins) maintaining oxidizing conditions. Temporally, in a soil setting that normally is dominated by air-filled pores and oxidizing conditions, the soil may become saturated, "water-logged," during precipitation/recharge events. Since DO is more limited in water than in the soil atmosphere (air), and reducing conditions may be established and dominate temporarily.

Reducing conditions are commonly

Table 71. Summary of ammonium-N data related to well depth.

Well depth range in feet	% wells in depth range with ammonium-N > 0.1 mg/L	% all wells with ammonium-N > 0.1 mg/L, in depth range	ammonium-N conc. mean	median in mg/L	max
< 50 ft	15%	9%	0.2	<0.1	8.3
50-99 ft	37%	17%	0.5	<0.1	5.7
100-199 ft	70%	42%	1.1	0.6	6.4
200-399 ft	66%	25%	1.4	0.9	7.1
>= 400 ft	75%	7%	1.2	0.8	4.1

Table 72. Summary of ammonium-N data related to nitrate-N concentration.

Nitrate-N conc. range in mg/L	% wells in nitrate range with ammonium-N > 0.1 mg/L	% all wells with ammonium-N > 0.1 mg/L, in nitrate range	ammonium-N conc. mean	median in mg/L	max
< 0.1	87%	71%	1.6	1.1	11.0
0.1-2.9	47%	19%	0.6	0.2	3.9
3.0-4.9	36%	5%	0.1	0.1	1.6
5.0-9.9	12%	3%	< 0.1	< 0.1	1.3
10.0-19.9	7%	1%	< 0.1	< 0.1	0.1
>=20.0	7%	1%	< 0.1	< 0.1	0.2

Table 73. Summary of pesticide detections related to ammonium-N concentration.

ammonium-N conc. range in mg/L	% wells in ammonium range with pesticide detections	% all wells with pesticide detections in ammonium range
<0.1	16%	65%
0.1-0.9	10%	22%
1.0-1.9	6%	6%
>=2.0	7%	7%

Table 74. Mean and median ammonium-N, nitrate-N, organic-N, dissolved oxygen, and well depth in relation to nitrate-N, ammonium-N, and DO concentration ranges.

Nitrate-N	ammonium-N		DO		organic-N		well depth	
conc. range	mean	median	mean	median	mean	median	mean	median
	in mg/L						feet	
<0.1	1.6	1.1	2.5	1.8	0.3	<0.1	192	160
0.1-0.9	0.6	0.2	3.9	2.8	0.4	0.1	144	112
1.0-2.9	0.1	<0.1	4.6	4.1	0.1	<0.1	115	60
3.0-9.9	<0.1	<0.1	5.7	5.5	0.2	0.1	87	50
10.0-19.9	<0.1	<0.1	6.2	5.9	0.2	0.2	68	49
> =20.0	<0.1	<0.1	6.4	6.0	0.3	0.2	62	45

ammonium-N	nitrate-N		DO		organic-N		well depth	
conc. range	mean	median	mean	median	mean	median	mean	median
	in mg/L						feet	
<0.1	11.7	7.0	5.6	5.5	0.2	0.1	96	50
0.1-0.9	1.7	<0.1	3.3	2.1	0.2	<0.1	141	110
1.0-1.9	0.3	0.2	2.1	1.8	0.2	<0.1	206	167
> =2.0	0.6	<0.1	2.3	1.9	0.6	0.2	210	188

DO	nitrate-N		ammonium-N		organic-N		well depth	
conc. range	mean	median	mean	median	mean	median	mean	median
	in mg/L						feet	
0.1-0.9	0.8	<0.1	1.4	0.9	0.5	0.2	165	135
1.0-1.9	1.3	0.6	1.3	0.6	0.5	0.1	178	131
2.0-4.9	5.7	0.3	0.8	0.1	0.2	0.1	143	100
> =5.0	11.7	7.0	0.2	<0.1	0.2	0.1	107	60

established, and dominant within two settings in the soil-groundwater system (e.g., Lowrance and Pionke, 1989; Hallberg, 1989a). In wetlands, poorly drained soils, and many young alluvial soils, organic matter is abundant. In these shallow groundwater environments common saturated conditions, coupled with the degradation of organic matter, lends to rapid depletion of DO, establishing reducing environments. Such settings are locally important in alluvial environments, along streams and rivers. Ongoing efforts suggest they are important over broad areas in the North-Central Region. This is the area of the youngest glacial deposits in the state (the Des Moines Lobe), where the the Clarion-Nicollet-Webster soils formed, and relatively large areas of poor drainage occur.

The second setting occurs at more substantial depth. As groundwater moves through geologic materials to depth, the DO in the groundwater is depleted through chemical reactions and/or biological processing. Typically in natural groundwater settings when DO is depleted reducing conditions are established. There also may be a transition zone where other factors, such as pH, change to set up conditions appropriate for reduction reactions to occur (Matthes, 1982; Howard, 1985). Chemical or biochemical reactions involving ferrous iron and sulfur compounds may enhance denitrification also (Strebel et al., 1989).

Nitrate Reduction, Denitrification At Depth

The consistency in the relations among NO_3, NH_4, and DO with depth clearly suggest that nitrate reduction and/or denitrification occur with depth in groundwater systems in Iowa. The greatest DO and NO_3-N concentrations occur with well depths typically <50 feet (Tables 70 and 74). At depths <100 feet, <50% of sites have detectable ammonium. Also, the median well depth associated with the greatest NH_4-N concentrations (>2 mg/L) is about 190 feet; the median well depth associated with the lowest detectable DO is about 140 feet (Table 74). This again illustrates the consistency of these well data, with the chemical evolution expected in the groundwater system. At shallow depth, oxidizing conditions dominate and NO_3 is the dominant N form. With greater depth, DO is depleted, reducing conditions are established, and NH_4 is the more stable form in the groundwater environment (Lowrance and Pionke, 1989; Howard, 1985). The NH_4 may be derived from the break down of organic matter within the aquifer or from denitrification. An analysis of the estimated stability fields implied by the water quality data, suggest that nitrite, NO_2, as well as NH_4, may be stable. NO_2 was not directly analyzed, but is converted to NO_3 in the sample preparation and hence, is included in the NO_3 data. Past work, which analyzed samples for NO_2-N, show that NO_2 is rare at shallow depths, and that NH_4 and NO_3 co-exist in samples from relatively shallow depths (Hallberg et al., 1984; Libra et al., 1984). Pumping wells, or observation wells, however, can draw water from various microenvironments.

The combined concentrations of NH_4-N plus NO_3-N (NO_3 + NO_2-N) is still quite low, compared with NO_3-N concentrations in shallow groundwater. Mean and median NH_4 + NO_3-N concentrations for all data are 4.1 and 1.9 mg/L. For the mean, the value for nitrate-N is dominant, at 3.4 mg/L (NH_4-N = 0.7 mg/L), but for the median NH_4-N dominates, at 1.1 mg/L (NO_3-N = 0.8 mg/L). The SWRL data, with relatively low N concentrations at depth, are consistent with other studies. These data, and other studies, raise the question of whether or not the deeper groundwater environment is capable of denitrifying the mass of nitrate-N apparent in shallow groundwaters across Iowa (as exhibited in the data from the wells <50 feet deep)? These waters with higher nitrate concentrations will be the recharge for deeper groundwater with time. The lowerconcentrations at depth suggest that these waters are relatively old and did not originally contain much NO_3-N. However, it is unclear whether NH_4-N is the principle stable product, or whether denitrification at depth in the groundwater environment generates N gases, as it does in the shallow soil environment (Howard, 1985; Lowrance and Pionke,1989). If such gases are generated and can escape from the groundwater system, further N might be accounted for at depth (e.g., Vogel et al., 1981). Various studies suggest that the low concentrations at depth are a function of water age, and that the capacity of the deep groundwater environment may be limited (e.g., Hallberg, 1989a). Further work in this area is needed.

NITRATE/CHLORIDE RATIO AND WELL DEPTH

The ratio of nitrate:chloride (NO3-N/Cl) in relation to well depth was also examined because many previous small-scale studies have suggested this may provide possible evidence of denitrification. As was previously discussed, of the major ions only nitrate and chloride concentrations decrease with increasing well depth. This is suggestive of predominantly surficial sources of these ions (e.g., KCl fertilizers and road salt for Cl).

Both Cl and NO_3 are very mobile in groundwater. Chloride is considered to be more conservative than NO_3; Cl is not involved in any appreciable biological or physical-chemical reactions or transformations in most groundwater systems. Nitrate concentrations, as discussed, are altered by various microbiologically mediated processes. Denitrification (the reduction of nitrate-N), for instance, transforms and effectively removes NO_3, and hence, nitrate concentrations would be decreased relative to other constituents. Hence, an appreciable decrease in the nitrate:chloride ratio may be evidence that denitrification is occurring (e.g., Saffinga and Keeney, 1977; Hill, 1986).

The NO_3-N:Cl ratio does decrease with increased well depth (Table 75). However, median values drop to "zero" in wells greater than 100 feet deep, but this is largely an artifact of changes in Cl. After an initial decrease at shallow depths, Cl then increases in concentration in deep wells because of natural contributions. Most of the previous studies that have used NO_3/Cl ratios have looked at this relationship on the scale of feet or tens of feet below the water table, in localized areas. At the scale of the SWRL data, with the complexity of natural sources of Cl at depth, these ratios would not seem useful as strong evidence for denitrifaction.

Similarly, increases in SO_4 have been noted accompanying denitrification in settings where oxidation of iron-sulfide takes place as part of the reaction series (Strebel et al., 1989). Again these observations are on too local a scale to relate to the SWRL data, and natural sources of sulfate occur in many settings.

Table 75. Mean and median nitrate-N:chloride ratios by well depth intervals for the SWRL study. Ratio is mg/L nitrate-N to mg/L chloride.

Well Depth Range (ft)	mean NO_3-N/Cl ratio	median NO_3-N/Cl ratio
<50	0.57	0.40
50-100	0.29	0.20
101-200	0.14	<0.01
201-400	0.11	<0.01
>400	0.01	<0.1

If denitrification is a major mechanism in the loss of nitrate, an accompanying increase in alkalinity (HCO_3^-) and hardness (Mg and Ca) may also be observed. However, no strong correlation between alkalinity and hardness, and nitrate-N was found either.

POTASSIUM/SODIUM RATIOS

Some studies, again in local small-scale areas, have suggested that a potassium to sodium (K/Na) ratio greater than 0.3 may be indicative of water-quality impacts from septic tank effuent (e.g., Daly and Daly, 1982). These values have been reported for an area in Ireland where land use is mixed pasture and cropland. The geologic setting is similar to many portions of Iowa; Paleozoic carbonate bedrock aquifers overlain by glacial deposits.

Coliform bacteria results were also compared with K/Na ratios to assess the ratios value as a predictor of contamination from septic tank effluents. Two-way contingency tables and chi^2 statistics were calculated to compare results. There was no significant relationship between K/Na ratios >0.3 and total coliforms, or between the ratios and fecal coliforms.

Groundwater contamination from septic tank effluents occurs at shallow depths, and will be most prevalent in shallow wells. Mean K/Na ratios increase with well depth, however. The lack of relationship between presence of bacteria and a K/Na ratio as well as an increase in K/Na ratios with depth, indicate that for the SWRL study, K/Na ratios are not useful as predictors of contamination from septic tank effluents.

The utility of K/Na, NO_3/Cl ratios as well as alkalinity-hardness-nitrate relationships are probably of limited use on such a large scale as the SWRL survey.

SITE CHARACTERISTICS AND WATER QUALITY

This section of the report will present first-stage analyses of the relationships among site inventory characteristics and water-quality findings. Further analysis will be performed for future reports, but this first-stage analysis will review basic relationships among water quality, land-use, local chemical use, and siting and construction features. For the first-stage analysis, the site inventory observations will be evaluated individually, to provide initial insights to relationships. This will also provide direction for later multi-factor analysis.

Local Environmental Effects And Potential for Point-Source Contamination

The potential causes of groundwater contamination with agricultural chemicals are numerous: infiltration through the soil from field application, surface run-off into sinkholes or agricultural drainage wells, product spills near well sites, back siphoning of product into wells during mixing operations, and tile line discharges into seepage wells, to name a few. As part of the SWRL study, detailed site inventory questionnaires were completed for each well location. Many questions were asked of participants concerning the local setting, common agricultural practices, and accidental occurrences that could lead to groundwater contamination. Also, during sampling DNR and PM&EH field staff personally conducted on-site inspections of the well, water system, and local environment.

Local Setting

The presence of sinkholes or agricultural drainage wells on the property near a well are two factors often associated with groundwater contamination. These factors are not common on a state-wide basis: sinkholes were identified in the vicinity of only 2.1% of the sites and only 0.6% of the sites were potentially affected by agricultural drainage wells.

The presence of sinkholes on the property, or immediately adjacent land was not a good predictor of pesticide contamination of the private, rural drinking water supply. Only three of the 14 sites with reported sinkholes had pesticide contamination. Similarly, only two of the 14 wells had >10 mg/L nitrate-N. Neither proportion is significantly different than the state-wide average. *None* of the sites with reported agricultural drainage wells had any pesticide detections or nitrate-N >10 mg/L.

Tile lines discharging directly into wells were identified at <0.6% of sites, state-wide. Only one of these four sites had pesticides detected. Two metabolites of carbofuran were observed at low concentrations at this site. Nitrate concentrations >10 mg/L NO_3-N were not observed at any of the sites.

Observations by SWRL field staff concerning local landuse surrounding the wells are summarized in Table 76. These observations illustrate some of the complexities of assessing environmental factors and some conventions that will be used throughout this section. As noted, this first-stage analysis presents the results for each factor reviewed individually, to begin to assess if certain factors are associated with well contamination. Few factors occur in isolation, however. Typically there are many confounding, and inter-related variables. For example, most wells with a feedlot in the adjacent vicinity will also have rowcrops nearby; many wells with forested areas or woodlots nearby will also be counted in some other landuse category, and so forth. In future analysis, these factors, and others, will be reviewed interactively and in combination, to further isolate possible associations. Many of the categories of the inventory were set up to provide a gradient analysis, as shown on Table 76. If a given landuse occurring within a 0.5 mile radius exhibits a strong positive correlation with nitrate contamination for example, this association should be even stronger when that landuse is immediately adjacent to the well, if there is any real relationship.

To illustrate this initial assessment of possible associations, the tables in this section also show the state-wide proportions for the total SWRL

Table 76. Well-site landuse observations and water-quality data. + and - indicate relative proportions that are between 1 to 5% above or below the state-wide 95% confidence intervals; ++ or -- indicate proportions >5% above or below the 95% confidence intervals.

Observations:	Proportion of sites in category %	Wells with total coliform detected %	Wells with >10 mg/L nitrate-N %	Wells with any pesticide detection %	Wells with any atrazine detection %
State-wide proportion from total SWRL sample:	100.0%	44.6%	18.3%	13.6%	8.0%
Wells <50 ft deep:	27.9%	71.5%	35.1%	17.9%	9.6%
Landuse within a 0.5 mile radius adjacent to the well:		RELATIVE state-wide proportions in each category:			
feedlot;	40%	45%	26% +	15%	10%
farmland;	96%	45%	18%	13%	8%
rowcrop;	94%	44%	19%	13%	8%
pasture;	66%	47%	19%	15%	8%
forested;	22%	47%	18%	14%	10%
non-farm, "suburban" houses;	6%	31%--	20%	17%	16% ++
chemical handling/ storage facility;	2%	56% ++	29% ++	24% ++	14% +
other;	15%	50% +	22% +	13%	5% -
Landuse in the area immediately adjacent to the well:					
feedlot;	29%	44%	24% +	15%	11% +
farmland;	79%	43%	20%	14%	9%
rowcrop;	72%	42%	20%	15%	9%
pasture;	47%	47%	17%	11%	6% -
forested;	13%	47%	17%	7% -	6%
non-farm, "suburban" houses;	3%	14%--	8%--	13%	11%
chemical handling/ storage facility;	<1%	Can't compute, sample too small.			
other;	23%	49%	20%	13%	7%

sample and the proportions for wells <50 feet deep affected by a contaminant. The results shown for a given category are the relative state-wide, population-weighted proportions for that category only. These relative proportions afford comparison with the state-wide data. The state-wide proportion of sites in a landuse category is also shown for perspective; while an association between landuse and water-quality parameters may be significant it may explain only a small portion of the total state-wide occurrence. The results summarized are also annotated with ++ and -- symbols. Data marked by a symbol are outside the 95% confidence interval (CI) for the state-wide proportions; a single + or - indicates that the data are between 1 and 5% outside the 95% CI (i.e., within a 90% CI); and two ++ or --, indicates they are more than 5% outside the 95% CI. Thus, this also summarizes some level of significance of the deviations shown.

As shown on Table 76, most aggregate landuse is, in part, too complex, and, in part, too ubiquitous in Iowa to provide definitive results. A few trends are apparent. The most prominent trend is an association with suburban housing tracts; while a small proportion of the rural area, these areas show substantially fewer wells with detections of >10 mg/L NO_3-N and total coliform bacteria. This reduced occurrence becomes more pronounced moving from the 0.5 mile radius to the immediate vicinity of the wells. Curiously, these same sites show somewhat greater than average detections of atrazine, but when viewed from the immediate vicinity of the wells this association is no longer significantly different than the state average.

At the 0.5 mile radius, the occurrence of chemical handling and storage facilities is associated with a greater proportion of all contaminants, coliform bacteria, nitrate, and pesticides. There is no logical relationship for the bacteria with such sites, however. Along the gradient to the immediate vicinity of the wells, this association does not hold up. As shown on the Table 76, there are too few sites with chemical handling or storage in the immediate vicinity of the wells (< 1%) to make meaningful estimates. However, *none* of these sites had any pesticides detected nor did any have NO_3-N >10 mg/L. This suggests that the 0.5 mile association may be a function of an inter-correlation with other factors, such as well depth, for example.

Pasture and forested areas in the immediate vicinity of the wells were associated with slight reductions in pesticide detections (and nitrate-N). The associations are not very strong, nor consistent.

Well Location

Continuing with the gradient analysis, SWRL field staff also characterized other attributes of the local setting of the well location, through on-site observation and interviews. Some of these added associations are summarized on Table 77.

Wells located on floodplains, and likely finished in alluvial aquifers, exhibited slightly greater proportions of nitrate-N and atrazine contamination, particularly when compared to wells located in upland positions. The majority of private wells state-wide, 64%, are isolated from routine activities, yet exhibit contamination at the state-wide average proportions.

Wells located within 15 feet of chemical storage and handling areas are very uncommon, occurring at <0.6% of rural sites. Even with this proximity to potential point source problems, *none* of these wells contained pesticides or NO_3-N >10 mg/L.

Wells located directly within a current or former animal feedlots showed significantly more wells with high concentrations of nitrate. About 43% of the wells in this category exceeded health advisory levels, compared to the 18.7% average of all wells state-wide. However, such sites comprise only about 3% of wells state-wide and, thus, these sites explain only about 1% of the wells with >10 mg/L, NO_3-N. These wells also exhibited a slightly greater proportion total coliform positives and for pesticide detections, particularly for atrazine. While the increased incidence of high nitrate concentrations and bacteria are understandable, there is no direct reason these wells should exhibit greater pesticide contamination. This suggests some degree of inter-correlation with other factors. It would have been expected that the proportion of wells sited directly in feedlots would have exhibited much greater proportions of total coliform positives than were observed. Also, none of these sites were positive for fecal coliforms. This suggests that well construction, and leakage at the well head was not a major factor contributing to contamination of these wells.

Similarly, a slightly greater proportion of wells located within 50 feet of manure storage areas were associated with high nitrate concentrations than the state-wide average. At this scale, there was no increase in the proportion of total coliform positives and none of these sites were positive for fecal coliforms either.

Proximity to septic tank leach fields did not result in an increased proportion of nitrate contamination. Strikingly, these wells, within 50 feet of septic systems, showed significantly fewer positives for total *and* fecal coliform bacteria. This fits the gradient noted in Table 76, showing that wells in non-farm, suburban settings had significantly lower coliform and nitrate contamination. There is always concern for well contamination by septic systems in such settings and while cases are known in Iowa these data indicate that this is not a major regional concern.

Wells located less than 25 feet from an out-house (only 0.2% state-wide occurrence) and wells less than 25 feet from a barn (7% occurrence) showed less than the state average for all categories of contamination. Less than 0.2% of wells were located within 15 feet of a fuel storage tank and none exhibited contamination by bacteria, high nitrates, pesticides, or petroleum compounds.

Table 77. Well location observations and water-quality data. (See text for discussion of fecal coliform.) + and - indicate relative proportions that are between 1 to 5% above or below the state-wide 95% confidence intervals; + + or -- indicate proportions >5% above or below the 95% confidence intervals.

Observations:	Proportion of sites in category %	Wells with total coliform detected %	Wells with >10 mg/L nitrate-N %	Wells with any pesticide detection %	Wells with any atrazine detection %
State-wide proportion from total SWRL sample:	100.0%	44.6%	18.3%	13.6%	8.0%
Wells <50 ft deep:	27.9%	71.5%	35.1%	17.9%	9.6%
		RELATIVE state-wide proportions in each category:			
Well located:					
on upland/hilltop;	29%	46%	19%	11%	5% -
on floodplain;	9%	45%	24% +	16%	13% +
Well isolated from routine activity:	64%	46%	19%	15%	8%
Well <15 ft from chemical storage, handling area:	<1%	Can't compute; sample size too small.			
Well located in feed-lot/cattle-yard:	3%	51% +	43% + +	17% +	17% + +
Well <50 ft from manure storage:	2%	46%	25% +	17%	8%
Well >50 ft from manure storage:	95%	44%	18%	14%	8%
Well <50 ft from septic system:	11%	32% --	19%	10%	6%
Well >50 ft from septic system:	79%	48%	20%	14%	9%

On-Site Chemical Use and Handling

Considerable information was obtained on agricultural chemical and fertilizer use, and other chemical use at both farm and non-farm sites to assess how these factors may be associated with contamination of groundwater, well-water sources. First-stage results for the major questions are summarized on Table 78. Of primary interest are any apparent associations with pesticide contamination. The confounding aspect of these assessments is evident by the number of questions with significant associations with the proportions of total coliform bacteria detections, for which there is no logical, or necessary relationship.

Farm sites that did not apply any herbicides during the recent growing season showed slightly less pesticide detections and slightly less proportions of wells with >10 mg/L, NO_3-N than those that did apply herbicides. They also showed slightly greater proportions of coliform positives.

For the range of questions dealing with pesticide application, some associations stand out, but they do not provide a consistent, or logical pattern. For example, if local mishandling of herbicides is a major factor in well contamination, sites where <u>all</u> herbicides are applied by a custom applicator could be expected to show less incidence of herbicide contamination. However, these sites exhibit proportions of detections equal to the state average. Sites where some, but not all herbicides are custom applied exhibit slightly greater proportions of detections, particularly when compared to sites where the farm operator noted that they applied all of their own

Table 78. On site pesticide/chemical handling and water-quality data. + and - indicate relative proportions that are between 1 to 5% above or below the state-wide 95% confidence intervals; + + or -- indicate proportions >5% above or below the 95% confidence intervals.

Observations:	Proportion of sites in category %	Wells with total coliform detected %	Wells with >10 mg/L nitrate-N %	Wells with any pesticide detection %	Wells with any atrazine detection %
State-wide proportion					
from total SWRL sample:	100.0%	44.6%	18.3%	13.6%	8.0%
Wells <50 feet deep:	27.9%	71.5%	35.1%	17.9%	9.6%
		RELATIVE state-wide proportions in each category:			
Did you apply herbicides during the most recent growing season?					
yes:	60%	42%	20%	15%	10%
no:	7%	52% +	13% -	11% -	4% -
Do you apply all of your own herbicides?					
yes:	29%	44%	23% +	15%	9%
no:	44%	28% --	13% -	11%	8%
Apply some, BUT not all of your herbicides?					
yes:	15%	40%	18%	18% +	14% +
Do you have ALL your herbicides custom applied?					
yes:	16%	42%	18%	14%	8%
Do you have some, BUT not all, herbicides custom applied?					
yes:	15%	41%	19%	20% +	16% + +
Mix all herbicides used, at site:	28%	45%	24% +	15%	9%
Don't mix all herbicides used, at site; mix some;	14%	36% --	14% -	18% +	13% +
Mix herbicides within 15 ft of well;	3%	53% + +	7% --	23% + +	13% +
Mix herbicides at hydrant near well, BUT > 15 ft;	25%	37% -	15%	16%	8%
Mix herbicides in field where applied;	22%	38% -	23% +	18% +	12% +
Mix herbicides - other;	37%	48%	20%	14%	9%
Rinse equipment within 15 ft of well;	2%	46%	6% --	18% +	12% +
Rinse equipment at hydrant near well, BUT > 15 ft;	16%	38% -	17%	18% +	12% +
Rinse equipment in field where applied;	18%	40% -	18%	15%	11%
Rinse equipment - other;	18%	52% +	21%	13%	9%
Non-farm, rinse at faucet near house, or from hose on faucet;	1%	43%	6% --	6% --	0% --
Sites with reported pesticide spills or back-siphoning accidents;	5%	32% --	27% + +	29% + +	14% +

herbicides.

Similarly, sites where all herbicides used are mixed at the site exhibit fewer pesticide detections than sites where only some of the herbicides are mixed on site. Farm sites (and non-farm sites) where herbicides are mixed within 15 feet of the well do show greater proportions of pesticide detections -- 23% of sites, compared to 14% for a state-wide average for all sites. Sites where pesticide application equipment is rinsed within 15 feet of the well exhibit slightly greater detections as well. These examples illustrate how multifactor analysis will assist further understanding, because most of these sites are one in the same. While there is an apparent association here, the proportion of wells potentially influenced by these factors is quite low, about 3%, state-wide. Also, sites noting that equipment is rinsed at a hydrant near the well, *but >15 feet away* (distances reported range up to several hundred feet) and farms reporting that herbicides are all mixed in the field where they are to be applied exhibit the same higher proportions of pesticide detections.

The two most significant associations in these data are related to non-farm sites and farm sites with reported chemical accidents. For strictly non-farm sites where pesticides are used and equipment rinsed at a faucet on or near the house, show significantly less proportions of pesticide detections, and NO_3-N >10 mg/L, and no atrazine detections.

Well owners also reported pesticide or fertilizer spills near their wells at 4.7% of the sites state-wide. Other direct point sources of contamination include back siphoning accidents; these occurred at <0.8% of the sites. As noted, these percentages are nearly identical to those reported from other detailed inventories in local areas of Iowa. In aggregate, an estimated 5.5% of private drinking water wells in Iowa have experienced one or more agricultural chemical spills near the well. A greater proportion of these sites exhibit pesticide detections and high nitrate concentrations than the average state-wide, as might be expected.

At the locations where pesticide spills within 15 feet of the well were reported, seven (7) of the wells contained one or more pesticides. When a specific product name was provided, only two of the seven wells containing pesticides actually contained the product named in the spill. At two of the three sites reporting a fertilizer spill, the concentration of nitrate-N observed in the wells exceeded health advisory levels of 10 mg/L.

The correlation between reported back-siphoning accidents and observed pesticide contamination in the well during this study is better. Two of the four wells contained the product reportedly involved in the incident at rather high concentrations, as discussed in the Pesticide Detections section. The other two incidents occurred in 1975 with no traces of the named products (atrazine or alachlor) detected during the SWRL study.

Well Construction Factors

Well construction characteristics and well location have both direct and indirect associations with water-quality. In the discussions above, for example, while there were several apparent associations among landuse, chemical use, and well siting factors and water-quality results, few are stronger than the general association between shallow wells (<50 feet deep) and water quality. In this section, other characteristics of the wells inventoried will be reviewed in relation to water-quality findings.

Certain types of wells, and the type of pumps that the well design might use, are often restricted to certain hydrogeologic settings or depths. A well that might use a "suction" type of pump is limited by gravity in the depth from which it can lift water, for example. Table 79 summarizes the 3 basic categories of wells used for private water supplies in Iowa. "Other" types constitute <1% of systems, state-wide. Sand-point wells constitute only about 4% of wells in Iowa. They consist of a "sand-point," a slotted metal screen (typically 2-10 feet long), with a hardened pointed tip that is attached to a metal pipe above. The well-screen and pipe are simply driven into the ground to the desired depth, much like a metal fence post. Such wells can only be used for shallow water supplies, and are typically 10 to 30 feet deep. They are not common, because they are restricted, most typically, to alluvial aquifers along streams, areas with saturated, shallow sand or gravel aquifers.

Drilled, small diameter wells are the most common type of well used in Iowa, comprising 62% of private wells. As defined for this inventory, small diameter drilled wells were considered to be those drilled wells with diameters of <18 inches. Such wells always have some length of casing which forms the wall of the well. Wells drilled into bedrock may not always have casing continuing very far into the bedrock; the well bore may simply be left open in the bedrock, allowing water to flow into the well throughout the open interval. In some bedrock wells, and in wells drilled into uncemented deposits, such as loose sand and gravel, the well usually has a continuous pipe casing, with a slotted well screen as the final section. The area around the screen is typically packed with sand or gravel. The space between the casing and the borehole opened by drilling is typically filled (grouted) with cement, bentonite clay, drill cuttings, and/or a mixture of these. A properly constructed well should have the bore hole grouted using clay or cement, with at least the uppermost portion of the casing shaft cemented in place. Such wells may be of any depth, but are most typical of bedrock wells and wells drilled into horizons that produce enough water that the diameter of the boring is not a limiting factor.

Large diameter wells, those >18 inches in diameter, are typically the seepage wells discussed earlier in this report. Such wells comprise 34% of private wells in Iowa; they are also drilled, or bored, using very large diameter equipment or augers. They typically are 2.5 to 3.5 feet in diameter, and use large round cement pipe as their casing material. Such material generally comes in sections that are 2 to 3 feet in length, which are stacked in the well boring. Water seeps into the well between each section, or in some cases, the pipe sections are perforated. For some wells, large diameter stainless steel screens have also been used, typically in sandy deposits, or with a back-filling of sand or gravel around the screened portion. The large diameter of these wells, as earlier noted, is to allow a large contact area for seepage into the well. This type of design is used in materials not generally considered an aquifer, such as fine-textured glacial deposits of till and loess. Their large diameter, and depth also allow seepage to be stored in the well. Water is typically pumped from the well to another storage tank (or cistern), pressure tank arrangement and into the farm distribution system. Even though such a well may only be capable of producing water at a limited rate, when coupled with the storage, this provides enough water for most private rural homes and farm operations.

Seepage wells are typically used where there are no dependable aquifers at a reasonable depth. By design, such wells are often open to water seepage from the just beneath the land surface to maximize seepage. This is particularly true for older seepage wells. Older seepage wells are sometimes lined (cased) with stone, clay tile, or clay bricks, which allow seepage throughout. Some are sealed for the first 5 to 20 feet, and only allow seepage below that depth. Newer large diameter wells often use a "buried-slab" design, which seals off the large diameter boring at depth, often at depth below the water table, and only extends a narrow diameter casing to the surface for access and maintenance. Such a design would utilize a submersible pump and pitless adapter to pump the water from the well. Such large diameter wells are most typically 25 to 45 feet deep, but some do extend to 100 feet.

Such large diameter wells are sometimes referred to as "dug" wells, often with the inference that these are old, hand-dug wells. In reality, few wells still in operation in Iowa are truly dug wells. This has become a colloquial expression, and one can often hear reference to "digging" a new well while watching a modern drilling rig in operation.

The first well on many older farm properties were indeed hand dug. But most of these wells were less than 25 deep and used a bucket for retrieving water, or later a simple "pitcher" pump, or "jack-handle" pump. Such arrangements cannot lift water more than about 25 feet. These wells were very undependable in dry years. The drilling of wells has been in practice since the 1870s in Iowa. Many farm wells were drilled in response to the droughts of the late 1800s and the 1930s. Another impetus for newer and deeper farm wells was the emphasis placed on sanitation and health and the definition of methemoglobinemia in the 1940s. Nearly 75% of the private wells in use for water supplies have been drilled since World War II, with 40% drilled since 1970. As noted earlier the median year for development of rural private wells in Iowa is 1960. Only 20% of the primary wells in use where developed before 1940.

Table 79. Proportion of well types used for private water-supplies in Iowa; from SWRL staff site inventories and inspection.

Type of well construction:	sand-point (driven)	drilled shaft (small diameter, <18 inches)	large diameter (drilled, "bored," or "dug")	Median well-depth feet	% wells < 100 ft deep
	- - - - Percentage by hydrogeologic region		- - - -		
State-wide:	4%	62%	34%	110	49%
Northeastern (1):	2%	97%	< 1%	190	9%
Eastern (2):	5%	81%	14%	130	33%
South-Central (3):	3%	25%	73%	40	79%
Southwestern (4):	9%	31%	59%	40	80%
Northwestern (5):	9%	29%	63%	40	74%
North-Central (6):	<1%	80%	20%	130	39%

As noted earlier, the types of wells in use in Iowa are related to the hydrogeology and water availability across the state. In southern and western Iowa, large diameter, bored seepage wells dominate, comprising 60% to over 70% of the wells in use (Table 79). In eastern and north-central Iowa, where regional aquifers are readily accessible, drilled wells dominate, comprising 80% to 97% of the wells in use. The collective proportion of large diameter wells and sand-point wells correlates significantly with the average well depths in the regions (Table 79). Hence, there is an expected inter-correlation between well-type, well-depth, and the resultant water-quality.

This association is apparent on Table 80, which summarizes water-quality data in relation to the well types and other well-construction features. The sand-point and large-diameter seepage wells show significantly greater proportions of contamination than the total state averages in most categories, with values very similar to the state averages for wells <50 feet deep. The exception is for the proportion of total coliform positives from the sand-point wells which are significantly lower than the state average. This is likely because sand-point wells are in settings where the water table is very shallow, often 2-3 feet below the land surface and the sand point is easily driven a considerable distance below the water-table. Conversely, drilled shaft wells show slightly to significantly less contamination in most categories, because they tend to be considerably deeper.

There are also some expected associations of water-quality with other well-construction, water-system features. These could be expected because certain features are typical of the water-system developed with certain types and/or ages of wells. Wells with above ground pumps (or with pump jacks) are associated with greater proportions of wells contaminated with coliform, high nitrates and to a lesser degree, pesticides. Such pumps tend to be in use on older and, to a degree, more shallow wells. Sites with pitless adapters tend to show less contamination, and typically are associated with deeper, drilled shaft wells. The presence of unsealed frost pits around a well head was associated with greater proportions of contaminated wells than were sealed frost pits. This has been shown in many other studies, and in that context, what may be of note is that the wells with sealed pits were not significantly less contaminated than the state average. In future data analysis these site factors will be integrated with well type and depth to more clearly sort out all the inter-related effects.

A last category of observations is summarized on Table 81. The SWRL field staff, after their on-site inspection and inventory, and after interviewing the site participants, were asked to make a professional judgement from their observations: "Was it likely that surface water

Table 80. Well construction characteristics and water-quality data. + and - indicate relative proportions that are between 1 to 5% above or below the state-wide 95% confidence intervals; + + or -- indicate proportions >5% above or below the 95% confidence intervals.

Type of well construction:	Proportion of sites in category %	Wells with total coliform detected %	Wells with >10 mg/L nitrate-N %	Wells with any pesticide detection %	Wells with any atrazine detection %
State-wide proportion from total SWRL sample:	100.0%	44.6%	18.3%	13.6%	8.0%
Wells < 50 ft deep:	27.9%	71.5%	35.1%	17.9%	9.6%
		RELATIVE state-wide proportions in each category:			
sand-point wells:	4%	11%--	36% + +	23% + +	19% + +
drilled shaft wells:	62%	27%--	7%--	10%-	6%
large diameter wells:	34%	83% + +	38% + +	19% +	11% +
W/ pump above ground:	22%	61% + +	26% + +	17%	11% +
W/ pump-jack:	10%	77% + +	24% +	13%	8%
W/ submersible pump:	73%	39%-	16%	13%	7%
W/ pitless adapter:	45%	33%--	12%-	13%	7%
W/ suction line:	12%	68% + +	22% +	16%	9%
W/ sealed frost pit:	26%	49% +	18%	11%	7%
W/ unsealed frost pit:	12%	50% +	21%	21% + +	16% + +
W/ well-head sealed:	71%	34%--	12%-	11%	6%
W/ casing open in pit:	17%	79% + +	23% +	12%	10%
W/ casing above pit:	55%	38%-	15%	14%	10%

runoff could enter the well head, or well pit?" "Was it likely that shallow groundwater seepage could enter the well, or well head?" From their field assessment, was there any evidence that contaminants might enter the well, because of well-construction or placement features, or water-system layout, such as location of a cistern or storage tank? The proportion of sites generating "yes" answers are summarized on Table 81; about one-fourth of all private wells show signs that well-construction or well-placement factors may make them more susceptible to contamination. However, these factors include the assessment of well types --i.e., seepage wells and sand-point wells are in themselves evidence that the well is more prone to the entry of contaminated water. Again, it must be emphasized that these results are presented individually at this time. These are not additive factors; many of the same wells are counted repeatedly in each of the categories.

As would be expected, the wells in these categories show greater proportions of contamination. Again, however, for most categories the proportion of contaminated sites is less than, or not significantly different than simply for any wells <50 feet deep. For wells that may allow surface runoff or shallow groundwater into the well, the proportion of pesticide detections is significantly greater than the state average, but these proportions are just within the 95% CI for the wells <50 feet deep.

Table 81. Well-site observations and water-quality data. + and - indicate relative proportions that are between 1 to 5% above or below the state-wide 95% confidence intervals; + + or --indicate proportions >5% above or below the 95% confidence intervals.

Observations:	Proportion of sites in category %	Wells with total coliform detected %	Wells with >10 mg/L nitrate-N %	Wells with any pesticide detection %	Wells with any atrazine detection %
State-wide proportion from total SWRL sample:	100.0%	44.6%	18.3%	13.6%	8.0%
Wells <50 ft deep:	27.9%	71.5%	35.1%	17.9%	9.6%
		RELATIVE state-wide proportions in each category.			
Surface runoff can likely enter well pit / well head:	10%	68% + +	20%	23% + +	14% +
Shallow groundwater seepage can likely enter well, well head:	18%	68% + +	26% +	21% + +	13% +
"From your field assessment, is there any evidence of likely contamination from:"					
well-construction factors;	27%	65% + +	30% + +	16%	9%
well placement factors;	23%	64% + +	23% +	18% +	13% +
location of cistern;	3%	58% + +	11%--	12%	6%

There is also a high degree of inter-correlation among these factors and shallow wells that will be assessed in more detail during future analysis.

GROUNDWATER QUALITY AND WELL REMEDIATION

The SWRL site inventories identify that perhaps 25% of rural private wells may have construction or placement factors that could facilitate the entry of contaminants into the well. These wells exhibit greater proportions of contamination than the state-wide averages for the total SWRL data set, particularly for coliforms and nitrate (see Table 81). The conclusion could be drawn that significant improvements in water quality may be achieved by remediation of the construction/location problems observed at these wells. This is undoubtably the case for some wells. For example, the SWRL inventory data suggest that about 10% of the wells state-wide could be affected by surface runoff (Table 81). Where true surface runoff can actually enter a well, it may deliver relatively high concentrations of pesticides and bacteria, though typically this runoff would not contain significant nitrate (e.g., Hallberg et al., 1983; Hallberg, 1987). Elimination of surface runoff to wells therefore might lower concentrations or detections of pesticides and bacteria in the affected wells. In a similar fashion, redrilling and/or casing wells to greater depths, or repairing relatively shallow casing defects, would likely yield improvements in water quality in some wells. Data from Hallberg and Hoyer (1982), the SWRL data (e.g., Tables 67 and 68), and many other studies show a decrease in contamination with greater well depth, suggesting that a reduction in the amount of shallow groundwater produced by a given well should result in decreased contamination.

Perspectives On Well-Construction

In perspective, however, well construction or well placement are not CAUSES of contamination; the wells did not generate these contaminants. Certain factors of construction or placement, or the aging of materials (e.g., the rusting of a metal casing) and lack of maintenance, may allow relatively easy entry of shallow, contaminated water into the well, but if the contaminants were not in the environment they would not get into the soil water and groundwater, or the well. Remediation of well construction or replacing current wells with a deeper well would undoubtedly reduce nitrate and pesticide contamination in many locations, but this would do nothing to address the cause of the contamination to the groundwater system. Such remediation often merely masks the contamination in the shallow portion of the groundwater system. In many settings, groundwater flux is downward, recharging to depth, and eventually deeper wells will also be affected by surface-derived contaminants, just as shown by the greater depth of contamination in northeastern Iowa where deeper groundwater circulation occurs more readily than in other areas. Review of Iowa data has also shown that deeper, properly constructed wells in some areas have started to show increases in nitrate concentrations in recent years, as well (e.g., Hallberg, 1987). Considerable work has been done in Iowa and neighboring states, using properly-constructed water-supply wells, research wells and piezometers that clearly indicates that nitrates, pesticides, as well as petroleum compounds, and other chemicals, have leached into groundwater from routine practices of the past (e.g., Libra et al., 1984; Libra and Hallberg, 1985; Klaseus et al., 1988a,b; Hallberg, 1989a,b). Groundwater contamination is far from being a problem of well construction.

While the wells with some signs of construction and placement problems exhibited greater contamination than the total SWRL data set, they typically showed lower proportions of contamination than the composite data for wells <50 feet deep. The majority of the wells allowing runoff or shallow seepage to enter the well head are typically shallow wells, and many overlapping factors need to be addressed. As another example, systems with submersible pumps and pitless adapters are generally indicators of good well construction. Yet samples from these wells show the same proportions of pesticide contamination as the state average.

While case studies have indicated that many of the most acute situations (i.e., wells with high, persistent concentrations) are often related to local point sources or mishandling, this is clearly not always the case, particularly with shallow wells. Review of the SWRL sites where pesticide concentrations exceeded HALs is an example; only 25% of these sites can clearly be attributed to point sources of contamination. Also, while seepage wells show greater than average proportions of contamination, field surveys also show that most such wells are located in areas to enhance seepage, often quite distant from the farmstead or hydrants where handling accidents or even routine spills and rinsing occur.

Research Data

Many studies on Agricultural Experiment Station sites, using controlled management practices have contributed to our current understanding of the movement of nutrients and pesticides to groundwater. Reviews of controlled crop production studies that monitored tile-drainage water (i.e., shallow groundwater), from Iowa and elsewhere in the corn-belt (Hallberg, 1987, 1989a), have noted that the concentrations and mass of NO_3-N leached to subsurface drainage water increases in direct proportion to the nitrogen applied, at rates above about 50 lbs-N/acre (Baker and Laflen, 1983). At current rates of fertilization, it is not uncommon for the mass of nitrogen discharged in shallow groundwater, through tile drainage, to equal nearly half of the nitrogen applied in some years (Kanwar, Johnson, and Baker, 1983). Similarly, controlled plot studies have continued to verify that some of the commonly used herbicides, that are not highly adsorbed to the soil, can and do leach into shallow groundwater. Ongoing controlled plot studies from Experiment station sites in Iowa (Czapar, 1990), Illinois (McKenna et al., 1989), and the USDA National Agricultural Research Center (Gish et al., 1989; Helling et al., 1988; Isensee et al., 1988) indicate similar results: that concentrations of 0.1 to 5.0 ug/L of herbicides such as atrazine, alachlor, or cyanazine, move through the soil and into the shallow groundwater beneath these fields; concentrations similar to those found in shallow wells in the SWRL study. Much greater concentrations can result when rainfall and the resulting water movement through the soil occur shortly after herbicide application. Concentrations noted in shallow groundwater by these studies were in the 10 to 100 ug/L range. These studies were using typical rates of application, under controlled conditions. Hence, it should no longer seem a surprise that nitrates and pesticides are found in many drinking-water wells designed to tap water from at or near the water-table.

Well Remediation

While some degree of groundwater contamination has been increasing over broad areas from routine activities, that does not minimize the need to properly construct new wells or properly maintain or remediate old wells. Sealing of frost pits, conversion to submersible pumps and pitless adapters, or buried slab construction, coupled with grouting and casing maintenance can help to improve the quality of many water supplies, and help protect them from some aspects of contamination, as noted. Particularly in Eastern and Northeastern Iowa, deep bedrock wells with shallow casing should be avoided; casing should be extended to depth to eliminate the influx of shallow groundwater that may be contaminated. Similarly, wells should not be left open to multiple formations where this may allow downward movement of shallow, contaminated groundwater into deeper aquifers.

Wells that are affected by surface runoff involve the greatest problems from health considerations. Wells that are "open" to the surface environment risk significant, and rather direct contamination from a wide variety of sources and parameters of concern: spills of chemicals, petroleum products, or other materials; viral and other biological pathogenic contamination that may not survive movement through the soil; and routine agricultural practices and use of chemicals. Remediation of these wells often involves little beyond repair and sealing of the well-head; raising the top of the casing or curbing a foot or two above the landsurface; and mounding the area around the well with soil. Wells identified as likely taking surface water showed greater contamination than the state-wide averages (Table 81). However, few of these wells likely received any surface runoff during the SWRL survey because of the drought conditions. The potential to receive surface runoff does not cause contamination, actual runoff does. A majority of these wells also take shallow groundwater, as well.

Wells that produce shallow groundwater also present risks to public health, from the same sources and parameters outlined above. As noted, field studies have shown that shallow groundwater below row-cropped fields often contain detectable herbicides and relatively high concentrations of nitrate (e.g., Baker and Johnson, 1981; Hallberg et al., 1986) and coliform bacteria (e.g., Hallberg et al., 1983). Remediation of wells that produce shallow groundwater is a complicated, and often expensive proposition, relative to the simple steps that may alleviate the entry of surface runoff. Some wells in this category are small-diameter wells with iron-casing that are drilled to depths of 100-300 feet, but have defects in the upper parts of the casing that allows the entry of shallow groundwater. In these situations, the replacement of the uppermost casing, coupled with grouting, may eliminate the effects of shallow groundwater. However, as monitoring from properly-constructed private wells and research piezometer wells has shown (e.g., Libra et al., 1984; Libra and Hallberg, 1985; Klaseus, 1988a; Thompson, 1990), this will not eliminate the presence of surface-related contaminants in many settings where they are transported to depth by the natural circulation of the groundwater.

Simple aging and deterioration of the casing materials are the cause of some of these defects. The SWRL findings showed that the age of Iowa's private rural wells were younger than expected; over 40% have been installed since 1970. However, another 40-45% are 30 years or more ago and some maintenance is undoubtably needed.

The majority of private wells in Iowa that produce shallow groundwater are the relatively large-diameter "seepage" wells described in preceding sections. As discussed, these wells are *designed* to produce shallow groundwater from fine-grained, slowly permeable glacial deposits, and are employed, to a large degree, in areas where there are no reliable deeper aquifers available. Improvements in water quality for these wells may occur if the they are sealed to some depth below the water table. Typically, sealing to about 20 feet is suggested. In particular, this would likely remove many detections of coliform bacteria that enter such wells from the uppermost saturated zone particularly if converted to a buried slab design. Such remediation *may* decrease nitrate and pesticide concentrations, but only if the natural circulation of groundwater does not deliver these contaminants to depth below the water table. In many instances this has not eliminated chemical contaminant problems. Sealing of seepage wells to depths greater then 20 feet may increase protection from contamination, however, there are practical limitations. A sufficiently large interval of the well must be open to receive water if it is to produce adequately. In the fine-grained glacial materials that these wells are usually completed in, permeability generally decreases with depth; sealing of the uppermost saturated zone eliminates the materials most capable of yielding water to the well. These factors, and the relatively shallow depth of most seepage wells, limits the depth to which they might be sealed. Further, in such settings, water-levels may fluctuate to depths greater than the typical seal. As the water level drops, the well can once again draw water from near the water-table.

In summary, well remediation would likely result in improved water quality at some of the wells identified as having construction/placement problem. In particular, wells that can receive significant surface runoff are at risk of direct contamination from the landsurface. Repairing casing defects in small-diameter steel-cased wells, and sealing large-diameter seepage wells to some depth below the water table, would likely improve water quality in some wells. While the wells identified has having construction/placement problems showed greater contamination than state-wide SWRL averages, they also showed less contamination than the state-wide averages for wells categorized simply as less than 50 feet deep. As was noted in the previous section, overlapping factors are involved, in that most of the wells identified as taking seepage are also less than 50 feet deep. This again emphasizes the simple, yet over-riding importance of well depth in relation to the occurrence of contaminants.

COMPARISON WITH OTHER DATA

Questions frequently raised in the discussion of the SWRL data include: How do the SWRL findings compare to other sampling studies in Iowa? How does it compare with data from other states? There is no ready answer because the SWRL survey is one of the first comprehensive, statistical studies of its kind. However, for general reference the findings can be compared with other available data.

HISTORIC WATER-QUALITY DATA AND TRENDS

Past reports have summarized various nitrate data collected from the same well, water system, or surface water sites over time, with some analyses dating to the early 1900s (Hallberg et al., 1984b; Hallberg, 1987a; 1989a; McDonald and Splinter, 1982). These data have clearly shown that nitrate-N concentrations in groundwater, and perennial surface waters that are fed by groundwater discharge, have increased dramatically, primarily since the early 1960s, and that these increases are regional in character. The regional increases occurred concurrently with the increased acreage of row-cropping and increased rate of use of nitrogen fertilizers in Iowa. Similar data have been reported across the corn-belt and in intensive agricultural regions world-wide (e.g., Keeney, 1986; Hallberg, 1987b; 1989a; Power and Schepers, 1989; Strebel, et al., 1989; Jurgens-Gschwind, 1989).

Nitrogen is ubiquitous in the environment, however, and local sources of nitrate contamination have existed as long as society has concentrated organic wastes. Point source problems, related to organic wastes or other nitrogenous materials, such as fertilizers, have been known for many decades as well (see Hallberg, 1987a; 1989a). At times the occurrence of nitrate contamination from historic data is cited and misconstrued as an indication that there has been little change in the scope of such problems. While localized problems have long been known, it is the regionally significant increases in nitrate-N contamination of water resources that has raised the issue to the current point of public concern.

1930s Nitrate Data

One such historic data set was reported by Johnson and others (1946) in the Journal of the Iowa State Medical Society. They summarized 1,379 nitrate analyses from private well-water supplies submitted in 1939, and some from 1944 and 1945, that were analyzed by the UHL. The report, and references cited, notes that most of these analyses came from worst case situations: sites where problems had been reported; from truly hand-dug wells, particularly ones with no mortar and wood or brick linings; sites where tree roots were reported growing into the well; from wells next to outhouses and manure storage, and so forth. It is difficult to make comparisons to these data because there are few other details given. No well depths are reported, but from the descriptions and references, it is evident that the majority of the wells were quite shallow, clearly <50 feet deep, and producing shallow groundwater. Use of these data is problematic, also: various percentages cited in the text and referenced data make it unclear if all the concentration data are expressed as nitrate-N, as noted in the text, or whether some might be as nitrate. However, simply taking the data at its reported face value, some comparisons can be made.

For the entire 1939 data set reported, 17.3% of the analyses exhibited >10 mg/L, nitrate-N, slightly, but not significantly less than the SWRL findings of 18.3%. However, the nature of the wells in the data sets are very different. The 1939 data are comprised of 53% dug and bored wells, probably all effectively <50 feet deep, and 47% drilled wells. From the SWRL survey, only 28% of wells in Iowa are now <50 feet deep. While the depths of the drilled wells from the 1939 data are not known, a comparison can be made using the shallow dug and bored wells. For the dug wells, 33.5% of analyses were >10 mg/L NO_3-N; for the bored wells, 20.5% exceeded 10 mg/L; and in composite these shallow wells showed 28% >10 mg/L. These figures are all less than the SWRL findings. For SWRL, 35% of rural wells <50 feet deep were >10 mg/L NO_3-N.

It should not be surprising that many of these early hand-dug wells exhibited nitrate contamination, particularly since the majority of the wells were, as stated, worst case settings. As noted, it was over the next 2 to 3 decades that most of these dug wells were replaced by better and deeper wells, and outhouses and privies replaced with other forms of septic systems. The SWRL survey shows that the mean and median year of well construction for rural private wells currently in use was 1960; with about 80% of currently used wells developed since 1940.

A better defined set of data, and of similar vintage, was summarized by the Iowa State Planning Board in 1938. This report summarized various water chemical data from 1,357 samples, the majority collected in 1934 through 1936, but some are summarized from earlier reports. These samples were collected from public and private wells, of various construction, and considerable information about each site, such as well depth and aquifer, were also compiled. These data are summarized and compared with the SWRL findings in Table 82. The Planning Board samples reflect an emphasis that was placed on study and sampling of the deep and very-deep wells constructed in the state at that time. Yet there is a significant portion of the samples obtained from more shallow wells that provide for comparison with recent data. For wells <50 feet deep, the 1930s data exhibit a mean nitrate-N concentration of 1.5 mg/L, compared to 11.2 mg/L for the late-1980s SWRL data; data for wells 50-99 feet deep show a similar 10 fold increase. Even the SWRL median NO_3-N of 6 mg/L for wells <50 feet deep is significantly greater than the 1930s *mean*. In total, the historic data show a median well depth of 200 feet and a mean nitrate-N concentration of 0.5 mg/L, compared to SWRL data, with a median of 110 feet and 6.2 mg/L NO_3-N. The 1930s data do show the same expected decline in nitrate concentrations with well depth, expressed either as the means or in the maxima.

The 1930s Planning Board data were also compared with summary data compiled from UHL samples analyzed during the 1980s (e.g., Table 25). Statistics were compiled by county to compare regional variations in well depths and nitrate-N concentrations. Across the state there were 30 counties that had 1930s median well depths of <200 feet, and where the median well depth for the 1930s data and the 1980s data were: 1. either within the same 50 foot depth class as used in this report (e.g., Table 20); or 2. the 1930s data exhibited a more shallow ("worse case") depth. The mean nitrate-N concentrations

Table 82. Comparison of nitrate-N concentration data by well depths for data compiled through 1936 (Iowa State Planning Board, 1938) and the SWRL data.

	Data Through 1936			SWRL	
Well Depth feet	% samples	Mean nitrate-N mg/L	Max. nitrate-N mg/L	% samples	Mean nitrate-N mg/L
<50 ft	17%	1.5	14.4	28%	11.2
50-99 ft	13%	0.9	27.8	21%	11.0
100-199 ft	19%	0.4	10.0	28%	1.5
200-299 ft	14%	0.2	6.0	13%	1.3
$\geq$300 ft	37%	0.2	7.3	10%	2.3

for these counties for the 1930s was 0.6 mg/L; for the 1980s the mean was 9.6 mg/L. In no case did the 1930s NO_3-N data exhibit a greater mean or median (or maxima) than the 1980s data; the differences in means ranged from an increase of 2.9 to 23.8 mg/L NO_3-N from the 1930s to the 1980s. These summary data used "mixed" statistics by design to provide a conservative comparison. The median well depths were used, but the means were used for the nitrate-N concentrations, because the median (50th percentile) well depth is a better comparative indicator of the data distribution, because the mean is skewed by (and overemphasizes) high values. Hence, using the mean nitrate-N provides a 1930s reference skewed toward greater values. Even the 1980s median is greater than the 1930s mean for most of these counties. Nearly any comparison made between these data exhibit a 10-fold increase in summary nitrate data between the 1930s and the 1980s.

To reiterate, the concern with nitrate contamination in Iowa, and many agricultural regions, is not that the problem was unknown until recent years, nor that high concentrations were not present locally. The problem is manifested in the regional increases in concentrations and the extension of greater concentrations to depth. Nitrate has become a nearly ubiquitous contaminant in the shallow groundwater system.

Temporal Changes With Well Depth

In a discussion of UHLs water-quality findings over the years, Morris and Johnson (1969) noted that even though better constructed wells were being installed, and that well depths were increasing, their data continued to show rather consistent proportions of wells with >10 mg/L NO_3-N. Beyond the data summarized above, one other data set may be useful to illustrate. UHL data on NO_3-N from private water samples submitted in 1962 from Buena Vista County were reviewed by staff of the UHL, Iowa Geological Survey, and the University of Iowa Department of Preventive Medicine in 1963 and 1964. The data with reported well depths are summarized in Table 83 and compared to UHL data from Buena Vista County for the 1980s. In this western Iowa County the options for groundwater supplies are limited, and hence, a majority of wells tend to be shallow, large diameter seepage wells. In comparison, over the past 20-30 years the data *suggest*: 1. there has been a 10% shift from wells <100 feet deep to deeper wells, with the largest decrease (7%) in wells between 50 and 99 feet deep, and the greatest increase (5%) in wells >300 feet deep, which would primarily be into the Dakota sandstone aquifer; and 2. that there has been a continued increase in the nitrate-N concentrations in wells to 100 feet deep. The total 1962 county data show a median well depth of 92 feet, a mean of 50 feet; with a mean NO_3-N concentration of 13.3 mg/L, and a median of 4.0 mg/L. In the 1980s the county data show a median well depth of 110 feet, a mean of 64 feet, with a mean NO_3-N concentration of 16.9 mg/L, and a median of 5.4 mg/L.

LOCAL SAMPLING IN IOWA

Considerable water-quality sampling has been done in various areas of Iowa (e.g., Detroy et al., 1988; Detroy and Kuzniar, 1988; Kelley and Wnuk, 1986; Libra et al., 1987; Hallberg, 1989a, b). Most of these studies are local in nature and sites were selected to provide controlled hydrologic observations, precluding meaningful comparisons with the SWRL data. One study of comparable design was conducted in Floyd and Mitchell counties in 1986 and 1987 (see Libra et al., 1987; Hallberg et al., 1987; Hallberg, 1989a, b). In this study, 10% of all rural farm wells were systematically selected for repeated sampling, in what became a fore-runner of the SWRL design. The results of this study showed that the proportion of wells with pesticide detections ranged from 29% to 42%, during individual quarterly sampling; for the entire year 60% of all wells exhibited detection of at least one pesticide during at least one sampling period (Hallberg, 1989b). About 19% of the wells exhibited NO_3-N >10 mg/L. As shown earlier, data from such a more localized area exhibit much more clearly defined predictive relationships between water-quality parameters such as nitrate concentration and pesticide detections (Fig. 49).

Floyd and Mitchell counties are in the SWRL Eastern Hydrogeologic region; the values for the detailed sampling in these counties are much higher than the summary results from the Eastern Hydrogeologic region, but as noted earlier, such data are not strictly comparable. While the SWRL data are applicable for regional and state-wide summary, they cannot be representative of more local areas, such as a county. Also, the Floyd-Mitchell data were collected prior to the drought under which SWRL was conducted. As discussed earlier in this report, and indicated in Figures 8 and 10, detections during the drought period were considerably reduced in the Floyd and Mitchell counties area, which may, in part, explain the magnitude of these differences.

MONITORING IN OTHER STATES

Considerable monitoring has been undertaken in other states as well (e.g., Hallberg, 1989b), but again few studies are comparable to SWRL. Statistical surveys have been conducted in Nebraska and Kansas, but they were limited in scope and design compared to SWRL. Both surveys were solely of farm wells. Both states are less intensively farmed than Iowa, and have lower pesticide and fertilizer use as well.

Table 83. Comparison of UHL nitrate-N concentration data for private well-water supplies from 1962 and the 1980s from Buena Vista County.

Well Depth feet	1962 % samples	1962 Median nitrate-N mg/L	1980s % samples	1980s Median nitrate-N mg/L
<50 ft	49%	10.0	46%	18.0
50-99 ft	35%	5.1	28%	9.2
100-149 ft	3%	<	6%	1.0
150-199 ft	4%	<	3%	1.0
200-299 ft	3%	<	6%	<
≥300 ft	6%	<	11%	<

In 1985, the Nebraska Department of Health conducted a statistical sampling of 451 rural farm wells, but only in 21 counties in central Nebraska (Nebraska Department of Health, 1985). Approximately 8% of the wells had NO_3-N >10 mg/L, and 4% showed pesticide detections (3.2% with atrazine).

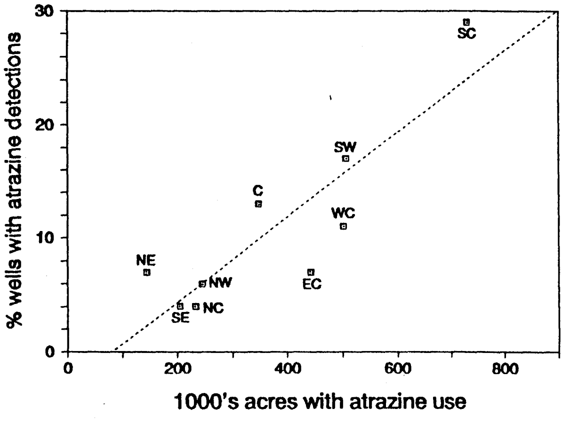
The Kansas survey was a state-wide design, but sampled only 104 wells in 50 counties (Koellicker et al., 1987; Snethen and Robbins, 1987a; Steichen et al., 1986). In this study, 28% of farms had NO_3-N >10 mg/L and 9% had pesticide detections (4.5% for atrazine). For eastern Kansas, where farming and pesticide use is more intensive, approximately 15% of wells had pesticide detections. In a second phase study, which targeted toward older and more shallow wells (Snethen and Robbins, 1987b) 32% of farm wells showed NO_3-N >10 mg/L, and 11% had pesticide detections.

The Minnesota Department of Agriculture and Department of Health have conducted a variety of cooperative studies. They sampled 725 wells from 1985 through 1987. These included observation/monitoring wells, and public and private water-supply wells; most were targeted toward agricultural areas thought to be susceptible to contamination, but some were distributed throughout the state to provide some overview. For all the wells sampled, 39% had detections of one or more pesticides, with atrazine occurring in 37% of the wells. Only 9% of the wells exhibited NO_3-N >10 mg/L. For the targeted private wells sampled, 46% had pesticide detections. The Minnesota surveys exhibit greater pesticide detections than the SWRL study, but this might be expected from a more targeted survey. Also, these studies were conducted under more normal climatic conditions than SWRL. The Minnesota data, overall, are more similar to the findings from the Floyd and Mitchell counties survey in Iowa.

The Wisconsin Department of Agriculture, Trade, and Consumer Protection has conducted a statistical survey of dairy well-water supplies (LeMasters and Doyle, 1989). During August, 1988, through February, 1989, water samples were collected from a random sample of 534 wells from dairy farms in Wisconsin. This population based survey found that 13% of dairy farms had wells with pesticide detections; 12% with atrazine; 10% of the wells exhibited NO_3-N >10 mg/L. These findings are quite comparable to the SWRL data. There are several differences in the surveys: the Wisconsin survey sampled only dairy-farm wells; and it did not include wells from some of most intensive row-crop areas, and heaviest atrazine use areas in the state. Climatic conditions were also closer to normal than in Iowa.

When summarized by crop reporting districts, the Wisconsin data show a very strong association, and significant linear correlation (p-value=0.002; r=0.87) between the proportion of wells with atrazine detections and the proportion of acres in the district treated with atrazine (Fig. 50). This might imply a logical relationship between overall use patterns and intensity and the detections in groundwater. The SWRL data, summarized by crop reporting district, do not show any such significant relationship between landuse, row crop acreage, corn acreage, or atrazine use and pesticide or atrazine detections (p-values typically >0.2; r values typically <0.2). This could be, in part, because well depth is such a dominant factor and there is such a large difference in well depths across the state. Linear regression models of the proportion of wells <25, <50, and <100 feet deep vs. the proportions of wells with pesticide detections, atrazine detections, or NO_3-N > 10 mg/L consistently produce significant results, p-values from 0.01-0.001, and correlation coefficients ranging from 0.55-0.95. This also raises the question of whether or not the Wisconsin data are fortuitous, or whether well depth is uniform enough across these areas to provide validity to the model? Or if well depth and crop reporting district exhibit some inter-correlation or co-linearity?

Figure 50. Linear regression relationship between the proportion of wells with atrazine detections and the crop acreage with atrazine use, for Wisconsin crop reporting districts (data from LeMasters and Doyle, 1989).



Additional state-wide surveys are being conducted in Wisconsin, that include private wells from all types of farms. In some areas, much greater proportions of wells are exhibiting detections. In Dane and Lafayette counties, for example, over 50% of wells sampled have shown atrazine detections (Wisc. Dept. of Agriculture, Trade and Consumer Protection release, June 1990). These data are analogous to results from detailed local studies in Iowa.

NATIONAL ALACHLOR WELL-WATER SURVEY

The Monsanto Agricultural Company has conducted a national survey of private well-water supplies, focused on areas of use of their herbicide alachlor (the active ingredient in trademarked products such as Lasso). Samples were collected from 1,430 wells, in 89 counties with high alachlor use in 26 states (Monsanto, 1990). Pesticides were detected in 13% of these wells, with atrazine detected in 12%; 5% of wells contained NO_3-N >10 mg/L. For farm wells, 10% had NO_3-N >10 mg/L. Regional data are not yet available, but the findings are quite comparable to the SWRL study. Nationally, about 0.8% of the wells sampled contained alachlor, compared to 1.6% of wells in Iowa, as shown through the SWRL survey. In addition, 0.4% of Iowa wells showed detections of hydroxy-alachlor; as discussed, this is a minimum level because methods research showed that most hydroxy-alachlor was likely lost in processing.

SUMMARY

As part of the implementation of the Iowa Groundwater Protection Act of 1987 the Iowa Department of Natural Resources (DNR), in conjunction with the University of Iowa, Center for Health Effects of Environmental Contaminants (CHEEC), conducted a survey of the quality of private drinking-water supplies used by rural Iowans. Overall responsibilities for project management were shared by co-principal investigators from the DNR and The University of Iowa, Department of Preventive Medicine and Environmental Health. The State-Wide Rural Well-Water Survey (SWRL) was conducted between April 1988 and June 1989, under pronounced drought conditions. This report reviews the design, the hydrologic conditions during the survey, and summarizes the basic water-quality results.

STUDY DESIGN

The primary objectives of SWRL were to answer two questions: 1. What proportion of the private rural wells in Iowa are affected by various environmental contaminants? 2. What proportion of rural Iowa residents are utilizing well water containing various environmental contaminants?

To provide a statistically valid framework to answer these questions, a systematic sample, stratified by rural population density, was designed using every 5-minute intersection of latitude and longitude to provide an independent. grid. This scheme selected 698 sites, and the drinking-water well closest to each selected intersection was chosen as the primary target for sampling. Iowa State University Cooperative Extension Service (CES) county staff identified eligible participants, based on design criteria. Less than 8% of rural Iowans contacted were unwilling to participate.

Temporal variability of groundwater quality during the survey was addressed by two elements of the survey design. The first element chose 10%, or 68, of all selected sites for a one-time repeat sampling during a different season. The second element divided Iowa into six general hydrogeologic regions with broadly similar soil, landscape, and geologic characteristics. These characteristics affect the general nature of the susceptibility of aquifers to contamination, well construction practices, and water availability. A county or counties typifying each of these regions was selected, and the 62 sites within these counties were sampled quarterly. Also, the SWRL field sampling schedule did not allow more than two contiguous counties to be sampled within four weeks of each other. The final SWRL sample of raw well water was 686 sites. Overall, 92 of 99 counties were sampled at 100% of the initial design. Ninety-four percent of the 10% repeat sites were resampled, and 93% of the quarterly sites were sampled four times. In total 1,048 well water samples were collected and analyzed during SWRL.

Standardized procedures for field activities were employed during SWRL. Information about the wells selected for sampling, local landuse, farming practices, chemical use, characteristics of the residences associated with these wells, and basic health information on the participating residents were collected using questionnaires designed specifically for SWRL. Sampling points were chosen as close to the well as possible while avoiding water treatment devices. Wells were purged prior to sample collection, until repeated temperature and specific conductance measurements stabilized. Alkalinity, pH, and dissolved oxygen concentrations were determined in the field. Field-related QA/QC (quality assurance and quality control) protocols included blank, spiked, and blind-duplicate samples. Custody forms documented the movement and custody of each well-water and QA/QC sample, from each laboratory to the field and back.

All primary well-water samples were analyzed for total coliform bacteria; nitrate (+nitrite)-N, ammonia-N, and organic-N; major inorganic ions; 27 commonly-used pesticides; and selected pesticide metabolites. Analyses were performed by three laboratories: the University Hygienic Laboratory; the Analytical Toxicology Laboratory, with the UI Department of Preventive Medicine and Environmental Health; and the Environmental Engineering Laboratory, with the UI Department of Civil and Environmental

Engineering. Methods were developed and tested for additional pesticide metabolites, for organic and toxicity screening of groundwater, and for fecal coliform bacteria during the project. Therefore, not all samples were analyzed for these parameters.

The labs had U.S. EPA QA/QC plans in place, and the SWRL plan utilized and verified their implementation. The Final Work Plan and Quality Assurance Plan for SWRL included procedures for field work and data management. Different facets of the program were audited both quarterly and following completion of specific work efforts. No problems were encountered that significantly affected the quality or integrity of the SWRL data.

DROUGHT EFFECTS

Interpretation of the SWRL results requires an understanding of the climatic and hydrologic conditions that prevailed during the survey. The survey was conducted during the driest consecutive two-year period on record in Iowa. The state-wide average precipitation was about 14 inches below normal during the April 1988 through June 1989 period of sample collection. Only three months during SWRL showed above normal precipitation, and only one of these was more than one inch above normal. Most months showed precipitation deficits of over one inch.

In many areas, from a perspective of groundwater recharge, the drought began in 1987. As the drought progressed, recharge was very limited; the water table declined in elevation, which resulted in declines in water levels in wells and the discharge rates of rivers and springs across the state during this period. The lack of recharge was prolonged enough that it also affected water levels in deeper bedrock aquifers.

The SWRL results may present a "best-case" situation because of the temporal coincidence with the drought. With reductions in recharge, the movement of contaminants to groundwater was also reduced. Longer term monitoring studies indicate that the frequency of wells with pesticide detections or with high nitrate concentrations were lower during the period of SWRL than during past periods. Hence, the SWRL sampling likely presents a conservative estimate of the proportion of wells affected by various contaminants. Never the less, the SWRL study provides an important base-line measure of the condition of the state's rural water supplies. Use and interpretation of these results must keep in mind the pronounced drought conditions.

Seasonal changes, as well as drought-related changes in water quality are evident in the SWRL results. These variations in water quality were not significantly different in direction or magnitude among the six hydrogeologic regions, and the SWRL regional data are directly comparable.

DEMOGRAPHICS

Comparison of traits of the SWRL participants shows that the population and the farm sites inventoried were a typical sample of rural Iowans. Of the SWRL sites, 69% were sites where residents operated the surrounding farm, and 31% were non-farm, "suburban" sites or sites where the residents were not farming at the immediate location. The SWRL demographic and farming practices data are very comparable to other state-wide surveys and census statistics. The comparative results for many factors, ranging from farmer age distributions to farm chemical usage, indicate that the SWRL population is clearly a representative sample of rural Iowans and of Iowa farmers and their agricultural practices.

WATER QUALITY RESULTS

The SWRL well-water data provide a population based summary of the drinking water used by rural Iowans. The data also provide a cross section of the quality of Iowa groundwater. The variations in water quality exhibited in the SWRL data, both regionally and particularly with depth, show consistent and predictable geochemical patterns, related to contaminant sources and transport, and age effects.

Average values for general parameters for Iowa well-waters were: temperature, 12°C; pH, 7.1; hardness, 430 mg/L, and alkalinity of 200 mg/L (as $CaCO_3$). These values show that Iowa well-waters are near neutral and on average have low to moderate values for total dissolved solids.

Table 84 summarizes the water-quality parameters of greatest interest related to health concerns and for inferences about geochemical processes in the groundwater system.

The data reported are population-weighted values. When comparison summaries are presented only data from sites that have analytical results for all the water-quality parameters compared are used. This avoids any aberrations related to missing data. As a consequence, however, the relative values presented may differ slightly among different sections of the report.

Regional Hydrogeology, Well Construction, and Water Quality

To understand the distribution of contaminants shown by the SWRL survey requires an understanding of the regional availability of groundwater in Iowa and, related to this, the nature of wells used. Considerable past work has shown that a depth of 50 feet serves well as a definable approximation between "shallow" and "deep" aquifer regions, and between shallow and deep wells that exhibit significant differences in water quality for surficially derived contaminants, such as nitrates and pesticides.

Well depth is clearly one of the most important variables affecting the quality of the derived well water. The depth (of the well and its casing) affects the potential for contaminants from land-surface activities to enter a well. The shallow portion of the groundwater system is affected more immediately and significantly by contamination related to land-surface activities than deeper parts of the system. Hence, shallow wells should be expected to exhibit a much greater degree of such contamination. Poor construction or maintenance may allow the contamination of a deep well through the entry of younger, shallow groundwater into the well. If accidents, such as well head spills, chemical disposal or back-siphoning are the cause of most contamination, such a relationship to well depth would not, of necessity, be apparent because such factors can affect wells of any depth. Deeper groundwater is typically much older and unaffected by recent contaminants. It has spent more time traveling in contact with various geologic materials and, thus, has had more time to dissolve minerals. Hence, groundwater from deeper wells tend to have higher dissolved solids concentrations, which may also impair their use.

State-wide, private wells exhibit a mean depth of 136 feet, a median of 110 feet, with a maximum of 880 feet. As the maximum depth suggests, there is considerable variation. Much of this variation in well depth occurs between regions in Iowa, related to variations in the hydrogeologic setting. Approximately 28% of private wells state-wide are <50 feet deep; 49% are <100 feet deep and 51% are ≥100 feet deep; and about 10% are ≥300 feet (Table 84).

The proportion of wells <50 feet deep varies from about 6% in Northeastern (NE) Iowa to over 50% in the South-Central (SC), Southwestern (SW), and Northwestern (NW) regions of Iowa. In the SC, SW, and NW nearly 75% of all wells are also <100 feet deep. These well depths reflect the regional hydrogeology. In NE Iowa bedrock aquifers occur at a shallow depth, but the local relief is the greatest in the state. Hence, the potentiometric water surface can be deep. In the NE wells tend to be deeper than in other regions, but because of the hydrogeology many wells are still drawing young water from the top of the groundwater system because the depth of groundwater circulation is much greater in NE (and portions of Eastern [E]) Iowa. In this region the relationships between water quality and well depth would be less well-defined, because of the greater depth of groundwater circulation and related well construction considerations (e.g., deep wells, but shallow casing).

The SC, SW, and NW regions are dominated by shallow wells because regional aquifers are often very deep, and hence, drilling may be prohibitively expensive. The water in these deeper aquifers also can be undependable, partly because natural water quality may be poor (e.g., high sulfates, high dissolved iron, etc.). Hence, the southern and western regions are dominated by "water-table" wells; wells which by design tap the most shallow portions of the groundwater system. Such wells often are not drilled into an aquifer, but are simply completed in the low-permeability glacial deposits, which in a strict sense, would be considered an aquitard. In these settings large diameter "seepage" wells are installed, often only 30-40 feet deep.

The degree of contamination in Iowa is far greater in shallow wells. As shown on Table 84,

Table 84. Summary of SWRL water-quality results.

Water-Quality Parameter	All Sites	Wells < 50 ft Deep	Wells 50-99 ft Deep	Wells ≥ 100 ft Deep
Wells:				
% wells of known depth		28%	21%	51%
median well depth	110 feet			
Nitrate-N:				
% sites > 10 mg/L (HAL)	18%	35%	32%	4%
% sites > 3 mg/L	37%	67%	56%	15%
% sites > 0.1 mg/L	58%	88%	72%	35%
mean conc., mg/L	6.2	11.2	11.0	1.6
median conc., mg/L	0.6	6.0	4.3	<0.1
max conc., mg/L	100.	95.	100.	37.
Total Coliform Bacteria:				
% sites positive:	45%	72%	52%	27%
Fecal Coliform Bacteria:				
% sites positive	7%	8%	12%	2%
Pesticides:				
% sites with detections	14%	18%	14%	9%
% sites with ≥2 detections	5%	9%	5%	4%
% sites > HAL	1.2%	2.0%	1.4%	0.6%
Atrazine (total):				
% sites with detections	8%	10%	12%	6%
% sites > 3.0 ug/L (HAL)	0.6%	0.4%	1.4%	0.3%
mean conc., ug/L (of detections)	1.1	1.7	1.0	0.7
median conc., ug/L	0.6	0.9	0.6	0.3
max conc., ug/L	7.7	7.7	4.8	4.9
Fluoride:				
% sites > 2 mg/L	3%	1%	1%	4%
% sites > 4 mg/L (MCL)	1%	1%	1%	2%
mean conc., mg/L	0.3	0.2	0.2	0.4
max conc., mg/L	7.0	5.2	4.0	7.0
Sulfate:				
% sites > 250 mg/L	15%	7%	10%	22%
mean conc., mg/L	130.	88.	90.	190.
max conc., mg/L	1940.	730.	800.	1940.
Ammonium-N:				
% sites > 0.1 mg/L	45%	14%	35%	68%
mean conc., mg/L	0.7	0.2	0.5	1.2
median conc., mg/L	<0.1	<0.1	<0.1	0.6
max conc., mg/L	11.0	8.3	5.7	7.1
Dissolved Oxygen:				
mean conc., mg/L	4.2	5.2	4.4	3.6
median conc., mg/L	3.2	5.0	3.1	2.4

Cumulative Proportion of Sites With NO_3-N > 10 mg/L, and/or any Pesticide Detected, and:
and/or Total Coliform Bacteria -- 55%; and/or Fecal Coliform Bacteria -- 31%

the SWRL data indicates the major difference in well contamination occurs deeper than 50 feet; the most significant difference occurs between wells <100 feet deep and those ≥100 feet deep. Wells <100 feet deep comprise nearly 50% of wells state-wide, and these wells account for about 70% of total coliform positives, 80% of fecal coliform positives, 64% of pesticide detections and total atrazine detections, and 89% of wells with NO_3-N >10 mg/L.

This significant change in water quality is evident in every region, except the NE. Because of the deep groundwater circulation that occurs in the NE hydrogeologic region, there is not a pronounced depth relationship, and high nitrate concentrations, for example, extend to great depth. Similar settings provide greater depth penetration of nitrate in portions of the E region, as well.

The greatest proportions of contaminated wells occur in the SC, SW, and NW regions of Iowa, coincident with the predominant use of shallow wells, primarily seepage wells. While such wells are highly vulnerable to contamination from land-surface activities, most were installed years ago when such contamination was not a consideration. They continue to be installed today as a practical matter because, as noted, there may be few economic options for the rural resident in these areas.

As described, the variations in water quality exhibited in the SWRL data, show consistent and predictable geochemical patterns, related to contaminant sources, transport, and age effects. These patterns have been the subject of controlled studies over many decades. The SWRL data implies that the majority of rural wells are of sound construction; well depths reflect waters with a predictable geochemistry. This does not mean that all wells meet today's standards for construction or siting; indeed they do not. In perspective, however, neither well construction, placement, nor well depth are CAUSES of contamination; the wells did not generate these contaminants. Certain factors of construction or placement, such as depth, or the aging of materials (e.g., the rusting of a metal casing) and lack of maintenance, may allow relatively easy entry of shallow, contaminated water into the well, but if the contaminants were not in the environment they would not get into the soil water and groundwater, or the well.

The median (50th percentile) year of development for Iowa's rural private wells is 1960. Nearly 75% of the wells in use for water supplies have been drilled since World War II, with 40% drilled since 1970. Even so, all but the newest wells were drilled before the current concerns with groundwater contamination. Remediation of well construction or replacing current wells with a deeper well would undoubtedly reduce nitrate and pesticide contamination in many locations, but this would not address the cause of the contamination to the groundwater system. The sources of contamination must be addressed because these shallow groundwaters will be the recharge for deeper groundwater with time. Sanitary and structural improvements of private water-supply systems and improvements in new well development must also be a priority. Current standards for well construction are outlined in the Iowa Administrative Code, Environmental Protection [567], Chapter 49, *Nonpublic Water Wells*.

In summary, on a state-wide basis, about 75% of all the wells in the SC, SW, and NW regions are <100 feet deep; these areas contain nearly 65% of all the wells <100 feet deep state-wide, with seepage wells comprising 65-75% of wells in these 3 regions. In contrast, over 75% of all the wells ≥200 feet deep are located in the NE, E, and NC regions of Iowa; narrow diameter, drilled shaft wells make up 80-97% of wells in these regions. These general observations provide a basis for evaluation and understanding of the regional and depth variations in water quality.

Dissolved Solids: Ion Concentrations

The chemical profile of Iowa groundwater is dominated by the dissolved ions of calcium, magnesium, bicarbonate, and in some cases, sulfate. Mean concentrations for all ions, except chloride and nitrate, increase or remain fairly constant with depth. The highest concentrations of chloride and nitrate at shallow depths are related to their surficial sources. Agricultural applications are the largest contributor of nitrate and probably chloride (related to potassium chloride used for K fertilizers). Road salt use and leaching of geologic materials are other sources of chloride, as well.

Fluoride (F), sulfate (SO_4), and chloride (Cl) are ions (in addition to nitrate) for which drinking water standards have been set. F in drinking water is known to be beneficial at <2 mg/L for decreasing dental cavities, and F has commonly been added to public water supplies. The EPA has established a Maximum Contaminant Level (MCL) for F at 4 mg/L to protect against skeletal and dental fluorosis. State-wide, well-waters showed a mean concentration of 0.3 mg/L with approximately 1.3% of wells exceeding 4 mg/L (Table 84). Secondary standards for F have also been set at 2 mg/L to protect against dental fluorosis (tooth mottling and staining). Approximately 2.5% of Iowa private well-waters exceed this level. In rural areas F supplements are often recommended for children because it is assumed that they are not using fluoridated water. However, the drinking water should be analyzed to assess the natural F content before a supplement is prescribed, to avoid problems with dental fluorosis.

Secondary standards have been set for SO_4 and Cl, primarily for objectionable taste problems, though sulfates can also have a laxative effect. The standards for both are set at 250 mg/L. About 15% of Iowa well-waters exceed this concentration for SO_4; only 0.3% exceed the limit for Cl.

Ion concentrations vary regionally, with the highest values occurring in SC and NW Iowa, and the lowest average values occurring in NE Iowa. The NW and NC regions of Iowa exhibit the highest concentrations of both SO_4 and F.

Nitrates

Nitrogen is ubiquitous in the natural environment, and its movement and transformation is part of the natural functioning of any ecosystem. In a natural ecosystem, however, little nitrogen is wasted. The natural background concentration of nitrate in aquifers in Iowa is typically less than 2 mg/L, often less than 1 mg/L NO_3-N. Concentrations over 3 mg/L are typically considered indicative of anthropogenic pollution related to agricultural practices, fertilizer use, manure, septic tank wastes, sewage sludge, or other sources. Results of the SWRL study reinforce other Iowa data showing that nitrate contamination of groundwater is an important environmental problem, of potential significance for public health state-wide, and more pronounced within specific regions.

Approximately 18% of Iowa's private, rural drinking water wells contain nitrate at concentrations exceeding the recommended health advisory level (HAL) of 10 mg/L NO_3-N. It is noteworthy that 40% of private rural wells exhibited no detectable nitrate (i.e., <0.1 mg/L). Yet, nitrate contamination is a widespread problem; in every region but NC Iowa more than one-third of the wells exhibit >3 mg/L NO_3-N.

As noted the degree of contamination is far greater in shallow wells; for wells <50 feet deep, 88% have detectable NO_3-N, 67% are >3 mg/L, and state-wide, 35% are >10 mg/L. In the three southern and western regions over 50% of the shallow wells are >3 mg/L, ranging to 97% of the wells in NW Iowa. In these regions from 30-50% of wells < 50 feet deep have >10 mg/L NO_3-N.

Coliform Bacteria

The occurrence of total coliform-positives in 45% of private water supplies in Iowa is cause for concern because of the potential health implications and should not be understated. However, single, positive total coliform samples from groundwater supplies do not provide unequivocal inferences, and the implications should not be overstated either. Total coliforms are ubiquitous constituents of soils, surface water, and shallow groundwater and the mere presence of *total* coliforms in drinking water does not mean a water supply is contaminated with fecal material. Fecal coliforms are a subset of the total coliform group, **but** the two cannot be equated. As the comparative SWRL data show, only about 7% of water systems were positive for fecal coliform bacteria.

Well conditions should not be interpreted solely on the presence of *total* coliform bacteria in a water-supply sample, particularly a single sample. Any interpretation is confounded by the variance of the testing and the high percentage of positives that typically originate from some point in the water-system other than the well. Hence, "poor" well construction or maintenance, or "seepage" around the well head, cannot be an automatic interpretation. The only sound, general interpretation of a persistent presence of total coliforms is that the water-system is allowing interaction with soil, soil-water, shallow groundwater, or possibly surface water. This could indicate that the system might be prone to other forms of contamination. However, total coliforms are not good predictors of chemical contaminants.

When total coliform positives can be attributed to the water from the well itself (which appears to be the case in only about 1/4 of positives) it can be suggestive of well construction deficiencies, at least for deeper wells. The origin of total coliforms in shallow wells is commonly just the shallow groundwater environment itself, though well defects may also be involved. With today's water-quality concerns, such construction appears deficient, but these concerns were not apparent when most of these wells were developed.

The occurrence of total coliforms in shallow groundwater is reflected in the relationship between the total coliform positives and the well depth of the SWRL water systems. Approximately 72% of wells <50 feet deep were positive for total coliforms (but only 8% were positive for fecal coliform bacteria). About 70% of all the total coliform positives, and about 80% of the fecal coliform positives, came from systems with wells <100 feet deep. No systems using wells ≥200 feet deep were positive for fecal coliforms. The proportions of total coliform positive sites vary regionally, paralleling the use patterns of shallow wells and related well construction practices.

Pesticides

In total, 16 pesticide compounds were detected in the SWRL well-water samples, including 11 parent compounds and 5 environmental metabolites. Sixteen of the parent pesticides analyzed for were not detected. Multiple residues were detected in wells in all regions of the state.

The mean concentrations of these pesticides were generally less than one part per billion; typically below recommended lifetime health advisory levels (HAL). Lifetime HALs were exceeded in an estimated 1.2 % of the private, rural drinking-water wells in Iowa. This included eight sites: five with atrazine, two with alachlor, and one with trifluralin.

These sites exceeding HALs for pesticides occur throughout the state. The sites are dominated by shallow wells or water sources (88% of sites); one deep well was involved, and this was a point source case which could affect any depth of well. Two of the sites, 25%, are clearly "point source" cases, a spill or back-siphoning accident (alachlor and trifluralin); the majority, 62.5%, are probable nonpoint sources related to pesticide occurrences in shallow groundwater (alachlor and atrazine); 1 case, 12.5%, is equivocal (atrazine).

Additive Water-Quality Effects

Individually, NO_3-N >10 mg/L occurred alone in about 4% of the water supplies state-wide; pesticides were present alone in about 5%. Total coliform positives occurred alone at 27% of the sites, which is over 60% of the total coliform positives. In a cumulative sense, these three contaminants, individually or in combination, were detected in nearly 55% of rural private water supplies. If fecal coliform occurrences are used in lieu of total coliforms, this reduces to about 30% of well-water supplies. No sites with fecal coliforms had pesticide detections, and only about 2.5% were positive for both fecal coliforms and NO_3-N >10 mg/L.

Environmental Health Implications: Population Exposure

The SWRL study was designed to allow a population exposure estimate to be made. Based on 1980 Census data, about 130,000 rural Iowa residents (or about 17.9% of the rural population) are consuming drinking water from private, rural wells with concentrations of >10 mg/L, NO_3-N, the recommended health advisory level. Based on the rural population census, about 94,000 rural Iowa residents (or about 13.1 percent of the rural population) are consuming drinking water that contain one or more pesticides. About 5,400 rural residents (or about 0.7 percent of the rural population) are consuming drinking water from wells that contain a pesticide concentration above an HAL.

These percentages are slightly less than the percentage of contaminated wells noted because some rural Iowans do not use their wells as their primary source of drinking water. Rather they obtain their drinking water from rural water systems (RWS). In portions of the state RWS utilize surface waters. Past surveys have indicated these systems often have pesticide concentrations greater than those found in groundwater.

At ten SWRL sites RWS water was sampled, along with the most often used well on site. These sites were included in SWRL because all the sites contacted, within the primary target area, used the RWS for their primary drinking water source. (Only the well data is included in the SWRL analysis.) Herbicides were detected in 40% of the RWS samples. A maximum NO_3-N concentration of 5 mg/L was measured in the RWS samples; median NO_3-N was < 1 mg/L, and no coliform bacteria were detected.

TEMPORAL VARIATIONS

Some temporal variations are evident in the SWRL data. In the midst of the drought, the fall and winter of 1988 and 1989 was a period of near normal precipitation, with some months slightly above normal. The peak of nitrate-N, pesticide detections, and total coliform positives was concurrent with the greater precipitation. These trends are unusual; most temporal monitoring would show peaks in the spring and early summer months, and lower or average conditions in the winter. The most pronounced trend that is apparent in the data, however, is the very low value of all parameters in the April-June, 1989 period, normally a time of peak values. This parallels the return of drought conditions in the late winter and spring of 1989. As seen in other studies, the appearance of contaminants in shallow groundwater appears related to precipitation and recharge patterns.

Upper Limits of Contamination: Inferences From Temporal Sampling

The added water-quality observations from repeated temporal sampling can also be used to assess an upper bound on the estimates of private-well contamination. Simply adding observations from the 10% and the quarterly repeat samples to the primary data, provides a measure of the proportion of wells state-wide that had detections over the course of one year. These data increase the state-wide proportions to: 1. wells >10 mg/L NO_3-N = 19% (compared to 18.3%); 2. wells with any pesticide detections = 18% (compared to 13.6%); and 3. wells with total atrazine detections (atrazine and/or metabolites) = 9% (compared to 8%).

Extrapolating from the temporal samples to the complete study data provides an estimate of an upper limit of wells likely to exhibit detections, at sometime over the course of 1 year (but not necessarily continuously), based on the climatic conditions during SWRL. This approach estimates these proportions: 1. wells >10 mg/L NO_3-N ≃ 21%; 2. wells with any pesticide detections ≃ 30%; 3. wells with atrazine detections ≃ 15%.

Potential Monitoring Network

The systematically selected 10% repeat sites provide a very consistent representation of the state-wide data, including proportionately representative detections of pesticides down to about a 1% occurrence interval. These wells can provide a very good subset for continued monitoring for assessing trends over time. Using both the 10% sites and the quarterly sites would improve the confidence interval of the state-wide estimates. In either case, such a monitoring subset would only be appropriate for state-wide summaries.

RELATIONSHIPS

Relationships among water-quality parameters were analyzed to assess possible predictive capabilities, contamination and transport mechanisms, and chemical changes occurring within the groundwater system. A first-stage analysis of the relationships among site inventory characteristics and water quality findings was also conducted, evaluating each factor individually. Few factors occur in isolation, however. Typically there are many confounding, and inter-related variables. This first-stage analysis will provide direction for further multi-factor analysis that will be performed for future reports.

Statistical analyses show significant associations between many water-quality parameters but the associations are not strong predictors based on state-wide data. By far the most significant factor explaining water-quality variations is well depth, as illustrated in Table 84. The strongest suite of associations between water-quality variables are relations among nitrate-N, ammonium-N, well depth, and dissolved oxygen (DO). As well depth increases the concentrations of NO_3-N and DO decrease, but NH_4-N increases. The consistency in these relationships with depth clearly suggest that nitrate reduction and/or denitrification occur with depth in groundwater systems in Iowa.

These relationships are an illustration of the consistency of the SWRL well data with the chemical evolution expected in the groundwater system. At shallow depth, oxidizing conditions dominate and NO_3 is the dominant N form. With greater depth, DO is depleted, reducing conditions are established, and NH_4 is the more stable form in the groundwater environment. It is not clear from these data if the deeper groundwater system has the capacity to denitrify the nitrate loads currently being delivered to the groundwater system, however.

Total coliform bacteria are not good indicators of other forms of contamination. While there are significant, but weak associations between total coliform and nitrate-N, there is no relationship between total coliform and pesticides; the best association is with well depth. As noted, 60% of all sites with total coliforms did not exhibit pesticide detections and/or NO_3-N >10 mg/L. If a prediction were to be made based on the presence of total coliform, the probability is better that these other contaminants <u>would not</u> occur in the water supply, than that they would occur. There is, however, an association among these contaminants. For water supplies that have NO_3-N >10 mg/L and pesticides, about 63% also had total coliforms. Similarly, for any site with NO_3-N >10 mg/L, about 70% also had total coliforms. Conversely, however, 70% of supplies that were total coliform positive did *not* exhibit NO_3-N >10 mg/L. For pesticides, sites having pesticide detections nearly split at 50-50 between those with and without total coliforms. Conversely, 84% of total coliform positive supplies did not have any pesticides detected. The apparent relationship between total coliforms and other contaminants, such as nitrate-N, is primarily a function of their co-occurrence (inter-correlation) related to well depth. As discussed, any shallow well, particularly if it is open to the water-table, will likely have total coliforms present, at least intermittently, from the influx of naturally occurring coliforms in the soil environment. Such shallow wells are also those most prone to other contamination from activities at the land surface. In short, high NO_3-N concentrations or pesticides are a moderately good predictor of the presence of total coliforms, but total coliforms are a very poor predictor of NO_3-N or pesticides.

The presence of sinkholes or agricultural drainage wells on the property near a well are two factors often associated with groundwater contamination. These factors are not common on a *state-wide basis*: sinkholes were identified in the vicinity of only 2.1% of sites and only 0.6% of sites were near agricultural drainage wells.

The presence of sinkholes on the property, or immediately adjacent land was not a good predictor of pesticide contamination of the private, rural drinking water supply. Only three of the 14 sites with reported sinkholes had pesticide contamination. Similarly, only two of the 14 wells had >10 mg/L nitrate-N. Neither proportion is significantly different than the state-wide average. *None* of the sites with reported agricultural drainage wells had any pesticide detections or nitrate-N >10 mg/L.

Tile lines discharging directly into wells were identified at <0.6% of sites, state-wide. Only one of these four sites had pesticides detected. Two metabolites of carbofuran were observed at low concentrations at this site. Nitrate concentrations >10 mg/L NO_3-N were not observed at any of the sites.

Landuse and Management

Aggregate landuse is, in part, too complex and, in part, too ubiquitous in Iowa to provide definitive relationships with water quality. A few trends are apparent. The most prominent is an association with non-farm, suburban housing tracts; proportionately, these areas show substantially fewer wells with >10 mg/L NO_3-N and total coliform bacteria.

Proximity to septic tank leach fields did not result in an increased proportion of nitrate contamination. Wells within 50 feet of septic systems, actually showed significantly fewer positives for total *and* fecal coliform bacteria. This fits the gradient noted, that wells in non-farm, suburban settings had significantly lower coliform and nitrate contamination. There is always concern for contamination by septic systems in suburban settings and while cases are known in Iowa these data indicate that they are not a major regional concern.

Wells located directly within current or former animal feedlots showed significantly higher concentrations of nitrate. About 43% of the wells in this category exceeded 10 mg/L, compared to the 18% average for all wells. However, such sites comprise only about 3% of wells state-wide and, thus, these sites account for only about 1% of the wells with >10 mg/L, NO_3-N. These wells only exhibited a slightly greater proportion of total coliform positives. It would have been expected that the proportion of wells sited directly in feedlots would have exhibited greater proportions of total coliform positives than were observed. Also, none of these sites were positive for fecal coliforms. This suggests that well construction, and leakage at the well head was not a major factor contributing to contamination of these wells.

Wells located less than 25 feet from an out-house (only 0.2% state-wide occurrence) and wells less than 25 feet from a barn (7% occurrence) showed less than the state average for all categories of contamination. Less than 0.2% of wells were located within 15 feet of a fuel storage tank and none exhibited contamination by bacteria, high nitrates, pesticides, or petroleum compounds. There were no reports of well contamination from fuel compounds in the SWRL survey; most petroleum leak and spill cases are in urbanized areas. Also, there were no obvious incidents of rural well-water contamination from landfills or other waste-disposal activities.

Farm sites that did not apply any herbicides during the recent growing season showed slightly less pesticide detections and slightly lower proportions of wells with >10 mg/L, NO_3-N than those that did apply herbicides. They also showed slightly greater proportions of coliform positives.

Similarly, sites where all herbicides used were mixed at the site exhibited fewer pesticide detections than sites where only some of the herbicides were mixed on site. Sites where herbicides were mixed within 15 feet of the well showed greater proportions of pesticide detections than state-wide averages, as did sites where pesticide application equipment was rinsed within 15 feet of the well. This example illustrates how multifactor analysis will assist further understanding, because most of these sites are one and the same. While there is an association here, the proportion of wells potentially influenced by these factors is quite low, about 3%, state-wide. Also, sites noting that equipment is rinsed at a hydrant *>15 feet away* from the well, and farms reporting that herbicides are all mixed in the field where they are to be applied, exhibit the same higher proportions of pesticide detections.

Wells located within 15 feet of chemical storage and handling areas are very uncommon, occurring at <0.6% of rural sites. Even with this proximity to potential point source problems, *none* of these wells contained pesticides or NO_3-N >10 mg/L.

The two most significant associations in these data are related to non-farm sites and farm sites with reported chemical accidents. Non-farm sites where pesticides are used show significantly lower proportions of pesticide detections, and NO_3-N >10 mg/L, and no atrazine detections.

Well owners also reported pesticide or fertilizer spills near their wells at 4.7% of the sites state-wide. Other direct point sources of contamination include back siphoning accidents; these occurred at <0.8% of the sites. As noted, these percentages are similar to those reported from other detailed inventories in local areas of Iowa. In aggregate, an estimated 5.5% of private drinking water wells in Iowa have experienced one or more agricultural chemical spills near the well. A greater proportion of these sites exhibit pesticide detections and high nitrate concentrations than the state-wide average, as would be expected.

At locations where pesticide spills within 15 feet of the well were reported, seven (7) of the wells contained one or more pesticides. When a specific product name was provided, only two of the seven wells with pesticides actually contained the product named in the spill. At two of the three sites with a fertilizer spill, the nitrate-N in the wells exceeded 10 mg/L.

The correlation between reported back siphoning accidents and observed pesticide well contamination is better. Two of the four wells contained the product reportedly involved in the

Table 85. Proportion of pesticide detections in SWRL well-water samples, related to nitrate-N concentration ranges and well-depth ranges.

Well Depth feet	A. Percentage of total pesticide detections in combined nitrate-N range and well-depth range categories:					% of all pesticide detections in well-depth range:
	- - - -	nitrate-n range, mg/L			- - - -	
	<0.1	0.1-2.9	3.0-4.9	5.0-9.9	>10.0	
as nitrate;	<1	1 -13	13 -22	23 -44	>45	
<50 ft	6%	8%	4%	8%	14%	39%
50-99 ft	3%	6%	3%	4%	9%	24%
100-199 ft	5%	1%	0%	8%	4%	18%
200-299 ft	8%	3%	0%	3%	0%	13%
≥300 ft	4%	3%	0%	1%	0%	8%
% of all pesticide detections in nitrate-N range:	25%	20%	6%	23%	26%	
% of all detections, as above, including sites with unknown well depths:	23%	20%	7%	23%	27%	

Well Depth feet	B. Percentage of samples in combined nitrate-N range and well-depth range categories with pesticide detections:					% of all sites in well-depth range with pesticide detections:
	- - - -	nitrate-n range, mg/L			- - - -	
	<0.1	0.1-2.9	3.0-4.9	5.0-9.9	>10.0	
as nitrate;	<1	1 -13	13 -22	23 -44	>45	
<50 ft	24%	17%	17%	17%	20%	19%
50-99 ft	6%	22%	25%	17%	18%	16%
100-199 ft	4%	3%	0%	50%	50%	9%
200-299 ft	13%	11%	0%	29%	0%	14%
≥300 ft	7%	22%	0%	33%	0%	10%
% of sites in nitrate-N range with pesticide detections:	8%	14%	14%	24%	20%	
% of sites, as above, including sites with unknown well depths:	9%	15%	17%	28%	24%	

incident at rather high concentrations, as discussed in the Pesticide Detections section. The other two incidents occurred in 1975 with no traces of the named products (atrazine or alachlor) detected during the SWRL study.

Well Factors; Local Sources

Wells located on floodplains, and likely finished in alluvial aquifers, exhibited slightly greater proportions of nitrate-N and atrazine contamination, particularly when compared to wells located in upland positions. Sand-point and large-diameter seepage wells show significantly greater proportions of contamination than the total state averages in most categories. The majority of private wells state-wide, 64%, are isolated from routine activities, yet exhibit contamination at the state-wide average proportions. These factors are highly inter-related with well depth, and few individual well factors exhibit any greater contamination than simply wells <50 feet deep. For wells that may allow surface runoff into the well, the proportion of pesticide detections is considerably greater than the state average, but even these values were within the 95% confidence interval for wells <50 feet deep.

While case studies have indicated that many acute situations (i.e., wells with high, persistent concentrations) often are related to local point sources or mishandling, this is clearly not always the case, particularly with shallow wells. Controlled research and plot studies continue to verify that high nitrate concentrations and some commonly used herbicides, that are not highly adsorbed to the soil, can and do leach into shallow groundwater from routine field use. Review of the SWRL sites where pesticide concentrations exceeded HALs is an example; of these sites only 25% of these sites can clearly be attributed to point sources of contamination; 63% are apparent nonpoint sources. Also, while seepage wells show greater than average proportions of contamination, field surveys show that most such wells are located in areas to enhance seepage, often quite distant from the farmstead or hydrants where handling accidents or even routine spills and rinsing occur.

Nitrates, Well Depth, and Pesticide Detections

Well owners and public agencies commonly ask for guidance on whether or not a particular well is likely to be affected by pesticide contamination or whether to go to the expense of analyzing well-water for pesticides. The analysis of the predictive relationships among various factors in the SWRL data may afford some insights and guidance. A significant statistical relationship was developed between well depth, nitrate-N concentration and the probability of a well having a detectable pesticides. The actual mathematical expression, while significant, has a very large confidence interval, because of the relatively low numbers involved for any particular combination of well depths, nitrate, and pesticides. Hence, it is not pragmatically useful. Table 85, however, summarizes these data in two ways that may provide some insight to these relationships. Table 85a, shows the percentage of sites with pesticide detections in the combined nitrate-N range and well depth range categories, as a proportion of all pesticide detections. The matrix, or the summary rows or columns, add to 100%.

Table 85b, shows the percentage of sites within a combined nitrate-N range and well depth range that had pesticide detections, as a proportion of all sites in the nitrate-N, well depth category. These data help to illustrate the generally increased probability of pesticide contamination as well depth decrease and nitrate-N increases. Because of the number of categories in the matrix, the number of sites are quite low for some, which affects the confidence of any interpretation. For example, for a well that is 170 feet deep, with 12 mg/L NO_3-N, Table 85b shows that 50% of SWRL samples in that category had pesticide detections. However, there were few samples in that category; it only accounted for 4% of all pesticide detections (Table 85a), so the likelihood of a pesticide detection is likely somewhat smaller than 50%. Table 85 provides a summary and, hopefully, some insight to these inter-relationships.

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