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Impact of Agricultural Drainage Wells on Groundwater Quality

Completion Report
1981-1983

Project 2450
ISU-ERI-Ames-85183

Sponsored by EPA Grant No. G007228010
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report

College of Agriculture
College of Engineering
Iowa State University

257

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SUMMARY

Agricultural drainage wells (ADWs) are used in certain locations in Iowa to provide outlets for subsurface and/or surface drainage systems. Good drainage is necessary in these highly productive and intensively farmed areas to allow field operations to be done at or near optimum times (soil trafficability) and to provide a good soil medium for crop growth (avoiding excessive wetness). The ADWs exist where natural drainage is not present, as in the pothole country, and where artificial drainage outlets via tile mains or drainage ditches have not been provided. In addition, the ADWs, which are drilled or dug holes from the soil surface to an underground aquifer, require a location where the underground strata is capable of receiving large quantities of water quickly.

The common construction and setting for an ADW is an underground cistern (½ to 2 m in diameter) with a well (10 to 25 cm in diameter) at the bottom, with the whole assembly located near the bottom of a pothole. Subsurface drainage tiles enter most cisterns, which are generally constructed from poured concrete, grouted bricks, large diameter clay tiles or metal culverts. A field survey showed that most ADWs either had surface inlets connected to the subsurface drainage systems, or the cisterns were low enough such that surface drainage could enter the wells directly when pondage occurred. For ADWs located near roads, most had surface inlets for roadway and ditch drainage.

Because of the large cost involved in providing artificial drainage, areas where ADWs exist are almost always used for row crop produc-

tion with fertilizers and pesticides applied to achieve near maximum yield. In intensively farmed areas in Iowa, essentially all the land is treated with herbicides, and an average of over 175 kg N/ha is applied annually for corn production (overall, the state average is somewhat lower at about 140 kg N/ha). Insecticides are also commonly used on continuous corn for rootworm control.

Soil adsorption of agricultural chemicals is important in determining differences in quality between surface runoff and subsurface flow from intensively farmed land. In general, as soil adsorption increases, less surface-applied chemical is transported with subsurface flow and more with surface runoff water and sediment. Nitrate-nitrogen (NO_3-N) is transported from the field mainly with subsurface flow because it is very soluble and not adsorbed by the soil. Anionic herbicides such as chloramben, 2,4-D and dicamba are only slightly adsorbed by soil and are soluble enough that they can be lost with subsurface flow. Most other pesticides used in Iowa are either moderately or strongly adsorbed to soil and are lost from the field mainly with surface runoff water and sediment.

The quality of water draining to the four ADWs monitored in this study showed the effects of the source of drainage water. During periods between runoff events when all the drainage to the ADWs was subsurface flow, NO_3-N concentrations were the highest, commonly in the range of 10-30 mg/L, while during periods of snowmelt or rainfall-runoff, concentrations often dropped below 10 mg/L for water draining to the wells receiving both surface runoff water and subsurface flow. On the other hand, pesticide concentrations in subsurface flow were lower (from undetectable to < 1.0 μg/L) than during rainfall-runoff events, when, for example, the two highest values measured, 55 and 80 μg/L (for alachlor and cyanazine, respectively), were detected in water draining to the wells receiving both subsurface flow and surface runoff. Lower bacteria and sediment levels were also measured during subsurface flow because of the filtering provided by the soil profile. On the average, the NO_3-N concentration in water draining to ADWs was 16 mg/L (exceeding the 10 mg/L drinking water standard for 85% of the samples) and alachlor, atrazine, cyanazine, and dicamba concentrations averaged 0.9, 0.02, 3.3, and 0.3 μg/L, respectively. These concentrations closely agree with those measured in other studies of agricultural drainage in Iowa, for the Des Moines River at Boone, and in the Big Spring Karst area of northeast Iowa. Pesticide and NO_3-N concentrations in agricultural drainage as predicted by mathematical models using data for the ADW area in north-central Iowa were also in reasonable agreement with those measured. At no time did pesticide concentrations measured in drainage to the ADWs exceed established or proposed criteria for pesticides in drinking or ground water.

Mathematical modeling of pollutant transport in the groundwater system under typical flow and NO_3-N loading conditions indicated that the areal influence of an ADW would be localized within about 2 km of the well, with dilution and dispersion of inflow with existing groundwater quickly reducing NO_3-N concentrations in inflow to below the drinking water standard. Sampling of farm water supply wells in areas with and without ADWs showed that NO_3-N concentrations were uniformly low in the area without ADWs, whereas in the area with many ADWs, some water supply wells exceeded the drinking water standard, while other supply wells in the same vicinity had near zero levels of NO_3-N. The depth to bedrock is an important parameter in nitrate movement into the aquifer. In those areas monitored, the NO_3-N concentrations where the glacial drift was greater than 15 m indicated higher levels of NO_3-N where ADWs were present over those areas where no ADWs were present. The differences in average NO_3-N concentrations were statistically significant, indicating an impact due to ADWs.

One of the difficulties in assessing the impact of ADWs on the groundwater is the determination of the proper criteria to be used in evaluating these impacts. There are no specific water quality criteria that apply directly to the ADWs situation. Some of the questions that are raised are:

1. Should the water being recharged by ADWs meet the drinking water standards?
2. Should the standards be applied to the highest concentrations recorded or the average concentrations?
3. Should there be no acceptable levels of pesticides (zero concentrations) allowed in the recharge water?

It is recognized that these questions are complicated and beyond the scope of this study. As the ULC program implementation begins, there will need to be further clarification of the proper criteria to be used by the EPA and the implementing states.

If the criteria to be applied to the recharge water entering ADWs is the current existing drinking water standard, then the ADWs would not meet the criterion for NO_3-N. In fact, some 85% of the samples of the recharge water exceeded the drinking water standard. The results from the individual well survey also indicated an elevated level of NO_3-N in the vicinity of high concentrations of ADWs. This indicates a localized impact that is degrading the water quality in the vicinity of large numbers of ADWs. Bacteria levels during high recharge events would also exceed the current drinking water standard, even in some samples from those ADWs that are believed to be receiving only subsurface drainage. However, the bacteria levels in the recharge water are not in excess of those in surface waters in the region that are used for public drinking water after proper treatment.

The only pesticide used extensively in Iowa that has a drinking water standard is 2,4-D. The maximum concentration measured in this study (0.4 μg/L) is well below the drinking water standard for 2,4-D (100 μg/L). The EPA is proposing a method for calculating maximum advisable levels (MALs) for pesticides in groundwater from established acceptable daily intake (ADI) values. If this method is used, MALs, for alachlor, atrazine, 2,4-D, and dicamba would be 1000, 215, 125, and 12.5 μg/L, respectively, while maximum values measured were 55, 0.5, 0.4, and 12 μg/L. Only the mobile, less strongly adsorbed dicamba approached its proposed MAL value.

As the recharge water moves into the groundwater system, dispersion and dilution will occur. The proposed criteria for implementing the Resource Conservation and Recovery Act (RCRA) recognizes this diluting effect in establishing the maximum contaminant levels. It is possible that the same approach should be taken with ADWs.

The ADWs have offered an economic benefit by allowing intensive agricultural development in portions of north-central Iowa. ADWs are sources of NO_3-N, bacteria, and pesticides entering the groundwater system. There is strong evidence that ADWs are contributing to increased levels of nitrate-nitrogen in the shallow groundwater system. These impacts are concentrated in the area of high ADW use and have not been observed to impact on the public water supplies in the region, but individual farm wells have been affected. This is probably because most ADWs in this region are recharging the Mississippian aquifer while most community water supplies obtain their water from the deeper Silurian-Devonian formations.

There are several options to reduce the impacts of ADWs, depending on what pollutant is involved. More careful N management could be used to reduce the amount of NO_3-N leached and transported to a ADW. Modeling showed that lower and/or better-timed N applications could reduce NO_3-N concentrations in drainage to below 10 mg/L. Decreasing the N application rate, however, from 150 to 75 kg/ha, would decrease net return for corn about \$26/ac at current corn and N prices (1 kg = 2.20 lb; 1 ha = 2.47 ac). Pesticide incorporation at application and the use of soil conservation practices, along with more strongly adsorbed pesticides, could decrease pesticide losses. For bacteria, moderately or strongly adsorbed pesticides, and sediment itself, closing the surface inlets and forcing surface water to infiltrate through the soil would decrease their transport to the aquifer (although the ponding that would result from slower drainage would increase any wetness problems). Transport of the slightly adsorbed anionic herbicides with subsurface flow, or the even lesser movement of other pesticides, would have to be solved by banning the pesticides of concern, or closing the ADWs if this transport was deemed a problem.

If the ADWs were closed and no alternative drainage outlets were provided, it is estimated that average (and highly weather-dependent) crop losses would be at least \$128/ac a year. If alternative drainage outlets were provided through tile mains and drainage ditches, with pumps used where necessary, it is estimated from known locations of 54 ADWs in Humboldt and Pocahontas counties, draining about 5500 acres, that one-time capital costs would average \$236/ac (range from \$90 to \$320/ac); there also would be some additional annual maintenance and fuel costs with the pumped drainage.

Much of the initial work on the project was conducted by Jack Musterman, Robert Fisher, and Lon Drake, then of the Department of Environmental Engineering, University of Iowa.

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I. INTRODUCTION

A. Background

It has been estimated that approximately 17 million waste disposal facilities are placing over 1,700 billion gallons* (6,400 billion liters) of contaminated liquids into the groundwater each year (U.S. EPA, 1977). This waste load represents a serious threat to the millions of Americans who are dependent on groundwater as a drinking water source. The sources of contamination can range from industrial wastewater impoundments to landfills and uncontrolled dumps to underground injection wells. Figure I-1 shows the variety and magnitude of the potential sources of contamination by subsurface disposal. This variety means almost unlimited possibilities of pollutant transport and associated impacts.

In an effort to avoid future contamination of existing groundwater supplies, the Federal Government has instituted various programs. The Safe Water Drinking Act states that "...underground migration of injected wastes cannot be determined accurately; therefore, highly toxic compounds should not be injected" (Musterman et al., 1980). The Federal Government, in adopting this policy, has taken a preventative rather than a 'clean-up' approach to groundwater contamination. The establishment of the Underground Injection Control (UIC) program is a major effort to prevent underground injection of waste which may endanger drinking water sources.

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*In some cases English units are used to improve readability: for example, gallons (3.78 liters), acres (0.405 hectares), pounds (0.454 kilograms), inches (2.54 centimeters).

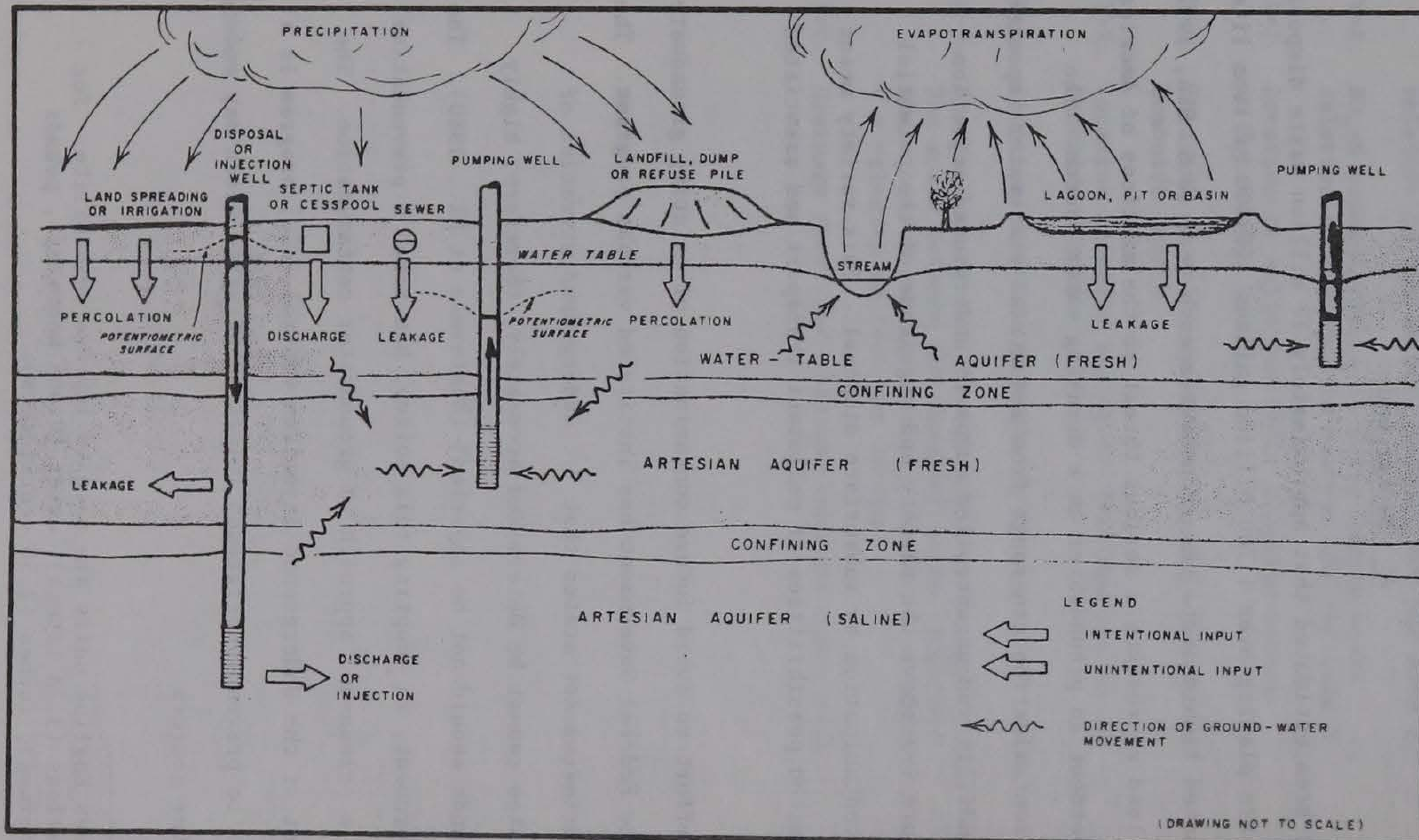

Figure I-1. How waste disposal practices contaminate the groundwater system (from U.S. EPA, 1977).

The UIC program was to be based on minimum federal standards and regulations and was to be administered by the state. It was left to the individual states to develop a program that satisfied the following minimum requirements:

1. Prohibit unauthorized underground injection, effective within three years after enactment of the program;
2. Require the injection applicant to bear the responsibility for assuring protection of underground sources of drinking water;
3. Provide assurance that no regulation would allow endangerment of underground sources of drinking water;
4. Provide inspection, monitoring, record-keeping and reporting requirements for injection wells;
5. Provide control over injection by federal agencies, whether or not the injection occurs on property owned or leased by the federal government; and
6. Provide non-interference with oil and gas production, unless such requirements are essential to assure protection of underground sources of drinking water (Musterman et al., 1980).

The final UIC regulations grouped all injection wells into five major classes depending on the nature of the injected fluid, the zone of injection, and its potential for groundwater contamination. The injection well classes are generally defined as follows:

Class I: Wells that inject hazardous waste below an underground source of drinking water;

Class II:	Wells used for brine disposal or enhanced recovery processes in the production of oil and gas;
Class III:	"Special process wells" used in in situ mining of copper, sulfur, etc.;
Class IV:	Wells that inject hazardous waste above or into an underground source of drinking water; and
Class V:	Other injection wells, such as hydrocarbon storage wells, cooling water return wells, and agricultural drainage wells (Musterman et al., 1980).

An inventory of subsurface injection activity in Iowa, prepared for the Office of Drinking Water, Environmental Protection Agency, by the Department of Environmental Engineering at the University of Iowa (Musterman et al., 1980, 1981), indicated the following:

1. Class I, II, and III Injection Wells

No Class I, II, or III injection wells were identified in Iowa during the inventory.

2. Class IV Wells

Initially, an estimate of 50 Class IV wells was made but was later reduced to 13 in a report received by the UIC group from the EPA (U.S. EPA, 1980). These 13 facilities were then contacted by phone where possible and asked to clarify the response indicated on the notification submitted by the EPA. All persons contacted characterized their initial responses to the EPA either as mistakes or as septic tanks receiving only domestic wastes. It therefore appears that few if any Class IV wells currently exist in Iowa.

3. Class V Wells

The inventory of Class V wells broken down by type is as follows: 5 cooling water wells, 9 cesspool wells, and 690 ADWs, for a total of 704 Class V wells. The current estimate of Class V wells is reduced from the number in an earlier survey because volatile hydrocarbon storage wells have been eliminated from UIC control. With this elimination of volatile hydrocarbon storage wells from UIC control, the agricultural drainage wells (ADW) become the only Class V injection activity of any significance in Iowa (Musterman et al., 1981).

B. Agricultural Drainage Wells

Injection wells have been used in Iowa for more than 100 years for subsurface disposal of surface runoff and tile waters from agricultural lands (Musterman et al., 1980). The use of these agricultural drainage wells (ADWs) has provided a simple, rapid and economical means of drainage control that has increased the amount of tillable acreage and the crop productivity in Iowa. In north-central Iowa, there are areas that could not be farmed economically without the drainage currently provided by ADWs.

Although ditches and tile drainage systems using surface outlets have provided artificial drainage in many areas of Iowa, they have not satisfied all the drainage needs throughout the state. The agricultural drainage well (ADW) has provided an alternative to surface ditch drainage by utilizing as an outlet a drilled, driven or dug hole extending from the surface to an underground strata that is capable of taking surplus water. In order for these wells to be operationally and cost effective, they must be drilled into a shallow, high-capacity aquifer such as a fractured carbonate strata (limestone or dolomite) with high secondary permeability. This assures good recharge capacity with a minimum of plugging by the drainage silt load (Harris, 1945).

The use of ADWs apparently flourished in the 1950s, with a report of as many as seven wells draining a single 400-acre farm (Hunt, 1961). However, the enactment of Chapter 455A.25 of the Code of Iowa in 1957 curtailed new ADW construction by requiring a permit from INRC for new ADW installations or expansion of the drainage area of existing wells. No permit was required for existing wells if they did not "...create waste or pollution." As a result of these regulations, only two new wells have been permitted since 1957. No permits have been issued for wells existing prior to 1957, and there are no currently active permits for ADWs in the state (Musterman et al., 1981). Because of the limited number of permits, there is a common belief that ADWs are illegal. In fact, however, they are illegal only if they were constructed after May 1957, without a permit, or if they "...create waste or pollution."

In 1981 the Iowa legislature reorganized the state agencies dealing with water and environmental quality. The new legislation, Iowa Code Chapter 455.B, resulted in several changes relating to ADWs. Under chapter 455.A, all drainage wells constructed prior to 1957 did not need a state permit if they were not creating waste or polluting the groundwater. In Chapter 455.B this "grandfather" clause is rescinded. The current rules and regulations for the Department of Water, Air and Waste Management (DWAWM) (Sec. 900-51.5) require a diversion permit for any diversion of surface waters into aquifers. Section 900-51.6 indicates tile drainage is considered as surface water. Although not specifically referenced, ADWs would fall under this rule; thus, it now appears that a diversion permit is required for all ADWs, both new and existing. Section 900-62.9 of the DWAWM rules state that "...there shall be no disposal of a pollutant other than heat into wells within Iowa." ADWs would also be covered by this requirement. Chapter 455B.171 (13) defines pollutant as sewage, industrial waste or other waste and subsection (3) defines other waste as heat, garbage, municipal refuse, lime, sand, ashes, offal, oil, tar, chemicals, and all other wastes that are not sewage or industrial waste.

It now appears that ADWs could still be constructed provided the owner obtains a diversion permit from DWAWM and the owner can show it does not pollute the groundwater. Further clarification of the DWAWM rules are needed to determine proper criteria to judge if an ADW is polluting.

1. Geological Setting in Iowa

About 150 years ago when settlers first arrived in Iowa territory, a large part of the upper Midwest area was considered unfit for human habitation. At that time, flat areas in north-central Iowa frequently were flooded. Many of the depressions were marshes and wetlands; the only farmable areas were the higher lands surrounding the marshes. The soils in this area are dominated by Wisconsin glacial till-derived soils developed under a native vegetation of prairie grasses. A recent survey (Schult et al., 1981) indicated that 60% of the soils in north-central Iowa are considered poorly drained soils, and another 20% are somewhat poorly drained soils. Therefore, without drainage, north-central Iowa would not now be one of the most productive agricultural areas in the country.

Research has proven that agricultural drainage is a valuable production practice. Adequate drainage allows a longer growing season, efficient use of water and fertilizer, reduced miring of agricultural equipment, and reduced fuel consumption.

Drainage of agricultural lands began in earnest in Iowa with the enactment of the 1906 drainage law. Most of the drainage systems were installed between 1906 and 1925 by organizing the natural watersheds into legal drainage districts (SCS, 1983). The drainage methods were diverse, including open ditch and tile drain combinations or underground systems only. Many of these drainage systems are inadequate today, and many were recognized as inadequate the year they were completed. The areas that could not be included in the legal drainage districts had to depend on other kinds of artificial drainage outlets such as agricultural drainage wells.

Agricultural drainage wells drain surface runoff or subsurface drainage from tiles, and in many cases both. The majority of the ADWs in Iowa are located in the north-central portion of the state in areas that were most recently covered with continental glacial ice sheets. Five geomorphic areas in Iowa were reported by Musterman et al. (1980) as candidate locations where ditch drainage was sufficiently difficult to offer incentives to drill ADWs. These potential landscapes are shown in Fig. I-2. The area outlined on Fig. I-2, indicating that drainage wells are probably abundant, was determined using topographic and soil considerations only. Before ADWs could be constructed, an acceptable aquifer to receive the recharge must also be present.

The prime aquifer units for recharge are likely to be the carbonate formations with sufficient joints, pore space, bedding planes, and solution channels, in order for ADWs to be able to recharge drainage water over the long term. Several major bedrock aquifers (sandstones and carbonates) exist in the region of the majority of ADWs. These aquifers include the Dakota Sandstone aquifer, the Mississippian aquifer (upper bedrock) the Silurian-Devonian aquifer (middle bedrock), and the Cambro-Ordovician aquifers (lower bedrock). All of these aquifers are major aquifers used for community and farm water supply in this region.

The major bedrock aquifers in north-central Iowa consist of sandstones and carbonates (limestones and dolomites) containing thin interbedded layers of shales and cherts. Several of the carbonate layers contain only limited secondary porosity. Musterman et al. (1981) presented several east-west (and one north-south) cross-sections of the bedrock aquifers in northcentral Iowa. Figures I-3 and I-4 show generalized lithologic cross-sections through north-central Iowa (see Fig. I-2 for cross-section locations).

As shown in Figs. I-3 and I-4, the glacial drift thickness varies significantly across the area, from in excess of 300 feet to places where the drift may be only a few feet thick. The thickness of the drift plays an important role in the quality of the shallow ground-

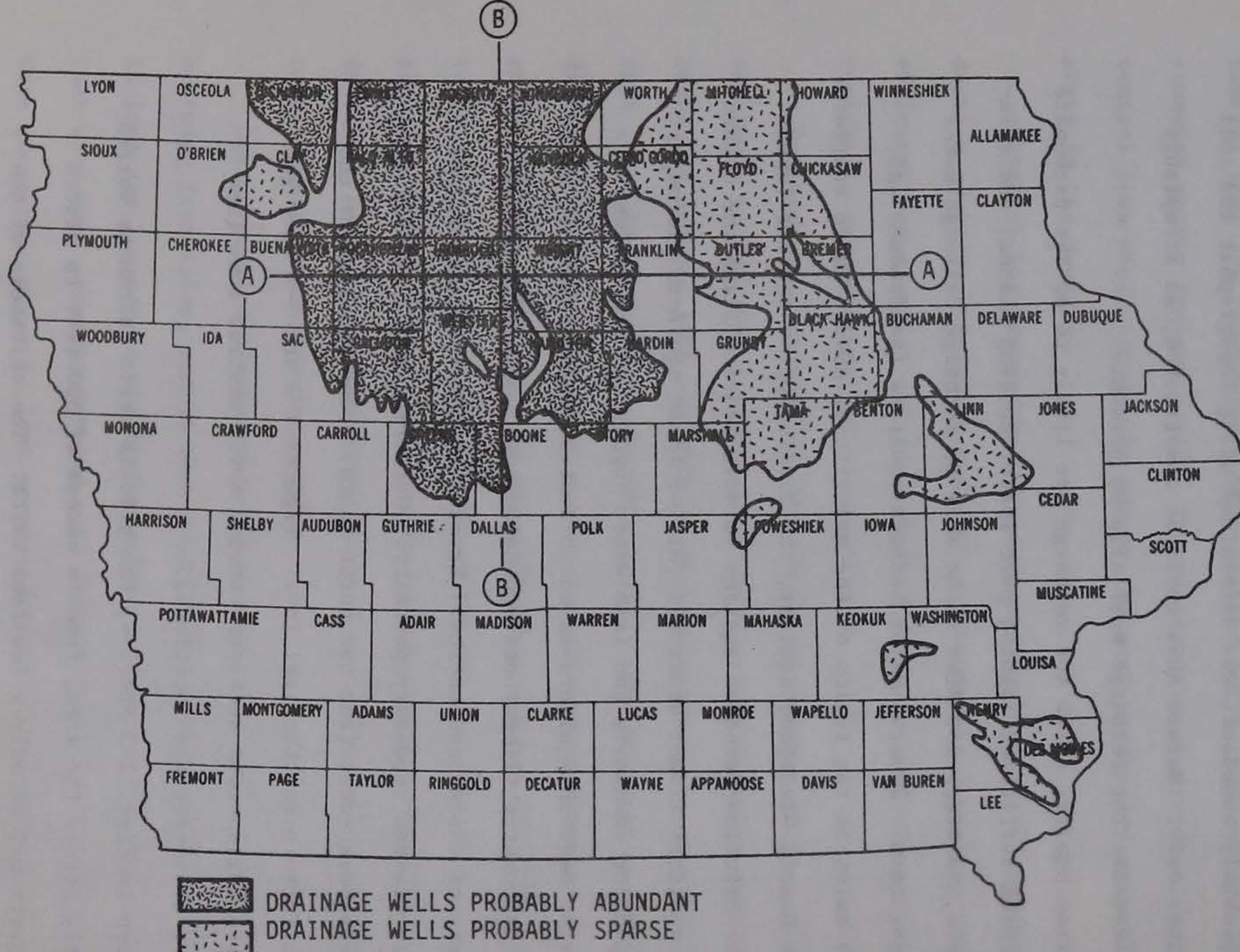

Figure I-2. Landscapes for drainage wells based on drainage problems. (After Musterman et al, 1980.)

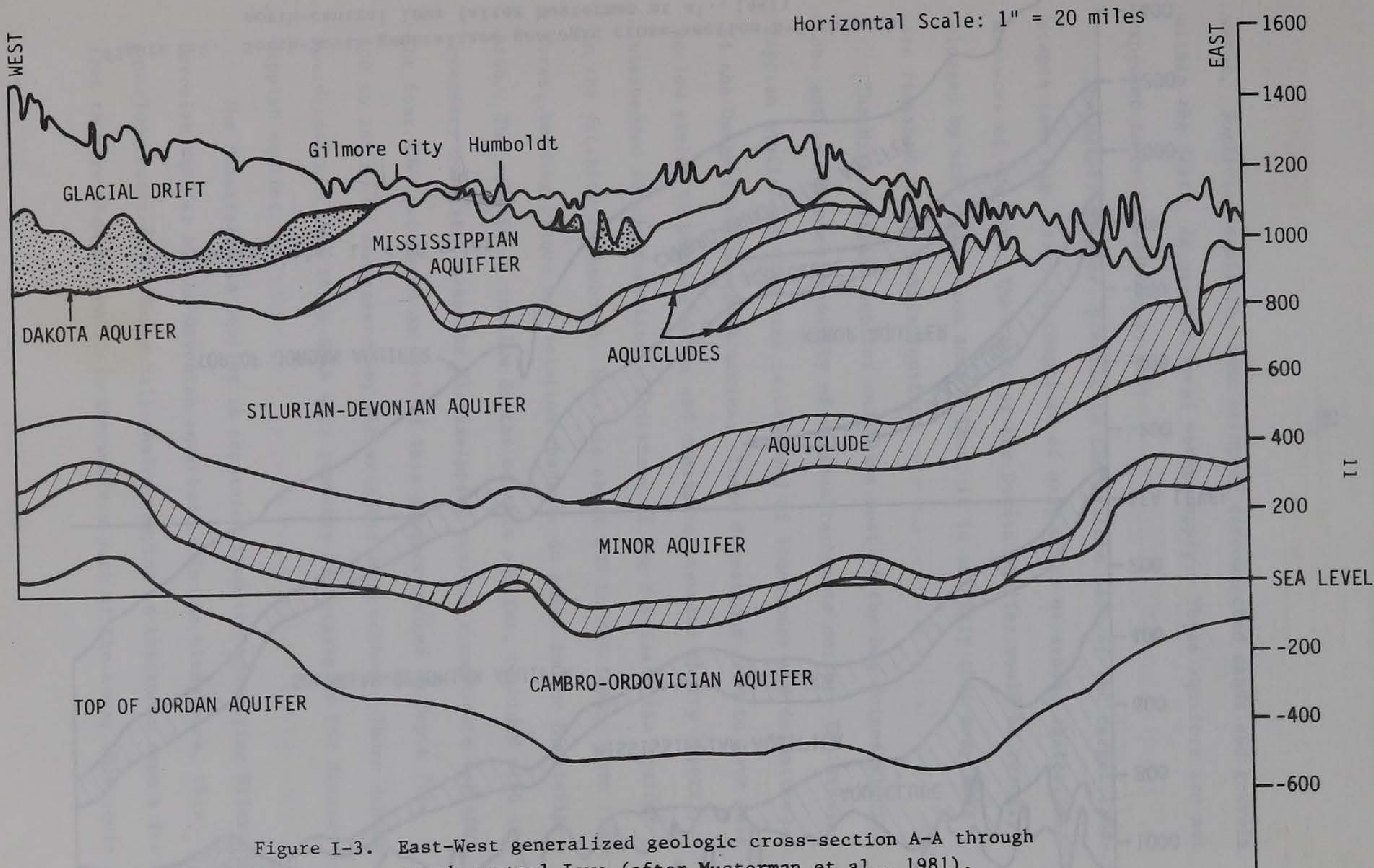

Figure I-3. East-West generalized geologic cross-section A-A through north-central Iowa (after Musterman et al., 1981).

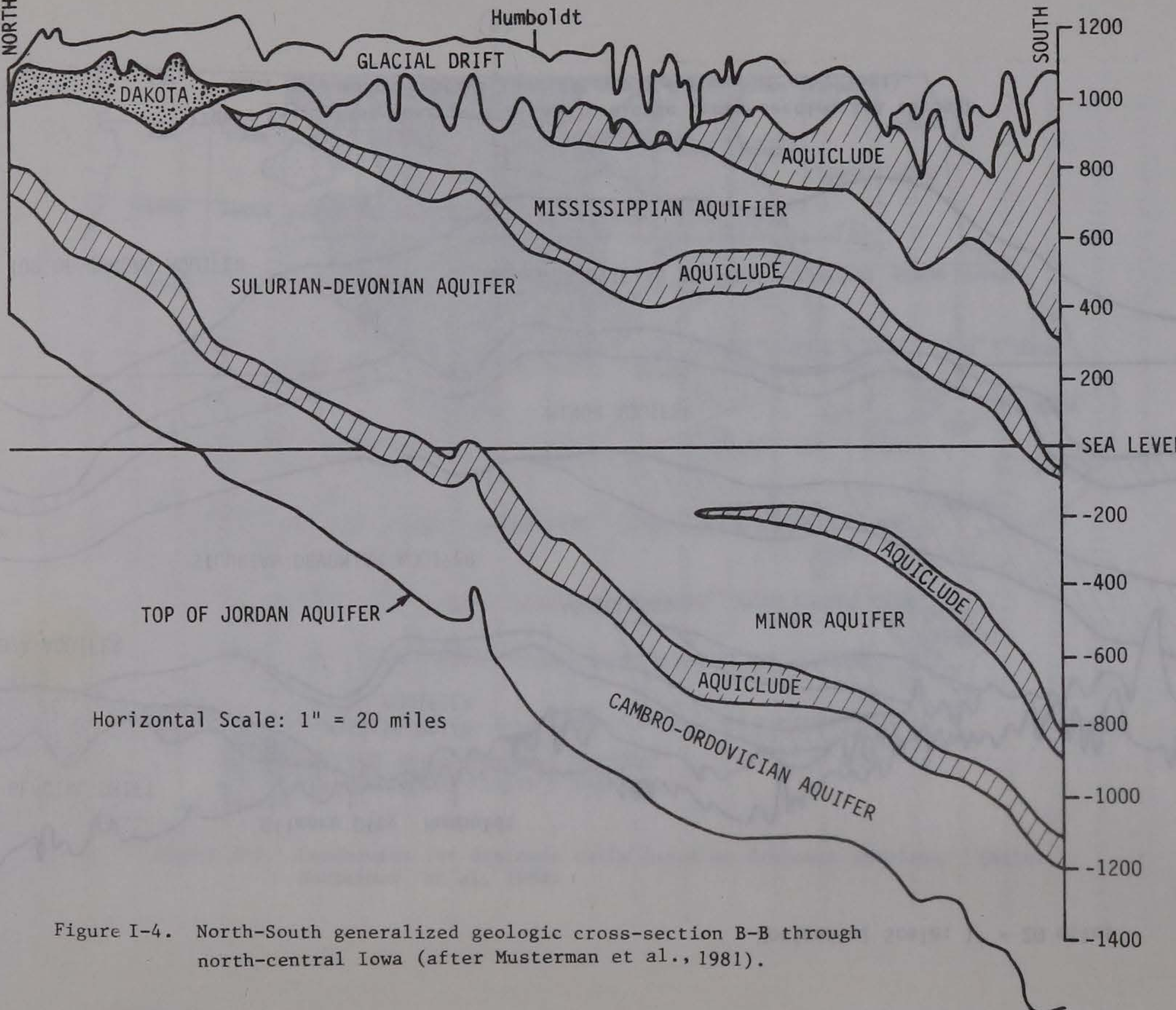

Figure I-4. North-South generalized geologic cross-section B-B through north-central Iowa (after Musterman et al., 1981).

water. Surficial aquifers, consisting of disconnected sands and gravels within the drift, do provide local water supply. These aquifers are not expected to be influenced by ADWs.

The Dakota aquifer, present in the western part of the target landscapes (see Fig. I-3), is comprised of many units of sands, shales, and mixtures of the two. The sands of the Dakota aquifer would be readily clogged by silt loads from ADWs. Thus, it is unlikely that many ADWs are finished in the Dakota aquifer.

The Mississippian aquifer underlies most of the Des Moines Glacial Lobe and is composed primarily of various carbonate units. The Mississippian aquifer in this area is composed of limestones and dolomites of the Osage and Kinderhook series. These formations tend to have fair to low yields to pumping wells and consist of several cherty layers interbedded in the formation. Thickness of the Mississippian aquifer in the vicinity of Humboldt, Iowa, is about 200 to 350 feet. Some fractures, bedding planes and solution channels do exist in the Mississippian. The relatively shallow depths of this aquifer, combined with the secondary porosity, make the Mississippian aquifer a candidate for ADWs. The four ADWs monitored as part of this project ranged in depth from 100 to 285 feet, as measured by the weighted line method. These depths would indicate that these ADWs were probably recharging to the Mississippian aquifer.

The Mississippian aquifer is separated from the underlying Silurian-Devonian aquifer by the Devonian aquiclude. In the study area, this aquiclude consists of Maple Mill shales varying in thickness from a few feet to more than 60 feet. In the western part of the area, this aquiclude is not apparent and the Mississippian and Silurian-Devonian aquifers are in hydrologic connection.

The Silurian-Devonian aquifer consists of a series of limestones and dolomite units with localized chert. Yields to wells in this aquifer are high to fair with pumping rates in excess of 500 gpm possible. The Silurian-Devonian generally has more secondary porosity than the Mississippian aquifer. Much of the sink hole development in northeast Iowa occurs in the Silurian-Devonian aquifer. The thickness of the Silurian-Devonian aquifer in the study area varies from less than 200 feet in extreme north-central Iowa to more than 800 feet in the south-central areas (see Figs. I-3 and I-4). Because the depth of the Silurian-Devonian aquifer increases to the southwest, use of this formation for ADWs is probably confined to the northern portion of the target landscapes. More municipalities use the Silurian-Devonian aquifer than the Mississippian aquifer in this region of the state.

The Silurian-Devonian aquifer is separated from the underlying Cambro-Ordovician aquifer by the Maquoketa aquiclude. This aquiclude consists of shales and dolomites. Below the Maquoketa aquiclude is the Galena formation, which is a minor aquifer of limestones and dolomites with low yields. The Maquoketa aquiclude does not underlie all the study region (see Figs. I-3 and I-4). The Decorah/Platteville formation is a series of thin limestone layers interbedded with shales forming an aquiclude covering the entire area. Immediately below this aquiclude is the St. Peter Sandstone, a formation with fair yields, the Prairie du Chien formation, and the Jordan Sandstone, formations with high yields. The Jordan sandstone is the most widely used formation in Iowa. It is doubtful that the Jordan is used for ADWs because of its lack of secondary porosity and its depth below ground surface.

Table I-1 shows a generalized geologic column in Humboldt County, Iowa. This column shows the sequence of geologic formations to be found and an approximate depth below ground surface.

Figure I-5 indicates the target areas for ADWs as classified by the first bedrock aquifer below the glacial drift. The extreme north and west portions of the target areas have no suitable, near-surface aquifers, and ADWs will not be expected in these areas. In the other areas, the target aquifers are either the Mississippian or Silurian-Devonian aquifers.

The combination of the flat topography and the availability of shallow carbonate aquifers makes north-central Iowa the prime location for large concentrations of ADWs; although isolated ADWs could be found anywhere in Iowa, the most significant potential for groundwater impacts will be in areas of large ADW concentrations.

2. Construction of ADWs

The details of the construction of ADWs in north-central Iowa are highly variable. Since some of the ADWs are reported to be more than 100 years old, a variety of designs has been used.

Figure I-6 is a sketch of a typical ADW showing the three types of flow into the well. In general, an ADW consists of a buried collection basin or cistern, one or more tile lines entering the cistern, and a cased drilled or dug well. In some ADWs, surface water can enter directly into the tile lines through surface inlets or through cracks

Table I-1. Generalized geologic column near Humboldt, Iowa (after Musterman et al., 1981).

Depth (ft)	Age	Rock Units	Hydrogeologic unit	Water bearing characteristics
0-50	Quarternery	Glacial Drift	Scattered sands and gravels	low yields
50-300	Mississippian	Osage and Kinderhook series	Aquifer	fair to low yields
300-320	Devonian	Maple Mill Shales	Devonian Aquiclude	does not yield water
320-830	Devonian	Lime Creek, Cedar Valley, Wapsipinicon	Silurian-Devonian Aquifer	high to fair yields
Not present	Silurian	Niagaran Series		
830-1120	Ordovician	Galena	Minor aquifer	low yields
1120-1190	Ordovician	Decorah Platteville	Aquiclude	does not yield water
1190-1270	Ordovician	St. Peter SS	Aquifer	fair yields
1270-1530	Ordovician	Prairie du Chien	Cambro-Ordovician Aquifer	high yields
1530 +	Cambrian	Jordan		

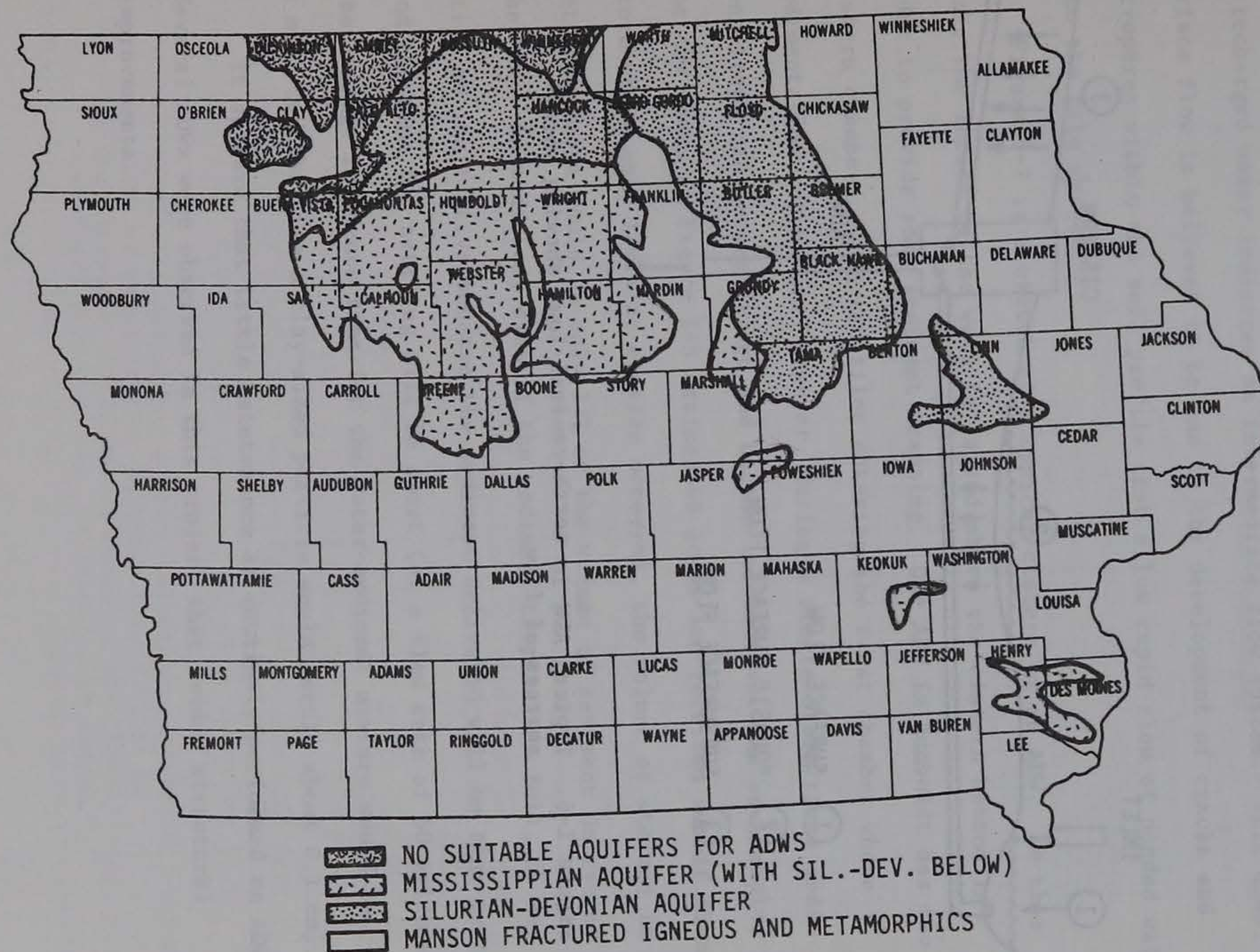

Figure I-5. Target aquifers beneath target landscapes.

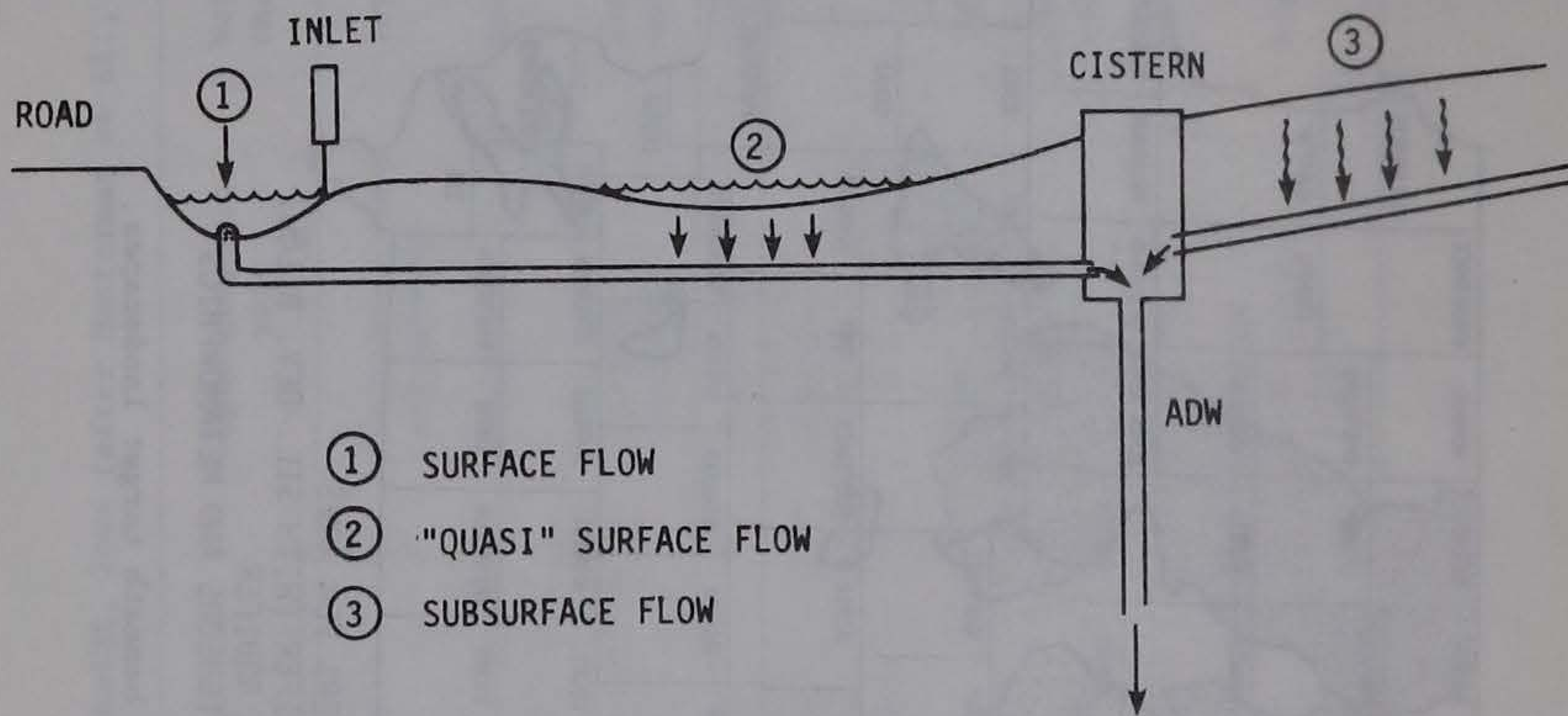

Figure I-6. Typical ADW in north-central Iowa showing three sources of flow.

in or overtopping the cistern. Others appear to collect only subsurface tile flow from agricultural tiles. In some cases the water quality of recharged water indicates an intermediate flow system. This quasi-surface flow is believed to be due to the development of cracks and macropores within the soil profile that allow rapid flow of ponded water into the tile system.

Figure I-7 is a sketch of a typical cistern of an ADW. In this case, the intake to the well is raised above the cistern bottom in order to provide some sediment trapping. One ADW in Humboldt has two cistern chambers, with the tiles entering the first chamber where sedimentation can occur. However, during a single storm these sediment traps can become filled and become ineffective. As an example, if the ADW shown in Figure I-5 drained 40 acres and received runoff from a storm where only 1/4 T/ac erosion occurred, the volume of storage in the settling basin would be only 17% of the volume of sediment delivered to the ADW. In addition, even if the sediment trap is not full, it may still be ineffective because the sediment delivered will be fine-grained, and the time for settling will be short (at a flow rate of 60 gal/min, the average residence time for the water-sediment mixture would be 5 min; in that time a clay-sized particle would settle about 0.1 cm).

It appears that little maintenance is routinely performed on ADWs. Several ADWs were observed in this project that needed structural improvements.

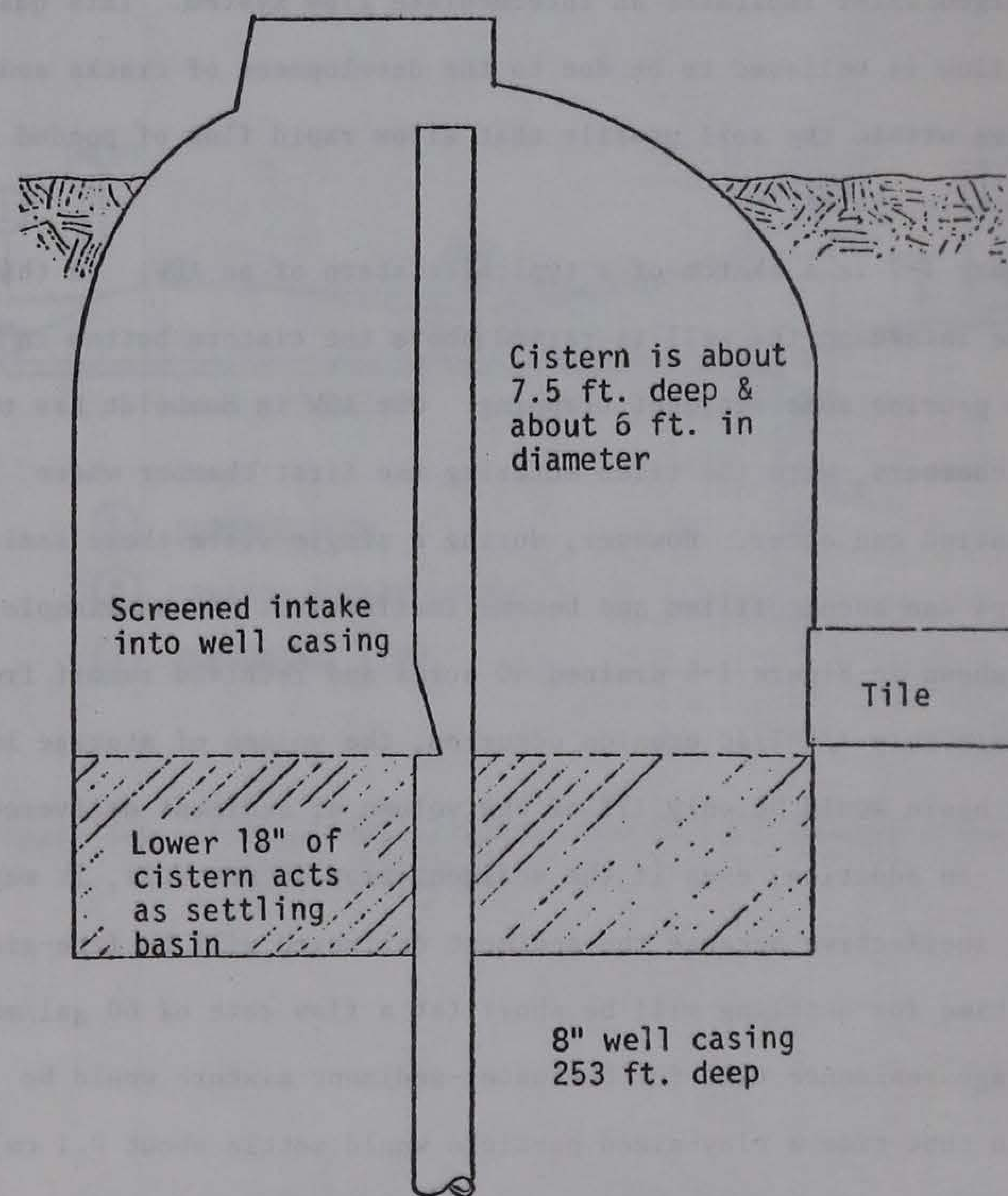

Figure I-7. Sketch of typical cistern and intake structure for an agricultural drainage well prepared by INRC staff from evidence of record for determination on permit application No. 55-R-4-W-1879 (Musterman et al., 1981).

C. Agricultural Drainage Water Quality

The quality of agricultural drainage from Iowa cropland is dependent on the management of that land and which type of drainage, surface runoff or subsurface flow, is being considered. Of the approximately 36 million acres in Iowa, about 22 million acres are in row-crop production of corn and soybeans. On the average, for the five-year period 1978-82, there were 13.8 million acres of corn and 8.2 million acres of soybeans. Because soybeans are usually rotated with corn, with essentially no continuous soybeans, there were about 5.6 million acres continuous corn. Throughout the state this row-cropped land is intensively farmed with an average of 128 lb nitrogen (N), 26 lb phosphorus (P), and 52 lb potassium (K) applied to each acre of corn ground and 1 lb N, 1 lb P, and 4 lb K applied to each acre of soybean ground in 1980 (Hargett and Berry, 1981). One lb/ac equals 1.12 kg/ha. A pesticide use survey (USDA, 1982a) showed that herbicides were used on 99% and insecticides were used on 44% of the corn grown in Iowa in 1980. Another survey (USDA, 1982b) showed that 97% of the soybeans grown were treated with herbicides. Application rates for most pesticides range from 0.5 to 2 lb/ac of active ingredient. (Throughout the text of this report, common names of pesticides will be used; for cross reference between common and trade names of pesticides, see Tables I-2 and I-3.) For corn, five herbicides--alachlor, atrazine, butylate, cyanazine, and 2,4-D--represented at least 90% by weight of the herbicides used; five insecticides--carbofuran, chlorpyrifos, fonofos, phorate, and terbufos--represented over 94% by weight of the insecticide used. For soybeans,

Table I-2. Use and properties of dominant pesticides in Iowa (1979).*

Name: Common	Name: Trade	Use (lb)	Persistence (weeks)	Solubility (ppm)	Adsorption class+	Toxicity LD_{50}** mg/kg
<u>Insecticides</u>						
terbufos	Counter	2,299,500	23+	10 to 15	II	
fonofos	Dyfonate	1,633,330	52+	13	II	
carbofuran	Furadan	1,134,920	7 to 54	700	II	
phorate	Thimet	992,990	7 to 20	50	II	
chlorpyrifos	Lorsban	670,271				
ethoprop	Mocap	343,221				
others	-	76,780				
Total		7,151,012				
<u>Herbicides</u>						
butylate	Sutan	13,596,600	6 to 8	45	II	3880
alachlor	Lasso	11,357,550	4 to 8	242	II	1200
cyanazine	Bladex	8,513,100	8 to 12	171	II	334
atrazine	AAtrex	6,644,254	8 to 32	33	II	3080
trifluralin	Treflan	4,535,420	12 to 24	1	III	3700
propachlor	Ramrod/Bexton	1,714,500	4 to 6	580	II	710
metochlor	Dual	1,674,350	4 to 12	430	II	2780
chloramben	Amiben	1,606,380	6 to 8	700	I	3500
metribuzin	Sencor/Lexone	1,594,080	4 to 16	1220	II	1940
2,4-D	-	1,373,192	4	900	I	300-1000
dicamba	Banvel	884,780	12 to 48	4,500	I	1140

Table I-2. Continued.

Name: Common	Name: Trade	Use (lb)	Persistence (weeks)	Solubility (ppm)	Adsorption class+	Toxicity LD_{50}** mg/kg
bifenox	Modown	551,040	6 to 8	1	III	6400
bentazon	Basagran	459,626	6	500	I	1100
others	-	2,124,622				
Total		56,629,494				

*Source: Becker and Stockdale (1980).

+Absorption class I represents weakly adsorbed pesticides, readily leached in sandy soil low in organic matter (<1%), but some resistance to movement in other soils. Class II represents mediumly adsorbed pesticides with moderate movement in sandy soils low in organic matter, but little or no movement in other soils. Class III represents strongly adsorbed pesticides with slight movement in sandy soils low in organic matter, with negligible movement in other soils.

**LD_{50} is single oral dose that is lethal to 50% of the test animals, usually white rats; for comparison the values for aspirin and table salt are 1200 and 3320 mg/kg, respectively (note, the smaller the value the more toxic the compound).

Table I-3. 96-Hour LC_{50} values for selected pesticides and fish species (μg/l).

	Rainbow Trout	Fathead Minnow	Channel Catfish	Bluegill	Largemouth Bass
<u>Organochlorines</u>					
Aldrin	2.6	8.2	53	6.2	5
DDT	8.7	12.2	21.5	8.6	1.5
Dieldrin	1.2	3.8	4.5	3.1	3.5
Heptachlor*	7.4	23	25	13	10
Toxaphene*	11	18	13	2.4	2.0
<u>Organophosphate</u>					
Diazinon*	90			168	
Fenitrothion	2400	3200	4300	3800	
Malathion*	200	8650	8970	103	285
Parathion	1430	2350	2650	400	620
Terbufos* (Counter)	9.4	270		1.7	
Fonofos* (Dyfonate)	20			7	
Phorate* (Thimet)	13		280	2	5
Chlorpyrifos* (Lorsban)	7.1		280	2.4	
<u>Carbamates</u>					
Carbaryl* (Sevin)	1950	14600	15800	6760	6400
Carbofuran* (Furadan)	380	872	248	240	

Table I-3. Continued.

	Rainbow Trout	Fathead Minnow	Channel Catfish	Bluegill	Largemouth Bass
<u>Herbicides</u>					
Cyanazine* (Bladex)	9000			22500	
Alachlor* (Lasso)	2400			4300	
Trifluralin* (Treflan)	41			58	75
2,4-D*	3100			7400	

*Commonly used in Iowa.

five herbicides--alachlor, bentazon, chloramben, metribuzin, and trifluralin--represented 84% by weight of the herbicide used (Becker and Stockdale, 1980). No aldicarb was reported to have been used in Iowa. (See Table I-2 for the usage and properties of the dominant pesticides.)

Although there are more than 3 million acres of oats and hay grown in Iowa each year, these crops are usually produced on marginal, less intensively farmed land. Because of the high cost of agricultural drainage wells (ADWs) and the subsurface drainage systems associated with them, land draining to ADWs is usually intensively farmed. For example, in Humboldt County, a county known to have a significant number of ADWs, over 95% of the cropland is in row-crops. For these reasons, in considering the quality of drainage to ADWs, only row-cropped land will be considered. Because of the much lower chemical inputs to hay and oat crops and the lower erosion potential of close-grown crops, any error as a result of this assumption will result in an overestimate of a potential pollution problem.

The chemicals of concern relative to nonpoint pollution from agricultural drainage generally involve the nutrients N and P. For surface water resources which must support aquatic life and serve as potentially potable water sources, there is concern for total N and P entering the system as well as for the specific ions NH_4^+ (NH_3, un-ionized NH_4^+ being toxic to fish), NO_3^- (conversion to NO_2^- causing methemoglobinemia in infants), and $PO_4^{\equiv}$ (nutrient often limiting algal growth). In the case of groundwater resources that are being protected as sources of potable water, it is contamination with NO_3-N above 10 ppm that is of the greatest concern. High levels of NH_4-N (greater than 0.5 ppm) would also be of concern where chlorine is used as a disinfectant, because NH_4 reactions with chlorine result in compounds with much lower disinfecting efficiency than free chlorine.

With pesticides, there is also concern for both aquatic and human life. Acute toxicities in the form of lethal doses to 50% of the test animals (usually rats), LD_{50}'s, are given in Table I-2 for pesticides commonly used in Iowa. In general, herbicides are much less toxic than insecticides to both mammals and fish. Table I-3 gives 96-hr LC_{50} concentrations (lethal concentrations to 50% of the test species in a 96-hr test) for selected pesticides.

The problem with pesticide contamination of water resources is that information is not available on the chronic effects of exposure to low levels of pesticides. The U.S. EPA (1976) has published concentration criteria on domestic water supply and freshwater and marine aquatic life for only two herbicides and 15 insecticides out of more than 1000 known pesticides. Domestic water supply concentrations for 2,4-D and 2,4,5-TP are set at 100 and 10 μg/L, respectively. Aquatic life concentrations for most chlorinated hydrocarbon insecticides (aldrin, dieldrin, and DDT, all now banned) are extremely low, in the 0.001 μg/L range. Because of their persistence and potential carcinogenicity, human exposure to these should be minimized. The organophosphorus insecticides listed (guthion, malathion, and parathion) are apparently one or two orders of magnitude less toxic to aquatic life; the concentration criteria range from 0.01 to 0.1 μg/L. The EPA has set no domestic water supply criteria for the organophosphorus insecticides, although the Federal Water Pollution Control Administration (1968) earlier had established a criterion of 100 μg/L for organophosphorus plus carbamate insecticides.

Because there is a need to establish guidelines for pesticide residues in drinking water, the hazard evaluation division of EPA's Office of Pesticide Programs has announced plans to establish maximum advisable levels (MAL) for pesticides in ground water. The basis will likely be the toxicology data base and scientific expertise currently utilized to establish tolerances for pesticide residues in food. The concept of acceptable daily intake (ADI) would be extended for the assessment of potential hazard. The ADI is the daily exposure level of a pesticide residue that, taken during the lifetime of a man, appears to be without appreciable risk. It is generally expressed as mg of pesticide per kg of body weight per day. Under the EPA plan, the maximum advisable level in one liter of water would be set as equal to the ADI for the pesticide of interest times 10 kg. This is equivalent to the mass of a 22 lb child (children are probably the group most sensitive because of a higher water consumption per unit weight (0.1 liter/kg/day)).

In equation form:

$$\text{Maximum Advisable Level} = \frac{\text{ADI} \times 10\ \text{kg}}{1\ \text{liter/day}}$$

As an example, the National Academy of Sciences (1977) has established 0.0215 mg/kg/day as the ADI for atrazine, therefore:

$$\text{MAL} = \frac{0.0215\ \text{mg/kg/day} \times 10\ \text{kg}}{1\ \text{liter/day}} = 0.215\ \text{mg/L}$$

or (using the conversion factor 1 mg/L equals 1000 ppb) 215 ppb. MALs for pesticides in use in Iowa, plus some others, are given in Table I-4, calculated from the available ADIs.

The quality and differences in quality of surface runoff and subsurface flow from row-crop land will be discussed in the following sections. The effects of different management practices on drainage water quality will also be discussed.

1. Surface Runoff

On soils that are not extremely wet or impermeable, initial rainfall preceding a runoff event continues to completely infiltrate until the infiltration rate decreases and/or the rainfall rate increases to the point that the infiltration rate is less than the rainfall rate and runoff begins. Under conditions common to Iowa this means that probably at least 10 mm of rain, and usually much more, infiltrates before runoff begins. It is also believed (Ahuja and Lehman, 1983; Frere et al., 1980; and Donigian et al., 1977) that the depth of mixing or interaction between soil and rainfall-runoff water is less than 10 mm (or mixing possibly decreases exponentially with depth to depths as great as 20 mm).

Therefore, the interaction between the soil and a chemical (i.e., soil adsorption), which affects how quickly the chemical is leached from the surface soil, is very important in determining chemical concentrations and losses in surface runoff. The other important factors are the amount, location, and persistence of a chemical in the soil profile. As a result of leaching from this surface mixing zone, concentrations of very soluble, non-adsorbed chemicals such as NO_3 are

Table I-4. Proposed maximum advisable levels for pesticides in groundwater.

Pesticide	No Observed Effect Level mg/kg/day	Safety Factor	Acceptable Daily Intake* mg/kg/day	Maximum Advisable Level ppb
Insecticides				
phorate	0.01	100	0.0001	1.0
diazinon	0.02	10	0.002	20
carbaryl	8.2	100	0.082	820
parathion	0.043	10	0.0043	43
malathion	0.2	10	0.02	200
methoxychlor	10	100	0.01	1000
toxaphene	1.25	1000	0.00125	12.5
captan	50	1000	0.05	500
aldicarb	0.1	100	0.001	10
Herbicides				
alachlor†	100	1000	0.1	1000
atrazine	21.5	1000	0.0215	215
trifluralin	10	100	0.1	1000
propachlor	100	1000	0.1	1000
chloramben	250	1000	0.25	2500
2,4-D	12.5	1000	0.0125	125
dicamba	1.25	1000	0.00125	12.5
2,4,5-T	10	100	0.1	1000
2,4,5-TP	0.75	1000	0.00075	7.5
butachlor	10	1000	0.01	100
propanil	20	1000	0.02	200
paraquat	8.5	1000	0.0085	85
propazine	46.4	1000	0.0464	464
simazine	215	1000	0.215	2150

*From National Academy of Science (1977).

†Values for alachlor will likely be revised downward when new data become available.

not very high in runoff water (as opposed to subsurface flow) and decrease with time, even though there may be over 100 kg NO_3-N/ha in the soil profile at the time of a runoff event. As shown in Table I-5, NO_3-N concentrations in surface runoff from corn and soybean fields in the Midwest are typically less than or equal to 5 mg/L, one-half the current drinking water standard. Because NO_3-N is not adsorbed, all of the NO_3-N losses in surface runoff are associated with the water phase and are usually less than 5 kg/ha per year.

Chemicals such as NH_4-N and PO_4-P are adsorbed by the soil, are not flushed as easily from the surface mixing zone, and are lost in surface runoff with both water and sediment. In one study (Baker and Laflen, 1983), the concentration ratio of NH_4-N in sediment to that in runoff water was 42 and for PO_4-P was about 2300. On the gentle slopes ($\leq$ 2 percent) expected in the areas where ADWs are used, the ratio of water to soil lost in runoff is probably on the order of 500 to 1. For NH_4-N then, most would be lost with runoff water, whereas for PO_4-P, most would be lost with sediment. Total losses of NH_4-N in surface runoff are usually less than 2 kg/ha which is less than the (approximate) 7 kg/ha that comes down with precipitation in Iowa (Tabatabai and Laflen, 1976). Total losses of PO_4-P are usually less than 1 kg/ha.

As shown in Table I-2, different pesticides show a wide variation in the degree of soil adsorption. A majority of the pesticides fall into Class II and are moderately adsorbed. This means in most soils they are not easily leached and are lost mostly in surface runoff. However, like NH_4-N, most of the losses in surface runoff are with

Table I-5. Soluble nutrients in surface runoff from Midwestern corn and soybean fields.

Crop	State	Flow	NH_4-N	NO_3-N	PO_4-P	Reference
		cm	-----------	mg/L[*]	-----------	
Corn (rotated with soybeans)	Iowa	3.1	0.8	4.5	0.16	9
Soybeans (rotated with corn)		2.9	0.1	3.5	0.03	9
Cont. corn (448 kgN/ha-yr)		5.6	1.0	2.4	0.20	2
Cont. corn (168 kgN/ha-yr)		4.9	1.1	1.3	0.18	2
Corn (terraced)		3.6	0.6	3.0	0.26	2
Corn and soybeans		4.4	--[†]	5.2[†]	0.06	32
Cont. corn (no-till)		3.2[**]	0.2	0.6	0.73	38
Cont. corn (conventional tillage)		5.4[**]	0.2	0.7	0.18	38
Cont. corn	Minnesota	8.0	0.3	1.5	0.22	23
Corn (in rotation)		4.6	0.4	1.0	0.24	23
Corn	So. Dakota	1.3	1.0	0.6	0.2	65
Cont. corn	Missouri	18.2	---[†]	5.3[†]	0.41	55
Cont. soybeans		17.4	---[†]	4.1[†]	0.74	55

[*] Flow-weighted average concentrations; kg/ha losses = mg/L × cm/10.

[†] NH_4-N is included with NO_3-N.

[**] Growing season runoff (late April-October).

water and not sediment. For example, in one study (Baker and Laflen, 1979), 82-89% of the atrazine, alachlor, and propachlor losses were in solution. Pesticides in Class III are strongly adsorbed and are lost primarily with sediment in surface runoff. Pesticides in Class I are only weakly adsorbed and in surface runoff are lost mainly with water; they also potentially could be leached.

Pesticides do not naturally exist, do degrade with time, and are usually surface applied or only incorporated to a shallow depth. Class I and II pesticides can be leached from the surface mixing zone. The result of these facts is that the pesticide concentrations in the first runoff event after application usually represent the maximum concentrations. In a review of pesticide runoff data, Wauchope (1978) noted that maximum concentrations of Class II compounds like atrazine, alachlor, cyanazine, metribuzin, propachlor, fonofos, and carbofuran in bulk runoff (sediment and water) were usually less than 1000 μg/L. Maximum concentrations of Class I compounds like 2,4-D and dicamba were in the 2000-5000 μg/L range when applied mainly to foliage, but were a factor of three less when applied to bare soil. Maximum concentrations for Class III compounds like trifluralin were less than 25 μg/L. As Wauchope states, the most soluble pesticides tend to give the highest runoff concentrations, unless they are applied to bare soil where leaching into the soil interior reduces the amount available for wash-off.

Because pesticides shown in Table I-2 that are used in Iowa are not persistent, annual average concentrations in runoff and losses are much less than those estimated from the maximum values just discussed.

It was estimated (Harmon and Duncan, 1978) that 0-1% of that applied would be lost in surface runoff water for Class I pesticides, 0-5% for Class II pesticides, and 0-0.5% for Class III pesticides. That lost with sediment was estimated as 0-0.1%, 0-1%, and 0-2% for Class I, II, and III pesticides, respectively.

It may seem illogical that losses in surface runoff water of the least strongly adsorbed, Class I pesticides are smaller than those for Class II. However, for most Iowa soils under most conditions, a significant amount of rainfall infiltrates before runoff begins during a storm. This infiltration can flush much of a non- or slightly-adsorbed pesticide from the soil surface, whereas stronger adsorption can hold a pesticide on the surface longer to be lost later with surface runoff.

2. Subsurface Flow

Soil adsorption, which is important in determining chemical concentrations in surface runoff, is also an important factor in subsurface drainage. The lack of adsorption that allows NO_3-N to be leached from the thin surface mixing zone also allows NO_3-N to be leached completely through the soil profile with excess precipitation and lost with subsurface flow. As shown in Table I-6, NO_3-N concentrations in subsurface flow from row-cropped areas usually exceed the 10-mg/L drinking water standard, and losses commonly exceed 20-kg/ha. It is also evident that the higher the application rates, the higher the NO_3-N concentrations and losses.

The adsorption of NH_4^+ and $PO_4^{\equiv}$ that causes some NH_4-N and PO_4-P to be lost with sediment in surface runoff also prevents NH_4-N and PO_4-P from readily leaching and being lost with subsurface flow.

Concentrations of NH_4-N and PO_4-P in subsurface flow (Table I-6) are less than those in surface runoff (Table I-5).

Soil adsorption (and low solubility in some cases) also prevents the significant leaching of Class II and III pesticides from non-sandy soils. However, in one study (Muir and Baker, 1976), the Class II herbicides atrazine, cyanazine, and metribuzin, applied to corn ground (sandy loam), were detected at low levels (relative to surface runoff) in subsurface flow from tile drains. Herbicide concentrations ranged from 0.30 to 1.49 μg/L for atrazine, 0.0 to 0.68 μg/L for cyanazine, and 0.0 to 1.65 μg/L for metribuzin. Heavy rainfall and high flow rates resulted in the appearance of herbicides in tile drain water only six days after herbicide application. This rapid flushing of even low levels of herbicides through the soil profile (the drain tile were at depths of 1.2 to 1.6 m) must have resulted from water movement through macropores. Macropores exist in most soils at some time because of soil cracking, root channels, or holes caused by worms or other insects. Dao et al. (1979), in a laboratory study, found that at a high water flow rate of 2 cm/hr, atrazine was leached much more quickly than at 0.08 to 0.04 cm/hr. They believed the herbicide was rapidly displaced through the soil column because of movement through macropores under nonequilibrium conditions where there was minimal chance for contact and adsorption of the herbicides by the soil matrix.

3. Best Management Practices

Agricultural chemicals can be classified in three different groups depending on their major mechanism for transport from the field with agricultural drainage. There are chemicals that are lost primarily

Table I-6. Soluble nutrients in subsurface flow from Midwestern corn and soybean fields.

Crop	State	Flow	NH_4-N	NO_3-N	PO_4-P	Reference
		cm	-----------	mg/L*	-----------	
Mixed cover watersheds	Iowa	--	0.1	12.1	0.12	9
Cont. corn (448 kgN/ha-yr)		9.9	0.2	21.0	--	22
Cont. corn (168 kgN/ha-yr)		11.8	0.2	5.8	--	22
Corn (terraced, 448 kgN/ha-yr)		17.6	0.3	20.0	--	22
Corn-oats-corn-soybeans (low fert.)		14.6	0.3	21.0	0.005	5
Corn-oats-corn-soybeans (high fert.)		12.4	--	40.5	--	5
Corn and soybeans		7.3	--†	13.0†	0.01	32
Mixed cover watersheds		--	--	19.0	--	66
Corn (20 kgN/ha-yr)	Minnesota	7.8	--	17.5	--	30
Corn (112 kgN/ha-yr)		8.2	--	21.5	--	30
Corn (224 kgN/ha-yr)		8.3	--	37.3	--	30
Corn (448 kgN/ha-yr)		9.3	--	61.2	--	30
Mixed cover watersheds	Illinois	--	Range:	5-22	--	34

*Flow-weighted average concentrations (if flow was measured); kg/ha losses = mg/L × cm/10.

†NH_4-N is included with NO_3-N.

with sediment--the strongly adsorbed chemicals such as PO_4-P (and other forms of P) and the Class III pesticides; chemicals lost primarily with surface runoff water--the less strongly adsorbed chemicals such as NH_4-N and the Class I and II pesticides; and chemicals lost primarily with subsurface flow--the non-adsorbed chemicals such as NO_3-N and other ions such as Cl^-. Management practices to control chemical losses must be chosen with these three groups in mind.

Since losses are determined from the product of concentration and the volume of the carrier, losses can be decreased by decreasing concentrations and/or the carrier. Practices such as rates, timing, and methods of chemical application affect concentrations; practices such as conservation tillage affect volumes of carriers (and sometimes concentrations because of differences in methods of chemical application). However, a practice used to control losses from one chemical group may increase losses from another chemical group, as illustrated later.

A Best Management Practice (BMP) by definition must be effective in controlling nonpoint source pollution, but also must be a socially and economically acceptable practice. These criteria severely limit the number of realistic BMPs. One obvious method of decreasing concentrations for all three groups of chemicals is to decrease the rate of application. This has been shown to work for both pesticides (Hall et al., 1972; Barnett, 1967; Bovey et al., 1978) and nutrients (Moe et al., 1967; Dunigan et al., 1974; Baker and Laflen, 1982; Baker and Johnson, 1981; Gast et al., 1978; Timmons and Dylla, 1981; Burwell et al., 1976; Benoit, 1973; Romkens and Nelson, 1974; Romkens et al., 1973; Zwerman et al., 1972; Bolton et al., 1970). In some cases the chemical form applied also affects losses (Barnett, 1967; Moe et al., 1968). Timmons and Dylla (1981) showed that improved timing of N additions through multiple applications versus a single application decreased NO_3-N leaching losses. Any increase in the time interval between chemical application and the first runoff event should decrease concentrations and losses. Chemical placement has been shown to be important, with insecticide loss from a broadcast application being greater than from an in-furrow application (Caro et al., 1973) and losses from a surface application of pesticides (Baker and Laflen, 1979) or nutrients (Timmons et al., 1973; Baker and Laflen, 1982) being much greater than for a soil-incorporated application. One problem with tillage incorporation of chemicals is that soil protecting surface crop residue is also incorporated, increasing the potential for erosion and losses of sediment-associated chemicals.

The use of soil conservation practices such as terraces (Burwell et al., 1974; Schuman et al., 1973a,b; Laflen et al., 1972), grassed waterways (Asmussen et al., 1977), and filter strips (Aull et al., 1980) reduces the loss of sediment and the chemicals sediment carries. However, in one study (Burwell et al., 1974), the use of level terraces also increased infiltration and therefore the losses of NO_3-N with subsurface flow. The use of conservation tillage which leaves some or all of the previous crop's residue on the soil surface is very effective in decreasing erosion and sediment-carried chemicals. However, there is some concern that herbicides broadcast-sprayed on the crop residue may be more susceptible to runoff losses under some conditions (Baker et al., 1978; Martin et al., 1978), and decreased fertilizer incorporation with conservation tillage, particularly for no-till where fertilizer is surface-applied, results in increased nutrient losses (Johnson et al., 1979; McDowell and McGregor, 1980; Romkens et al., 1973; Barisas et al., 1978; Siemens and Oschwald, 1978).

It is very difficult to affect significantly the total volume of drainage water, although in some cases it is possible to shift water from the surface runoff route to the subsurface flow route. The effect of level terraces has already been mentioned, and installation of subsurface drains short-circuits the subsurface flow path and increases subsurface flow (Baker and Johnson, 1976). In a review of conservation tillage effects on water quality, Baker and Laflen (1983) noted that conservation tillage systems generally reduce the volumes of surface runoff by 25%, although the degree of reduction is highly variable. Presumably, subsurface flows would increase by approximately the same volumes. In general, it is evident that attempts to control chemical losses due to either surface runoff or subsurface flow should focus mainly on practices that decrease chemical concentrations.

In pothole areas where ADWs are used to provide the only drainage outlet, surface inlets to the underground drainage system (or direct access to the ADW, e.g., through a grated opening at the soil surface) are provided. In these unique cases the opportunity exists to force all drainage to be subsurface flow by closing the surface inlets. This case is somewhat analogous to level terraces, except that level terraces pond water over larger areas somewhat uniformly distributed across the landscape, whereas in the pothole area a few smaller but deeper ponds would form in scattered depressions. Closing the surface inlets would cause percolation of large volumes of water through the soil in the ponded area and leach out any NO_3-N present (this extra loss would be less than for level terraces because of the smaller area involved; it would also be possible to avoid application of N to this area). Another major disadvantage would be the land taken out of production in the pondage areas and the inconvenience of trying to farm around the resulting scattered wet areas. The advantage of forcing runoff water that collected in the depression to percolate through the soil would be that much of the particulate matter (sediment and bacteria) would be filtered from the water before it entered the subsurface drainage system. There would also be opportunity for the extraction of soil-adsorbed chemicals; however, the higher flow rates caused by the hydraulic head of the ponded water and rapid movement of water through macropores under non-equilibrium conditions (as discussed earlier) would still allow a portion of even the strongly adsorbed chemicals to pass through quickly. Depending on the rates of chemical dissipation in the soil in the pondage area, it would also be possible for chemical concentrations in the soil column in that area to become high enough to decrease the efficiency of extraction or even to release chemical to percolating water that initially had little or no chemical in it, e.g., pesticides to runoff water late in the growing season.

To summarize BMPs, soil conservation practices should be used to control losses of sediment-carried chemicals in group one and soil incorporation of applied chemicals will reduce losses for chemicals in group two that are lost mainly in surface runoff water. Better timing of N applications to match crop needs and reduce amounts present in the soil profile at any one time should reduce leaching losses of NO_3-N, the most important chemical in group three that is lost mainly in subsurface flow. Although decreased application rates would decrease losses of all chemicals, because of the high cost of chemicals very few farmers are putting on amounts in excess of what is needed with current technology. If improved application equipment and procedures are developed, more efficient use of chemicals and resultant lower rates may be possible.

D. References

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II. ADW MONITORING

A. Wells Monitored

Water draining to four ADWs in Humboldt County was monitored during periods of flow in 1981 and 1982. All four wells took drainage from row-cropped areas, and emphasis was placed on sampling during the spring periods when flow was highest and agricultural chemicals had just been applied. Figure II-1 shows the approximate locations of the wells relative to each other (the principal investigators especially appreciate the cooperation of the landowners who allowed their wells to be monitored). Although it was not possible to determine the exact extent of the area of drainage or the amounts of agricultural chemicals used on these areas, it is known that in a watershed similar in intensity of row-crop production to those in the Humboldt area (Johnson and Baker, 1983) that 99% of the corn and 28% of the soybeans were fertilized in 1980. Also, 99% of the row-cropped area in that study received herbicide treatment and 70% of the corn received an insecticide treatment.

Well no. 1 was located on the south side of Highway 3, west of the city of Humboldt. It took drainage from an area that was in soybeans in 1981 and in corn in 1982 and had a circular brick cistern of about 1.5 m in diameter and about 0.9 m high surrounding it. However, the cistern had been moved from its foundation such that the well took both surface runoff and subsurface flow. Sediment had deposited within the cistern to the point that the well and the subsurface tile (about 0.3 m in diameter) draining to it were not readily accessible. During

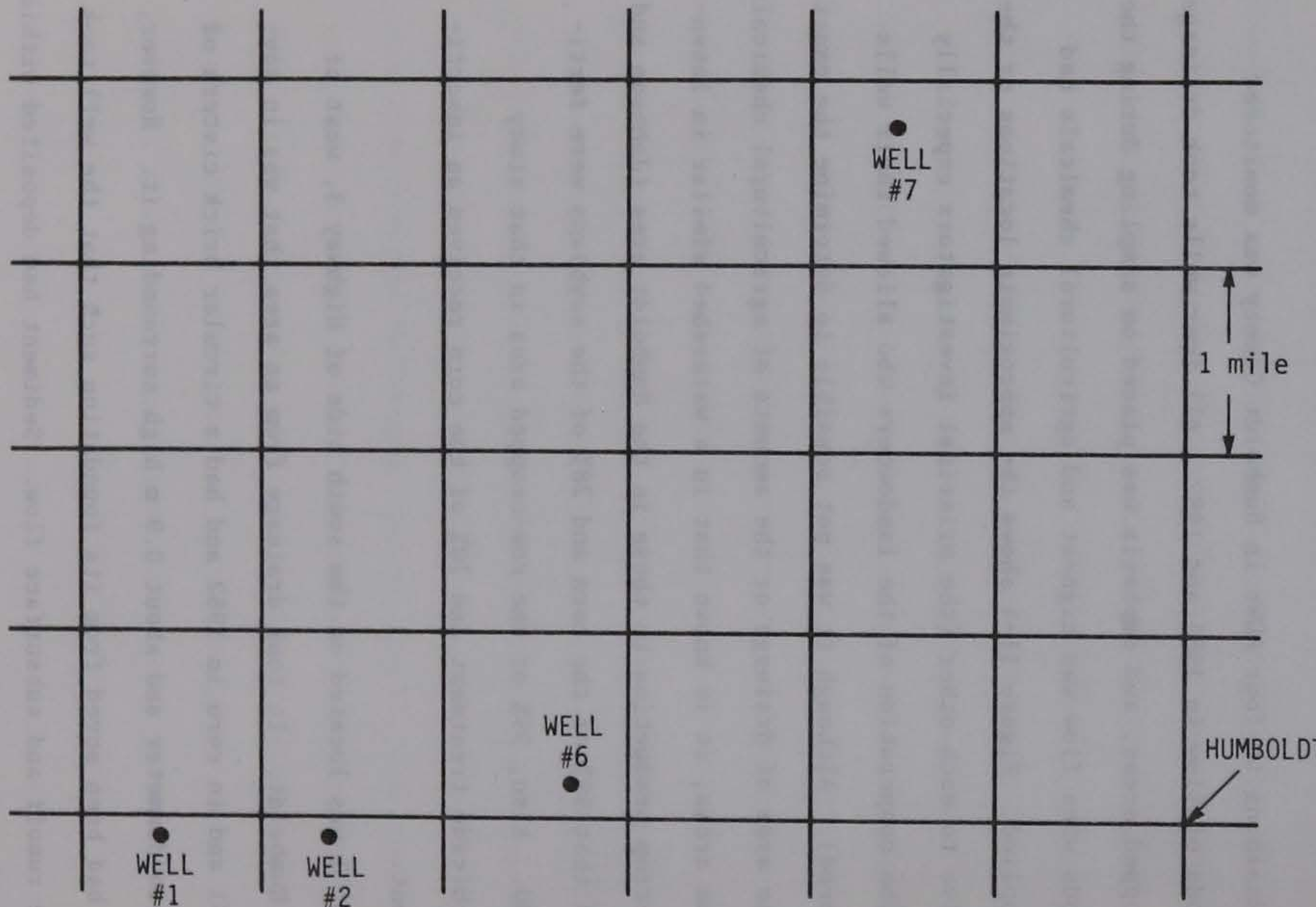

Figure II-1. Relative locations of four monitored ADWs.

recharge events it was possible to hear subsurface drainage reaching the well and, at times, to sample it. It was, of course, possible to sample the surface runoff draining to the well. No depth could be obtained from this well.

Well no. 2 was also located on the south side of Highway 3, west of Humboldt, but east of well no. 1. Well no. 2 took drainage from an area that was believed to be about 36 ha in area, 3/4 of which was in corn and 1/4 of which was in soybeans in 1981, with those areas rotated in 1982 (i.e., 3/4 soybeans, 1/4 corn). This well had a 20 cm diameter casing in the bottom of a brick cistern, with the well inlet protected from clogging by a trash rack. The cistern was about 1.8 m in diameter and 2.7 m high extending 2.4 m below the ground surface (about 0.3 m above the ground surface). There was no evidence that surface water was entering this well as no surface inlets were observed and water would have to pond quite deeply before overtopping the wall of the cistern. The subsurface drains that came in through the side of the cistern were 25 to 30 cm in diameter. It was observed during one period of high recharge that the well would take water for a time and then air would bubble back out. Apparently the well pipe was not flowing full and air being trapped by the entering water was building up pressure that periodically had to be released. A weighted line was used to estimate the depth of this well. Well no. 2 appears to be about 49 m deep and would be recharging into the Gilmore City limestone formation of the Mississippian aquifer.

Well no. 6 was located north of Highway 3, west of Humboldt and east of well no. 2. It took drainage from an area that was in corn in

1981 and soybeans in 1982. It had a unique construction with two small concrete cisterns about 75 cm in diameter, one into which three tile drains emptied and a second, connected to the first through a large diameter clay tile, which had a 15 cm, unguarded well at its bottom. It is believed this well is nearly 100 years old, making it one of the oldest ADWs in Humboldt county. Its location on a side-slope would prevent surface water from entering the well directly, but from February 1982 observation of muddy water in both cisterns at a level equal to that of snowmelt ponded around the well, it was concluded that a surface inlet(s) was connected to this well. A weighted line was used to determine the approximate depth of this well. Well no. 6 appears to be about 87 m deep and would be recharging the Maynes Creek formation of the Mississippian aquifer.

Well no. 7 was located two miles north and two miles east of Rutland, south of a county road. It took drainage from an area that was in corn in 1981 and soybeans in 1982. It had a corrugated metal culvert, 1.2 m in diameter and about 3.5 m long (1 m of which was above ground surface) used vertically, for a cistern which had a 30 cm drain till entering one side and a 15 cm tile entering from the opposite side. There was no evidence that the well took any surface drainage. The subsurface flow entered a 20 cm unguarded drilled well. A weighted line was used to determine the approximate depth of this well. Well no. 7 appears to be about 37 m deep and would be recharging the Gilmore City limestone of the Mississippian aquifer.

In addition to the four ADWs monitored, a water supply well in Sheldon Park (just a few hundred feet west of the Des Moines River at

Humboldt and south of Highway 3) was monitored on about a weekly basis. The Sheldon Park well was drilled in 1967 to a depth of about 54 m (178 feet) into the Mississippian aquifer. Mississippian limestone was reached at a depth of 12 m below the land surface. Two water supply wells on farms close to ADWs 6 and 7 were also each sampled twice. No hydrogeologic data on these two wells could be obtained.

B. Methods and Procedures

The methods used to analyze the water samples for various water quality parameters are as follows:

Parameter	Method	References
Total coliform (TC)	membrane filter	AWWA (1976)
Fecal coliform (FC)	membrane filter	AWWA (1976)
Fecal streptococcus (FS)	membrane filter, two-stage test	Millipore (1972)
pH	potentiometric	AWWA (1976)
NH_4-N	automated phenate method	EPA (1979)
NO_3-N	cadmium reduction method	EPA (1979)
PO_4-P	ascorbic acid reduction	EPA (1979)
Cl	ferric thiocyanate method	EPA (1979)
Ca	flame photometric method	AWWA (1976)
Fe	flame photometric method	AWWA (1976)
Suspended solids (SS)	glass fiber filter	AWWA (1976)
Total solids (TS)	evaporation and weighing	AWWA (1976)

Pesticides were divided into four classes: carbamate insecticides, organophosphate insecticides, chlorinated hydrocarbon insecticides, and herbicides.

Carbamate and organophosphate insecticides are analyzed by the methods in National Pollutant Discharge Elimination System, Appendix A. Fed. Reg., 38, No. 75, PL II. The carbamate are analyzed by G.C. using an N.P. detector and the organophosphates by G.C. using an FPD detector.

The chlorinated hydrocarbon and acid herbicides are analyzed by Methods For Organochlorine Pesticides and Chlorophenoxy Acid Herbicides in Drinking Water and Raw Source Water. EPA - EMSL, Cincinnati.

The acid herbicides were methylated using diazomethane, not boron trifloride, as choramben and dicamba are also determined by using this procedure.

The other (non-acid) herbicides (such as atrazine, cyanazine, alachlor, trifluralin, metribuzin, etc.) were determined using G.C. with a combination of dual column electron capture and N-P detection.

Sampling of water draining to the four monitored ADWs was performed in one of two ways. If there was slowly receding flow to the wells (i.e., no heavy rains or snowmelt in the recent past), then a weekly grab sampling was performed. At the same time automatic samplers were in place (ISCO), capable of taking an aliquot of flow every hour after being triggered by a stage-activated switch. This switch was positioned such that a few cm rise in the water level in the cisterns on ADWs 2, 6, and 7, resulting from rain and an increase in drainage to the wells, would trigger it. For ADW 1, the switch was positioned so that surface runoff ponding around its cistern would trigger it. Samples taken by the automatic samplers were then a composite of several hours of flow during increased recharge rates. The Sheldon Park well was grab-sampled on approximately a weekly basis.

C. Data and Discussion

Data in Tables II-1 and II-2 give arithmetic averages (and ranges and standard deviations) for the water quality parameters measured for water entering the four monitored ADWs and the Sheldon Park well. All of the actual data are given in Tables A-1 through A-5 in the Appendix; these data include some of that taken by Musterman et al. (1981). As expected (see the literature review of Section I), different sources of water had an effect on the quality of water entering the ADWs. In particular, there were differences between snowmelt, rainfall-runoff, and subsurface drainage. Samples taken following at least a week without rainfall (e.g., 6/8/81, 6/16/81, 4/27/82, and 5/4/82) represent strictly subsurface drainage. As such, the NO_3-N, Cl, Ca, and dissolved solids (TS-SS) levels were usually near their highest values, although NO_3-N levels sometimes decreased with time and decreasing flow during a dry period following a wet period (see the Appendix, Table A-6 for rainfall data). For subsurface drainage samples, bacterial levels as represented by fecal coliform were the lowest, usually <10/100 ml. Similarly, pesticide concentrations in subsurface drainage water were either low (<1 ppb) or below the limit of detection of about 0.01 ppb.

The samples taken 2/22/82 and 3/19/82 during snowmelt show higher levels of NH_4-N, PO_4-P, Fe and suspended solids in surface water samples than for subsurface drainage, but lower levels of NO_3-N, Ca, dissolved solids, and sometimes Cl. Suspended solids concentrations in excess of 1000 mg/L in water draining into well no. 6 during snowmelt would indicate that it, as well as no. 1, was taking surface water directly.

Table II-1. Concentrations of nutrients, dissolved solids, and sediment in, and pH of water entering monitored ADWs.

Species	Well No. 1	2	6	7	Sheldon Park Well
NH_4-N mg/L, avg.	.14	.12	.39	.06	.03
(range; S.D.)	(.01-.49; .20)	(.01-1.12; .27)	(.01-3.78; .89)	(.01-.26; .07)	(.01-.11; .03)
NO_3-N mg/L, avg.	10.8	14.8	20.1	17.9	1.5
(range; S.D.)	(1.5-26.0; 7.0)	(2.3-25.0; 5.7)	(1.7-34.0; 8.7)	(6.3-26.0; 5.6)	(.1-11.0; 2.8)
PO_4-P μg/L, avg.	64	183	333	180	121
(range; S.D.)	(20-137; 41)	(10-1882; 410)	(40-1992; 488)	(20-1715; 402)	(40-520; 127)
Cl mg/L, avg.	7.6	40.1	36.0	29.2	7.7
(range; S.D.)	(1.0-12.0; 4.0)	(22.0-120.0; 16.7)	(9.5-49.0; 9.7)	(19.0-38.0; 5.8)	(1.0-39.0; 10.6)
Ca mg/L, avg.	47	121	110	96	84
(range; S.D.)	(13-62; 19)	(60-150; 20)	(18-140; 29)	(36-120; 20)	(53-93; 6)
Fe mg/L, avg.	.15	.11	.24	.14	.58
(range; S.D.)	(.03-.44; .15)	(.01-1.50; .31)	(.01-2.60; .61)	(.01-1.70; 40)	(.06-6.50; 1.65)
Dissolved solids mg/L, avg.	238	581	557	456	379
(range; S.D.)	(78-369; 102)	(249-709; 103)	(75-745; 146)	(137-604; 100)	(272-636; 63)
Sediment mg/L, avg.	1340	17	166	17	3
(range; S.D.)	(1-5360; 2177)	(0-159; 37)	(1-2260; 511)	(1-130; 37)	(1-18; 4)
pH, avg.	7.2	7.5	7.5	7.5	7.4
(range; S.D.)	(6.6-7.8; .4)	(7.1-8.2; .3)	(7.1-7.8; .2)	(7.2-7.8; .2)	(6.8-7.9; .2)

Table II-2. Concentrations of pesticides and bacteria in water entering monitored ADWs.

Species	Well No. 1	2	6	7	Sheldon Park Well
Atrazine µg/L, avg. (range; S.D.)	.02 (0-.12; .04)	.01 (0-.18; .04)	.01 (0-.11; .02)	.03 (0-.50; .12)	<.01 (0-.11; .02)
Cyanazine µg/L, avg. (range; S.D.)	11.8 (0-80.0; 27.9)	.43 (0-7.4; 1.35)	.54 (0-7.5; 1.76)	.49 (0-5.6; 1.36)	0 -
Alachlor µg/L, avg. (range; S.D.)	.15 (0-.70; .24)	.21 (0-2.8; .63)	3.01 (0-55.0; 12.6)	.08 (0-1.2; .29)	.08 (0-2.7; .45)
Dieldrin µg/L, avg. (range; S.D.)	.004 (0-.028; .010)	.001 (0-.009; .002)	.001 (0-.011; .004)	.001 (0-.016; .004)	0 -
Metribuzin µg/L, avg. (range; S.D.)	0 -	0 -	.06 (0-.41; .14)	0 -	<.01 (0-.15; .03)
Dicamba µg/L, avg. (range; S.D.)	.11 (0-.90; .32)	.21 (0-6.1; 1.11)	.78 (0-12.0; 2.78)	.12 (0-1.8; .44)	0 -
Fecal coliform* #/100 mL, avg. (range; S.D.)	105 (20-330; 150)	23 (<10;250; 59)	15800 (<10-180000; 44000)	134 (<10-2000; 520)	5 (<10-20; 8)
Total coliform* #/100 mL, avg. (range; S.D.)	24800 (500-90000; 43500)	860 (<10;6000; 1650)	32100 (<10-260000; 75800)	460 (<10-4000; 1140)	13 (<1-120; 24)

*Results recorded as <10 were taken as zero; results recorded as too numerous to count (>) were not used to compute an average.

Concentrations in excess of 100 mg/L were observed for wells no. 2 and 7, indicating possible short-circuiting of ponded snowmelt water to subsurface drains through macropores in the soil profile (i.e., quasi-surface runoff).

Samples taken 5/4/81, 5/24/81, 6/24/81, 4/20/82, 5/7/82, 5/18/82, and 5/27/82 closely followed one or several rainfall events totaling at least 20 mm when surface runoff (or ponding) could be expected to take place. It was for these samples that the highest levels of pesticides and bacterial counts were measured. The samples of 5/24/81 illustrate the difference between surface and subsurface flow with respect to sediment. Wells nos. 1 and 6 with surface runoff sources had 5360 and 2260 mg/L, respectively, while wells nos. 2 and 7, with subsurface flow only, had 100 and 28 mg/L. It was also during large recharge events that the lowest pH values were measured. However, the influence of surface water on the amount of NO_3-N (and dissolved solids) in water draining to the ADWs was not as pronounced for rainfall runoff as for snowmelt. This implies that under the wet conditions following rainfall, a significant portion of the drainage to the ADWs was still from subsurface drainage.

The source of total coliform bacteria (TC) in water draining to ADWs may be animal feces, soil, vegetation, etc. The presence and level of contamination give some indication of the efficiency of the subsoil to filter out these bacteria, which in turn depends on the travel time and distance of water coming from the more biologically active surface soil.

Fecal coliforms (FC) are inhabitants of warm-blooded animal intestines and may be considered indicators of recent fecal pollution. The presence of other coliform organisms as measured by total coliform suggests less recent pollution or contributions from other sources of nonfecal origins. The ratio of fecal coliform to fecal streptococcus (FS) can be used to determine the relative contributions of fecal organisms from humans, livestock, and small wild animals (Geldreich, 1966). Ratios of FC/FS above 4.0 indicate human sources, 0.4 to 0.1 indicate livestock and poultry sources, and less than 0.04 indicate small, wild animal sources. Differential die-off or analytical interference from other bacteria may reduce the significance of the measured ratios.

Samples taken 5/7/82, 5/18/82, 6/1/82, and particularly 5/27/82 had high bacterial levels. In general, these dates followed rainfall events when flows to the wells would be expected to increase. The high values of 4/13/82 inexplicably occurred after a five-day dry period. As shown in Table II-2, wells nos. 2 and 7, which do not have surface inlets, had much lower maximum and average values for total coliform than wells 1 and 6 with surface inlets. Wells 2 and 7 had fecal coliform and fecal streptococcus levels ≤50, with the exceptions of 5/18/82 and 5/27/82, while wells nos. 1 and 6 had levels often exceeding 100 (fecal coliform had a maximum value of 180,000 for well no. 6 on 5/27/82). For wells nos. 1 and 2, the FC/FS ratio averaged about 0.2, indicating that the source of fecal contamination was probably livestock. For well no. 6, the ratio varied from 0.06 to 4.3, indicating that at various times different sources dominated; sometimes human, sometimes livestock. It was not possible to obtain a ratio for well no. 7.

The maximum total coliform count for the Sheldon Park well was 120 per 100 ml on 5/25/82. This was also the day of maximum fecal coliform and fecal streptococcus levels (as well as maximum NO_3-N concentrations). Nine of the 13 samples analyzed for fecal coliform had less than 10 per 100 ml.

The presence and levels of NH_4-N, PO_4-P, Cl and Ca do not present much of a water quality concern (with the exception of hardness and the problem of excess NH_4-N reducing the efficiency of the chlorination process), but in some cases they can be used to estimate roughly the proportion of total flow to an ADW that is coming fairly directly from the soil surface. At the least, substantial increases in NH_4-N and PO_4-P (e.g. to values above 0.2 mg/L) would indicate that surface water is influencing the quality of water draining to an ADW. The reasons for the increase are that surface soils are usually more fertile than subsoils with respect to N and P because of fertilization and higher organic matter; hence, water contacting these soils has higher NH_4-N and PO_4-P concentrations. In addition, upon slow passage through the soil profile, as in the case of subsurface flow, NH_4-N can be removed from water by cation exchange, and PO_4-P can be removed by precipitation. As shown in Table II-1, well no. 6, which has a surface inlet, had the highest maximum and averages for NH_4-N and PO_4-P, although well no. 1, also taking surface drainage, had the lowest PO_4-P values.

For Cl and Ca, decreases in concentrations would indicate that there is a surface water influence. The Cl ion (plus the NO_3-N ion discussed later) is readily leached from the surface and the Ca ion is dissolved from subsoil sources; thus they have higher concentrations in subsurface flow than in surface runoff. The prime examples of increased NH_4-N and PO_4-P and decreased Cl and Ca concentrations indicating the influence of surface water on the quality of water draining to the ADWs are the samples of 2/22/82 and 3/19/82 taken during snowmelt (see Tables A-1 to A-4 in the Appendix). The maximums and averages in Table II-1 particularly show the influence of surface runoff quality on values of Cl, Ca and dissolved solids for well no. 1.

For iron, concentrations in excess of 0.3 mg/L may cause objectionable tastes or laundry staining, but the 0.3 mg/L standard is of aesthetic rather than toxicological significance. This value for drainage to the ADWs was exceeded only during snowmelt and for the samples taken 5/27/82 during an increase in flow following a rainstorm. The high values of 5/27/82 ($\geq$1.5 mg/L) may have resulted from the influence of surface water which had been in contact with lower pH surface soils.

Concentrations of NO_3-N entering the ADWs exceeded the 10 mg/L standard for 85% of the samples. As shown in Table II-1, average concentrations ranged from 10.8 to 20.1 mg/L (with an overall average of 16 mg/L) with a maximum of 34 mg/L for a single sample. The times at which concentrations were depressed, or below 10 mg/L, were usually during the influence of surface flows with their lower NO_3-N concentrations. During the wet spring of 1982 (April, May, June) with sustained subsurface drainage, NO_3-N concentrations in water draining to the four ADWs averaged 21 mg/L, whereas they averaged 13 mg/L in the dry spring of 1981.

For the Sheldon Park well, just west of the Des Moines River a few hundred feet, NO_3-N concentrations were very low (average <1 mg/L) for all of 1981 and up to 4/27/82 (116 mm of rain fell between 4/1 and 4/27). From 4/20 to 5/25, NO_3-N concentrations in water from the Sheldon Park well increased from 0.2 to 11 mg/L. In that same period, Cl concentrations increased from 3 to 33 mg/L (concentrations measured for a weekly sampling of the Des Moines River downstream at Boone were 14.5 mg for NO_3-N and 33 mg/L for Cl on 5/19/82; Baumann et al., 1983). Then from 5/25 to 6/29, NO_3-N and Cl concentrations in the Sheldon Park well decreased to 1.4 and 10 mg/L, respectively. Linear regression techniques showed that the correlation between NO_3-N and Cl concentrations was significant at the 1% level for the Sheldon Park well (NO_3-N $= -0.4 + 0.25$ Cl; $r = .956$, $n = 42$), but was not for eight quarterly Cl samples taken from the Des Moines River in 1981 and 1982 (NO_3-N $= 21.3 - 0.36$ Cl; $r = -.038$, $n = 8$) (Baumann et al., 1982, 1983).

At times, several different pesticides were detected in water draining to the ADWs, but always at levels less than 100 ppb and usually less than 1 ppb. The pesticides detected were alachlor, atrazine, carbofuran, chlordane, cyanazine, 2,4-D, dicamba, dieldrin, and metribuzin at maximum concentrations of 55, 0.5, 0.6, 1.8, 80, 0.4, 12, 0.028, and 0.41 ppb, respectively. The detection of chlordane in the samples from wells nos. 2 and 6 on 3/30/82 was surprising, as chlordane is not heavily used in Iowa and was not detected in any samples other than those during snowmelt. Over half of the samples taken of water draining to the ADWs and analyzed for pesticides had no pesticides above detectable limits.

Of the 35 samples taken from the Sheldon Park well and analyzed for pesticides, only four were found to contain pesticides. The maximum concentrations were 2.7 ppb for alachlor, 0.15 ppb for metribuzin, and 0.11 ppb for atrazine. As a point of reference, atrazine has a higher (less toxic) acute oral LD_{50} than aspirin, and one would have to drink 3,000,000 L of water at 0.11 ppb to ingest as much atrazine as there is aspirin in a single five-grain aspirin tablet (more water than anyone consumes in a lifetime). Of course, chronic toxicity is of most concern, and little is known about the effects of long-term but very low-level exposure to most pesticides. However, if the proposed MALs in Table I-4 for alachlor (1000 ppb) and atrazine (215 ppb) are approved (none are yet proposed for metribuzin), there should be no problem with respect to these two heavily used herbicides unless concentrations in groundwater increase dramatically with time (however, it is likely that the MAL value for alachlor will be lower after new data are available).

On two occasions, in December 1980, and July 1981, a farm well within one-half mile of well no. 6 and another farm well within one-half mile of well no. 7 were sampled. The data (Table A-7 in the Appendix) show that no presticides were found in any sample, and the NO_3-N content in both wells averaged less than 5 ppm.

D. References

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III. ESTIMATION OF POLLUTANTS INJECTED BY ADWs

Knowledge of the volume of drainage is necessary to determine the pollutant loads from concentration data, and is also important in determining the expected dilution of the drainage water entering the total groundwater system through ADWs. The physical characteristics of ADWs, the method in which they function, a lack of knowledge about the area which they drain, and the variability caused by weather make flow measurements very difficult. Therefore, two different mathematical models were used to estimate the volumes of water delivered to the ADWs to be injected. One of these models, CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems), developed by the USDA, was also used to predict sediment, nutrient, and pesticide concentrations and loads in surface flow and NO_3-N in subsurface flow. The second model, developed at ISU and used as a check on the CREAMS model, was also used to predict NO_3-N concentrations and loads delivered to ADWs with subsurface drainage water.

A. CREAMS Model

The CREAMS model (Knisel, 1980) simulates three hydrologic systems as separate sub-models. The brief description of these three sub-models and their data input needs are given in the following sections.

1. Hydrology Model

The CREAMS hydrology model simulates the processes of infiltration, soil water movement, and soil/plant evapotranspiration between storms.

It is a continuous model using a day as a time step for evapotranspiration and soil water movement between storms, and using shorter time increments, dictated by the available rainfall records, during storms.

The hydrology model operates on a given rainfall data sequence plus a record of average monthly radiation and temperature, with information on crop, soil profile, and field shape, to generate a sequence of information on runoff, evapotranspiration, and seepage. This output information is produced on the hydrology pass file and is used by the erosion, pesticide, and nutrient models in simulating sediment and chemical transport.

Two options are available to the user: option one, which uses daily rainfall data; and option two, which uses the breakpoint or hourly rainfall data. Hourly rainfall data for Humboldt County (rain gauge station Humboldt 2) were used in this study. These data were recorded by the National Climatic Center, Asheville, NC, on magnetic tape, with a standard format, for the years 1957 through 1979. The data were corrected by comparing them with daily rainfall for Humboldt County and converted to the format required by the CREAMS model in hydrology option two (the same format as breakpoint rainfall data).

Figure III-1 illustrates the variations in the annual rainfall for the 23 years of record, with an average of 780.0 mm.

The hydrology model is designed to use physically related or easily estimable parameters as much as possible. It does not depend on extensive detail for soil or field topography. The simplifications used are dictated largely by data limitations rather than ignorance of the interrelations of the physical processes involved.

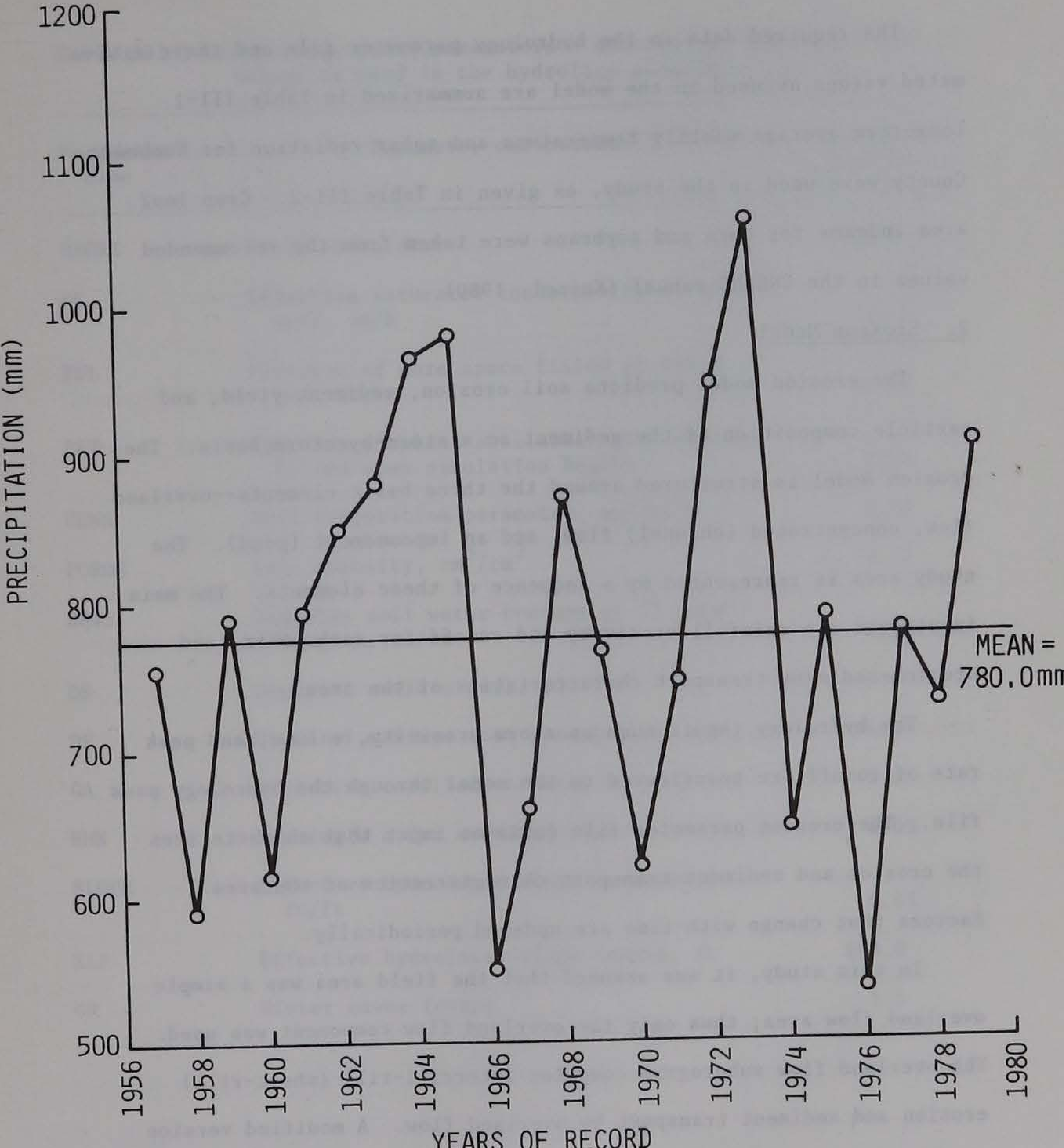

Figure III-1. Variation of annual rainfall, Humboldt 1957-1979.

The required data in the hydrology parameter file and their estimated values as used in the model are summarized in Table III-1. Long-term average monthly temperature and solar radiation for Humboldt County were used in the study, as given in Table III-2. Crop leaf area indexes for corn and soybeans were taken from the recommended values in the CREAMS manual (Knisel, 1980).

2. Erosion Model

The erosion model predicts soil erosion, sediment yield, and particle composition of the sediment on a storm-by-storm basis. The erosion model is structured around the three basic elements--overland flow, concentrated (channel) flow, and an impoundment (pond). The study area is represented by a sequence of these elements. The main input data are rainfall erosivity and runoff for each storm, and erosion-sediment transport characteristics of the area.

The hydrology inputs such as storm erosivity, volume, and peak rate of runoff are transferred to the model through the hydrology pass file. The erosion parameter file contains input that characterizes the erosion and sediment transport characteristics of the area. Factors that change with time are updated periodically.

In this study, it was assumed that the field area was a simple overland flow area; thus only the overland flow component was used. The overland flow subprogram computes interrill-rill (sheet-rill) erosion and sediment transport by overland flow. A modified version of USLE (Universal Soil Loss Equation) has been used. This relation uses input values for storms such as the rainfall erosivity factor, runoff volume, and peak discharge rate, and also factors such as soil erodibility, cover, management, and contouring.

Table III-1. Hydrology model parameters: description and calibrated values as used in the hydrology program.

Parameter name	Parameter definition	Estimated values
DACRE	Field area, acre	40.0
RC	Effective saturated conductivity of the soil, in/h	0.18
FUL	Fraction of pore space filled at field capacity	0.78
BST	Fraction of plant-available water storage filled when simulation begins	0.50
CONA	Soil evaporation parameter, mm/day$^{1/2}$	3.50
POROS	Soil porosity, cm^3/cm^3	0.47
BR15	Immobile soil water content at 15 bars tension, in/in	0.19
DS	Depth of surface soil, in	2.0
DP	Depth of maximum root growth layer, in	34.0
GA	Effective capillary tension of soil, in	9.0
RMN	Manning's n for overland flow	0.03
SLOPE	Effective hydrologic slope steepness, ft/ft	0.01
XLP	Effective hydrologic slope length, ft	200.0
GR	Winter cover (crop)	1.0

Table III-2. Average monthly temperature and solar radiation for Humboldt County, Iowa.

Month	Average temperature °F	Average solar radiation Langleys/day
January	18.0	174.0
February	22.0	253.0
March	32.5	326.0
April	48.0	403.0
May	60.0	480.0
June	69.5	541.0
July	74.5	536.0
August	72.0	460.0
September	63.5	367.0
October	52.5	274.0
November	35.5	167.0
December	23.0	143.0

The required hydrology input data are transferred to the erosion model through the hydrology pass file. Watershed characteristics and management practices are collected in the erosion parameter file. In this file, the default values suggested in the CREAMS manual were used for those parameters where no measured values were available. The numerical values of other parameters were estimated from the site data.

The original surface soil layers (Webster silty clay in Humboldt County) consist of 25% clay, 70% silt, and 5% sand. The fraction of organic matter was taken as 2.5%. Based on these soil data, the soil erodibility factor was determined by use of erodibility nomograph to be 0.38. A watershed area of 40.0 acres, with the slope length of 200.0 feet and slope steepness of 0.01, was used in the erosion model to be compatible with the hydrology model. Soil loss ratios were determined based on the cropping management of corn after corn with spring tillage and residue left on the ground, as given in Table III-3 (Wischmeier and Smith, 1978).

Erosion model output is transferred to the chemical model in the erosion pass file. The erosion pass file is the same as the hydrology pass file, except that the numerical values of the amount of soil loss and sediment enrichment ratio computed in the erosion model are substituted for the excess rainfall rate and rainfall erosivity factor in the hydrology pass file.

3. Nutrient Model

The major chemicals essential for the plant growth are nitrogen (N), phosphorus (P) and potassium (K). The CREAMS model considers only N and P as principal nutrient pollutants. The nutrient model takes the

Table III-3. Soil loss ratios for various periods during the growing season.

Period of growing season		Soil loss ratio
Period F:	rough fallow, plowing to secondary tillage (March 1 to April 15)	0.31
Period SB:	seed bed, secondary tillage to 10% canopy development (April 15 to June 1)	0.55
Period 1:	establishment, 10% to 50% crop canopy development (June 1 to July 1)	0.48
Period 2:	development, 50% to 75% crop canopy development (July 1 to July 15)	0.38
Period 3:	maturing crop, 75% canopy developed to harvest (July 15 to October 1)	0.20
Period 4:	residue or stubble, harvest to plowing or new seeding (October 1 to March 1)	0.23

required input data from the erosion pass file and the nutrient input file data and predicts the amounts of nitrogen and phosphorus in runoff water and lost with sediment, N mineralized, plant N uptake, nitrate leached, and N denitrified. It also estimates nitrogen storage; i.e., soil nitrate and solution N, so the model could be run sequentially for as many years as data are available.

The hydrology model provides estimates of the volume of runoff, percolation below the root zone, soil water content, plant growth, and water use. The erosion model estimates sediment loss on a field scale. These data are transferred to the nutrient model in the erosion pass file to be used along with the required nutrient input parameters to predict nitrogen and phosphorus losses.

The required hydrology and sediment input data are transferred to the nutrient model through the erosion pass file. The required nutrient input data and their numerical values as used in the model are summarized in Table III-4.

In the initial runs of the CREAMS nutrient model (version 1.6), two problems were observed:

1. Dentrification was unrealistically high
2. Plant nitrogen uptake for the first storm in the third year was also in error

Therefore, the computation of denitrification and plant N-uptake in the nutrient source program was checked, and the errors were corrected.

To account for nitrogen carryover from soybeans, it was assumed that actual fertilizer N applied would be adjusted down by 25-30%, but that fertilizer N plus carryover would still be 150 kg/ha (134 lb/ac).

Table III-4. Nutrient model parameter definitions and calibrated values as used in the nutrient program.

Parameter name	Parameter definition	Estimated value
SOLPOR	Soil porosity, percent by volume	0.47
FC	Field capacity, percent by volume	0.41
OM	Average organic matter in the total root zone (915 mm), percent of soil mass	1.25
SOLN	Initial soluble nitrogen in surface cm of soil, kg/ha	1.0
SOLP	Initial soluble phosphorus in surface cm of soil, kg/ha	0.2
NO3	Initial total nitrate in the root zone, kg/ha	50.0
SOILN	Content of total nitrogen in the surface soil, kg/kg	0.0012
SOILP	Content of total phosphorus in the surface soil, kg/kg	0.0006
EXKN & EXKP	Extraction coefficient for N and P	0.075
AN & AP	Enrichment coefficient for N and P	7.0
BN & BP	Enrichment exponent for N and P	-0.15
RCN	Nitrogen concentration in rainfall, mg/L	1.60
NF	Number of fertilizer applications	1 & 3
DEMERG	Julian date of emergence, 10 days after planting	See Table III-5
DHRVST	Julian date of harvest	270
CROP	Crop type	1.0 = corn 2.0 = soybeans
SNAJ	Soybean N proportion factor, kg/ha	100.0

Table III-4. Continued.

Parameter name	Parameter definition	Estimated value
RZMAX	Maximum depth of root zone, mm	915.0
Y_p	Potential yield, kg/ha	9400.0[a]
POTM	Potential mineralizable nitrogen, kg/ha	150.0
AWU	Actual water use, mm, variable with year	From hydrology output
PWU	Potential water use, mm	358.50[b]
C_1 & C_3	Cubic coefficient in N uptake equation	0.0209 & 0.0128[c]
C_2 & C_4	Cubic exponent in N uptake equation	-0.157 & -0.415[d]
DF	Date of fertilizer application, variable with year	See Table III-5
FN	Nitrogen applied, kg/ha	75, 150, 225[e]
FP	Phosphorus applied, kg/ha	33.0
FA	Surface fraction of application	0.10

[a] Y_p = 3020.0 kg/ha for soybeans.

[b] PWU = 476.50 mm for soybeans.

[c] $C_1 = C_3 = 0.0259$ for soybeans.

[d] $C_2 = C_4 = -0.104$ for soybeans.

[e] No nitrogen fertilizer was applied for soybeans; 75, 150, 225 kg/ha = 67, 134, and 201 lb/ac.

Therefore, the input value of 150 kg/ha N really represents about 110 kg/ha from fertilizer and 40 kg/ha from soybean carryover. Assumed fertilizer and pesticide application dates are given in Table III-5.

4. Pesticide Model

The pesticide model is designed for field scale application and provides estimates of pesticide mass and storm mean concentrations at the edge of the field. Foliar and soil-applied pesticides are separately described so that different decay rates can be used for each source of the chemical if necessary. In this study, only soil-applied pesticides are considered. Pesticide extraction by raindrop splash and interrill soil movement occurs in a very shallow layer, whereas extraction from rills may extend several centimeters deep. However, in the CREAMS pesticide model, these processes are conceptually combined for simplicity.

A hydrology pass file is used to generate an erosion pass file which also contains the hydrology data required by the pesticide model. Rainfall and runoff volume, and sediment yield and the sediment enrichment ratio, are obtained from the erosion pass file.

Additional pesticide model parameters and inputs which characterize individual pesticide application with their numerical values as used in the pesticide program are given in Table III-6.

The pesticide model was revised to print out the pesticide mass and percent lost in both water and sediment, besides the total values, at the end of each year. The average annual pesticide concentrations in runoff water and sediment were also calculated.

Table III-5. Planting, pesticide and fertilizer application dates, as used in the model, for the period 1957 to 1979.

Year	Date of planting and pesticide application	Date of fertilizer application 1st after April 1	2nd after June 1	3rd after July 1
1957	4/29	4/3	6/4	7/6
1958	4/27	4/9	6/6	7/7
1959	4/27	4/4	6/4	7/4
1960	4/27	4/7	6/4	7/4
1961	4/28	4/7	6/4	7/4
1962	5/3	4/9	6/8	7/10
1963	5/7	4/6	6/9	7/10
1964	4/25	4/8	6/4	7/8
1965	4/30	4/2	6/10	7/4
1966	4/26	4/5	6/18	7/17
1967	4/29	4/4	6/3	7/3
1968	4/28	4/6	6/3	7/10
1969	5/1	4/3	6/3	7/12
1970	4/27	4/4	6/3	7/9
1971	5/3	4/3	6/12	7/13
1972	4/26	4/7	6/10	7/11
1973	4/29	4/12	6/10	7/7
1974	4/28	4/3	6/2	7/6
1975	4/30	4/4	6/7	7/4
1976	4/28	4/3	6/4	7/4
1977	4/30	4/7	6/4	7/4
1978	4/28	4/8	6/4	7/9
1979	4/28	4/5	6/4	7/6

Table III-6. Pesticide model parameters; definition and calibrated values as used in the model.

Parameter name	Parameter definition	Estimated value
NPEST	Number of pesticide applications	3
PBDATE & PEDATE	Date the model begins and ends to consider pesticide (Julian dates)	57105 to 79365
APDATE	Date of pesticide application, variable with the year	See Table III-5
PSTNAM	Pesticide names	atrazine, alachlor and fonofos
APRATE[a]	Rate of application, kg/ha	2.24[b]
DEPINC[a]	Depth of incorporation, cm	1.0[c]
EFFINC	Efficiency of incorporation	1.0
FOLFRC	Fraction of pesticide applied to the foliage	0.0
SOLFRC	Fraction of pesticide applied to the soil	1.0
FOLRES	Amount of pesticide residue on the foliage prior to the application, μg/g	0.0
SOLRES	Amount of pesticide residue in the soil prior to application, μg/g	0.0
WSHFRC	Fraction on the foliage available for rainfall washoff	0.0

Table III-6. Continued.

Parameter name	Parameter definition	Estimated value
WSHTHR	Rainfall threshold for foliage washoff, cm	0.0
SOLH2O[a]	Water solubility, ppm	30.0[d]
HALIF	Eoliare residue half-life	0.0 (not applied)
EXTRCT	Extraction ratio, soil:water ratio in the mixing zone	0.10
DECAY[a]	Decay constant, day^{-1}	0.0154[e]
KD[a]	Distribution coefficient	4.0[f]

[a]The tabulated values are for atrazine.

[b]APRATE = 2.24 kg/ha for lasso and 1.12 kg/ha for dyfonate.

[c]DEPINC = 1.0 cm for lasso and 5.0 cm for dyfonate.

[d]SOLH2O = 242.0 ppm for alachlor and 13.0 ppm for fonofos.

[e]DECAY = 0.0495 d^{-1} for alachlor and 0.0347 d^{-1} for fonofos.

[f]KD = 6.0 for alachlor and 50.0 for fonofos.

Similar to the nutrient model, the user can specify the type of output desired from the pesticide model. It generates annual summaries only, monthly and annual summaries, or individual storm as well as monthly and annual summaries.

5. Results and Discussion

The CREAMS hydrology model, using hourly rainfall and the average monthly temperature and radiation data, predicts surface runoff, deep percolation, actual evapotranspiration, and soil moisture storage for the study period 1957 to 1979.

Figure III-2 illustrates the yearly variation in surface and subsurface flows with the corresponding precipitation. The annual average precipitation of 775 mm generated an annual average of 28 mm of surface runoff and 89 mm of subsurface flow (below the three-foot root zone). In the absence of actual data on flow to ADWs, which are difficult if not impossible to obtain, the predicted flow values can be combined with actual measured concentration data to estimate loadings.

Recorded stream flows for 1960 through 1979 for the rivers at stations in the vicinity of the study area (Humboldt County), including the Iowa River at Rawan, the Des Moines River at Humboldt, the Boone River near Webster City, and the Des Moines River near Stratford, were used to evaluate the model predictions (Table III-7). The average annual measured stream flows for these six rivers were compared with the total surface and subsurface flows predicted by the model as shown in Fig. III-3. While the predicted values for some years do not agree very closely with the measured values, the average values are reasonably close. The measured records had an average of 148 mm, whereas

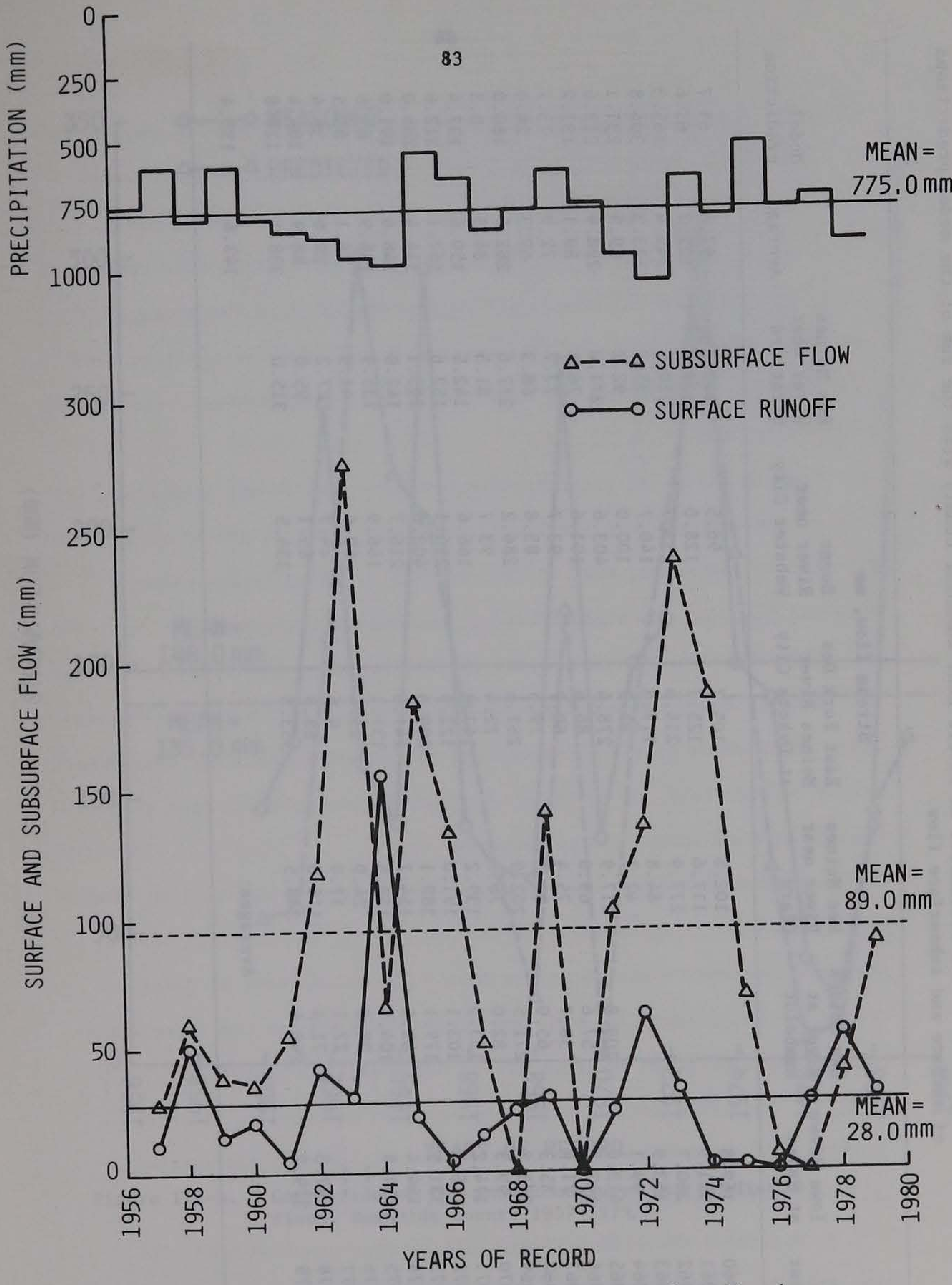

Figure III-2. Yearly variation in precipitation and predicted surface runoff and subsurface flow, Humboldt County 1957-1979 (CREAMS model).

Table III-7. Comparison of the streamflow of rivers near Humboldt County with the sum of the model predictions of surface and subsurface flow.

	Stream flow, mm							
Year	Iowa River at Rawan	Des Moines River at Humboldt	Des Moines River near Clare	East Fork Des Moines River at Dakota City	Boone River near Webster City	Des Moines River near Stratford	Average	Model prediction
1960	169.8		102.6	105.7	96.5	113.8	97.8	61.7
1961	121.9		117.6	125.7	128.0	117.6	122.2	67.6
1962	280.7		277.9	214.6	253.7	220.5	249.4	165.3
1963	153.9		84.8	117.3	140.7	95.7	113.3	304.8
1964	99.1		47.7	11.0	100.0	92.2	89.9	231.1
1965	413.0	209.8	217.9	278.4	403.6	271.0	298.9	213.6
1966	143.2	57.6	69.8	86.6	101.6	76.4	89.1	137.2
1967	71.4	58.7	75.4	60.7	93.7	77.7	72.9	73.7
1968	85.3	65.0	31.7	79.5	85.8	68.3	69.3	24.0
1969	273.8	271.5	202.9	261.6	286.2	277.6	262.4	180.0
1970	99.3	82.0	76.7	72.4	93.7	81.5	84.3	0.3
1971	174.2	133.3	139.2	147.3	166.6	142.5	150.6	137.4
1972	216.4	103.1	191.0	122.9	216.1	152.6	167.1	212.6
1973	331.5	179.1	389.1	299.5	402.6	295.1	314.4	286.0
1974	154.4	82.5	154.7	141.5	216.7	142.0	148.6	191.0
1975	151.9	104.6	130.5	131.1	166.9	135.9	136.9	83.6
1976	---	28.9	24.9	25.4	69.6	41.9	38.1	65.5
1977	---	22.1	17.0	9.6	24.9	27.2	20.0	39.4
1978	97.3	71.4	115.6	68.8	85.1	95.0	88.9	104.4
1979	379.2	298.4	348.5	353.3	334.5	315.0	338.1	128.8
			Averages:				147.6	135.4

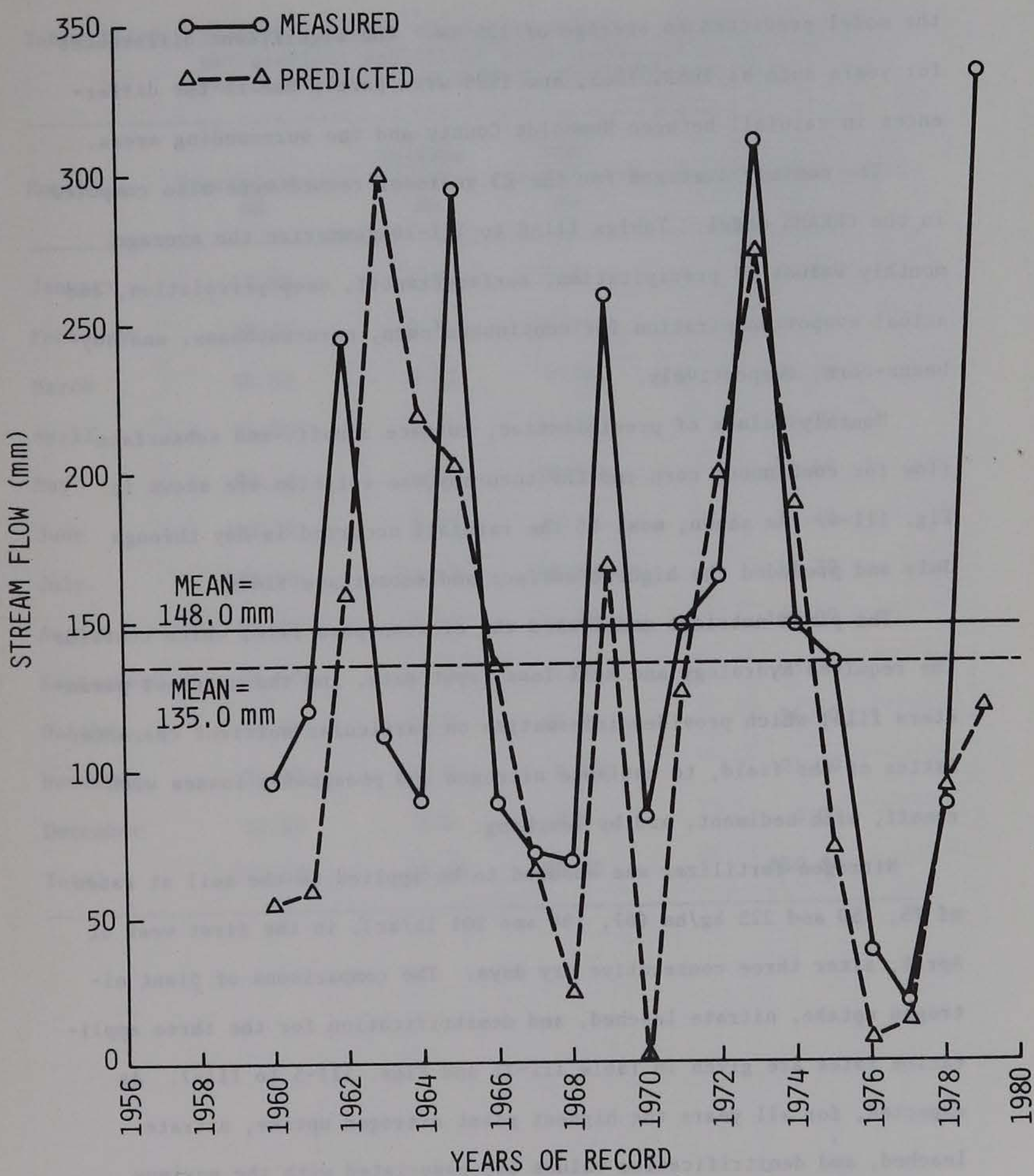

Figure III-3. Comparison of the annual measured and predicted flows, Humboldt County 1957-1979.

the model predicted an average of 135 mm. The significant differences for years such as 1963, 1969, and 1979 were partly due to the differences in rainfall between Humboldt County and the surrounding areas.

The monthly averages for the 23 years of record were also computed in the CREAMS model. Tables III-8 to III-10 summarize the average monthly values of precipitation, surface runoff, deep percolation, and actual evapotranspiration for continuous corn, corn-soybeans, and soybeans-corn, respectively.

Monthly values of precipitation, surface runoff, and subsurface flow for continuous corn and the corn-soybean rotation are shown in Fig. III-4. As shown, most of the rainfall occurred in May through July and produced the highest surface and subsurface flows.

The plant nutrient model used the erosion pass file, which contains the required hydrology and soil loss input data, and the nutrient parameters file, which provides information on particular nutrient characteristics of the field, to estimate nitrogen and phosphorus losses with runoff, with sediment, and by leaching.

Nitrogen fertilizer was assumed to be applied to the soil at rates of 75, 150 and 225 kg/ha (67, 134 and 201 lb/ac), in the first week of April, after three consecutive dry days. The comparisons of plant nitrogen uptake, nitrate leached, and denitrification for the three application rates are given in Table III-11 and Figs. III-5 to III-7. As expected, for all years the highest plant nitrogen uptake, nitrate leached, and denitrification values are associated with the maximum application rate (225 kg/ha).

To determine the effect of multiple nitrogen applications, the same amounts of fertilizer (75, 150, 225 kg/ha) were considered to be

Table III-8. Monthly averages of precipitation, surface runoff, deep percolation, and actual evapotranspiration for the 23 years of record, 1957-1979, for continuous corn.

Month	Precipitation mm	Surface runoff mm	Deep percolation mm	Evapotranspiration mm
January	17.90	0.0	0.0	0.40
February	23.30	0.0	0.0	0.0
March	50.50	0.23	9.95	29.90
April	75.00	0.74	14.35	54.85
May	93.50	0.86	16.35	63.65
June	113.70	5.75	26.70	74.40
July	110.40	13.40	18.80	113.30
August	92.20	5.35	0.90	162.00
September	88.50	2.30	1.95	93.90
October	51.40	0.0	0.0	36.40
November	33.20	0.0	0.20	21.10
December	22.80	0.0	0.0	0.75
Total	722.30	28.62	89.20	650.80

Table III-9. Monthly averages of precipitation, surface runoff, deep percolation, and actual evapotranspiration for the 23 years of record, 1957-1979, for corn-soybean rotation.

Month	Precipitation mm	Surface runoff mm	Deep percolation mm	Evapotranspiration mm
January	17.90	0.0	0.0	0.41
February	23.30	0.0	0.0	0.0
March	50.50	0.23	9.50	29.95
April	75.00	0.74	14.65	54.85
May	93.50	1.02	16.65	62.65
June	113.70	5.87	25.20	93.30
July	110.40	14.73	8.80	145.90
August	92.20	5.31	0.0	137.0
September	88.50	2.87	1.95	79.85
October	51.40	0.0	0.0	34.90
November	33.20	0.0	0.20	21.72
December	22.80	0.0	0.0	1.02
Total	722.30	30.25	77.0	661.65

Table III-10. Monthly averages of precipitation, surface runoff, deep percolation, and actual evapotranspiration for the 23 years of record, 1957-1979, for soybean-corn rotation.

Month	Precipitation mm	Surface runoff mm	Deep percolation mm	Evapotranspiration mm
January	17.90	0.0	0.0	0.43
February	23.30	0.0	0.0	0.0
March	50.50	0.23	9.80	29.95
April	75.00	0.73	14.25	54.90
May	93.50	1.05	16.45	62.60
June	113.70	5.80	22.90	93.75
July	110.40	14.15	12.40	149.50
August	92.20	5.30	0.90	129.85
September	88.50	1.90	2.0	82.25
October	51.40	0.0	0.0	34.65
November	33.20	0.0	0.0	21.80
December	22.80	0.0	0.0	1.04
Total	772.30	29.15	78.65	660.00

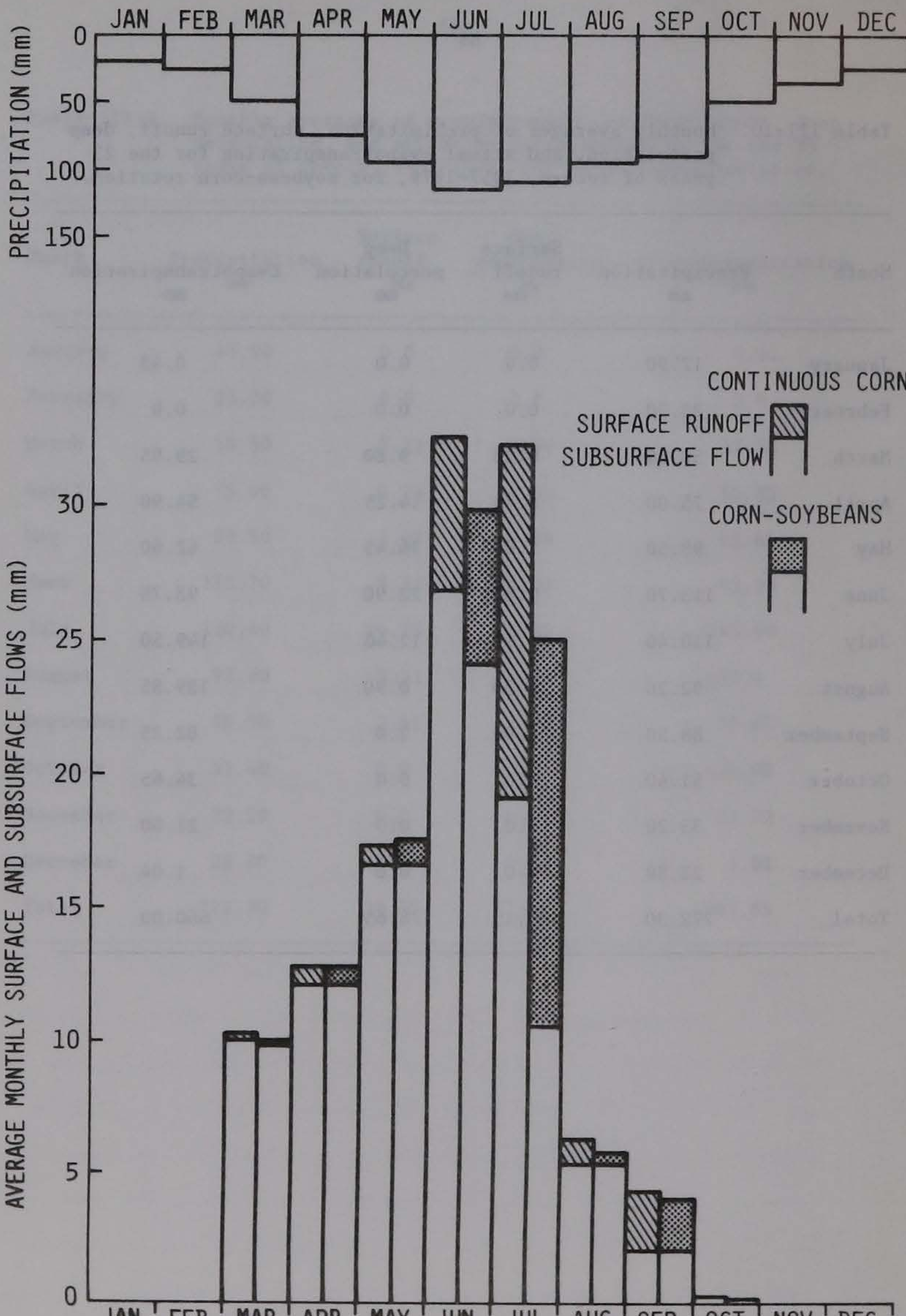

Figure III-4. Monthly variation in precipitation and predicted surface runoff and subsurface flow, Humboldt County 1957-1979.

Table III-11. Comparison of the plant nitrogen uptake, nitrate leached and denitrification for 75, 150, and 225 kg/ha fertilizer nitrogen application rates, using single application, Humboldt County, 1957-1979.

Year	Fertilizer N = 75 kg/ha[a]			Fertilizer N = 150 kg/ha[a]			Fertilizer N = 225 kg/ha[a]		
	N uptake	NO_3-N leached	Denitrification	N uptake	NO_3-N leached	Denitrification	N uptake	NO_3-N leached	Denitrification
		kg/ha			kg/ha			kg/ha	
1957	154.2	7.4	25.0	208.3	12.1	41.0	262.5	16.8	57.0
1958	126.1	7.7	9.5	179.7	17.2	21.2	215.8	26.8	32.8
1959	129.6	6.3	7.2	190.4	13.0	14.8	265.3	21.3	24.2
1960	120.2	6.0	19.4	170.6	11.8	38.1	221.1	17.6	56.8
1961	129.6	7.0	9.8	191.5	12.1	17.7	253.4	17.3	25.6
1962	113.9	16.9	21.1	150.6	33.0	43.0	187.3	49.1	65.4
1963	94.9	36.3	22.4	116.9	66.9	44.6	138.8	97.6	66.9
1964	130.3	6.5	11.1	179.1	17.0	26.5	227.4	27.7	42.2
1965	106.3	20.9	24.9	146.1	36.7	44.3	185.5	52.5	63.7
1966	116.9	13.8	17.9	169.7	23.2	30.8	222.4	32.5	43.8
1967	113.6	9.1	15.7	164.9	17.6	30.7	216.3	26.2	45.6
1968	137.7	0.0	0.0	212.3	0.0	0.0	272.0	0.0	0.0
1969	113.8	16.3	23.9	157.6	28.5	42.9	207.6	44.3	66.8
1970	137.7	0.0	0.0	212.0	0.0	0.0	213.4	0.0	0.0
1971	112.5	14.5	18.3	156.0	28.4	35.6	196.3	57.2	72.0
1972	112.9	15.2	26.4	151.6	29.7	48.3	201.5	55.5	90.4
1973	109.1	22.3	21.4	153.0	36.4	38.0	196.9	50.5	54.7
1974	109.4	19.5	21.6	155.2	33.3	37.1	200.9	47.0	52.6
1975	109.1	13.8	21.6	154.4	25.1	39.9	199.6	36.4	58.0
1976	134.5	1.7	6.1	199.7	3.0	10.8	199.7	4.4	15.6
1977	141.1	0.0	0.0	219.9	0.0	0.0	281.4	0.0	0.0
1978	128.2	6.1	11.3	181.9	13.0	25.5	269.0	27.3	55.3
1979	121.3	12.7	17.4	161.0	25.8	39.5	213.3	43.4	69.9

[a]Fertilizer was applied on the first week of April; 75, 150, and 225 kg/ha = 67, 134, and 201 lb/ac.

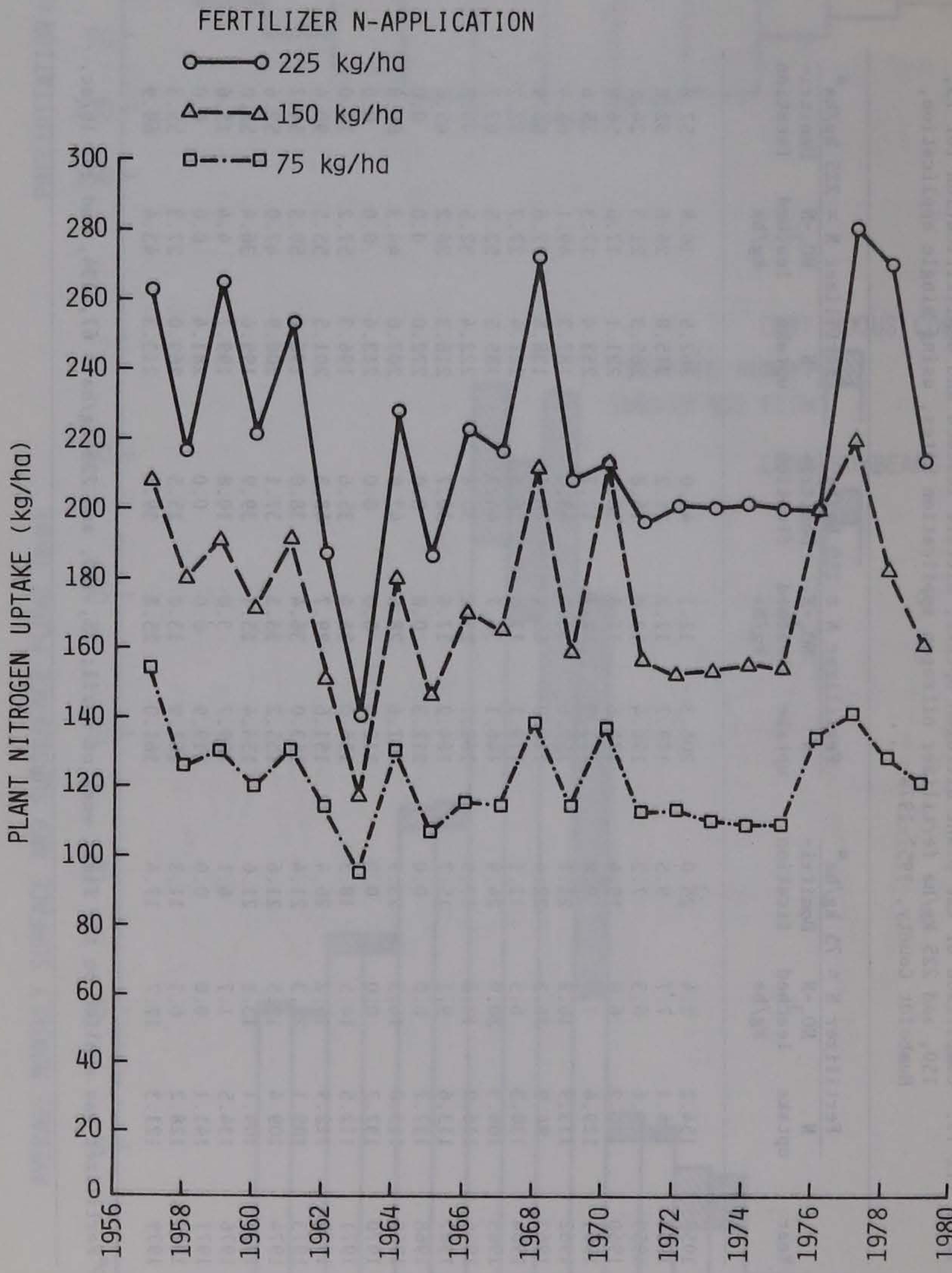

Figure III-5. Yearly variation in plant nitrogen uptake for three application rates, continuous corn.

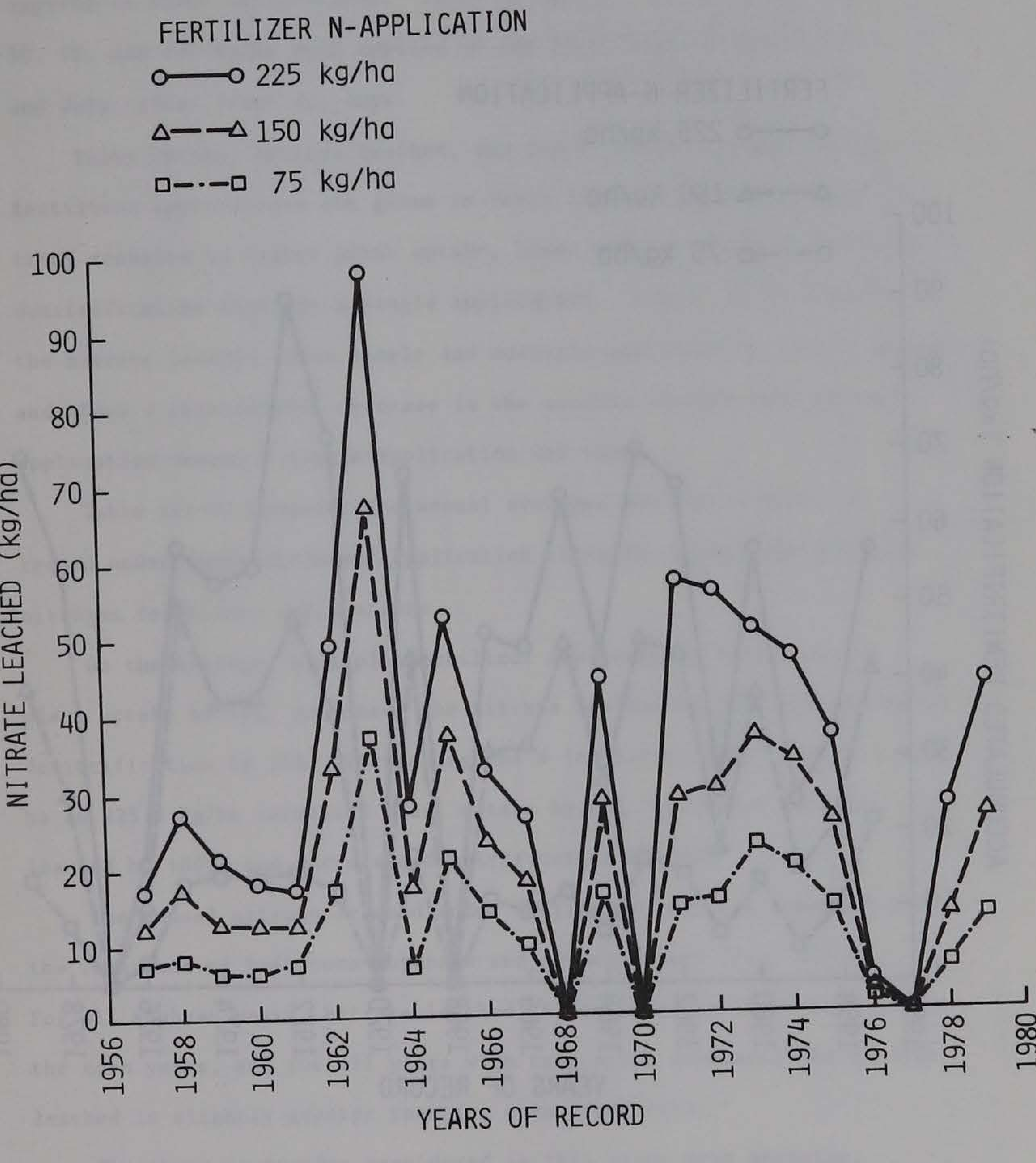

Figure III-6. Yearly variation in nitrate leached for three application rates, continuous corn.

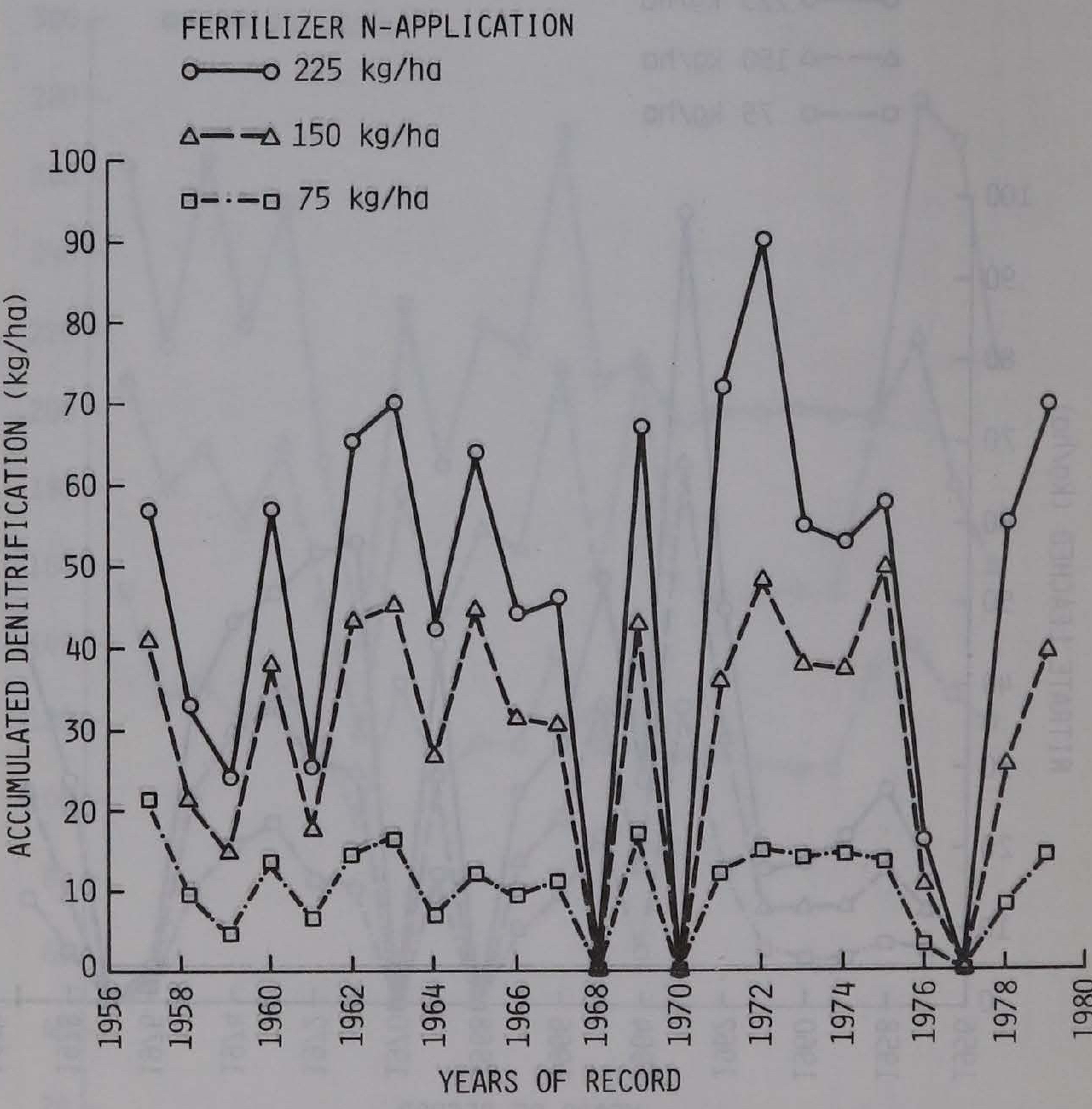

Figure III-7. Yearly variation in accumulated denitrification for three application rates, continuous corn.

applied in three applications. Three 25 kg/ha, three 50 kg/ha, and 50, 75, and 100 kg/ha were applied in the first week of April, June, and July, after three dry days.

Plant uptake, nitrate leached, and denitrification under multiple fertilizer applications are given in Table III-12. Multiple applications resulted in higher plant uptake, lower nitrate leached, and lower denitrification than for a single application. Figure III-8 compares the nitrate leached under single and multiple applications of 150.0 kg/ha, and shows a considerable decrease in the nitrate leached when multiple application versus a single application was used.

Table III-13 compares the annual averages for the 23 years of record under three different application rates for single and multiple nitrogen fertilizer applications.

On the average, multiple fertilizer applications increased the plant uptake by 10%, decreased the nitrate leached by 40%, and decreased denitrification by 35%. Increasing the N fertilizer rate from 75.0 kg/ha to 225.0 kg/ha increased plant uptake by 85%, increased nitrate leached by 180%, and increased denitrification by 135%.

The annual nitrate leached under continuous corn was compared with the rotations of both corn-soybeans and soybeans-corn (Fig. III-9). For all soybean years, nitrate leached is considerably lower than for the corn years, and for all years with corn after soybeans, the nitrate leached is slightly greater than for corn after corn.

The three pesticides considered in this study were atrazine, alachlor, and fonofos. The common pesticide parameters, including application rate, incorporation depth, water solubility, half-life, and

Table III-12. Comparison of the plant nitrogen uptake, nitrate leached, and denitrification for 75, 150, and 225 kg/ha fertilizer nitrogen application rates, using multiple application, Humboldt County, 1957-1979.

Year	Fertilizer N = 75 kg/ha[a]			Fertilizer N = 150 kg/ha[a]			Fertilizer N = 225 kg/ha[a]		
	N uptake	NO_3-N leached kg/ha	Denitri-fication	N uptake	NO_3-N leached kg/ha	Denitri-fication	N uptake	NO_3-N leached kg/ha	Denitri-fication
1957	159.7	5.85	20.96	219.3	9.07	32.98	10.8	10.8	41.0
1958	126.1	7.60	9.5	179.7	17.2	21.20	27.8	27.8	34.1
1959	134.4	4.10	4.6	200.1	8.5	9.70	12.8	12.8	14.6
1960	128.2	4.10	13.3	186.7	7.8	25.90	11.1	11.1	36.8
1961	135.3	4.50	6.5	203.0	7.2	11.1	8.2	8.2	13.1
1962	126.8	10.7	14.2	176.2	20.8	29.9	26.3	26.3	40.6
1963	117.2	20.2	16.1	161.4	34.8	32.1	40.8	40.8	43.4
1964	137.2	4.0	6.8	192.1	12.2	18.4	19.2	19.2	27.6
1965	129.7	10.3	12.0	193.0	15.6	18.4	16.0	16.0	18.9
1966	131.8	7.6	9.3	199.3	10.7	13.6	10.7	10.7	13.6
1967	121.4	6.3	10.8	180.7	11.9	20.7	16.9	16.9	29.4
1968	137.8	0.0	0.0	212.6	0.0	0.0	0.0	0.0	0.0
1969	125.1	11.8	17.1	180.2	19.4	29.3	32.9	32.9	48.1
1970	137.1	0.0	0.0	212.0	0.0	0.0	0.0	0.0	0.0
1971	112.5	10.0	11.9	177.5	19.4	23.0	39.0	39.0	47.3
1972	112.9	10.2	14.8	180.6	21.0	28.2	50.1	50.1	72.6
1973	109.1	13.4	13.7	186.5	18.7	22.7	19.3	19.3	26.2
1974	109.4	13.3	14.3	182.1	20.9	22.5	23.8	23.8	25.5
1975	109.1	8.3	13.5	181.4	14.2	23.7	16.2	16.2	27.8
1976	134.5	0.81	2.9	200.0	1.3	4.5	1.5	1.5	5.3
1977	141.1	0.0	0.0	228.2	0.0	0.0	0.0	0.0	0.0
1978	128.2	3.9	7.9	193.1	8.5	18.8	0.0	0.0	0.0
1979	121.3	9.5	14.1	173.9	19.5	32.9	20.8	20.8	46.8

[a]Three applications in the first week of April, June and July.

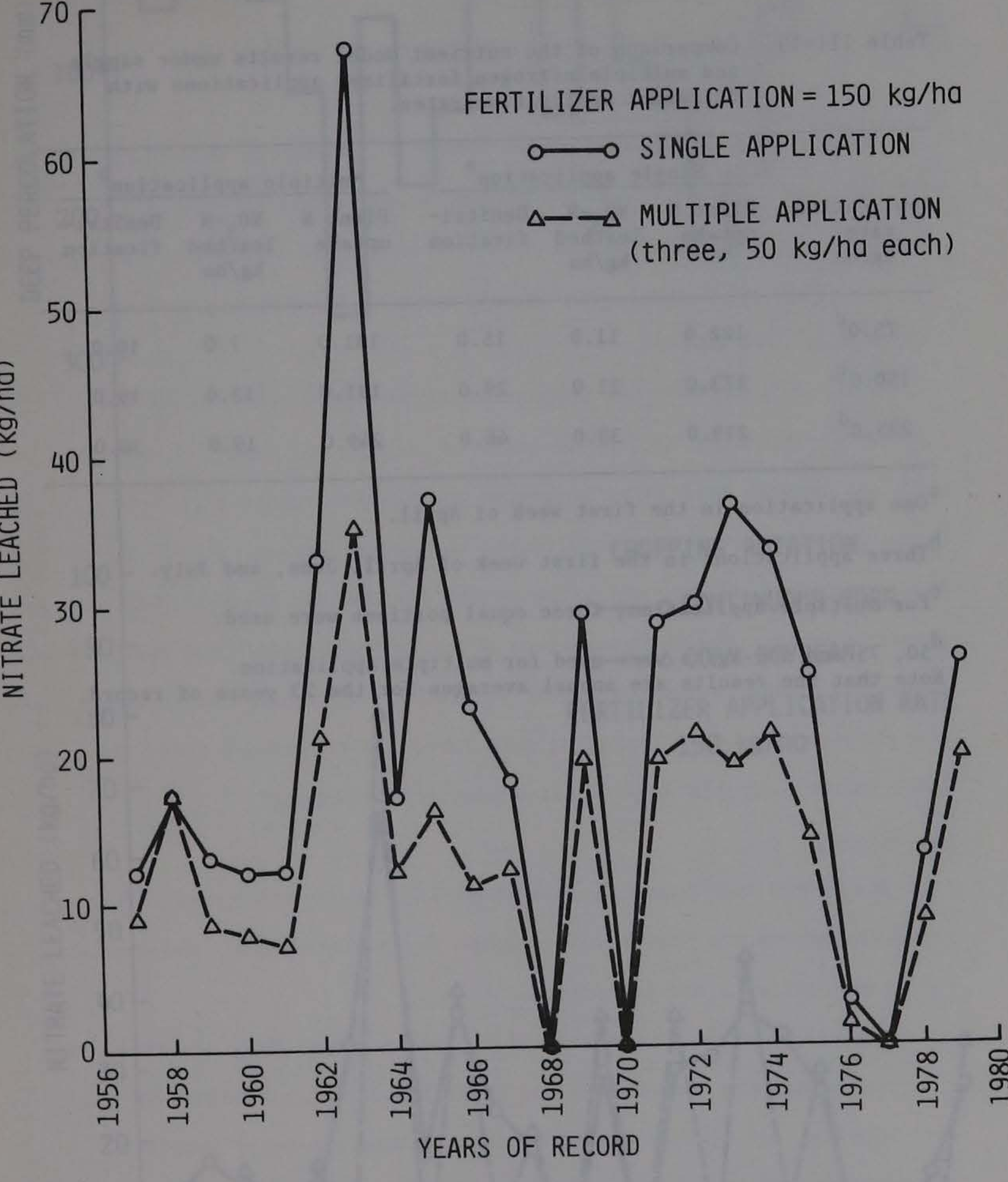

Figure III-8. Comparison of nitrate leached for single and multiple fertilizer applications, continuous corn.

Table III-13. Comparison of the nutrient model results under single and multiple nitrogen fertilizer applications with different application rates.

Application rate kg/ha	Single application[a]			Multiple application[b]		
	Plant N uptake	NO_3-N leached kg/ha	Denitri-fication	Plant N uptake	NO_3-N leached kg/ha	Denitri-fication
75.0[c]	122.0	11.0	15.0	131.0	7.0	10.0
150.0[c]	173.0	21.0	29.0	191.0	13.0	19.0
225.0[d]	219.0	33.0	46.0	249.0	19.0	30.0

[a]One application in the first week of April.

[b]Three applications in the first week of April, June, and July.

[c]For multiple application, three equal portions were used.

[d]50, 75 and 100 kg/ha were used for multiple application.
Note that the results are annual averages for the 23 years of record.

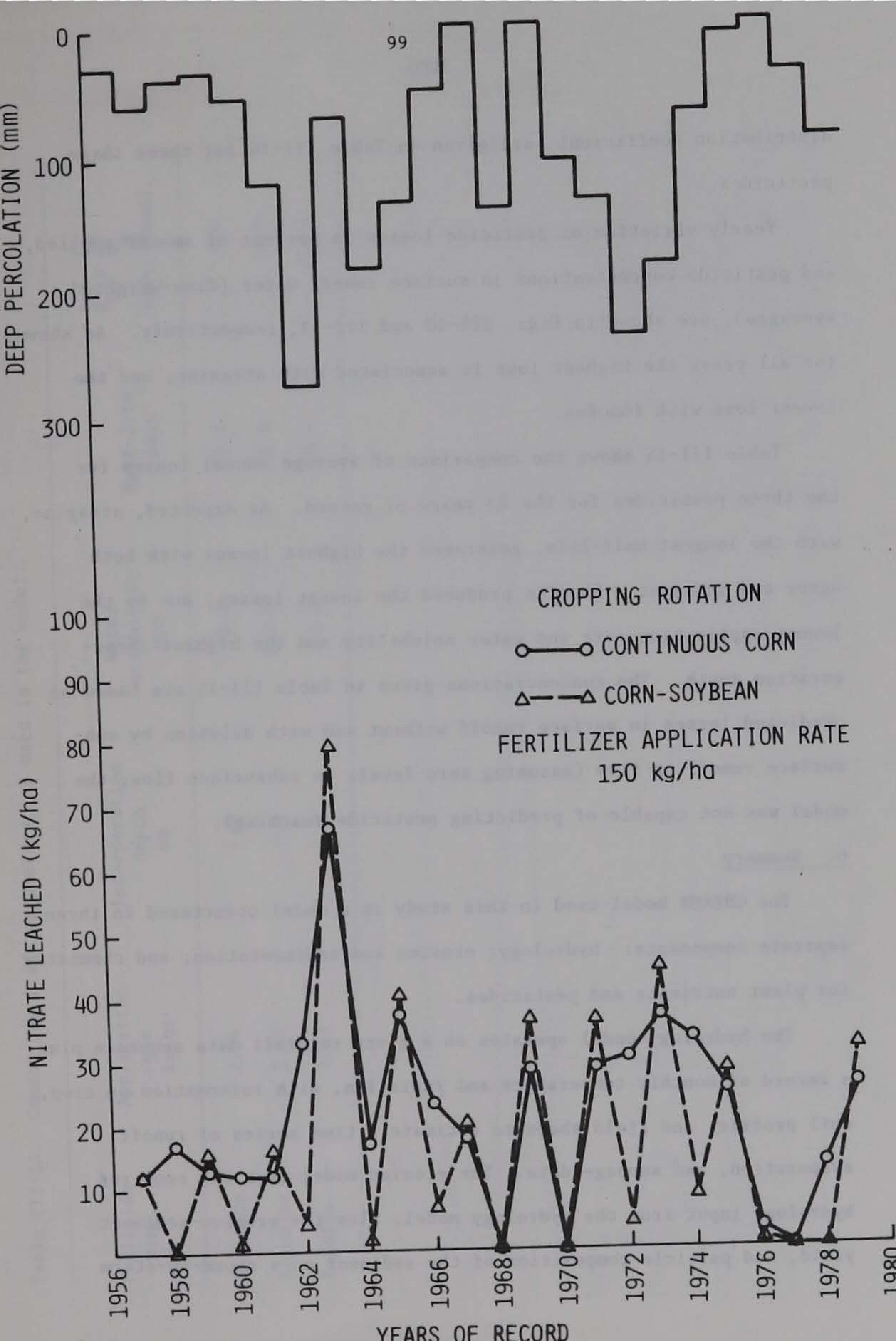

Figure III-9. Comparison of nitrate leached for continuous corn and corn-soybean rotations.

distribution coefficient, are given in Table III-14 for these three pesticides.

Yearly variation of pesticide losses in percent of amount applied, and pesticide concentrations in surface runoff water (flow-weighted averages), are shown in Figs. III-10 and III-11, respectively. As shown, for all years the highest loss is associated with atrazine, and the lowest loss with fonofos.

Table III-15 shows the comparison of average annual losses for the three pesticides for the 23 years of record. As expected, atrazine, with the longest half-life, generated the highest losses with both water and sediment. Fonofos produced the lowest losses, due to the lowest application rate and water solubility and the highest incorporation depth. The concentrations given in Table III-15 are based on predicted losses in surface runoff without and with dilution by subsurface runoff or flow (assuming zero levels in subsurface flow; the model was not capable of predicting pesticide leaching).

6. Summary

The CREAMS model used in this study is a model structured in three separate components: hydrology; erosion and sedimentation; and chemistry for plant nutrients and pesticides.

The hydrology model operates on a given rainfall data sequence plus a record of monthly temperature and radiation, with information on crop, soil profile, and field shape to estimate a time series of runoff, evaporation, and seepage data. The erosion model uses the required hydrology input from the hydrology model, plus the erosion-sediment yield, and particle composition of the sediment on a storm-by-storm

Table III-14. Common pesticide parameters, as used in the model.

Pesticide name	Application rate kg/ha	Incorporation depth cm	Water solubility ppm	Half-life days	$K_D = \frac{C_s}{C_w}$ Dist. Coeff.
Atrazine	2.24	1.0	30.0	45.0	4.0
Alachlor	2.24	1.0	242.0	14.0	6.0
Fonofos	1.12	5.0	13.0	20.0	50.0

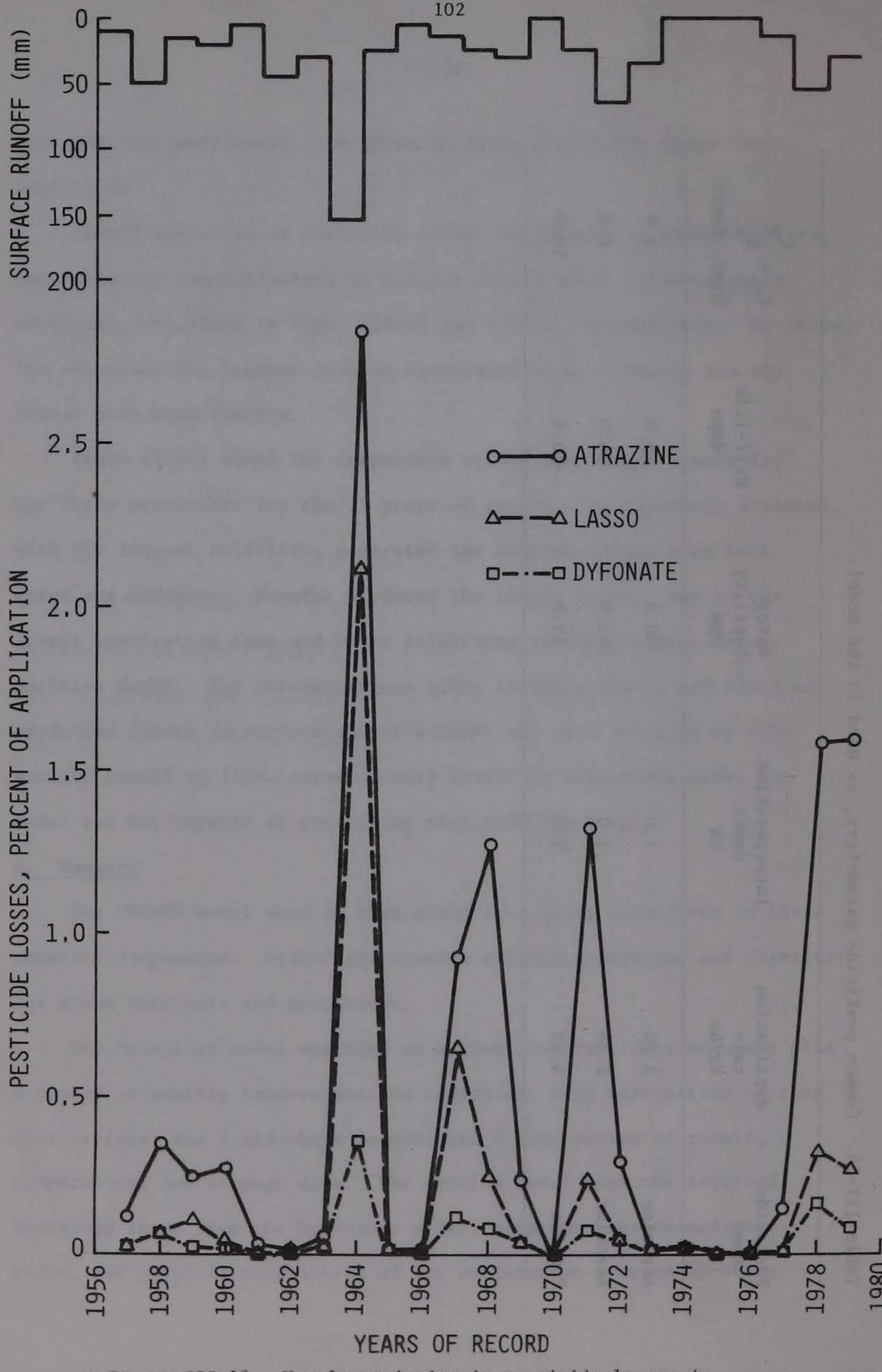

Figure III-10. Yearly variation in pesticide losses in water.

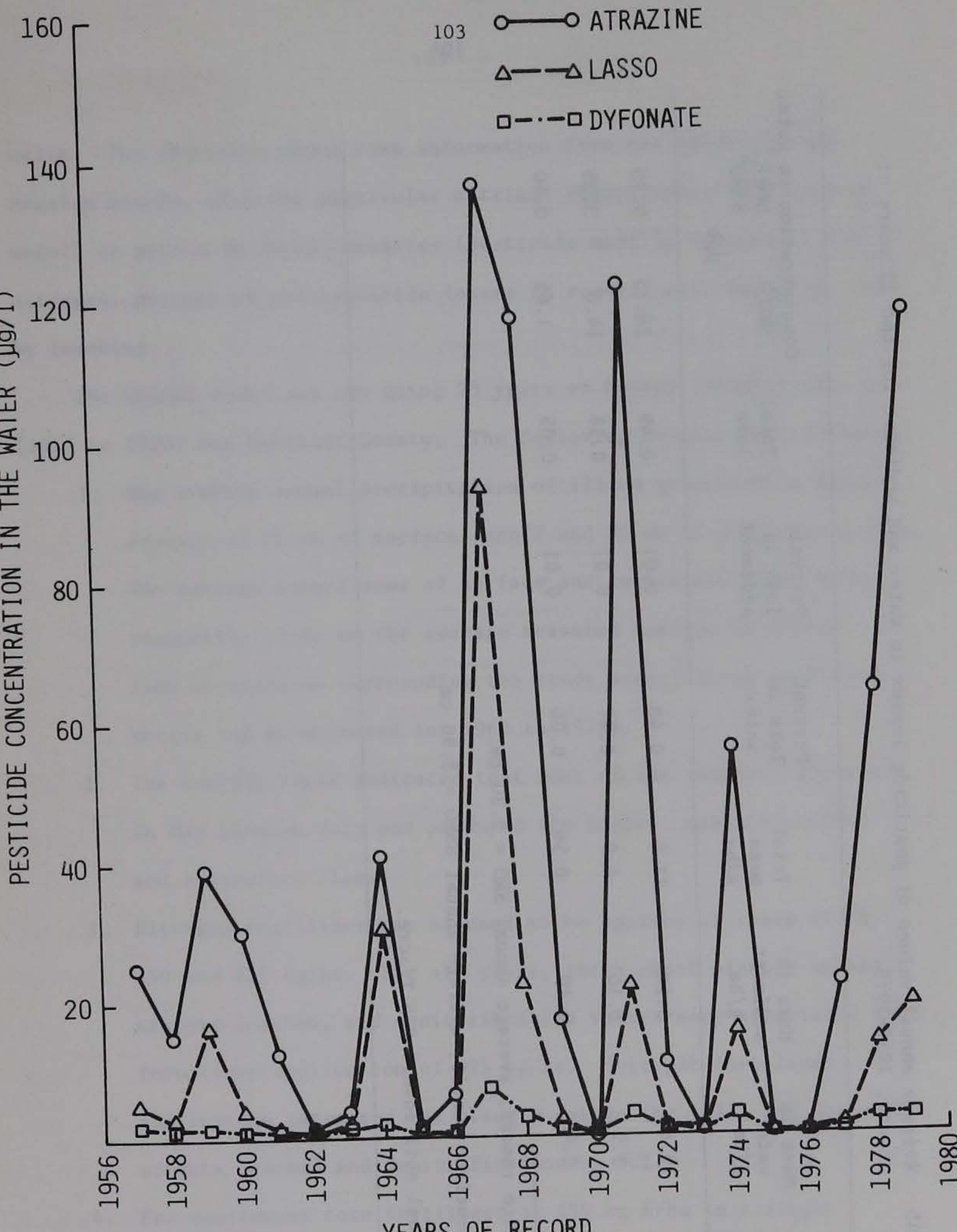

Figure III-11. Yearly variation in average pesticide concentrations in water.

Table III-15. Average annual values of pesticide losses in water and sediment for the 23 years of record, 1957-1979.

Pesticide name	Mass in water g/ha	Mass in sediment g/ha	Total mass g/ha	Percent loss in water	Percent loss in sediment	Total loss	Concentration in water SRO[a] ppb	SOR+ SSRO[b] ppb
Atrazine	10.8	0.20	11.0	0.48	0.01	0.49	38.75	9.25
Lasso	3.9	0.10	4.0	0.17	0.01	0.18	14.0	3.35
Dyfonate	0.46	0.10	0.56	0.04	0.01	0.05	1.65	0.40

[a]SRO = surface runoff; average annual SRO = 2.80 cm.

[b]SSRO = subsurface runoff; average annual SSRO = 8.90 cm.

Average annual soil loss = 0.5 T/acre.

basis. The chemistry model uses information from the hydrology and erosion models, plus the particular nutrient characteristics (Nutrient model) or pesticide characteristics (pesticide model), to predict the nitrogen, phosphorus and pesticide losses in runoff, with sediment, and by leaching.

The CREAMS model was run using 23 years of hourly rainfall data (1957 to 1979) for Humboldt County. The following results were obtained.

1. The average annual precipitation of 775 mm generated an annual average of 28 mm of surface runoff and 89 mm of deep percolation. The average annual sums of surface and subsurface flows were reasonably close to the average measured records of stream flow at stations surrounding the study area (135 mm predicted versus 148 mm measured for 1960 to 1979).
2. The monthly trend indicated that most of the rainfall occurred in May through July and produced the highest monthly surface and subsurface flows.
3. Nitrogen fertilizer was assumed to be applied at rates of 75, 150 and 225 kg/ha. For all years, the highest plant N uptake, nitrate leached, and denitrification values were associated with fertilizer application of 225 kg/ha. Multiple fertilizer application increased the plant N uptake (by 10%), and decreased nitrate leached and denitrification by 40%.
4. For continuous corn fertilized at 150 kg N/ha in a single application, the predicted average value for NO_3-N leached was 21.0 kg/ha; for comparison, the average annual NO_3-N loss estimated from predicted total flow (117 mm) and monitored ADW drainage was 18.7 kg/ha.

5. The nitrate leached under soybeans was considerably less than that leached under corn; for corn after soybeans, the nitrate leached was slightly more than for corn after corn.
6. Among the three pesticides used in this study, atrazine, with the highest half-life, generated the highest losses with both water and sediment. Fonofos, with the lowest application rate and highest incorporation depth, resulted in the lowest losses.

B. ISU Drainage Model

1. Hydrology and Nutrient Transport Model

A computer simulation model was developed to simulate the hydrologic and nitrogen flow processes occurring in a tile-drained agricultural field (Kanwar, 1981). This model allows prediction of the nitrate load from tile drainage as a function of various farm management practices and weather conditions. The major inputs to the model include daily precipitation and daily open-pan evaporation, planting and harvesting dates, days and amounts of fertilization, soil-water relationships, and plant growth relationships. The various outputs from the model are surface runoff, tile drainage, nitrate load in the tile effluent, and nitrate uptake by plants.

This model was calibrated and verified using eight years (1970 to 1978) of data on tile flow and the nitrate concentrations of the tile water from field experiments conducted at the Iowa State University's Agronomy and Agricultural Engineering Research Center near Boone, Iowa. The model predictions agreed reasonably well with the measured values of tile drainage water and nitrate losses in the tile effluent (Kanwar et al., 1983).

Daily rainfall data for Humboldt County were taken from rain gauge station Humboldt 2 and were used for the model simulations from 1957 to 1979. The open-pan evaporation data were not available at station Humboldt 2. Therefore, these data for 1957 to 1979 were taken from the Kanawha station, which was the nearest station to Humboldt 2. An assumption was made in the model simulations that the open-pan evaporation data obtained from the Kanawha station were the same as would have been taken at the Humboldt 2 station.

The planting and harvesting days for the corn were taken as May 15 and October 15, respectively. Data on moisture stress factors, crop development ratios for corn, and distribution of root system as a function of time are given by Kanwar et al. (1983). Corn growth rate function used in the model is similar to the one used by Duffy et al. (1975).

The data on initial soil moisture content, field capacity, wilting point, diffusivity, unsaturated and saturated hydraulic conductivities, and initial water table depth are needed as inputs for the model. The data on initial soil moisture content for April 15 were available for Kanawha station from 1957 to 1979. These data were converted to Humboldt 2 station using the available conversion factors. The data on field capacity and wilting points were taken from Kanawha station and have been used in the model. The initial water table depth was assumed to be at 150 cm on the day simulation started for all the years.

The saturated hydraulic condictivity was taken as 15 cm/day (from the calibration process as developed by Kanwar, 1981). The data on unsaturated hydraulic conductivity and diffusivity were taken from the literature for the Clarion-Webster soils.

The data on depth of tiles, drain spacings, depth of impermeable layer, and drain diameter are needed for simulation. These data as used in the model are given in Table III-16.

A fertilizer application rate of 150 kg/ha was used in the model. All the fertilizer was considered to be applied on May 1 of each year in a single application.

The model does predict that some water will not be intercepted by the tile lines and will move deep into the soil profile. However, it is assumed that no NO_3-N moves with deep percolation, but instead that any NO_3-N in that water is denitrified (although the amount of total denitrification in the surface soil plus in deep percolation is obtained by calibration).

Kanwar (1981) used a trial and error procedure to calibrate the selected parameters used in this model by using field data from the Agronomy and Agricultural Engineering Research Center near Ames in Boone County, Iowa. Some values of these parameters were used for the Humboldt County data which are given in Table III-16.

The initial nitrate concentration levels in the soil profile are also needed as inputs, and such data were not available for Humboldt County area. Therefore, constant nitrate concentrations were taken for April 1 of each year as 2, 2, 7, 5, 10, 8, 8, 10, 10, and 5 ppm for layers one through ten of the soil profile.

Table III-16. Parameter definitions and calibrated values used in the model.

Parameter	Definition	Value
D	Diffusion coefficient of nitrate in water	1.0 cm^2/day
DISP	Dispersion coefficient of nitrate in water	4.0 cm
WF	Weighting factor to account for cracks	0.8 for top 5 layers 0.9 for other layers
F	A factor for approximating the amount of transpiration from various layers	0.5 for layer 1 and 2 1.0 for other layers
TORT	Labyrinth factor	0.8
T	Thickness of nearly impermeable layer	170 cm
A	Vertical hydraulic conductivity of the nearly impermeable layer	0.4 cm/day
K_1	Lateral hydraulic conductivity	15 cm/day
S	Spacing between tiles	3658 cm
d	Distance between the impermeable layer and the tiles	290 cm

2. Results and Discussion

Model simulations were made for the study period 1957 to 1979. Predictions of tile flows, surface runoff, evapotranspiration, nitrate concentrations in the tile drainage water, and nitrate leached through tile water were made and are given in Tables III-17 and III-18.

Figure III-12 shows the tile flows and surface flows for the study period of 1957 to 1979. The model predicted an average of 66.4 mm of tile flow from April through November. Similar predictions for the surface flow gave 29.6 mm. From the survey of agricultural drainage wells in the Humboldt County area, it was found that most of the wells receive surface as well as tile flows. Therefore, according to the predictions made by this model, 96 mm on the average is going into the agricultural drainage wells. Since this model simulates the soil-water-plant system only between April and November, a good possibility exists that slightly more than 96 mm of surface and subsurface water may be entering into the wells on an annual basis. As discussed earlier for the CREAMS model, in the absence of actual flow data to the ADWs, the predicted values can be used with measured concentration data to estimate loadings.

Table III-18 gives the loss of nitrate with the drainage water and through denitrification. On the average, about 12.45 kg/ha of NO_3-N was lost through subsurface water during the eight-month period modeled each year. About 9.83 kg/ha of NO_3-N was being lost because of the denitrification process. The average NO_3-N uptake by the plants was about 227 kg/ha on an annual basis.

Table III-17. Tile flows for various months from 1957 to 1979.

Year	Tile flows (cm) for the months of								Total
	April	May	June	July	Aug.	Sept.	Oct.	Nov.	cm
1957	0.08	0.00	0.02	0.18	0.00	0.00	0.00	0.04	0.32
1958	0.19	0.03	0.05	4.72	0.93	0.00	0.00	0.00	5.92
1959	0.05	0.10	0.26	0.00	0.37	0.10	0.04	0.11	1.03
1960	0.07	0.51	0.76	0.10	0.01	0.13	0.00	0.11	1.59
1961	0.69	0.67	0.82	0.04	0.04	0.06	0.11	0.05	2.48
1962	0.42	0.74	1.66	1.95	2.01	6.85	1.07	0.13	12.80
1963	0.34	0.44	6.48	1.59	0.22	0.00	0.00	0.00	9.07
1964	1.55	3.82	1.04	0.17	5.69	1.21	0.37	0.00	13.87
1965	0.41	2.71	3.79	0.28	0.00	0.42	3.02	0.65	15.06
1966	0.07	0.11	1.49	0.28	0.00	0.00	0.00	0.00	1.95
1967	0.05	0.00	3.04	1.03	0.00	0.00	0.00	0.00	4.12
1968	0.06	0.00	0.04	0.15	0.00	0.00	1.31	0.83	2.39
1969	1.68	0.99	1.38	2.21	0.10	0.00	0.00	0.00	6.36
1970	0.09	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.19
1971	0.05	0.02	0.25	0.80	0.00	0.00	0.00	0.13	1.25
1972	0.59	3.53	1.41	1.34	3.66	0.59	1.99	3.20	16.31
1973	4.23	4.67	1.89	0.00	0.00	0.84	2.58	1.22	15.43
1974	4.11	2.42	2.43	0.53	0.00	0.00	0.00	0.00	9.49
1975	2.39	4.30	2.70	0.79	0.00	0.00	0.00	0.00	10.18
1976	0.44	1.75	0.90	0.00	0.00	0.00	0.00	0.00	3.09
1977	0.92	0.77	0.04	0.00	0.18	0.46	0.60	1.55	4.42
1978	0.43	1.03	1.98	3.91	0.35	0.00	0.00	0.00	7.70
1979	1.18	2.11	0.49	0.00	0.98	0.51	0.39	2.07	7.73
Avg.	0.87	1.40	1.44	0.87	0.63	0.49	0.50	0.44	6.64

Table III-18. Amounts of nitrate leached through tile water, nitrate uptake by plants, and nitrate lost through denitrification.

Year	NO_3-N lost through tile water kg/ha	NO_3-N lost as denitrification kg/ha	NO_3-N uptake kg/ha
1957	0.48	11.31	222.78
1958	15.85	11.62	233.51
1959	2.76	7.42	243.30
1960	0.90	7.56	241.22
1961	1.24	6.70	250.78
1962	28.22	15.74	205.40
1963	47.62	11.61	182.53
1964	44.73	11.67	208.20
1965	28.44	18.01	191.67
1966	0.97	3.50	266.94
1967	9.47	7.89	222.00
1968	4.65	11.06	244.16
1969	6.44	9.09	210.88
1970	0.12	7.20	246.64
1971	3.09	0.62	261.26
1972	29.80	14.72	194.09
1973	16.04	14.95	216.91
1974	6.02	7.67	236.23
1975	11.08	13.50	224.15
1976	1.90	3.90	234.10
1977	5.20	10.00	238.41
1978	14.98	8.50	217.40
1979	6.53	11.82	239.00
Avg.	12.45	9.83	227.46

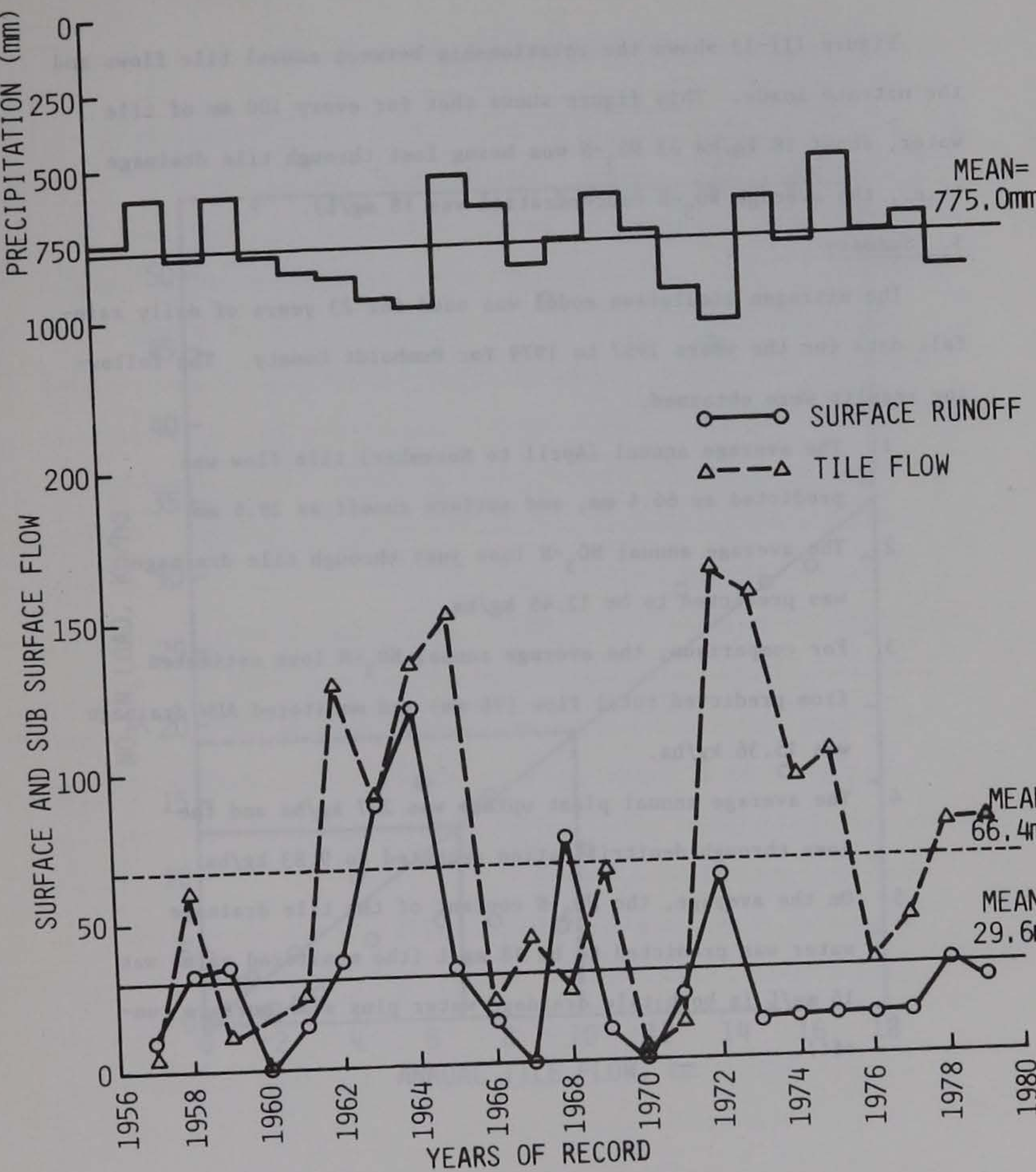

Figure III-12. Yearly variation in precipitation and predicted surface runoff and subsurface flow, Humboldt County 1957-1979 (ISU model).

Figure III-13 shows the relationship between annual tile flows and the nitrate loads. This figure shows that for every 100 mm of tile water, about 18 kg/ha of NO_3-N was being lost through tile drainage (i.e., the average NO_3-N concentration was 18 mg/L).

3. Summary

The nitrogen simulation model was used for 23 years of daily rainfall data for the years 1957 to 1979 for Humboldt County. The following results were obtained.

1. The average annual (April to November) tile flow was predicted as 66.4 mm, and surface runoff as 29.6 mm.
2. The average annual NO_3-N loss just through tile drainage was predicted to be 12.45 kg/ha.
3. For comparison, the average annual NO_3-N loss estimated from predicted total flow (96 mm) and monitored ADW drainage was 15.36 kg/ha.
4. The average annual plant uptake was 227 kg/ha and the loss through denitrification amounted to 9.83 kg/ha.
5. On the average, the NO_3-N content of the tile drainage water was predicted to be 18 mg/L (the monitored value was 16 mg/L in both tile drainage water plus some surface runoff).

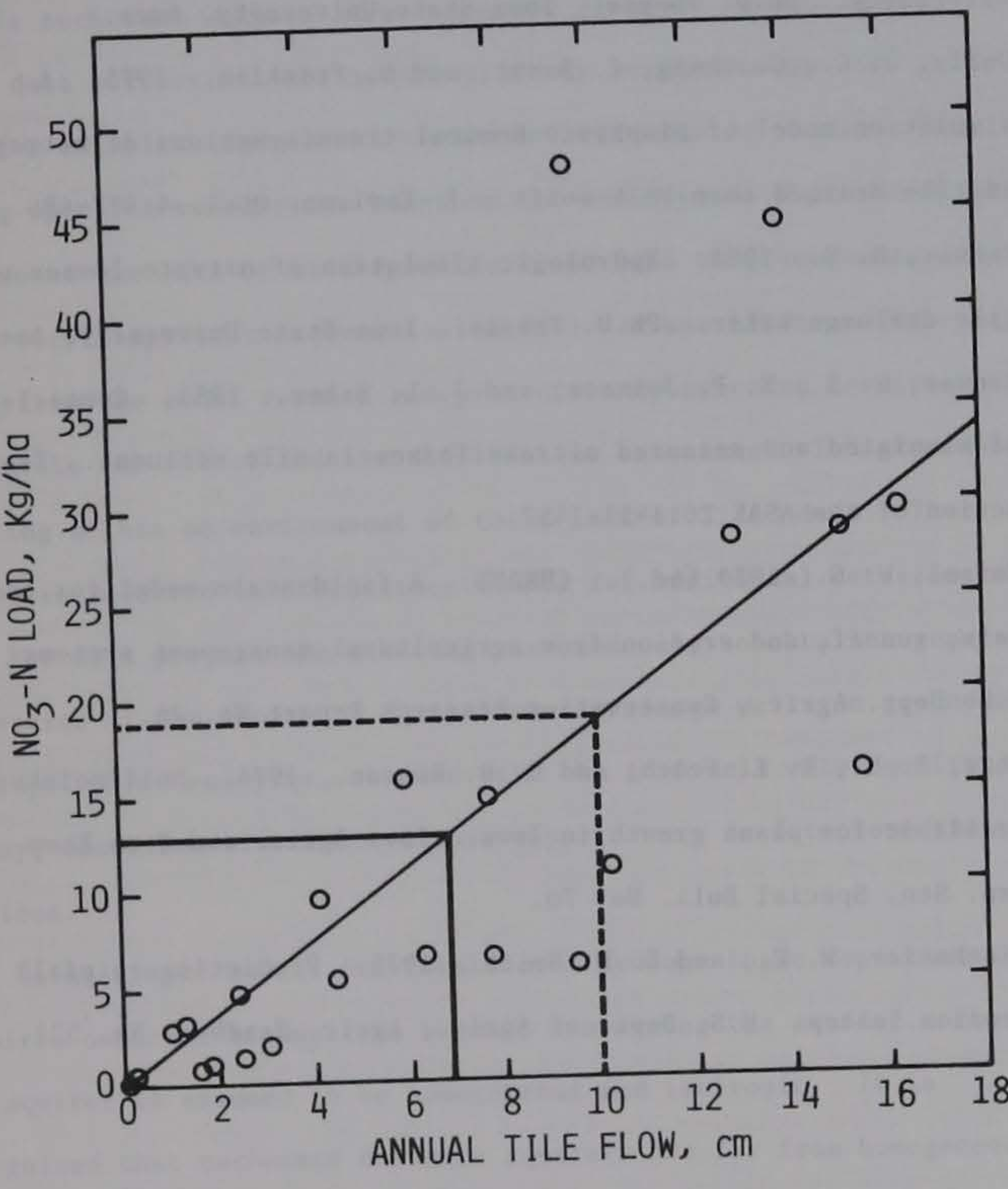

Figure III-13. Relationship between annual tile flows and nitrate loads.

C. References

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IV. MODELING GROUNDWATER IMPACTS

The state of the art of modeling pollutant movement through aquifers is such that without very extensive groundwater monitoring and field data collection, the accuracy of the results makes them useful only for establishing general trends. The reader is cautioned against making specific conclusions based on the model results presented here. Because of the lack of site-specific field data on groundwater dispersion, very simplistic mathematical models are use herein. This chapter summarizes the model results and presents only selected results to illustrate the use of and the difficulties associated with groundwater modeling within an environment of total lack of field data. The reader is referred to Overholtzer (1983) for additional detailed results.

For this project the modeling was used only to obtain a general impression of the impact of agricultural drainage wells on groundwater quality. These analyses, coupled with the farm well water quality survey, described elsewhere in this report, were used to form the conclusions.

Using simple models for complicated hydrogeologic conditions requires numerous assumptions be made concerning the aquifer. First, the aquifer is assumed to be homogeneous and isotropic. It is recognized that carbonate dolomite aquifers are far from homogeneous when viewed at close range. Flow in carbonate aquifers is concentrated along secondary porosity planes. Large fluctuations in hydraulic properties can occur in short distances in limestone formations. When viewed from a county or regional scale, limestone aquifers appear to be more homogeneous. The authors realize the possible misinterpretations that can result in modeling a limestone formation as a homogeneous and isotropic media; however, because very little field data from the study area were available and the project budget was not sufficient for field data collection, it was felt that analysis of the Mississippian aquifer in Humboldt County, Iowa using simple models could help establish the range of possible impacts due to ADWs.

A second major assumption deals with dispersion in a groundwater system. When a pollutant is injected into an aquifer, a plume will develop in the groundwater downstream of the source, spreading out to the sides and below. If the aquifer is relatively thin, the vertical extent of the plume is limited by the bottom impermeable boundary. The pollutant quickly mixes over the vertical direction, and its concentration becomes essentially uniform with depth. When that occurs, the plume can be regarded as essentially two-dimensional. As the plume moves away from the pollution source, the concentration of contaminants decreases because of dispersion and other attenuation effects. Longitudinal dispersion occurs in the direction of flow and is caused by the velocity distribution within the pores of the media, the variations in pore diameters and the tortuous path followed by the pollutant particle. For example, water entering at point B in Fig. IV-1 appears to pass directly through the medion and, will advance faster than that entering at point A. Transverse dispersion occurs normal to the direction of flow and results from the repeated splitting and deflecting of the fluid by the solid particles in the aquifer. This transverse dispersion is effective only at the edges of a pollution plume (see point C in Fig. IV-1).

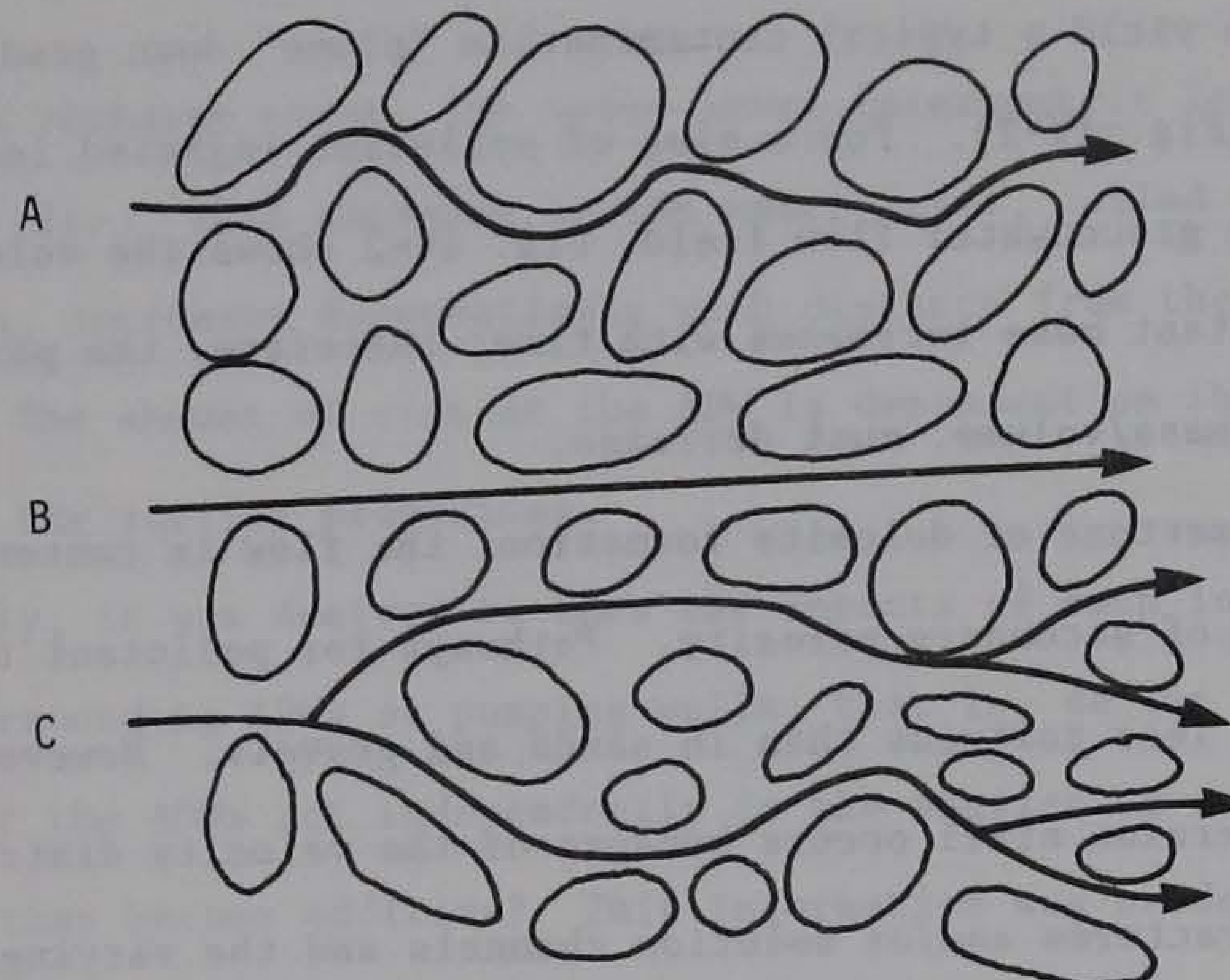

Figure IV-1. Schematic of pathlines causing longitudinal dispersion (A and B) and transverse dispersion (C) of a pollutant passing through a saturated porous medium (after Bouwer, 1978).

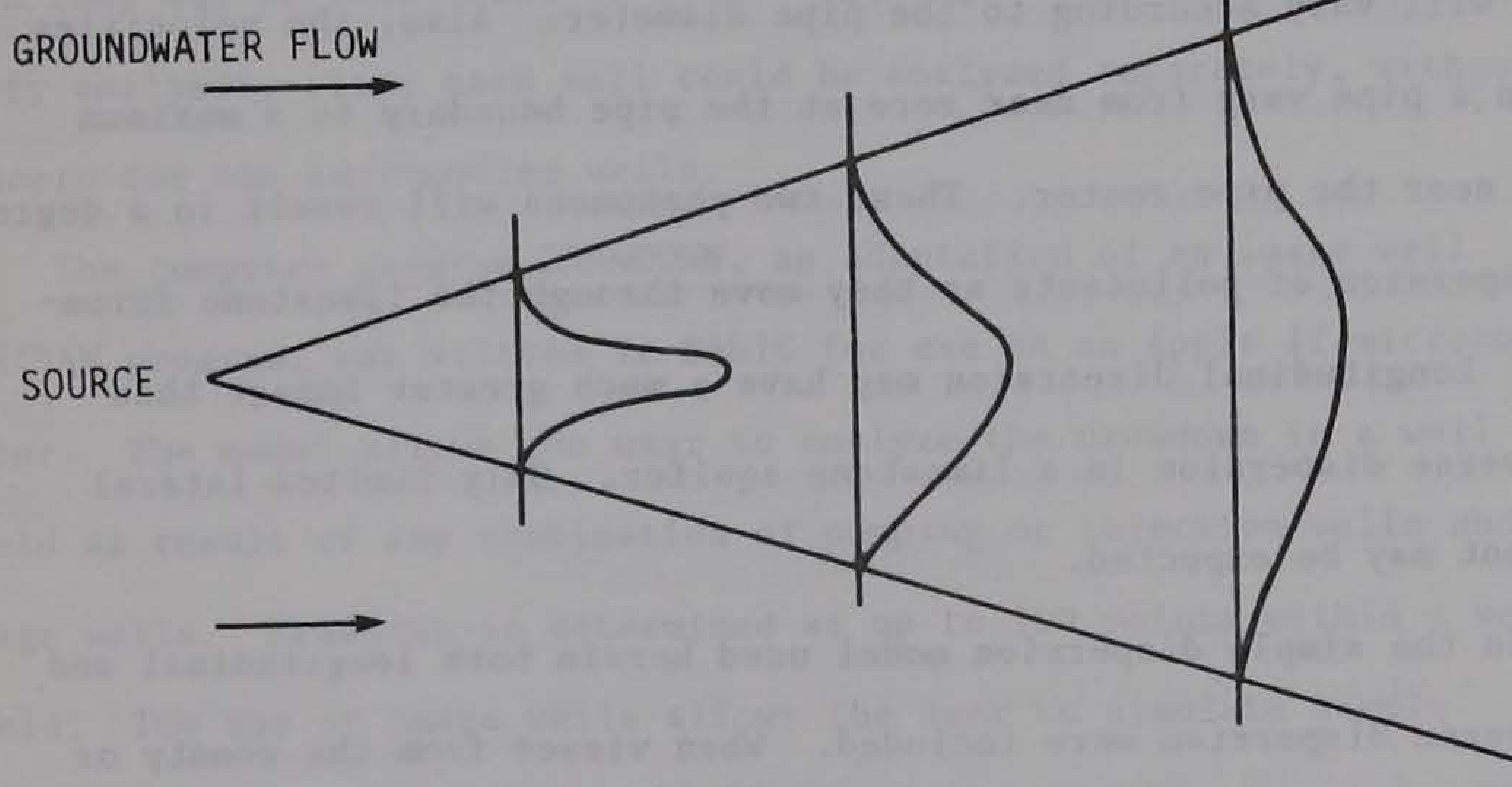

Figure IV-2. Schematic of a contamination plume downgradient from a point pollution source with Gaussian concentration distributions across plume (after Bouwer, 1978).

For a point source of pollution, both longitudinal and transverse dispersion work to yield a typical contamination "plume" down gradient from the source (Fig. IV-2). For a slug of pollutant injected instantaneously in a uniform groundwater flow field, Fig. IV-2 shows the volume occupied by the pollutant mass increases with time; therefore, the pollutant concentration, mass/volume, must decrease.

In a limestone or dolomite formation, the flow is concentrated along planes of secondary porosity. Pathways for pollutant transport appear to be less tortuous than in sands and gravels. However, longitudinal dispersion still occurs because of the velocity distribution within the fractures and/or solution channels and the varying diameters of the channels. Flow in a limestone aquifer can be thought of as similar to flow through a large number of variable diameter pipes oriented along planes of secondary porosity. Velocities between these pipes will vary according to the pipe diameter. Also, the velocities within a pipe vary from near zero at the pipe boundary to a maximum value near the pipe center. These two phenomena will result in a degree of dispersion of pollutants as they move through the limestone formation. Longitudinal dispersion may have a much greater impact than transverse dispersion in a limestone aquifer. Only limited lateral movement may be expected.

In the simple dispersion model used herein both longitudinal and transverse dispersion were included. When viewed from the county or regional scale and in view of the lack of site-specific data for the study area, the assumptions of this approach were considered justifiable.

A. The Well Drawdown Model

During a recharge event, the water level (piezometric level) in the ADW will rise. This increase in the water level, called the cone of impression, decreases exponentially with distance from the recharge well (ADW). The amount of rise at the ADW is dependent on the rate of recharge and the aquifer properties.

Initially, it was desired to know the impacts of each injection well upon surrounding ADWs or pumping wells; that is, do the cones of impression of the ADWs act independently in the aquifer or do they overlap and thus become additive? This information was needed for the later analysis of pollutant plumes by the dispersion model. If the cones of impression, caused by the wells injecting at a reasonable rate, did not significantly interfere with other injection wells in the area, then they may be considered to be acting singularly. This would simplify analysis, since each well could be analyzed separately, without concern for the surrounding wells.

The computer program DRAWDOWN, an adaptation of an image well FORTRAN program, was written in BASIC for use on an Apple II microcomputer. The model allows the user to analyze the drawdown in a well field as result of any combination of pumping or injection wells and image wells. Drawdown is determined at up to 100 points within a well field. The use of image wells allows the user to simulate simple aquifer boundary conditions and adds flexibility to the program. More than one well can be recharging or pumping simultaneously, and the total drawdowns (or rises, in the case of injecting wells) are calculated using the principle of superposition.

The model DRAWDOWN is based on the non-equilibrium well equation; thus, the assumptions for that equation also hold for the model. Of these, the assumption of a homogeneous, isotropic aquifer is the most limiting in this study. DRAWDOWN also assumes that an aquifer is infinite in areal extent and does not allow for varying aquifer thickness. Complex boundary conditions cannot be modeled accurately with this model, nor can those aquifers with a significantly sloping piezometric surface.

The DRAWDOWN computer model is generally accurate enough to provide a good first estimate of the cone of impression that occurs around an injection well system. The model is best suited to problems involving few recharging wells and uncomplicated aquifers with simple boundary conditions. For the purposes of this study, however, the DRAWDOWN model proved satisfactory.

Once the impact of an injection well on the water table was known, the next step was to model the effects of pollutant movement within that aquifer. This was accomplished by using a computer model, PLUME, that utilizes the dispersion equation developed by Wilson and Miller (1978).

B. The Dispersion Model

The purpose of pollutant modeling was to determine the extent of the groundwater plume from agricultural drainage wells at various times and recharge rates. Three major questions were addressed:

1. How much will injectant concentrations be reduced by mixing with the groundwater?

2. In what direction will the resulting plume travel, and how will its shape change along the travel path?
3. How far downstream will the plume travel before dispersion reduces the pollution concentration to below acceptable drinking water limits?

The PLUME model was adapted from a groundwater plume concentration program developed by Pettyjohn (1982). PLUME is a two-dimensional model of the pollutant movement within a porous media. It calculates and displays pollutant concentrations at a single point specified by the user or as a fixed spacing grid map.

As in the non-equilibrium well equation and the DRAWDOWN model, the assumptions of the two-dimensional dispersion equation also apply to the PLUME computer model. The aquifer is assumed to be homogeneous, isotropic, and of infinite areal extent. The pollutant is assumed to be completely mixed throughout the depth of the aquifer. The inability to simulate non-homogeneous aquifers and aquifers with varying thickness and complicated boundary conditions limits PLUME's use in advanced studies. Thus, the results given here can only be used to evaluate the range of influences of ADWs recharging limestone formations.

C. Model Parameters

For any model to give proper results, reasonable values must be used for the input parameters. Obtaining good estimates for these parameters can be difficult since on-site methods for determining various aquifer properties are often complicated, costly, and time-consuming. For the Humboldt County study area, it was determined early in this study that an intensive aquifer testing effort was not possible within funding levels, so other methods were used to obtain reasonable values for those aquifer parameters required for the modeling studies.

Since little actual data from the study wells were available, textbooks, manuals, and technical literature provided the bulk of the bracketing values for the aquifer properties. This gave 'typical' values which may or may not reflect the actual conditions in our test site. Examples of these are given in Tables IV-1 and IV-2. In the Humboldt region, the ADWs primarily penetrate carbonate (limestone and dolomite) formations with sufficient secondary porosity (cracks, joints and solution channels) to accept the surface and tile drainage without clogging (Musterman et al., 1981).

Table IV-1. Range of values of hydraulic conductivity (from Freeze and Cherry, 1979).

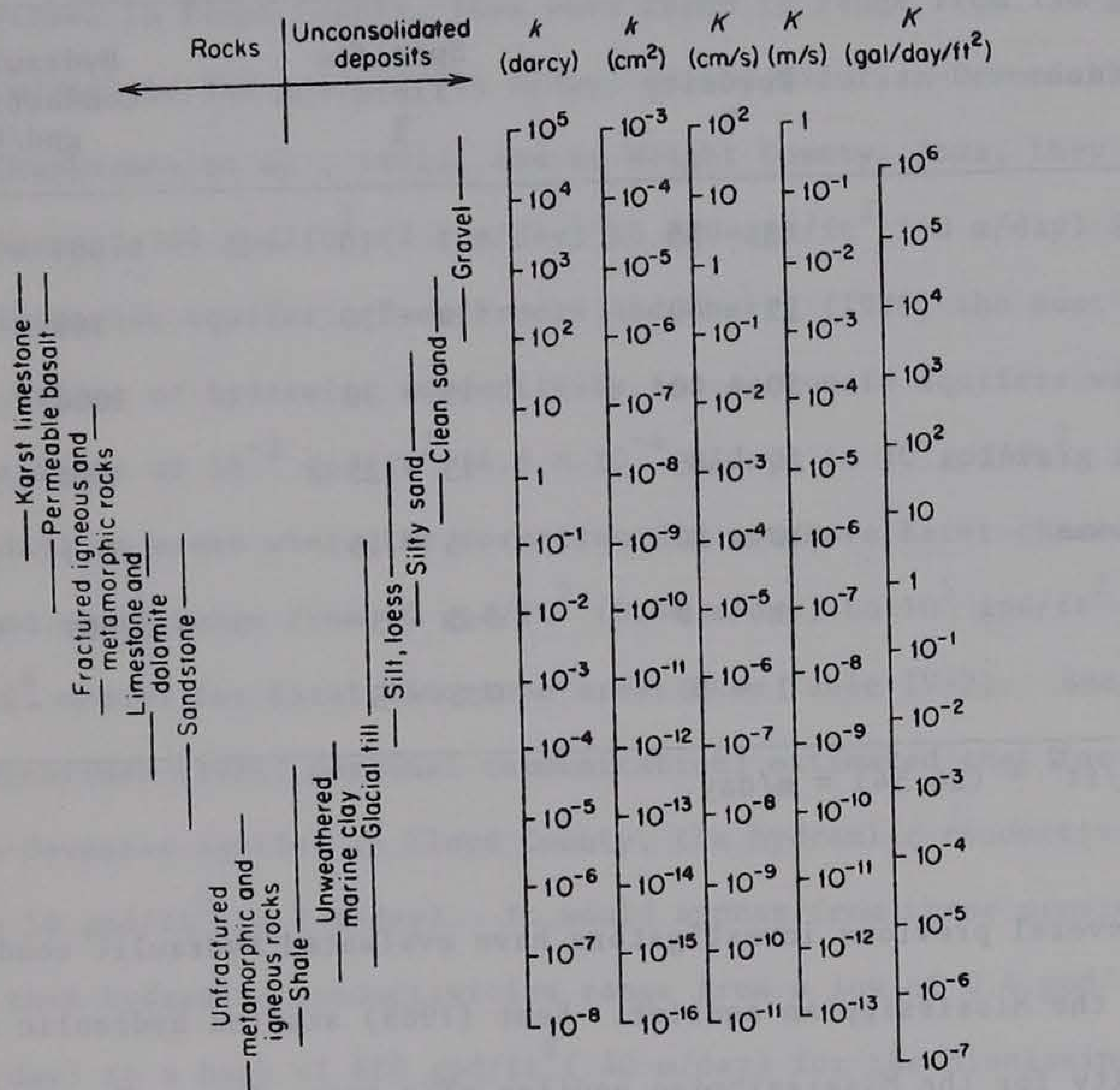

Table IV-2. Typical values of porosity, specific yield, and hydraulic conductivity for various materials (after Groundwater Management Manual, ISU Extension, 1982).

Formation	Porosity %	Specific Yield %	Hydraulic Conductivity gpd/ft^2 [a]
Clay	45 - 55	1 - 10	0.001 - 2
Sand	35 - 40	10 - 30	100 - 3000
Gravel	30 - 40	15 - 30	1000 - 15000
Sand & gravel	20 - 35	15 - 25	200 - 5000
Sandstone	10 - 20	5 - 15	0.1 - 50
Shale	1 - 10	0.5 - 5	0.00001 - 0.1
Limestone	1 - 10	0.5 - 5	-

[a] 1 gpd/ft^2 * (24.54) = m/day.

Several previous investigators have evaluated hydraulic conductivity of the Mississippian aquifer. Kent (1969) studied hydraulic conductivity for the Mississippian aquifer near Ames, Iowa. He reported values ranging from a maximum of 37 gpd/ft^2 (1.7 m/day) to 0.36 gpd/ft^2 (0.017 m/day). This resulted in transmissivities of 8100 gpd/ft (100 m^2/day) to 5780 gpd/ft (72 m^2/day). Munter (1980) made local estimates of hydraulic conductivities for the Silurian-Devonian aquifer near Charles City, Iowa. These estimates ranged from 1.5 gpd/ft^2 (0.07 m/day) to 540 gpd/ft^2 (24 m/day). Musterman et al. (1981), using a flow net analysis, estimated the hydraulic conductivities in Humboldt County for the Mississippian aquifer to range from 2.2 gpd/ft^2 (0.1 m/day) to 780 gpd/ft^2 (35 m/day). They also estimated hydraulic conductivities of 25 gpd/ft^2 (1.1 m/day) to 420 gpd/ft^2 (19 m/day) for the Silurian-Devonian aquifer (Musterman et al., 1981). Estimates of hydraulic conductivities in Floyd County, Iowa were found to range from 120 gpd/ft^2 (5.4 m/day) to 960 gpd/ft^2 (44 m/day) in the Silurian-Devonian aquifer (Musterman et al., 1981), and in Wright County, Iowa, they found ranges of 160 gpd/ft^2 (7.1 m/day) to 880 gpd/ft^2 (40 m/day) in the Mississippian aquifer. From Freeze and Cherry (1979) the most probable values of hydraulic conductivity for carbonate aquifers would be in the range of 10^{-2} gpd/ft^2 (4.6×10^{-4} m/day) to 10 gpd/ft^2 (0.46 m/day) in areas where the formation did not have Karst characteristics and would range from 10 gpd/ft^2 (0.46 m/day) to 10^5 gpd/ft^2 (4.6×10^4 m/day) for Karst limestone areas (see Table IV-2).

Steinhilber (1981, personal communication) estimated that for the Silurian-Devonian aquifer in Floyd County, the hydraulic conductivity would be 50 gpd/ft^2 (2.2 m/day). It would appear from these previous studies that hydraulic conductivities range from a low of 0.4 gpd/ft^2 (0.02 m/day) to a high of 880 gpd/ft^2(40 m/day) for the Mississippian aquifer and from a low of 1.5 gpd/ft^2 (0.07 m/day) to a high of 960 gpd/ft^2 (44 m/day) for the Silurian-Devonian aquifer in the study region. This range in hydraulic conductivity is three orders of magnitude, which is common in carbonate aquifers. For this study, it was assumed that hydraulic conductivity of the Mississippian aquifer varied from 30 gpd/ft^2 to 800 gpd/ft^2.

An attempt was made to determine the hydraulic conductivity in Humboldt County by using a slug test at one of the the ADWs being monitored. A weighted line was used to determine approximate depths of each well. Table IV-3 shows the ground elevation, water level elevation, and approximate bottom elevation for each of the ADWs monitored in this project. Depth estimates ranged from 100 feet (30.5 m) to 285 feet (87 m), which according to the generalized geologic cross-sections (Figs. I-3 and I-4), would indicate these ADWs are recharging the Mississippian aquifer.

The slug test was performed at well nos. 2 and 6 during August 1982. Both wells were recharging subsurface drainage water during the test. Measurements of tile discharge into the ADWs were made using a bucket and stop watch. The influent tiles were plugged and the time rate of decline in water levels in the ADW was recorded. A transmissivity of about 600 gpd/ft was found at both wells. Assuming the depths in Table IV-3 are correct and the wells recharge through the entire depth of bedrock penetrated, hydraulic conductivities of 6 gpd/ft^2 (0.27 m/day) and 2.7 gpd/ft^2 (0.12 m/day) were obtained. These appear smaller than most of the previously reported values. This may be due to plugging from sediments in the aquifer or errors in the slug test. Little confidence in the reliability of these data is warranted.

Table IV-3. Summary of ADW survey.

Well No.	Ground elev. (ft. msl)[a]	Water table elev. (ft msl)	Approx. bottom of well elev. (ft msl)
1	1140.18	1100.09	n/a
2	1135.34	1087.82	975.3
6	1133.53	1086.87	848.7
7	1140.73	1092.49	1020.7

[a] 1 ft = 0.3048 meters.

Thickness of the Mississippian aquifer in Humboldt County, Iowa is 200 ft (61 m) to 300 ft (91 m) (Figs. I-3 and I-4). The four ADWs measured did not appear to penetrate the entire thickness of the aquifer. Based on the depths of these ADWs, it was assumed that most ADWs would penetrate 75 ft (23 m) to 200 ft (61 m). Combining these depths with the assumed hydraulic conductivities leads to the most probable range of transmissivities from 6000 gpd/ft (75 m^2/day) to 60,000 gpd/ft (745 m^2/day) used in the drawdown model.

Typical values for the storage coefficient are very dependent on the physical configuration of the aquifer. The storage coefficient is a dimensionless term used to describe the volume of water yielded per unit area and unit drop in piezometric level. For a confined aquifer, storage coefficient values are quite small and can range from 0.005 to 0.00005. For unconfined aquifers, the storage coefficient is known as specific yield and typically ranges from 0.01 to 0.30 (Bouwer, 1978; Freeze and Cherry, 1979).

The depth to water in the ADWs monitored in Humboldt County ranged from 40 ft (12 m) to 48 ft (15 m) below ground surface. This appears to be at or near the interface between the glacial drift and the Mississippian aquifer. No site-specific data could be found for storage coefficient. Thus, the modeling was done with two storage coefficients, 0.02 and 0.002, which would be typical of that expected in an unconfined and confined aquifer, respectively.

Ranges for recharge rates and length of time of recharge events were estimated from earlier studies, which had shown that tile systems classified as providing poor drainage could be expected to drain

1/4 inch (0.64 cm) per day from the soil, while those systems with excellent drainage conditions could drain 1 inch (2.54 cm) per day or more. Assuming each ADW receives subsurface drainage from an average area of 40 acres (16.2 ha) per tile system, recharge rates were estimated to range from 180 to 750 gpm (0.68 to 2.84 cubic meters per min). Modeling reported previously (Fig. II-2) indicates the average annual subsurface drainage during the period 1957 to 1979 was estimated at 3.5 inches (8.9 cm) and the maximum annual value was 11.0 inches (28 cm). Using drainage coefficients of 0.25 inch/day and 1 inch/day, the estimated drainage times of 15 to 45 days were obtained (Baker and Austin, 1982). It is realized that constant flow at the maximum rates would not occur; in fact, a hydrograph would result; thus, the actual drainage times would be longer, but the flow rates would be smaller, and the shorter periods of higher constant flows were used in the modeling in order to simulate "worst case" conditions.

Porosity values for limestone usually range from 1 to 10%, but for the test area, where secondary porosity is prevalent, values could range from 5% to 30% or more. These values were used in the model.

The PLUME dispersion model can account for pollutant decay and adsorption within the aquifer, however. These options were not used in this analysis. Therefore, there is no change in pollutant mass as the plume moves through the aquifer.

The pollutant mass loading rate and the dispersivity values were perhaps the most elusive of the solute transport parameters. Longitudinal and transverse dispersivities are very difficult to measure in the field, and laboratory measurements of these values are complicated and time-consuming. Longitudinal dispersivity can be measured in the laboratory by passing a tracer through samples collected from boreholes or excavations, but the resulting values are generally viewed as providing little indication of the in situ dispersivity of the geologic materials. Dispersivity has the distinction of being a parameter for which laboratory-obtained values are commonly regarded as having little relevance in the analysis of problems at the field scale (Freeze and Cherry, 1979). It is, however, generally accepted that longitudinal and transverse dispersivities under field conditions are larger than those indicated on borehole samples. In other words, tracer or contaminant spreading in the field resulting from dispersion is generally greater than would be indicated by laboratory measurements.

Very little information could be obtained from the literature for the dispersion coefficients in limestone formations. Pinder (1973) reported values of longitudinal dispersivity, dispersion coefficient divided by average pore velocity, of 70 ft (21.3 m), and transverse dispersivity of 14 ft (4.27 m) for a glacial outwash aquifer in Long Island, New York. Oaks and Edworthy (1976) used a longitudinal dispersivity of 2 ft (0.6 m) for the Bunter Sandstone aquifer near Mansfield, England.

In a recent review of the use of models to simulate movement of contaminants through groundwater, Anderson (1979) discussed the problems of determining longitudinal dispersivity for various media. Longitudinal dispersivity for a limestone formation at Brunswick, Georgia was reported as 200 ft (61 m), the ratio of longitudinal to transverse dispersivities of 33.3 and porosity of 0.35 (Anderson, 1979). Also, data for a limestone aquifer near Culter, Florida, is reported as longitudinal dispersivity of 22 ft (6.7 m), ratio of longitudinal to transverse dispersivity of 10.0 and porosity of 0.25. Anderson (1979) and Gelhar (1983) showed longitudinal dispervisities for limestone aquifers to be between 3.3 ft (1 m) and 330 ft (100 m) with the most probable upper limit at 33 ft (10 m). Bakr reported longitudinal dispersivity of 125 ft (38.1 m) and porosity of 0.12 for fractured dolomite near Carlsbad, New Mexico (Anderson, 1979).

The lower the longitudinal dispersivity the less the effects of dispersion will be realized. For this study it was assumed that a low longitudinal dispersivity (1 ft or 0.3 m) and a high longitudinal dispersivity (100 ft or 30 m) would represent the range of conditions found in north-central Iowa.

Average pore velocity was determined using Darcy's equation and the range of hydraulic conductivities and porosities already mentioned. The groundwater gradient was estimated by assuming that the regional piezometric surface was approximately parallel to the ground surface.

This assumption was verified by determining water surface elevations at several quarries in Humboldt County. In addition, water levels were measured in the four ADWs monitored and at several points along the Des Moines River. Assuming these represent the regional piezometric elevations, the groundwater gradient was determined. The direction of groundwater flow seems to be toward the Des Moines River, which serves as a discharge point.

Nitrate nitrogen was chosen as the parameter for modeling because it represented the most significant contaminant in the recharge water, in terms of violation of drinking water standards. It was felt that modeling of other pollutants would give similar results. An average concentration of NO_3-N in the recharge water of 20 mg/ℓ was assumed. This led to mass loadings of 18 lbs/day (8 kg/day) for the 75 gpm recharge rate. No attempt to model pesticides or other parameters was made in this study. Table IV-4 summarizes the input parameters used in the models.

D. Results

A computer analysis of the cones of impressions from the three ADWs along Highway 3 west of Humboldt (Fig. IV-3) was conducted using the best estimates of the aquifer parameters from Table IV-4. Table IV-5 gives the results of these analyses using typical injection conditions while Table IV-6 gives results from extremely high flows for shorter durations. Each table shows the recharge rate and aquifer parameters used for each run and the arithmetic average of rises at each of the three wells for each time period. The distance to the 0.1 ft (3.0 cm) rise gives an estimate of the radius of influence of the cone of impression occurring for a single well operating under the modeled conditions at the times given in each heading. For example, in run 5, a single well injects 75 gpm (0.28 cubic meters/min) into an aquifer with a transmissivity of 6000 gpd/ft (74.5 square meters/day) and a storage coefficient of 0.002. After seven days, the rise at the well (the peak of the injection cone) is 23.5 ft (7.1 m) and the radius of the cone extends for 4990 ft (1520 m) around the well. After 28 days, the rise at the well

Table IV-4. Major input parameters needed by the computer models.

Parameter	Bracketing values	Units[a]
A. Drawdown Model		
storage coefficient	0.02-0.002	decimal
hydraulic conductivity	30-800	gpd/ft^2
transmissivity	6000-600000	gpd/ft[b]
well flow rate	75-750	gpm
B. Dispersion Model		
aquifer thickness	75-200	feet
porosity	0.05-0.3	decimal
velocity	1.31-131	ft/day[c]
longitudinal dispersivities	1-100	ft
dispersivity ratio	5-20[d]	-
mass loading rate	18-180	lb/day[e]

[a]Refer to units conversion table for conversion factors.

[b]Transmissivity, T, is found to T = KD, where K is the hydraulic conductivity (gpd/ft^2) and D is the height of the aquifer.

[c]The velocity of the groundwater flow within the voids can be estimated from: V = KI/n, where K is the hydraulic conductivity, I is the gradient of the groundwater flow, and n is the effective porosity of the aquifer.

[d] Ratio of longitudinal to transverse dispersivity.

[e]Estimated directly or obtained from the product of QC_o, where Q is the volume flow rate (L^3/T) and C_o is the initial concentration.

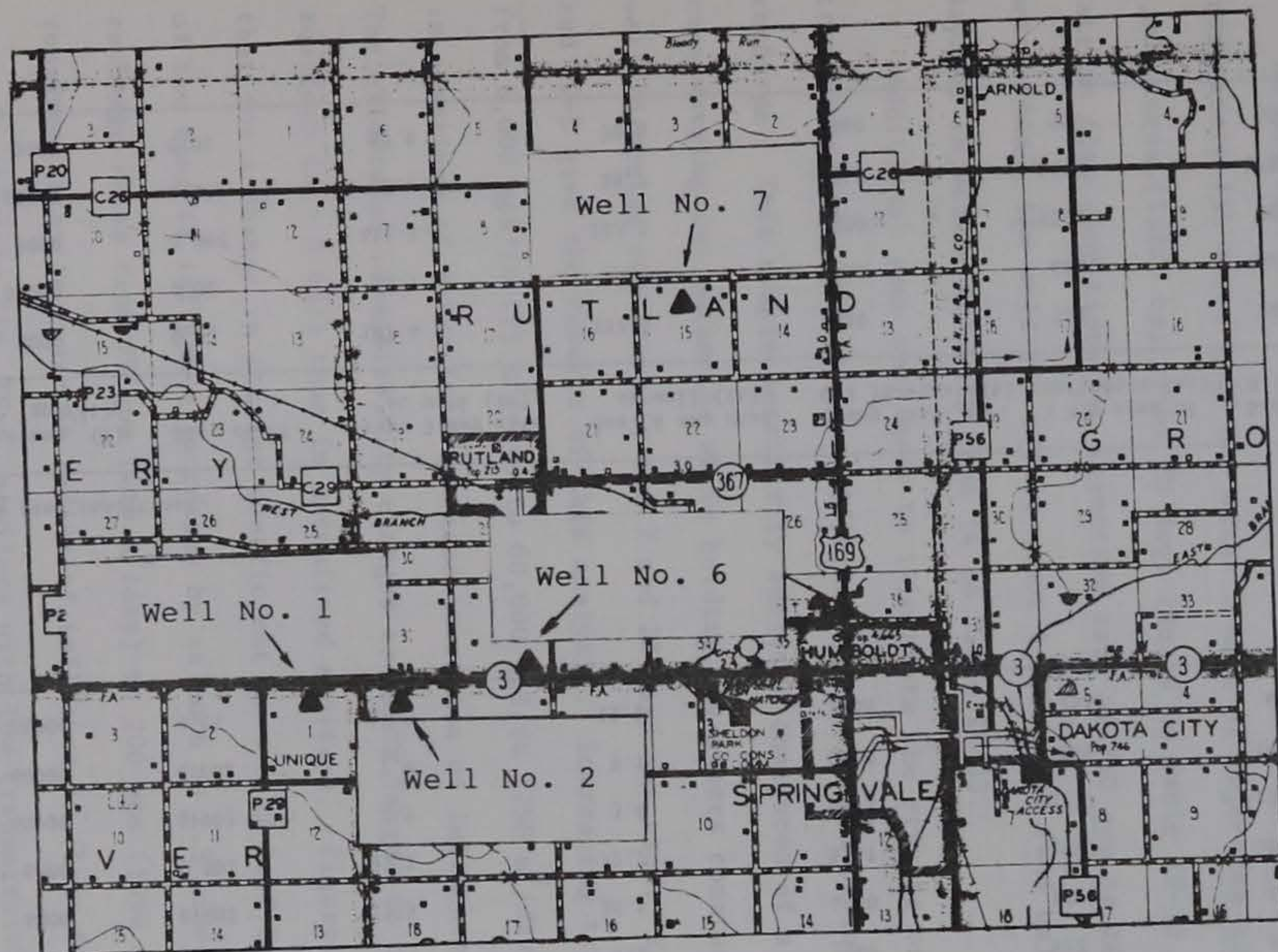

Figure IV-3. Location of agricultural drainage wells in Humboldt County.

Table IV-5. Typical flow conditions.

Run #	Flow rate (gpm)	trans. (T) (gpd/ft)	Sto. coeff.	ave. 7 day rise at well (ft)	Ave. 28 day rise at well (ft)	7 day dist. to 0.1 ft rise (ft)	7 day area of influence (ac)	28 day dist. to 0.1 ft rise (ft)	28 day area of influence (ac)
1	75	6000	0.02	20.0	22.0	1372	136	2916	613
2	200	6000	0.02	53.2	58.7	1838	244	1989	263
3	75	60000	0.02	2.4	2.7	1686	205	4187	1264
4	200	60000	0.02	6.3	7.2	2960	632	6222	2792
5	75	6000	0.002	23.5	26.5	4990	1796	6997	3531
6	200	6000	0.002	63.1	70.7	5257	1993	10560	8042
7	75	60000	0.002	2.9	3.4	6172	2727	13200	12566
8	200	60000	0.002	7.8	9.1	9900	7069	18480	24630
9	150	20000	0.01	14.3	15.5	3235	755	6475	3024

Table IV-6. Extreme flow conditions.

Run #	Flow rate (gpm)	trans. (T) (gpd/ft)	Sto. coeff.	ave. 3 day rise at well (ft)	Ave. 6 day rise at well (ft)	3 day dist. to 0.1 ft rise (ft)	3 day area of influence (ac)	6 day dist. to 0.1 ft rise (ft)	6 day area of influence (ac)
10	750	6000	0.02	187.4	197.3	1320	126	2640	503
11	750	60000	0.02	22.1	23.4	3080	684	4220	1284
12	750	6000	0.002	221.2	223.5	4620	1539	5940	2545
13	750	60000	0.002	26.7	28.7	9240	6158	17820	22902
14	750	2000	0.01	67.9	70.8	2860	590	4050	1183

is 26.5 ft (8.1 m) and the radius of influence is 6,997 ft (2,133 m). These radii of influences can be converted directly to areas of influence of 1,800 and 3,500 acres (730 and 1400 ha), respectively.

An ADW well in the Humboldt area can be expected to influence approximately 755 acres (305 ha) after 7 days of injecting and up to 3,000 acres (1,200 ha) after 28 days (Run 9 in Table IV-5). Under extreme flow conditions (an injection rate of 750 gpm or 28 cubic meters/min), the well can affect about 1,200 acres (486 ha) after 6 days of injection activity (Run 14 in Table IV-6).

Additional runs were made to illustrate the effects of changes in transmissivity and storage coefficient on the radius of influence during recharge. This simple sensitivity analysis consisted of a series of runs varying one parameter while holding the others constant. A single well injecting continuously for 7 and 28 days was used. Figures IV-4 and IV-5 give the results of this analysis. Increasing transmissivity from 6,000 gpd/ft (75 m^2/day) to 60,000 gpd/ft (750 m^2/day) resulted in a maximum of a two-fold increase in radius of influence (Fig. IV-4). The effects of transmissivity changes are much larger in confined aquifers (s = 0.002) than in unconfined aquifers. Figure IV-5 shows that for a change in storage coefficient from 0.02 to 0.002, the radius of the cone of influence increases by as much as 2.5 times. Increasing recharge rates from 75 gpm (410 m^3/day) to 200 gpm (1090 m^3/day) resulted in an increase in radius of influence of about 50%.

As shown in Fig. IV-6, aquifers with low transmissivity develop tight, high cones of impression, whereas aquifers of high transmissivity develop shallow cones of wider extent. The storage coefficient affects

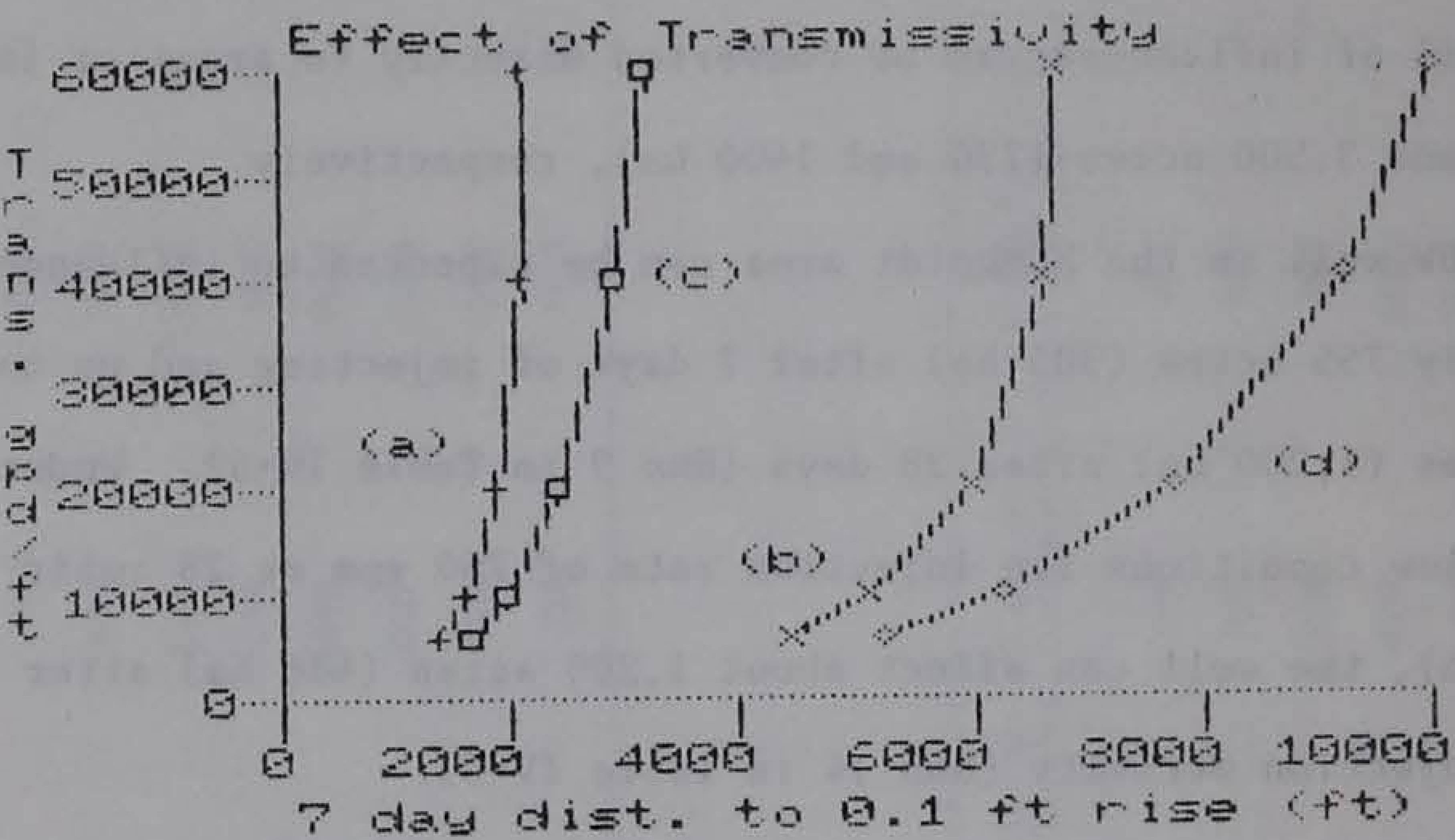

Figure IV-4. Effect of transmissivity on radius of influence given (a) S = 0.02, Q = 75gpm; (b) S = 0.002, Q = 75gpm; (c) S = 0.02, Q = 200gpm; (d) S = .002, Q = 200gpm. See conversion table for metric equivalents.

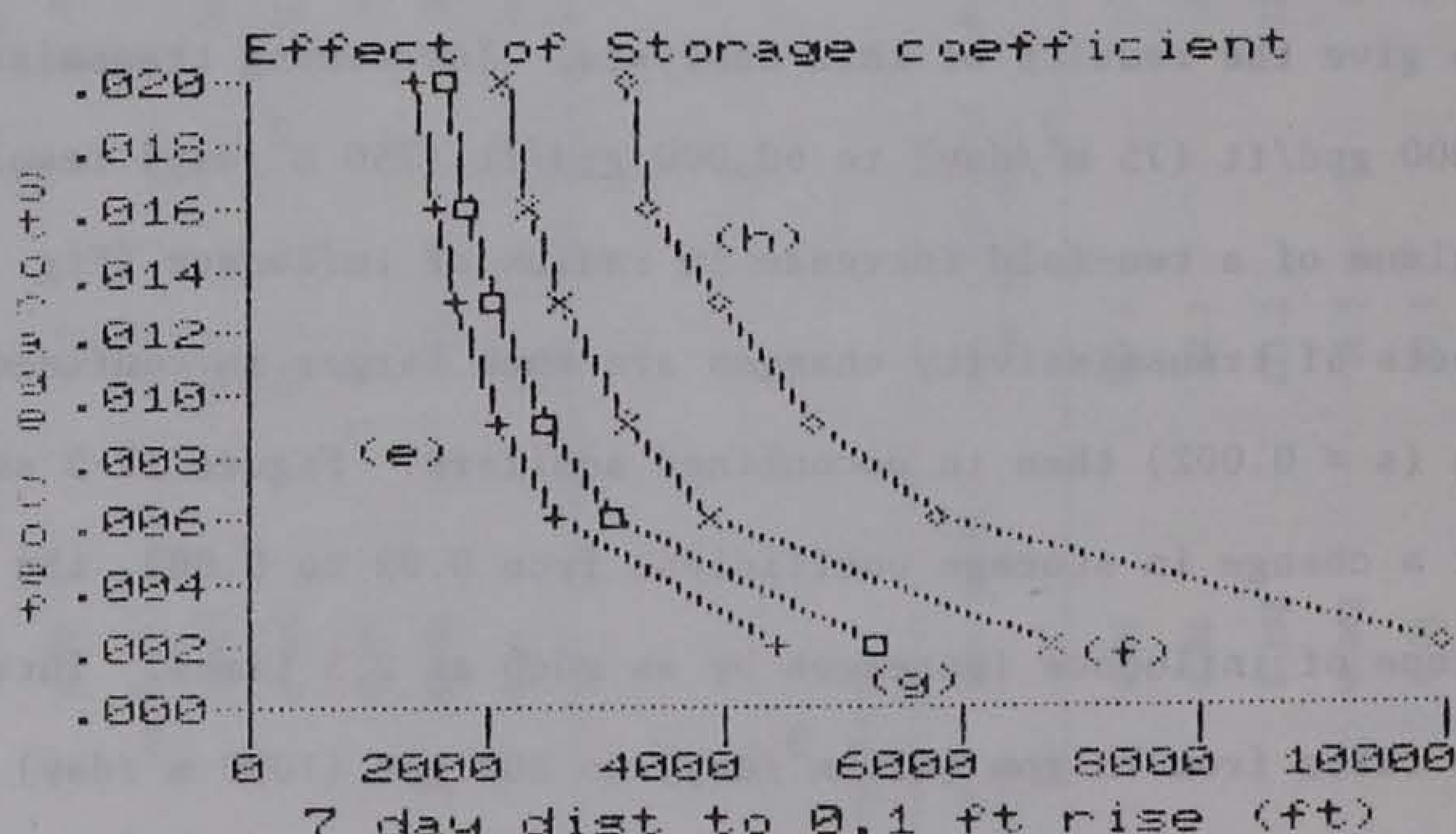

Figure IV-5. Effect of storage coefficient on radius of influence given (e) T = 6000gpm/ft, Q = 75gpm; (f) T = 6000gpm/ft, Q = 75gpm; (g) T = 6000gpm/ft, Q = 200gpm; (h) T = 60000 gpm/ft, Q = 200gpm. See conversion table for metric equivalents.

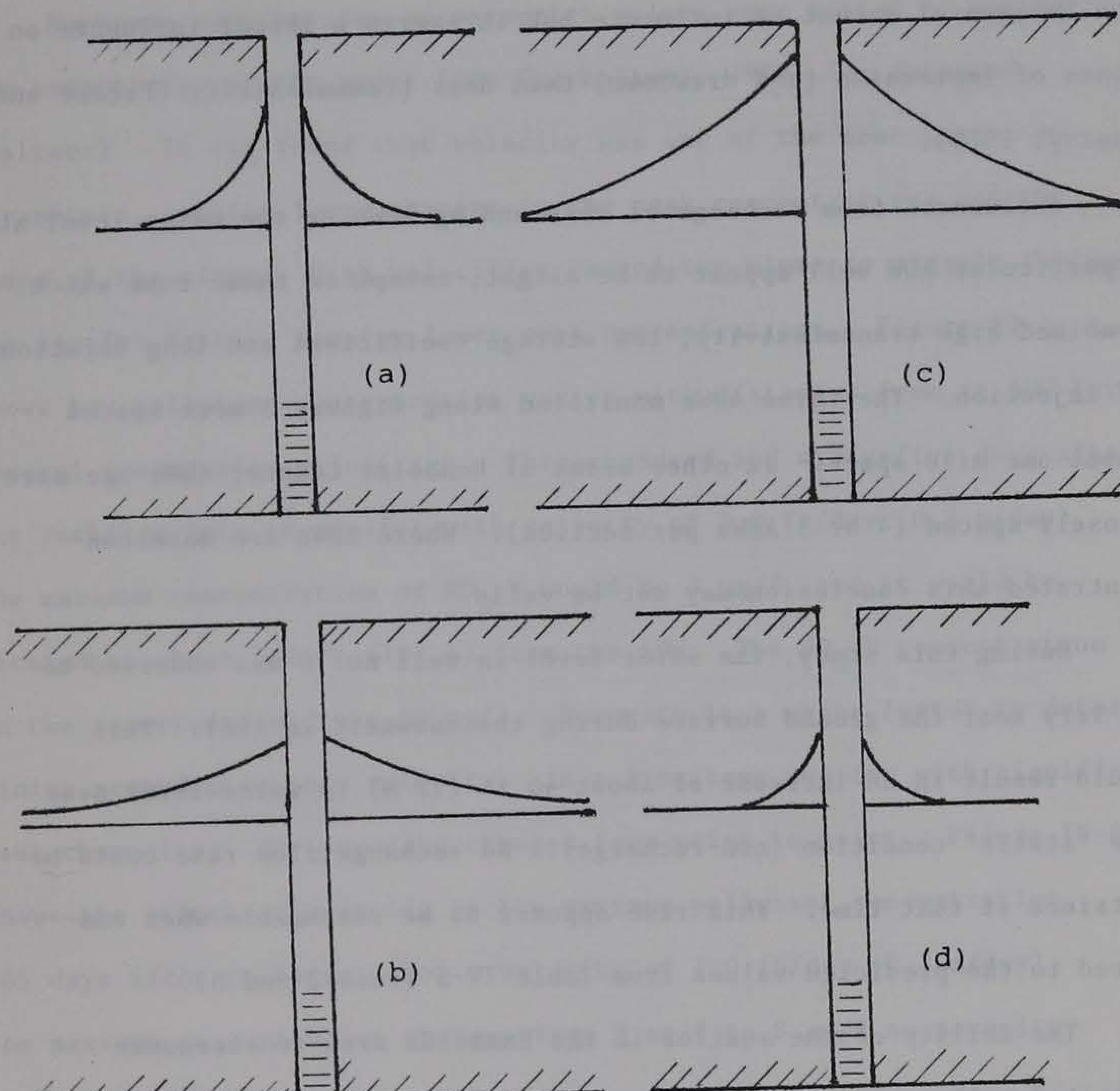

Figure IV-6. Comparisons of cones of influence at a given time for aquifers of (a) low transmissivity; (b) high transmissivity; (c) low storage coefficient; (d) high storage coefficient (after Freeze and Cherry, 1979).

the horizontal extent of influence but it exerts a lesser influence on cones of impression (and drawdown) than does transmissivity (Freeze and Cherry, 1979).

Influences from recharge of surrounding ADWs on the water level at a particular ADW well appear to be slight, except on those runs which combined high transmissivity, low storage coefficient and long durations of injection. The three ADWs monitored along Highway 3 were spaced about one mile apart. In other areas of Humboldt County, ADWs are more closely spaced (4 or 5 ADWs per section). Where ADWs are more concentrated this conclusion may not be valid.

During this study, the water level in well no. 6 was observed to be very near the ground surface during the snowmelt in 1981. This would result in an increase of about 40 ft (12 m) in water level over the "static" condition (non-recharge). No recharge flow rate could be obtained at that time. This rise appears to be reasonable when compared to the predicted values from Table IV-5 (runs 2 and 5).

The ability of the aquifer in the Humboldt area to attenuate injected pollutants was studied with the help of a computer model called PLUME. The model required information about the aquifer characteristics, rate of injection, and pollutant properties. Only nitrate-nitrogen was used as the pollutant in the studies since it is the principal contaminant injected by ADWs. The best estimates of the critical parameters are shown in Table IV-4. A 30-day recharge event was simulated, and output was obtained for day 365 after recharge began. In other words, a slug of NO_3-N was injected continuously for 30 days, and then the concentration was evaluated one year later.

Numerous combinations of parameter values were tested to determine the sensitivity of the model (see Overholtzer, 1983, for detailed analyses). It was found that velocity was one of the most sensitive parameters, causing large changes in the peak concentrations and the shape of the plume. High velocities caused the plume to migrate through the aquifer faster, causing lower peak concentrations. Figure IV-7 shows how pollutant concentration varies with distances from an ADW for several groundwater velocities. Three-hundred and sixty-five days after the recharge into an aquifer with velocity of 1.0 ft/day (0.3 m/day), the maximum concentration of NO_3-N would be 3 mg/ℓ, and it would be located at about 500 ft (152 m) from the ADW. The NO_3-N concentration in the injected fluid was 20 mg/ℓ. Porosity is a major factor in determining groundwater pore velocity. In a limestone aquifer with significant channeling, the porosity changes from point to point. Figure IV-8 shows the effect of porosity on the maximum pollutant concentration, 365 days after recharge. For a velocity of 1.0 ft/day (0.3 m/day), the maximum concentration changes from 3 mg/ℓ to 8 mg/ℓ as porosity changes from 0.3 to 0.1.

The effect of slug interval (length of recharge event) on the plume shape was evaluated (Fig. IV-9). For a 365-day recharge event, the concentration in the groundwater is equal to the concentration in the injected water for 500 feet (152 m) downstream. Drinking water standards were violated for about 750 ft (229 m). For a 120-day injection period, the maximum concentration on day 365 is 10 mg/ℓ at 300 ft.

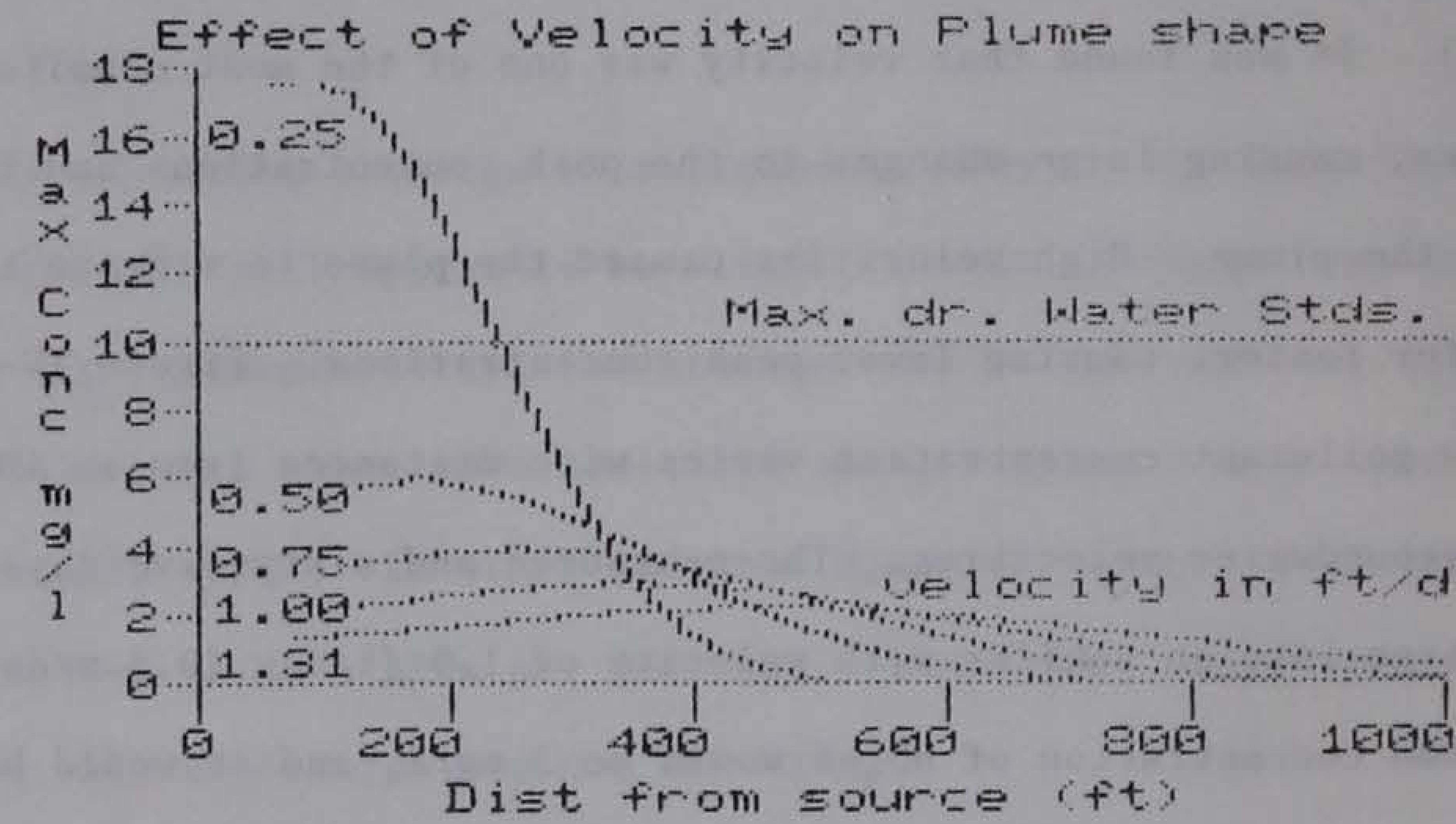

Figure IV-7. The effect of velocity on the shape of the pollution plume (mass loading rate, 18 lb/d; porosity, 0.30; aquifer thickness, 100 ft; x dispersivity, 100 ft; dispersion ratio, 20). See conversion table for metric equivalents.

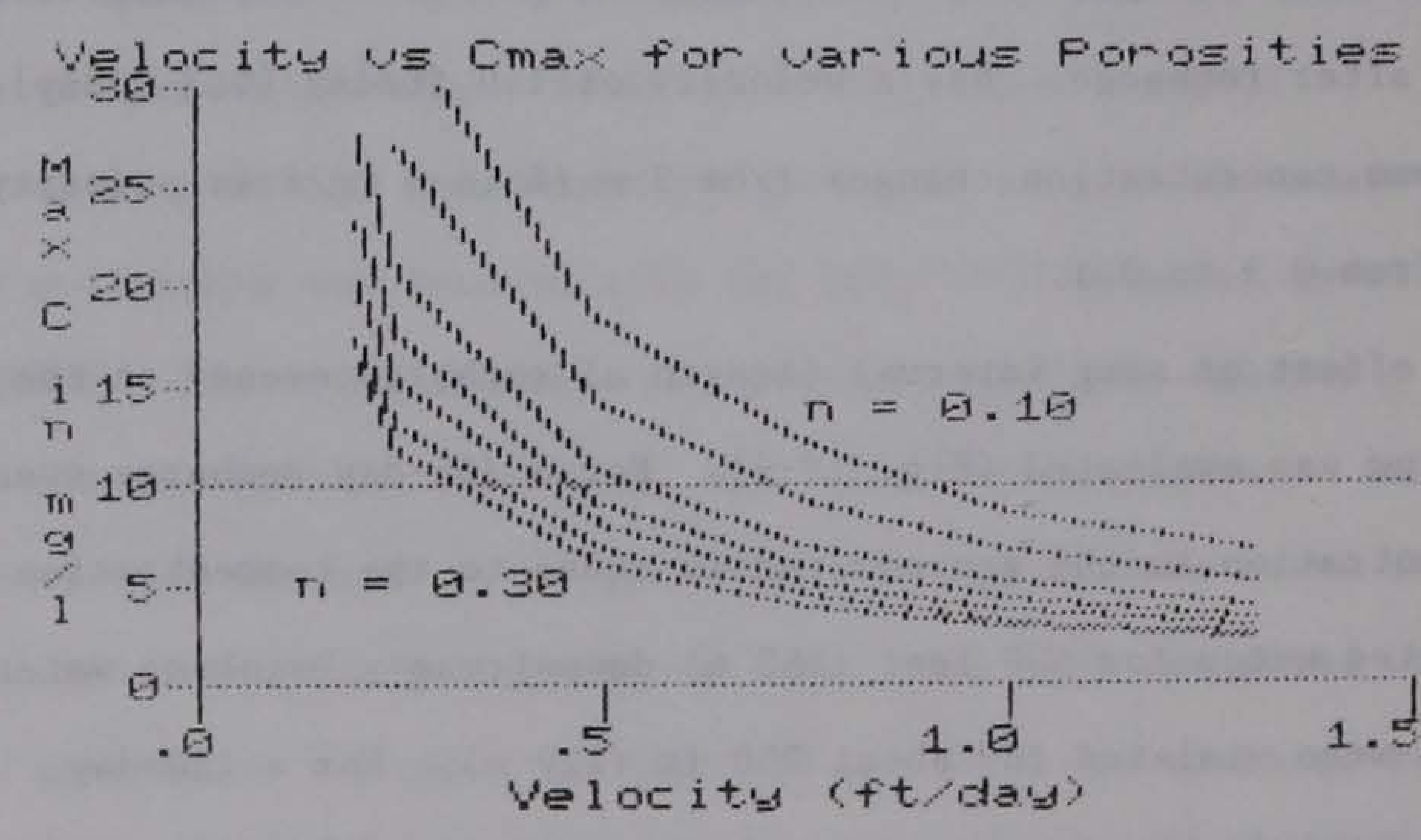

Figure IV-8. The effect of velocity on peak concentration for various porosities (mass loading rate, 18 lb/d; aquifer thickness, 100 ft; x dispersity, 100 ft; dispersion ratio, 20). See conversion table for metric equivalents.

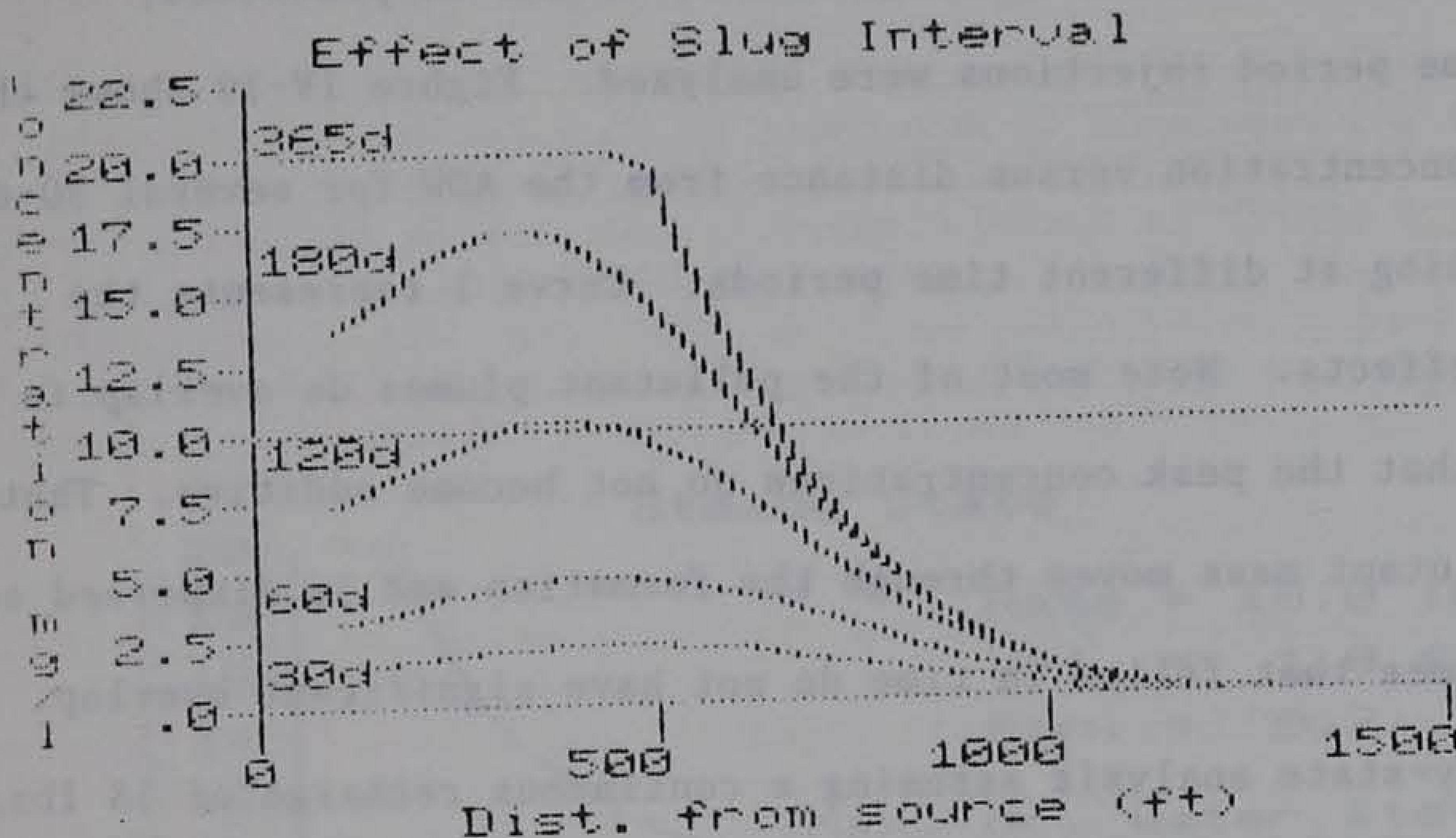

Figure IV-9. The effect of slug interval on plume shape (mass flow rate, 18 lb/d; velocity, 1.31 ft/d; porosity, 0.30; x dispersivity, 100 ft; dispersion ratio, 20). See conversion table for metric equivalents.

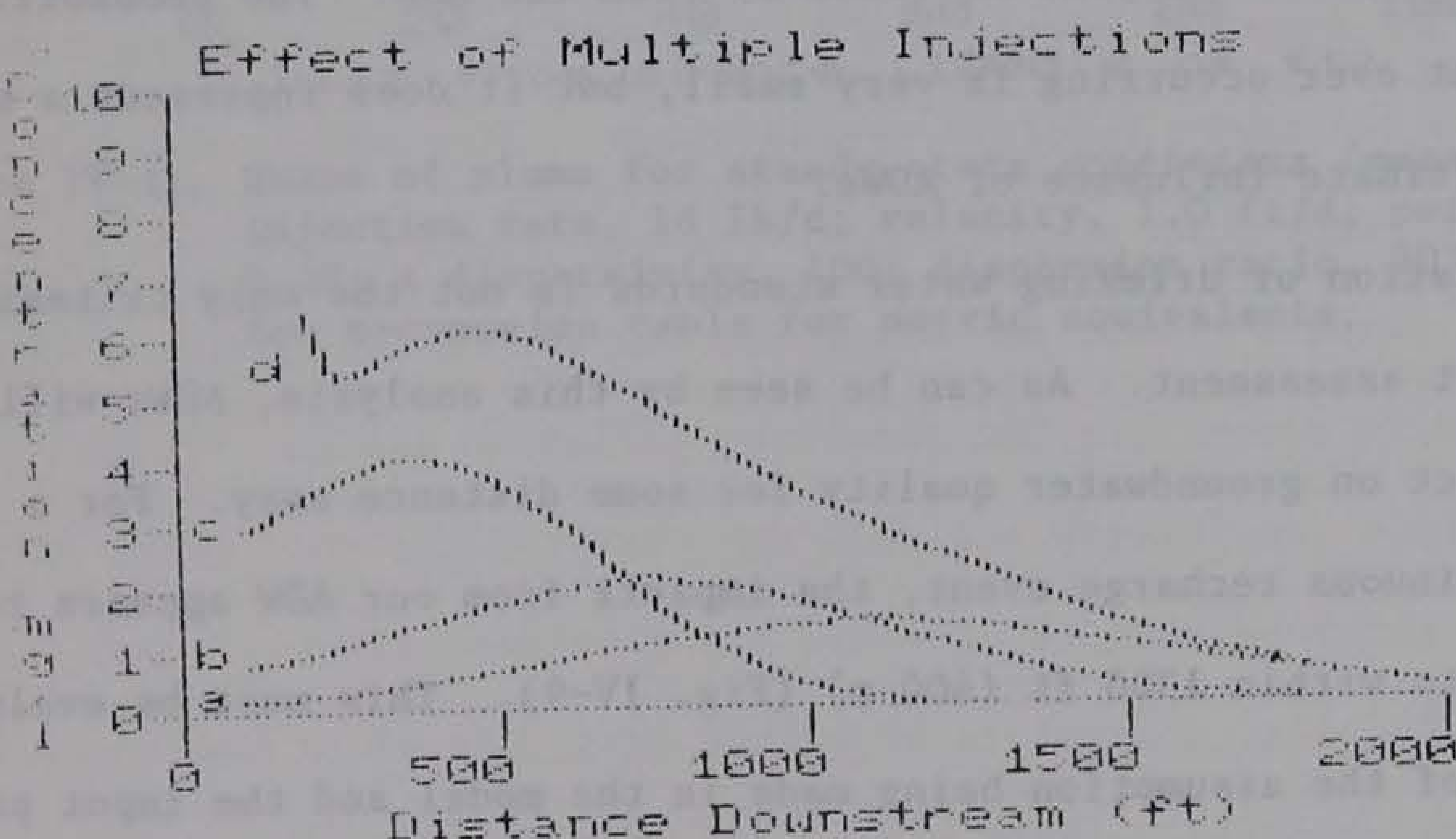

Figure IV-10. The effect of multiple injections on plume shape for (a) 30 day slug starting at day 0; (b) 30 day slug starting at day 365; (c) 30 day slug starting at day 730; and (d) combined effect from 30 day slugs starting at days 0, 365, 730, and 1095 (time of observation, 1125 d; mass flow rate 18 lb/d; velocity, 1.0 ft/d; porosity, 0.20; aquifer thickness, 100 ft; x dispersivity, 100 ft; dispersion ratio, 20).

Because the length of recharge events cannot be predicted, multiple time period injections were analyzed. Figure IV-10 shows the pollutant concentration versus distance from the ADW for several 30-day slugs beginning at different time periods. Curve 1 represents the cumulative effects. Note most of the pollutant plumes do overlap in such a way that the peak concentrations do not become additive. That is, the pollutant mass moves through the formation and is dispersed so that the plumes that follow in time do not have significant overlap.

A steady-state analysis assuming a continuous recharge of 18 lbs/day (8.2 kg/day) for an infinite time period was made. Figure IV-11 shows concentration versus distance from an ADW under steady conditions. Under these worst case conditions, drinking water standards would be violated for about 30,000 ft (9140 m) from the ADW. The probability of this event ever occurring is very small, but it does represent a measure of the ultimate influence of ADWs.

Violation of drinking water standards is not the only criteria for impact assessment. As can be seen by this analysis, ADWs will have some impact on groundwater quality for some distance away. For a one year continuous recharge event, the impacts from our ADW appears to be confined to within 1300 ft (400 m) (Fig. IV-9). This must be evaluated in light of the assumption being made in the model and the input parameters.

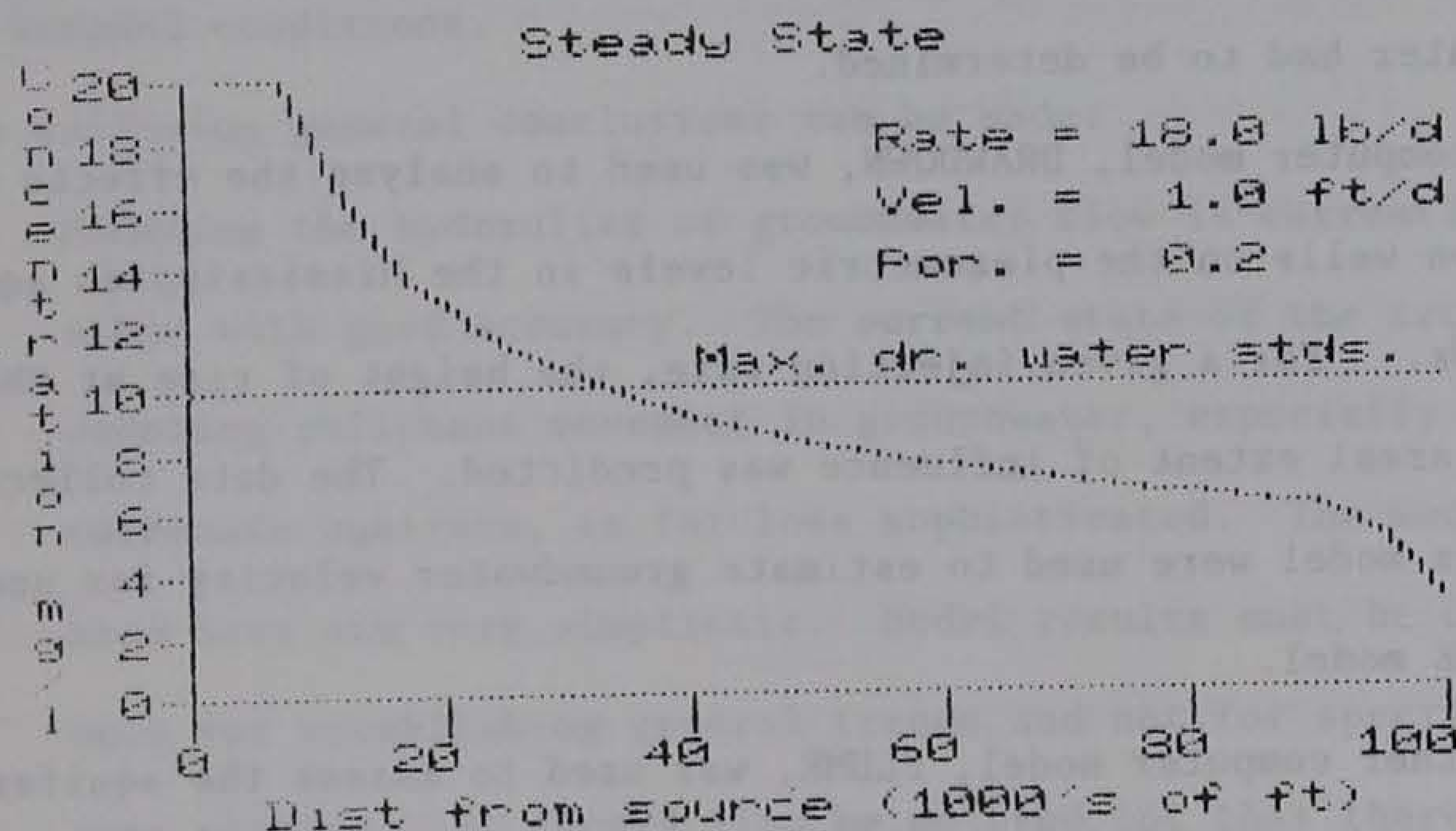

Figure IV-11. Shape of plume for steady state conditions (mass injection rate, 18 lb/d; velocity, 1.0 ft/d; porosity, 0.20; x dispersivity, 100; dispersion ratio, 20). See conversion table for metric equivalents.

E. Summary and Conclusions

A study has been made to determine to what extent agricultural drainage wells (ADWs) in Humboldt County, Iowa may be affecting the underground supply of drinking water in that area. To achieve that purpose, the impact of these wells on pollutant concentrations in the groundwater had to be determined.

A computer model, DRAWDOWN, was used to analyze the effects of injection wells on the piezometric levels in the Mississippian aquifer near ADWs. For a given injection rate, the height of rise at the well and the areal extent of influence was predicted. The data collected from this model were used to estimate groundwater velocity for use in the PLUME model.

Another computer model, PLUME, was used to assess the aquifer's ability to attenuate injected contaminants. This model considered the dispersive properties of the aquifer as its primary attenuation method. The model takes a number of parameters describing the aquifer, time and space, and it calculates the resulting pollution concentrations.

A simple sensitivity analysis was performed on each model to determine which parameters effected the greatest change in each model's results. Critical parameters were then tested in an effort to obtain "worst case" results. These worst case results were then used to make decisions regarding the impact of ADWs on the aquifer and the potential for pollution.

The modeling approach taken in this study is extremely simplistic. The Mississippian aquifer is a limestone dolomite formation with planes of secondary porosity. All models used herein assume homogeneous, isotropic conditions. Only limited field data were available from secondary sources. No new data were collected in this project. Verification of the models using field data was not possible. Thus, the best that can be expected is to determine a range of possible impacts for various assumed conditions.

The following general conclusions can be made:

1. Modeling the hydraulics of groundwater flow is currently possible with good accuracy. The current state of the art of modeling pollutant movement in groundwater, especially in carbonate aquifers, is far less sophisticated. The models used here are very simplistic. Model results must be used only for establishing general trends and not for specific case studies. It should also be pointed out that there were no field data to verify and calibrate these models, and the project budget did not permit field data collections.
2. The maximum concentration of pollutants during and immediately after a recharge event will be found near the ADW. Dispersion will occur, even in a limestone dolomite formation. It is not possible to estimate with any degree of accuracy what the limits of the pollutant plume will be, but the model results, presented here and by Overholtzer (1983), using a range of "best" estimates of parameters indicate the major impacts will be located near the ADW; that is, within about one mile (2 km). Violations of drinking water standards can be expected to occur within a few hundred feet of the ADW.

3. The model results indicate no significant regional (county-wide or greater) impact from a single ADW. However, in areas where a large number of ADWs exist in a small area, some overlap of the pollutant plumes can be expected. The ADWs monitored in this study were about one mile apart, and little interaction between these wells would be expected.

   The steady state solution (Fig. IV-11) for an infinite continuous recharge event shows NO_3-N levels in excess of drinking water standard as far as 30,000 ft (914 m) from the ADW. This event is not expected to ever occur since it involves continuous injections of drainage water from infinite time, but it should represent some indication of maximum areal extent of the impacts.

4. Modeling of other parameters was not performed in this study, but similar results would be expected.

For additional details on the models and analysis of results see Overholtzer (1983).

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h a maximum of 8 within any one section. The ADWs in
spersed throughout the sampling area. Area 2 has a
ty with a total of 24 ADWs inventoried in the vicinity,
em located on the north and west border of the sampling
id have a ground pit (half-circle) in the southwest
had no ADWs within the sampling area, with 13 ADWs in
ne mile west of the area. Also shown in Fig. V-1 are
edrock (ranging from less than 7.5 m to greater than 30 m)
ithin areas 1, 2, and 3. As indicated, a large portion

Table V-1. Farm home water supply well characteristics.

Area	Age* yr	Depth of well (m)	Depth to water (m)	% Drilled†	% Grouted
1-avg. (range; S.D.)	47.5 (2-93; 23.7)	40.6 (9.1-121.9; 24.2)	21.4 (1.8-76.2; 16.2)	88%	15%
2-avg. (range; S.D.)	35.6 (1-103; 24.6)	42.8 (9.1-152.4; 27.6)	26.5 (2.4-115.8; 25.2)	98%	35%
3-avg. (range; S.D.)	44.7 (3-83; 24.7)	36.6 (13.7-76.2; 15.0)	18.9 (5.5-51.8; 11.7)	100%	21%

*To 1983; data on well characteristics were available for about 70% of the wells sampled.

†Wells less than 30 cm (12 inches) in diameter were assumed to be drilled.

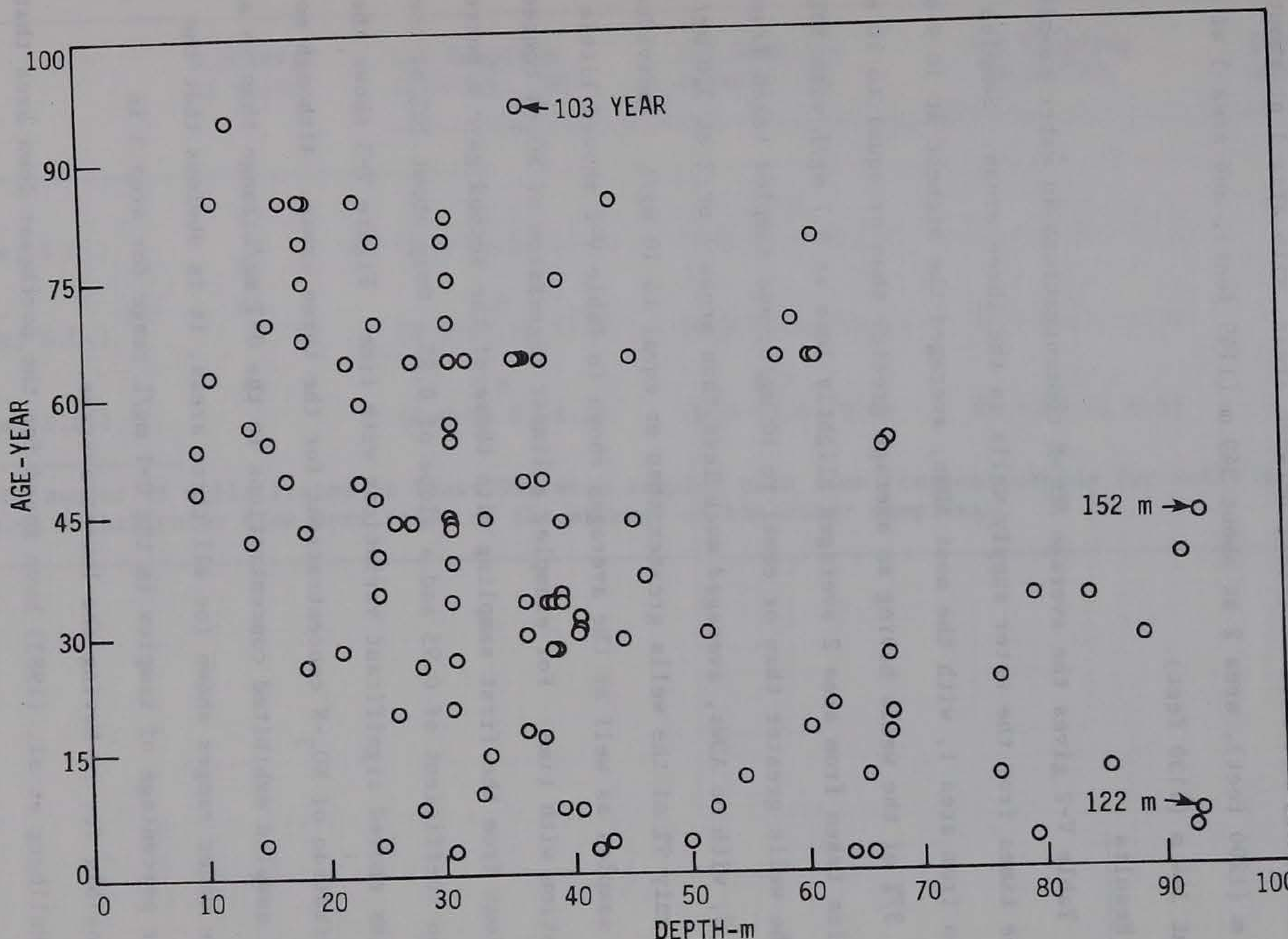

Figure V-2. Relationship between age and depth of farm water supply wells sampled in the three areas.

show a wider diversity within areas 1 and 2. The average land surface elevation was not much different between areas, with area 1 at about 366 m (1200 feet), area 2 at about 349 m (1145 feet), and area 3 at about 344 m (1130 feet).

2. Results

Table V-2 gives the average NO_3-N concentrations in water sampled three times from the water supply wells in the three areas. Samples taken from area 1, with the most ADWs, averaged the highest at 10.9 mg/L with 37% of the wells having an average greater than or equal to 10 mg/L; samples taken from area 2 averaged slightly less at 8.7 mg/L with 30% of the wells greater than or equal to 10 mg/L; and samples taken from area 3, with no ADWs, averaged much less than areas 1 or 2 at 3.0 mg/L with only 9% of the wells greater than or equal to 10 mg/L. Individual well samples as well as the averages shown in Table V-2 showed little variation with time. For example, a linear regression of NO_3-N concentrations from the first sampling with those of the second gave a correlation coefficient of 0.93 and a slope of 0.85. Only about 10% of the samples showed significant variations with time. Figure V-3 shows the distribution of NO_3-N concentrations for the three areas. Although more wells sampled exhibited concentrations in the 0-1 mg/L range than in any of the other ranges shown for all three areas, it is obvious that the larger percentage of samples in the 0-1 mg/L range for area 3 is responsible for it having the lowest average.

Hallberg et al. (1983) have found for the northeast Iowa area that where soil materials cover the aquifer by 15 m (50 feet) or more, infiltration of surface water percolating down from above is reduced. As a

Table V-2. NO_3-N concentrations in samples from farm home water supply wells.

Area	Number* of wells	1st Sampling July-Aug. '82	2nd Sampling January-'83	3rd Sampling July-'83	Overall average
		------mg/L------			
1-avg. (range; S.D.)	47	11.0 (0.0-93.6; 17.8)	10.8 (0.0-61.3; 15.2)	10.8 (0.0-60.8; 14.2)	10.9
2-avg. (range; S.D.)	66	7.8 (0.0-69.6; 13.8)	9.2 (0.0-82.1; 15.9)	9.2 (0.0-63.7; 13.9)	8.7
3-avg. (range, S.D.)	57	3.0 (0.0-39.5; 8.2)	2.7 (0.0-43.6; 8.4)	3.3 (0.0-47.5; 8.5)	3.0

*Of the number of wells listed at least 95% were sampled each time; concentrations for wells not sampled a particular time were taken as the average of the other samplings.

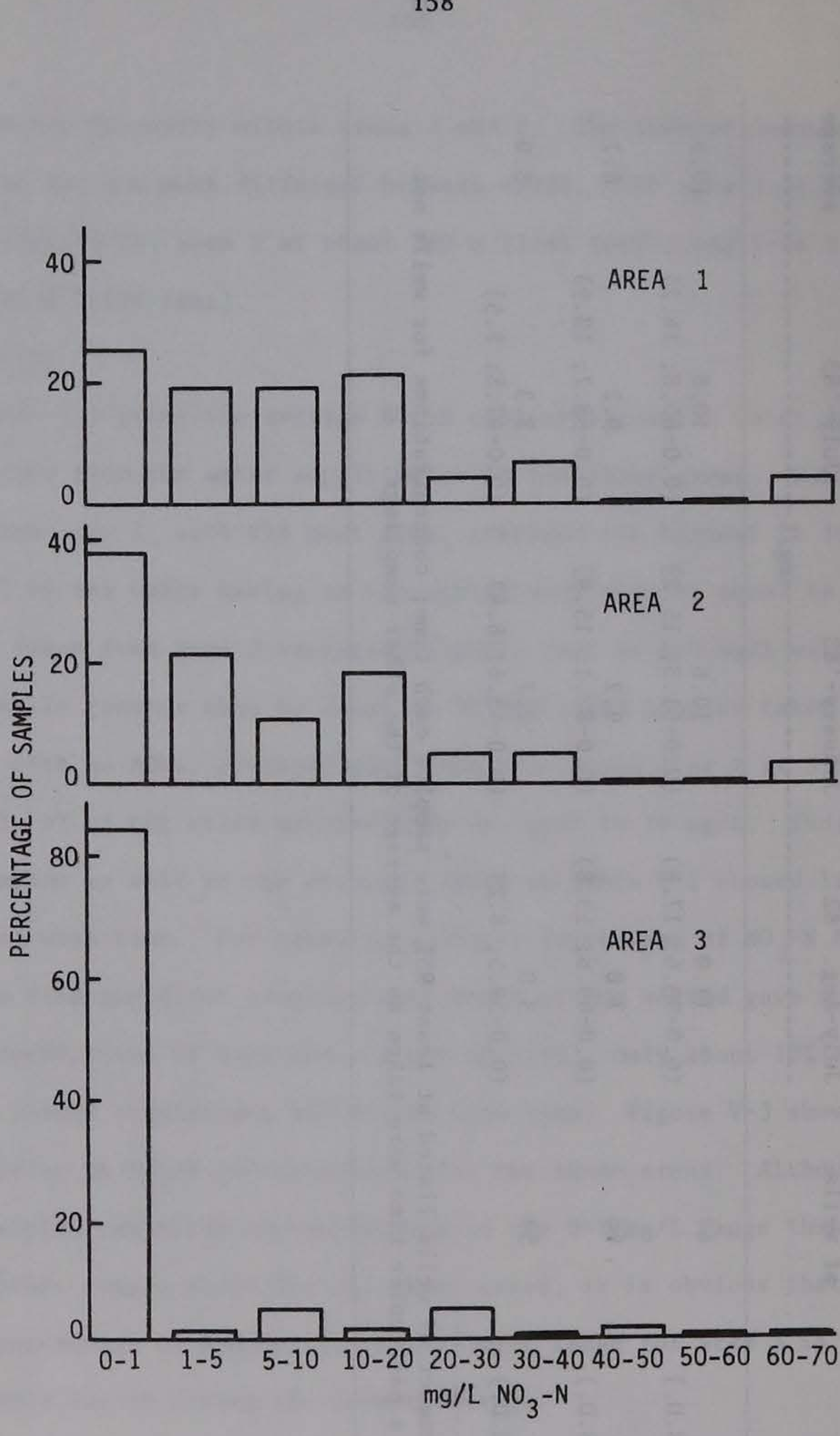

Figure V-3. Distribution of NO_3-N concentrations.

result, NO_3-N concentrations in the groundwater are nearly zero, and other surface contaminates are very low. In areas where the soil materials are less than 15 m thick, rainwater easily percolates through the soils carrying dissolved materials, with the thinner the soil layer, the greater the water movement. In aquifers under thin surface soil layers, NO_3-N contamination has become a significant problem.

Therefore, it is of interest to classify the NO_3-N contamination of the farm water supply wells by the depth-to-bedrock (or overburden) data shown in Fig. V-1. Average NO_3-N concentrations for the three areas for those wells with less than 15 m overburden and for those wells with greater than or equal to 15 m overburden are given in Table V-3. It is evident for all three areas where depth-to-bedrock is less than 15 m that NO_3-N contamination is a problem. Statistically, there were no significant differences between any of the areas for wells with less than 15 m overburden. For areas 1 and 2 in particular and possibly for area 3, direct surface contamination in addition to the influence of recharge from ADWs could cause the high NO_3-N concentrations measured. However, for areas where depth-to-bedrock is greater than or equal to 15 m, direct surface contamination is not believed to be likely. Hence, it would have to be concluded that the higher average NO_3-N concentration for areas 1 and 2, relative to the low value for area 3, are directly due to the influence of recharge from ADWs. For wells with more than a 15-m overburden there was a statistically significant difference between the average for area 1 and the average for area 3 (at the 1% level), and between the averages for area 2 and area 3 (at the 5% level); there was no significant difference between areas 1 and 2.

Table V-3. NO_3-N concentrations in farm water supply wells as affected by depth-to-bedrock.

Depth-to-bedrock	Area 1	Area 2	Area 3
	--------------------	mg/L	--------------------
<15 m	13.4	11.7	20.3
$\geq$ 15 m	10.4	6.4	1.3

Figure V-1 shows the location of ADWs relative to the location and average NO_3-N concentrations of farm water supply wells sampled (the water supply well is located at the position of the average concentration shown for it). From the preceding discussion and in appraising the locations of the ADWs and the NO_3-N concentrations for the three areas, it would have to be concluded that the ADWs are affecting the NO_3-N levels of some wells. However, from the knowledge of NO_3-N concentrations in agricultural drainage monitored in this study (discussed in Section II) and found in other studies (discussed in Section I), it is believed that any well averaging over 25 mg/L is likely also experiencing contamination from local high level sources from past disposal of organic wastes (e.g., feedlots, septic fields, etc.) in addition to possible effects from ADWs. The six wells in area 1 (all in the lower part) that averaged more than 25 mg/L were shallow wells and/or had high water levels. The data available for five of the wells showed a depth-to-water average of only 5 m. Three of the wells were apparently finished above bedrock because they were in an area where depth-to-bedrock was believed to be 15 m or more. Figure V-4 shows an inverse

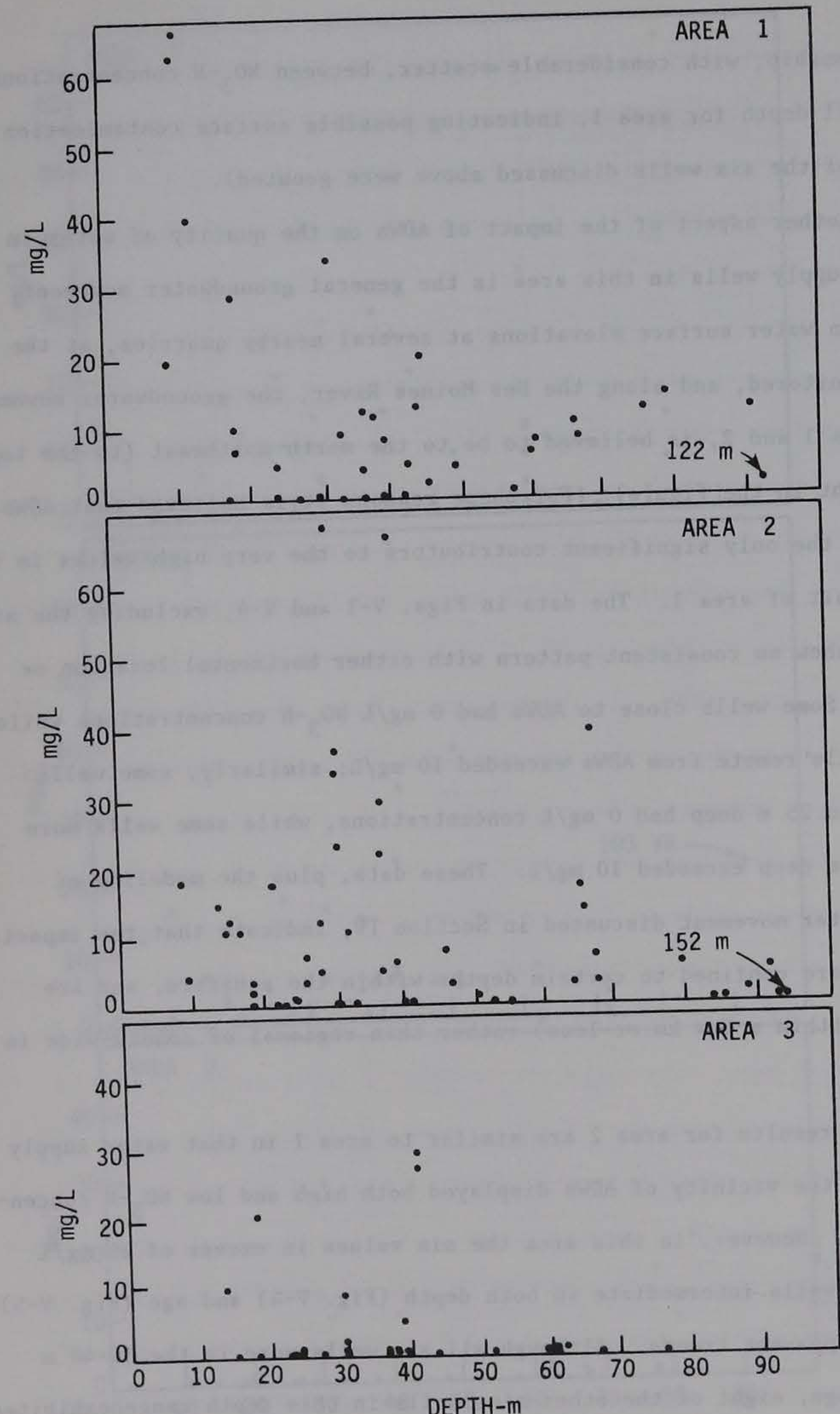

Figure V-4. Relationships between NO_3-N concentrations and depth for water supply wells sampled in the three areas.

relationship, with considerable scatter, between NO_3-N concentrations and well depth for area 1, indicating possible surface contamination (none of the six wells discussed above were grouted).

Another aspect of the impact of ADWs on the quality of water in water supply wells in this area is the general groundwater movement. Based on water surface elevations at several nearby quarries, at the 4 ADWs monitored, and along the Des Moines River, the groundwater movement, in Areas 1 and 2, is believed to be to the north-northeast (to the top and right in the figure). For these reasons it is believed that ADWs are not the only significant contributors to the very high values in the lower part of area 1. The data in Figs. V-1 and V-4, excluding the six wells, show no consistent pattern with either horizontal location or depth. Some wells close to ADWs had 0 mg/L NO_3-N concentrations while some wells remote from ADWs exceeded 10 mg/L; similarly, some wells less than 25 m deep had 0 mg/L concentrations, while some wells more than 75 m deep exceeded 10 mg/L. These data, plus the modeling of groundwater movement discussed in Section IV, indicate that the impacts of ADWs are confined to certain depths within the aquifers, and are local (within a few km or less) rather than regional or county-wide in nature.

The results for area 2 are similar to area 1 in that water supply wells in the vicinity of ADWs displayed both high and low NO_3-N concentrations. However, in this area the six values in excess of 25 mg/L were for wells intermediate in both depth (Fig. V-4) and age (Fig. V-5), with no apparent trends. Although all six wells were in the 30-40 m depth range, eight of the other nine wells in this depth range exhibited

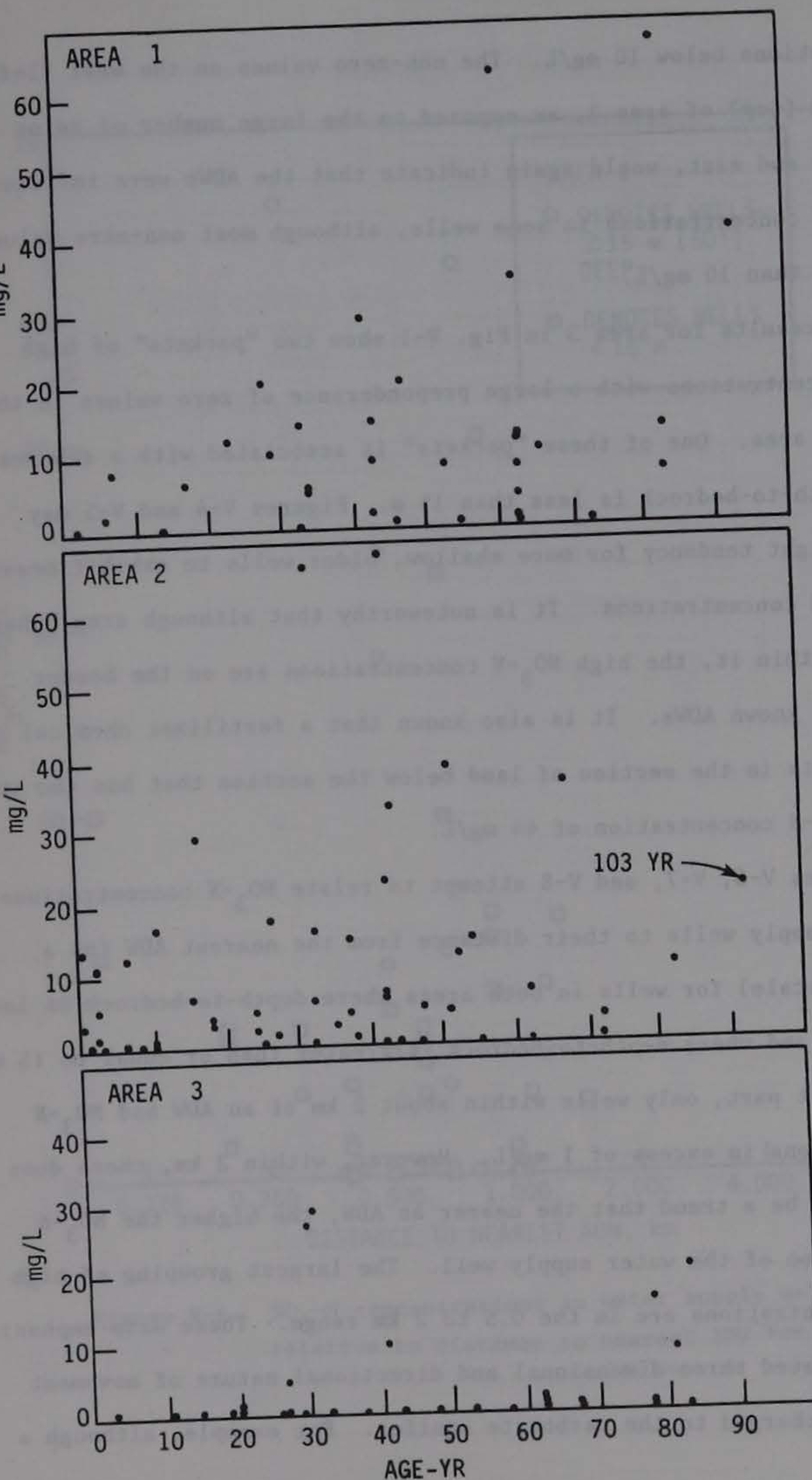

Figure V-5. Relationships between NO_3-N concentrations and age for farm water supply wells sampled in the three areas.

concentrations below 10 mg/L. The non-zero values on the west (left) and north (top) of area 2, as opposed to the large number of zeros to the south and east, would again indicate that the ADWs were influencing the NO_3-N concentrations in some wells, although most non-zero values were less than 10 mg/L.

The results for area 3 in Fig. V-1 show two "pockets" of high NO_3-N concentrations with a large preponderance of zero values in the remaining area. One of these "pockets" is associated with a subarea where depth-to-bedrock is less than 15 m. Figures V-4 and V-5 may show a slight tendency for more shallow, older wells to exhibit measurable NO_3-N concentrations. It is noteworthy that although area 3 has no ADWs within it, the high NO_3-N concentrations are on the border closest to known ADWs. It is also known that a fertilizer chemical plant exists in the section of land below the section that has the well with a NO_3-N concentration of 44 mg/L.

Figures V-6, V-7, and V-8 attempt to relate NO_3-N concentrations of water supply wells to their distance from the nearest ADW (on a logarithm scale) for wells in both areas where depth-to-bedrock is less than 15 m, and where depth-to-bedrock is greater than or equal to 15 m. For the most part, only wells within about 2 km of an ADW had NO_3-N concentrations in excess of 1 mg/L. However, within 2 km, there does not seem to be a trend that the nearer an ADW, the higher the NO_3-N concentration of the water supply well. The largest grouping of high NO_3-N concentrations are in the 0.5 to 2 km range. These data emphasize the complicated three-dimensional and directional nature of movement of water recharged to the carbonate aquifer. For example, although a

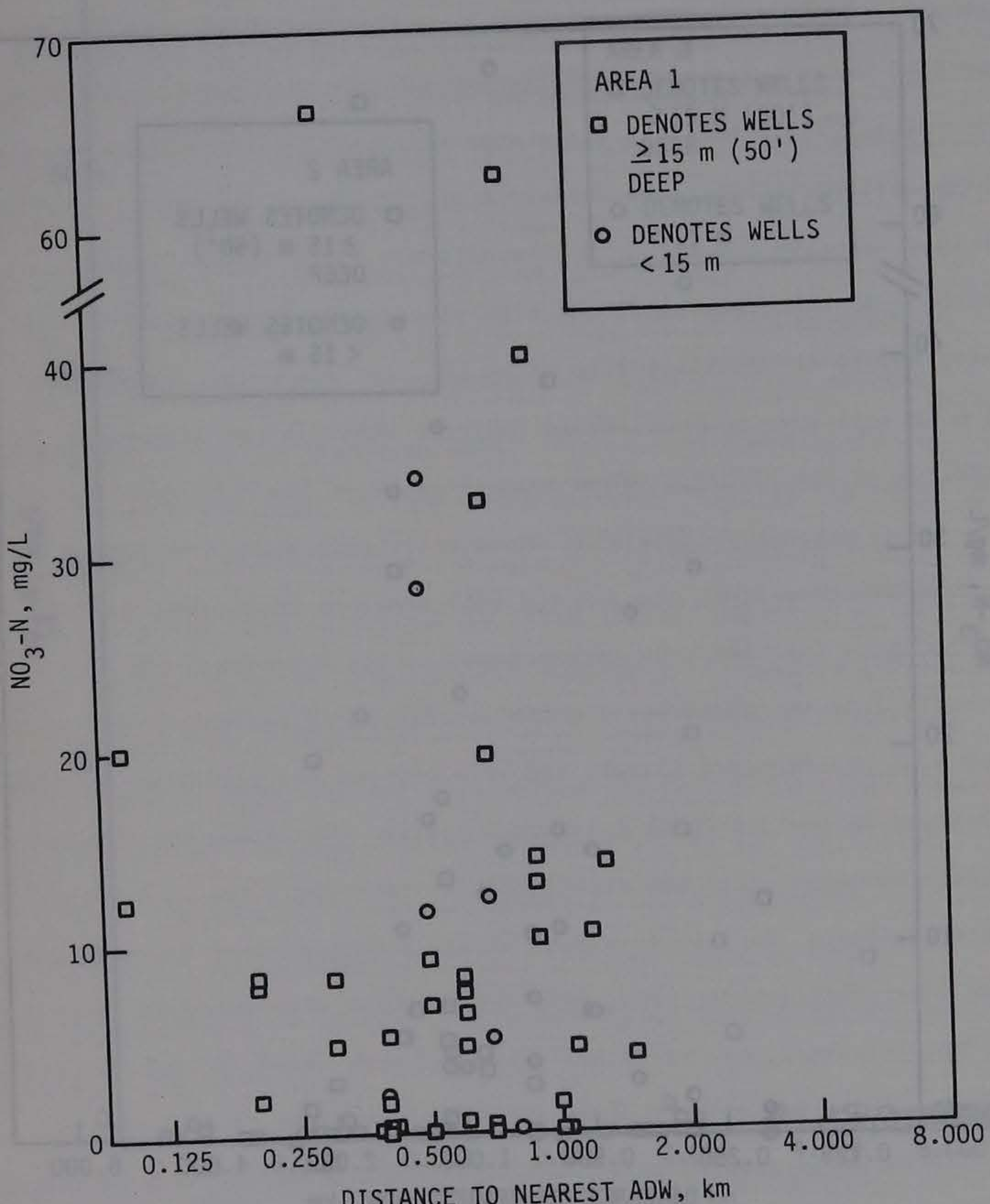

Figure V-6. NO_3-N concentrations in water supply wells relative to distance to nearest ADW for area 1.

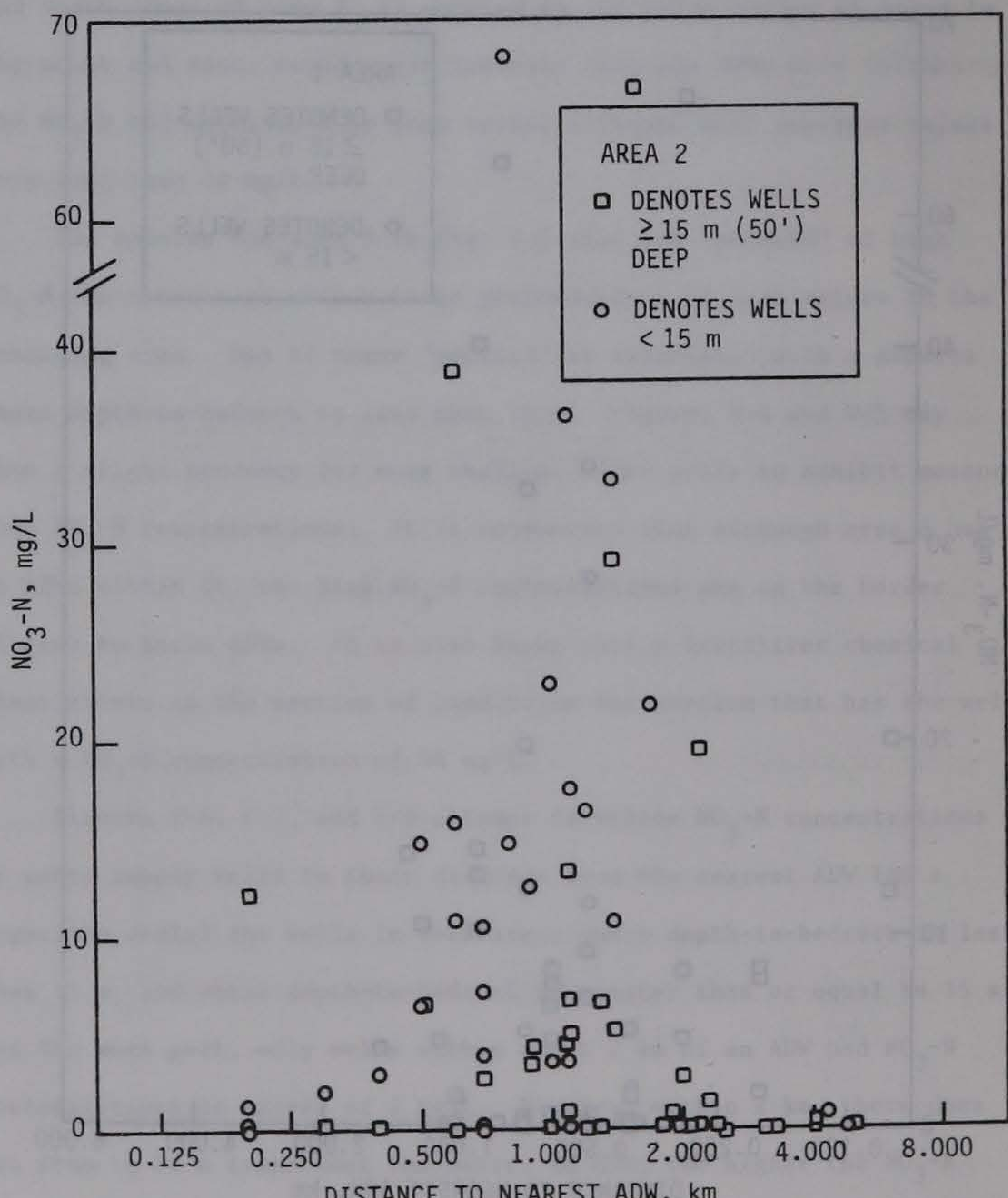

Figure V-7. NO_3-N concentrations in water supply wells relative to distance to nearest ADW for area 2.

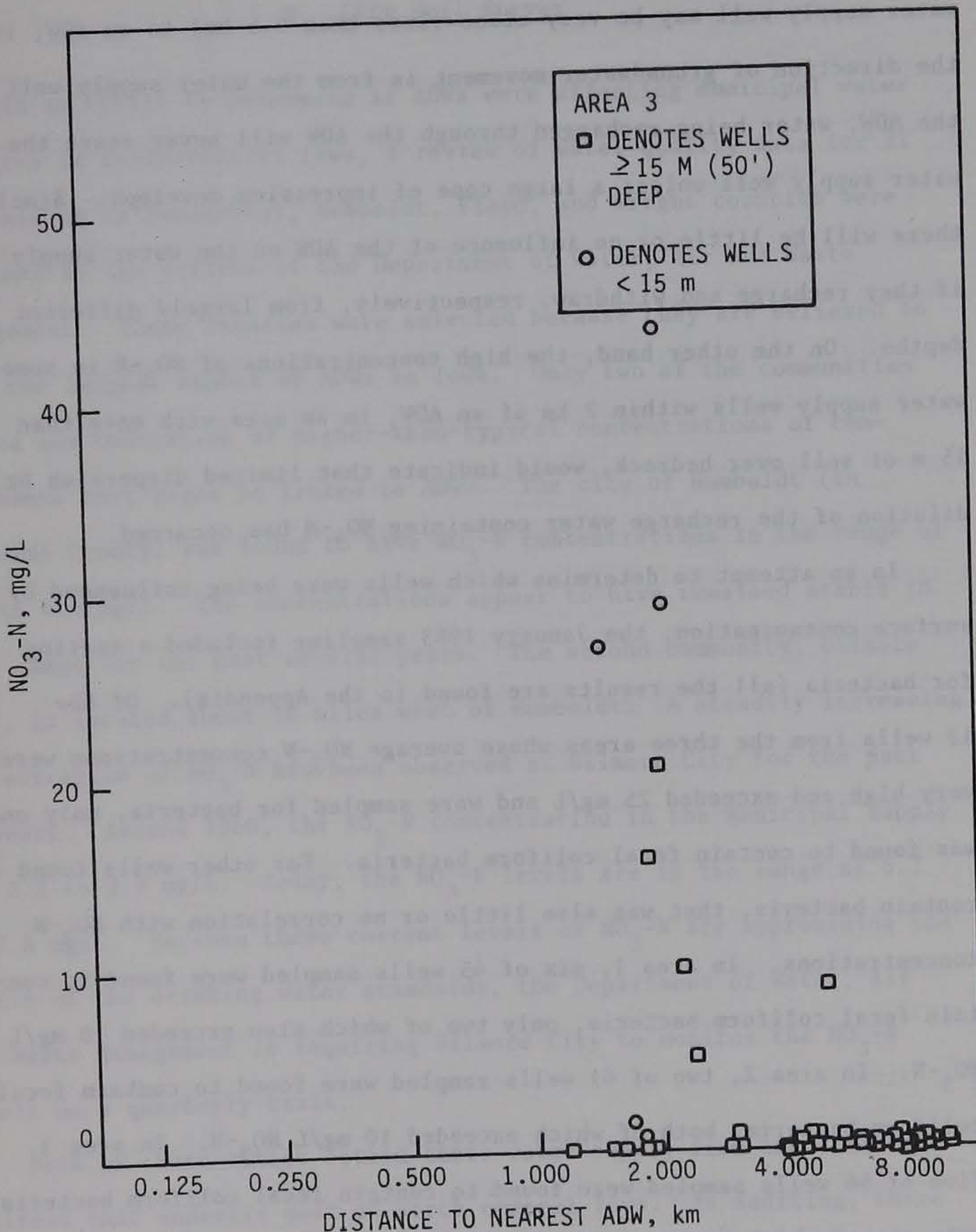

Figure V-8. NO_3-N concentrations in water supply wells relative to distance to nearest ADW for area 3.

water supply well may be very close (less than 0.5 km) to an ADW, if the direction of groundwater movement is from the water supply well to the ADW, water being recharged through the ADW will never reach the water supply well unless a large cone of impression develops. Similarly, there will be little or no influence of the ADW on the water supply well if they recharge and withdraw, respectively, from largely different depths. On the other hand, the high concentrations of NO_3-N in some water supply wells within 2 km of an ADW, in an area with more than 15 m of soil over bedrock, would indicate that limited dispersion or dilution of the recharge water containing NO_3-N has occurred.

In an attempt to determine which wells were being influenced by surface contamination, the January 1983 sampling included a testing for bacteria (all the results are found in the Appendix). Of the 13 wells from the three areas whose average NO_3-N concentrations were very high and exceeded 25 mg/L and were sampled for bacteria, only one was found to contain fecal coliform bacteria. For other wells found to contain bacteria, that was also little or no correlation with NO_3-N concentrations. In area 1, six of 45 wells sampled were found to contain fecal coliform bacteria, only two of which also exceeded 10 mg/L NO_3-N. In area 2, two of 61 wells sampled were found to contain fecal coliform bacteria, both of which exceeded 10 mg/L NO_3-N. In area 3, five of 56 wells sampled were found to contain fecal coliform bacteria, only one of which exceeded 10 mg/L NO_3-N.

B. City Well Survey

In an effort to determine if ADWs were affecting municipal water supplies in north-central Iowa, a review of water quality data for 21 communities in Pocahontas, Humboldt, Floyd, and Wright counties were examined at the offices of the Department of Water, Air and Waste Management. These counties were selected because they are believed to have the largest number of ADWs in Iowa. Only two of the communities showed any indication of higher-than-typical concentrations of constituents that might be linked to ADWs. The city of Humboldt (in Humboldt County) was found to have NO_3-N concentrations in the range of 2.2 to 4.4 mg/L. The concentrations appear to have remained stable in this range for the past several years. The second community, Gilmore City, is located about 10 miles west of Humboldt. A steadily increasing concentration of NO_3-N has been observed at Gilmore City for the past 20 years. Around 1960, the NO_3-N concentration in the municipal supply was 2.2 to 3.3 mg/L. Today, the NO_3-N levels are in the range of 6.7 to 7.8 mg/L. Because these current levels of NO_3-N are approaching the limits of the drinking water standards, the Department of Water, Air and Waste Management is requiring Gilmore City to monitor the NO_3-N levels on a quarterly basis.

Both of these cities obtain their water supply from the limestone aquifers that underlie most of north central Iowa. In addition, there are numerous ADWs around these cities that recharge into the limestone aquifers. The water supply well for Gilmore City is 63 m deep with the static water level around 15 m below ground surface. In an interview with the water superintendent of Gilmore City, he stated that he believed the municipal well was finished in a "limestone cavern underground." This is the same limestone formation that is mined at several locations northwest of Gilmore City. In addition, there are several abandoned limestone quarries in the area that would allow pollutants direct access into the aquifer.

Numerous ADWs exist five to ten miles northwest of Gilmore City. These are beyond the limestone quarries. The water superintendent stated that the area was originally a marshland and the ADWs were the most inexpensive measure to turn the marshlands into useable farmland. A survey of the water levels in the ADWs being monitored between Gilmore City and Humboldt, the water levels in the quarries northwest of Gilmore City, and the water level in the Gilmore City well, allowed the piezometric surface of the limestone aquifer to be approximated. This analysis showed the groundwater flow in the Gilmore City area to be to the north-northeast, generally toward the Des Moines River. Upon careful evaluation of the maps and aerial photos of the area, it appears that the NO_3-N levels in the Gilmore City easily could be related to the city's wastewater lagoon, instead of the ADWs. There were no ADWs found to the southwest of the city, and the wastewater lagoon was constructed in the mid 1960s, which corresponds roughly with the beginning of the increased levels of NO_3-N in the city's water supply. This information has been communicated to the Gilmore City council, and an engineering consultant has been retained to investigate this problem further. It is possible that the increasing NO_3-N levels could also be due to direct charge from the surface that is being influenced by increased fertilization.

In conclusion, there was no evidence from the municipal water supply water quality data of an increasing level of NO_3-N in the aquifer that could be related to ADWs. It must be recognized that the data were very sparse, particularly prior to about 1960. This makes conclusions concerning trends difficult. However, there were no municipalities in the counties surveyed where the NO_3-N levels exceeded the drinking water standards. The only community with an increasing NO_3-N level was Gilmore City; however, it is believed this increase is not due to ADWs.

C. Reference

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VI. ASSESSMENT OF ADWs' IMPACT ON GROUNDWATER QUALITY

A. NO_3-N

The current domestic water supply criterion for NO_3-N has been set at 10 mg/L (EPA, 1976). The rationale is that high intakes of nitrates are a hazard to man under conditions that favor nitrate reduction to nitrite. This can happen in the gastrointestinal tract; the nitrite formed then reaches the bloodstream and reacts with hemoglobin to produce methemoglobin which results in impairment of oxygen transport. This is particularly hazardous in infants under three months of age.

There is some potential for the presence of high levels of NO_3-N in water to enhance the formation of a family of organic compounds known as nitrosamines, some of which are known to be strongly carcinogenic. If this potential tie is proven or other unforseen problems arise, the NO_3-N standard would likely be revised downward; therefore, the standard is subject to change.

Monitoring of drainage water to ADWs, two of which are believed to take surface runoff, definitely showed that this water often exceeded the 10 mg/L NO_3-N standard (as shown in Table VI-1, NO_3-N concentrations in all samples averaged 15.9 mg/L, with 85% of them exceeding 10 mg/L). This was not surprising as a review of earlier work in Iowa (Table I-6) showed that the average NO_3-N concentration in subsurface flow from corn and soybean fields was 19.0 mg/L. Surface runoff averaged 2.6 mg/L (Table I-5). Table VI-1 also shows that a tile line monitored in the Big Spring study averaged 16.0 mg NO_3-N/L.

Table VI-1. Average nutrient and solids concentrations in various water sources.

Source	NH_4-N	NO_3-N	PO_4-P	Cl	DS	SS
	--------------------	--------------------	mg/L	--------------------		
ADWs	.18	15.9	.19	28.2	458	385
Des Moines River*	.20	8.3	.09	37.2	457	310
Tile line (108)†		16.0				
Big Spring		8.9				
Turkey River		6.2				

*Average of up to 47 samples taken at the Boone sampling station from April 1981 through June 1982 (Baumann et al., 1982, 1983).

†Data from Big Spring study for October 1981 through December 1982 (Hallberg et al., 1983).

The quality of total drainage, including surface runoff from row-crop land as well as drainage from non-row-crop land, is represented by the values in Table VI-1 for the Des Moines river for the ADW study area, and by the values for Big Spring and the Turkey River for the Big Spring study area. It is evident that the major difference in quality between total drainage and drainage to ADWs, or subsurface drainage, with respect to nutrients and solids was for NO_3-N, but total drainage still averaged about one-half the NO_3-N concentrations and sometimes exceeded 10 mg/L. Therefore, aquifers interchanging water directly with these rivers could be expected at times to have significant NO_3-N levels.

Modeling of the quality of water entering ADWs, using the USDA's CREAMS model and the ISU drainage model and historical weather data as discussed in Section III, predicted that the average NO_3-N concentrations in total drainage (surface plus subsurface flow) from corn ground fertilized at 150 kg N/ha would be between 13 and 18 mg/L. It was estimated that decreasing the rate to 75 kg/ha would decrease concentrations 48% (to below 10 mg/L), but that increasing the rate to 225 kg/ha would increase concentrations 57%.

A survey of the quality of water from farm water supply wells in three areas (three times) indicated that in several cases the presence of ADWs in the vicinity (within 2 km or less) resulted in elevated NO_3-N concentrations (see Section V). The fact that a large number of water supply wells with elevated concentrations, and near ADWs, were in areas that had 15 m or more overburden is strong evidence that the NO_3-N in recharge to the ADWs, and not surface infiltration, was increasing the NO_3-N concentrations in the aquifer.

Modeling of the predicted areal influence of an ADW under expected drainage conditions (see Section IV) indicated that the impact of an ADW would be localized (within a few km) and directional. There was a general trend toward the more shallow water supply wells having the highest NO_3-N concentrations. There were several cases where one water supply well in the vicinity of ADWs would exceed 10 mg/L NO_3-N, while another water supply well within one-half mile (0.8 km) of the first would have NO_3-N concentrations less than 0.5 mg/L. Whether this is due to the directional influence of ADWs or to a well depth factor (it is likely that recharge water from ADWs is layered over existing water in the aquifer because of the likely higher porosity of the surface layers of limestone), it indicates that the adverse influence of the ADWs should not be regional (county-wide) in nature as long as the whole region does not have ADWs. Overall, the current policy of not permitting more ADWs would seem appropriate.

B. Pesticides

Currently, domestic water supply criteria have been set for only a limited number of pesticides (EPA, 1976). They are: 100 ppb for 2,4-D; 10 ppb for 2,4,5-TP; 100 ppb for methoxychlor; 5 ppb for toxaphene; 4 ppb for lindane, and 0.2 ppb for endrin. One ppb equals 1 μg/L. For others such as aldrin/dieldrin, chlordane, DDT, and heptachlor, the EPA suggests that their persistence, bioaccumulation potential, and carcinogenicity caution human exposure to a minimum. Unfortunately, criteria have not been established for any of the six most used insecticides in Iowa and have been for only one (2,4-D) of the 13 most-used herbicides in Iowa (Table I-2).

Because there is a need to establish guidelines for residues of these and numerous other pesticides in drinking water, the hazard evaluation division of EPA's Office of Pesticide Programs has announced plans to establish maximum advisable levels (MAL) for pesticides in groundwater as discussed in Section I. Acceptable daily intake (ADI) values would be used to calculate MALs, which are given in Table I-4 for pesticides of interest to Iowa. ADI values for pesticides not listed (e.g., butylate and metribuzin) are yet to be released; there is also the possibility that some of those given in Table I-4 may be revised (particularily for alachlor).

Table VI-2 presents the average pesticide concentrations measured for water draining to ADWs. It is evident that relative to at least 2,4-D, (and atrazine and alachlor as well, if the EPA adopt their suggested guidelines for establishing maximum advisable levels and the numbers in Table I-4 are used) that water draining to ADWs had average concentrations at least 1000 times less than the 100 ppb, 215 ppb, and 1000 ppb values for 2,4-D, atrazine, and alachlor, respectively. Furthermore, the maximum concentration of 0.4, 0.5, and 55 ppb for 2,4-D, atrazine, and alachlor in water draining to ADWs was at least 18 times less than proposed or established criteria. However for dicamba, the average concentration was only 40 times less than the 12.5 ppb proposed value, with a maximum value of 12.0 ppb measured. Because water from the aquifers recharged in part by ADWs will have time to mix with previously recharged water plus water from other recharge areas (see

Table VI-2. Average pesticide concentrations in various water sources.

Source	Alachlor	Atrazine	Cyanazine	Dicamba	2,4-D	Metribuzin	Dieldrin
	ppb						
ADWs	0.863	0.018	3.320	0.305	0.001	0.015	0.002
Des Moines River*	0.430	0.360	0.855	0.185	0.050	0.075	0.002
Tile Line (108)†	0.018	0.494	0				
Big Spring	0.008	0.360	0.019				
Turkey River	5.140	10.400	1.310				

*Average of two samples taken at the Boone sampling station in June 1981 and July 1982 (Baumann et al., 1982, 1983).

†Data from Big Spring study for October 1981 through December 1982 (Halberg et al., 1983).

Section IV for discussion of modeling of pollutant movement in the aquifer), flow-weighted average concentrations rather than maximum concentrations in any one sample may provide a better comparison with the criteria.

The values for the ADWs in Table VI-2 are arithmetic averages with 69% of the samples taken in May and June, with extra samples taken during surface runoff and high flows, both times of expected maximum pesticide concentrations and losses. It is known from other studies (Baker et al., 1978; Johnson and Baker, 1982) that a large portion of agricultural drainage in Iowa occurs during snowmelt and early spring (prior to May 1 and pesticide application) and in the fall, at times when precipitation commonly exceeds evapotranspiration. Modeling (Section III) shows that the May-June periods account for about 23% of the total annual surface runoff and about 46% of the annual subsurface flow. Gauging of the Des Moines River (USGS, 1981) shows that May and June contribute 32% of the long-term average annual flow. Thus, if there is any difference between flow-weighted average concentrations and the arithmetic averages of Table VI-2, the arithmetic averages given are probably higher.

Considering that corn and soybeans were rotated in 1981 and 1982 in at least part of the areas for which the four monitored ADWs provided drainage outlets, and considering the measured pesticide concentrations, it appeared that alachlor, cyanazine, and dicamba were the major pesticides used with little or no atrazine or 2,4-D applied in those areas. Pesticide concentrations for the Des Moines River, which integrates both surface runoff and subsurface flow from a much larger area (including all of Humboldt County, most of Pocahontas County, and several other counties) that would receive more representative application of pesticides are therefore of interest and are given in Table IV-2 (Baumann et al., 1982, 1983). Concentrations of atrazine and 2,4-D in the Des Moines River were higher than for the ADWs, but concentrations for alachlor, cyanazine, and dicamba were all lower. These average values are at least 500 times below proposed or established criteria for 2,4-D, alachlor, and atrazine and 67 times lower for dicamba.

Average values from the Big Spring study area of northeast Iowa (Hallberg et al., 1983) are also given for comparison in Table VI-2. Water from a monitored subsurface tile line and from Big Spring also contained atrazine and alachlor at levels at least 500 times below proposed criteria. Four samples of water in the Turkey River, three of which were taken in May and June, had higher average values, but were still more than 20 times below proposed criteria. The maximum concentrations for atrazine and alachlor (for a sample taken June 7-8, 1982) were 37 and 20 ppb, respectively. Cyanazine (5 ppb), fonofos (1.6 ppb), and carbofuran (0.36 ppb) were also found in that sample.

Modeling, using 23 years of precipitation records, and assuming yearly application of atrazine and alachlor to continuous corn, showed that atrazine concentrations in total drainage (surface runoff plus subsurface flow) would be 9.25 ppb; for alachlor, 3.35 ppb. These values, like measured concentrations, are at least 20 times lower than the proposed criteria for these herbicides.

In summary, none of the samples of water draining to the ADWs (or from the Des Moines River or Big Spring sampling sites) were found at any time to contain pesticides in excess of proposed or established drinking water criteria. In addition, average concentrations, which should provide a better basis from which to make an impact judgment than maximum concentrations, were many times lower than maximum concentrations. Modeling also indicated low (below criteria) pesticide concentrations in agricultural drainage. Therefore, unless new criteria established for pesticides not now covered (e.g., for cyanazine, metribuzin, fonofos, and carbofuran) are exceeded, or currently proposed or established criteria are revised markedly downward, the pesticides in drainage to ADWs should not present a known health hazard.

C. Bacteria

The accurate determination of the presence of pathogenic (disease-causing) bacteria in water is a difficult and time-consuming process. Therefore, a group of non-pathogenic bacteria that are found in the intestines of all warm blooded animals, including man, is used as an indicator of possible pollution. Coliform bacteria are used frequently as indicators; however, coliform bacteria are also found in many soils so their presence in a water sample does not necessarily indicate a contamination due to fecal matter. Current water quality standards call for no more than 1 total coliform colony per 100 ml of water.

The presence of pathogenic bacteria and viruses could pose a potential health hazard. Viruses are difficult to isolate and identify so coliform bacteria will be used as an indicator. Until recently, groundwater, particularly deep groundwater, was thought to be free of bacteria and virsues. However, careful investigation has shown the subsurface groundwater regions are not totally hostile to microbial life (McNabb and Dunlap, 1974). The correlation between the presence of coliform bacteria and pathogenic bacteria in surface water has been established, but because of the different environmental conditions in the groundwater system, the use of coliform bacteria as an indicator may not be adequate. The use of fecal streptococcus organisms may give a more reliable indication of the presence of pathogenic bacteria in groundwaters. This test determines the presences of certain fecal bacteria (fecal streptococcus) that are more closely linked to animal enteric pathogens (Kabler and Clark, 1960).

Total coliform, fecal coliform and fecal streptococcus bacteria analyses were conducted on the recharge water at the four ADWs monitored in this study. Very high total coliform bacteria counts were found in all ADWs at some time during the monitoring. Those ADWs that receive surface water directly or through macropores exhibited much higher total coliform counts. In general, the high total coliform counts were accompanied by high fecal coliform and fecal streptococcus. Geldreich (1966) found that the ratio of the fecal coliform bacteria to fecal streptococcus bacteria was a good indicator of the source of the bacteria. For fecal matter from man, the ratio of fecal coliform to fecal streptococcus was found to be about 4.3; for animal waste, the ratio was found to be less than one. Most of the data from the ADWs showed ratios of fecal coliform to fecal streptococcus to be less than one, indicating the bacteria were from an animal waste source rather than human waste.

Samples from the farm wells were also analyzed for bacteria. The results were correlated with several well parameters, without success. At best the data were inconclusive in establishing a trend in the bacteria data. A fairly high percentage of the wells showed fecal streptococcus bacteria but did not show high levels of total coliform or fecal coliform bacteria. This was not expected and cannot be fully explained.

ADWs permit the direct recharge of waters with high bacteria levels, thus bypassing the natural filtration that occurs in the soil. Most bacteria are removed if the water is filtered through a few feet of soil. The fate and transport of bacteria in the groundwater system in north-central Iowa is unknown. The farm well survey showed increased levels of nitrate nitrogen in the areas of higher ADW concentration but the bacteria sampling did not show the same trends. In fact, there was no significant correlation between the nitrate nitrogen levels in the wells and the bacteria levels. This could be an indication of a rapid die-off of the bacteria that are entering the aquifer from the ADWs. This is consistent with many investigations that have found the groundwater not to be a good environment for bacteria. This result does not indicate a need for less concern about bacteria because virus and some bacteria spores can survive for long periods of time in an inactive state, only to begin to grow under better environmental conditions. The elimination of direct surface runoff entering the ADWs and the prevention of all septic tank effluents from entering ADWs would help eliminate the potential for injection of pathogenic bacteria and viruses.

D. References

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VII. POSSIBLE CONTROL ALTERNATIVES

From field surveys, it has been found that a majority of the large number of agricultural drainage wells (ADWs) located in Humboldt and Pocahontas counties are taking surface runoff directly. These ADWs are injecting surface and subsurface drainage water into the underlying aquifers that are used for drinking water supplies, mostly by rural communities. Two major problems associated with the continued use of ADWs are: (1) sediments, pesticides, and bacteria entering some of these wells along with surface runoff, and (2) subsurface drainage water, containing high NO_3-N levels, entering into the ADWs. From the water quality data taken from the farm water supply wells in the area, evidence has been found to relate the use of ADWs to higher NO_3-N levels in some of the groundwater near the ADWs. Although the evidence relating the deterioration of local groundwater quality to these wells may only be circumstantial at this stage, the potential for deterioration of water supplies certainly does exist. Therefore, possible control alternatives to control the direct injection of surface as well as subsurface waters into the groundwater systems of the region have been considered in case the determination is made that control is necessary.

One of the obvious alternatives is to eliminate the use of these wells. If this alternative is to be considered, then other drainage outlets would have to be provided or large reductions in crop yields would result. Open ditches and/or tile drains could be used as possible means to carry water to the closest natural outlet, i.e., a stream or river. Therefore, the details of these means were considered to determine if they are economically feasible. In cases where it is necessary to have very deep ditches or trenches (which are expensive) to provide drainage by gravity, the use of pumped drainage was considered.

The cost in terms of crop reduction of eliminating the use of ADWs without alternative drainage was also considered. Also, there is a possibility of reducing the volume of water draining to the wells through controlled drainage. In addition to these alternatives, there are a few other options such as local land treatment, chemical management, and modification of the construction of wells, which will be discussed.

A. Cost of Alternative Drainage

To make the comprehensive economic feasibility study of alternative drainage outlets, topographic maps showing contours at 10-foot intervals were purchased from the USGS, Denver, Colorado for the areas in and around Humboldt and Pocahontas counties. An inventory of more than 50 ADWs in Humboldt and Pocahontas county was completed by physically locating these wells in a field survey. Approximate site locations of all these ADWs were located on the topographic maps. Figure VII-1 shows the locations of ADWs in Pocahontas county, and Figures VII-2, VII-3, and VII-4 show the approximate site locations of ADWs in Humboldt county. These site locations on topographic maps were used to determine the elevations of the ADWs. Suitable routes between the ADWs and the natural outlets (stream or river) or existing main drains were then determined. These routes for tile drains were delineated on topo-

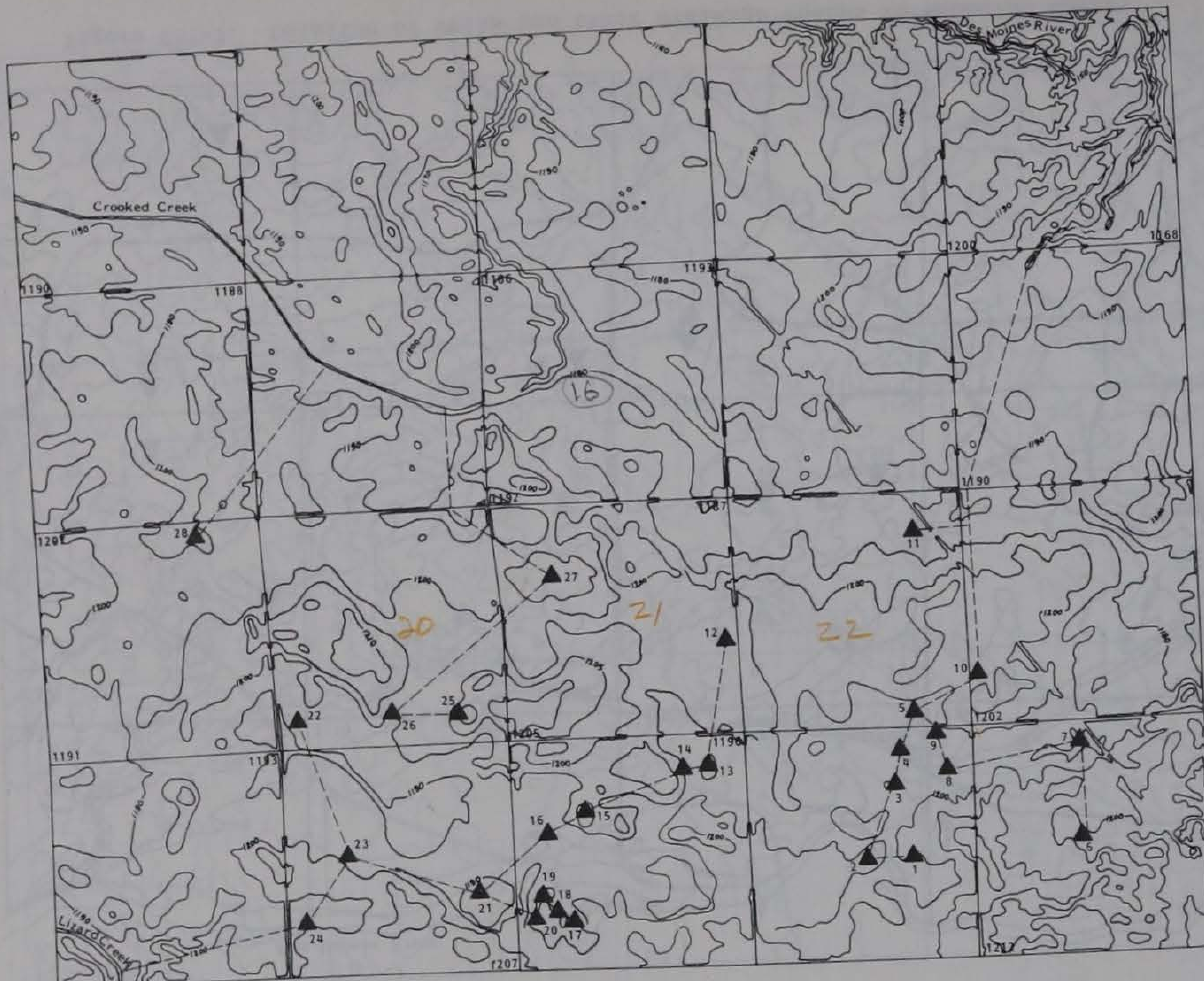

Figure VII-1. Location of wells and their drainage routes in Pocahontas County.

Figure VII-2. Location of wells and their drainage routes in Humboldt County.

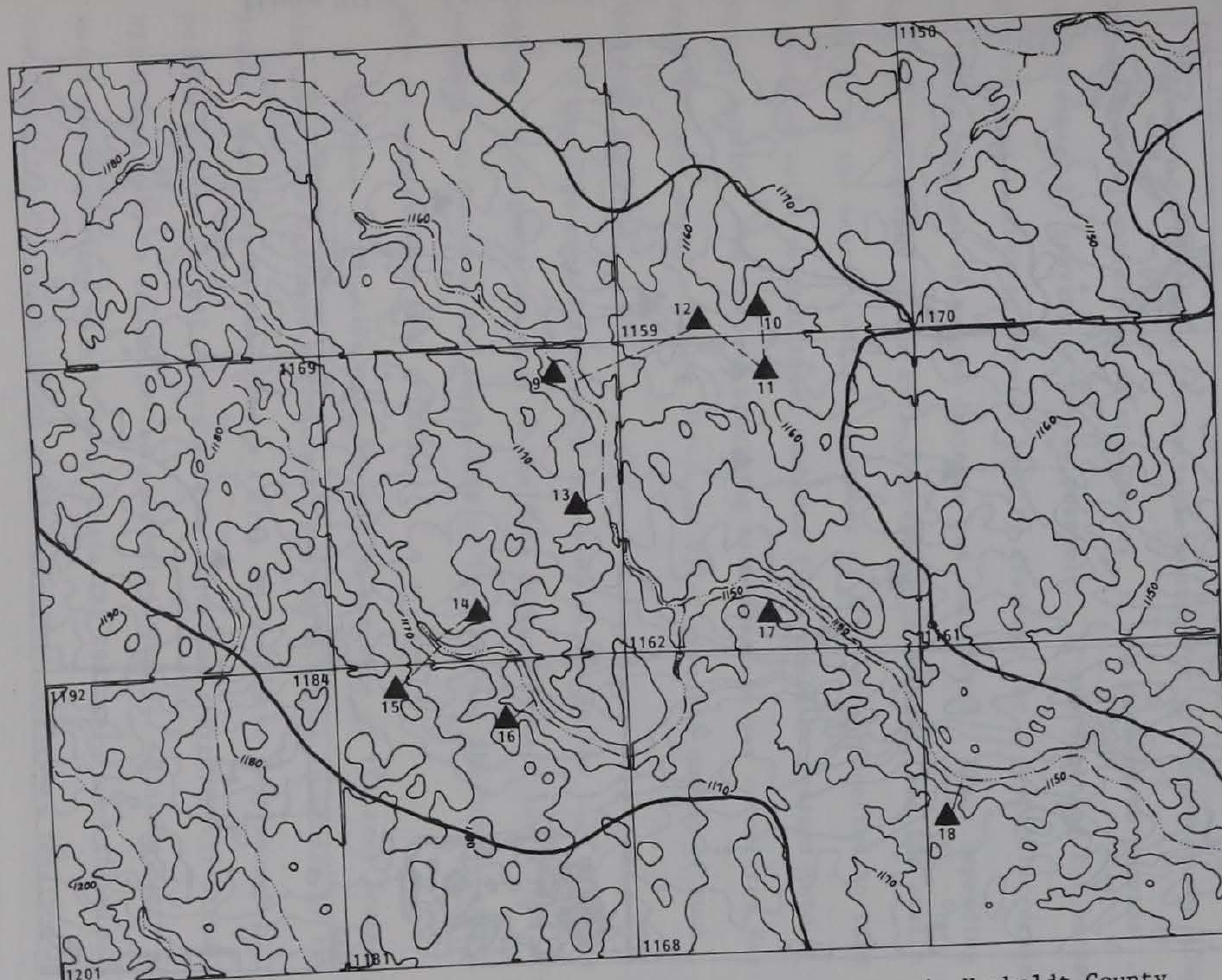

Figure VII-3. Location of wells and their drainage routes in Humboldt County.

Figure VII-4. Location of wells in Humboldt County.

graphic maps and are shown with dashed lines in Figures VII-1, VII-2, VII-3, and VII-4.

In order to determine the suitable routes for drainage between ADWs and the natural outlets or existing mains, it was decided to provide tile drainage by gravity as far as possible. If the depth of cut became greater than 15 feet to provide gravity drainage for a particular drainage route, the use of pumped drainage was considered. Also, depending upon the topography of the area, concentration of ADWs, and the availability of the nearest outlet, a number of ADWs were connected to form a group of wells so that their combined flows could be drained into a single outlet. The cost of providing tile drainage was calculated for each drainage group. Tables VII-1 and VII-2 give the number of such drainage groups.

A drainage coefficient of ½" was used to design the tile drainage system. A judgment was made on the area drained by each ADW. This judgment was based on the topography of the land and the number of ADWs in each section (640 acres).

The size of the tile main depends on the size of the area drained, the grade, the drainage coefficient, and the internal roughness of the drain. These drains should have a free outlet and be deep enough to provide outlets for all laterals to be installed. Figures VII-5 and VII-6 were used to determine the size of tile mains. Minimum depth of mains was maintained at 4.5 feet. Costs for 1982 were used to calculate the costs for the installation of the tiles and earth work required (Table VII-3).

Table VII-1. Cost of drainage in Pocahontas County.

Group	Well Number	Area (acres)	Vol. of Drainage (cfs)	Total Drainage (cfs)	Length (ft)	Slope (%)	Dia. of Tile (inch)	Avg. Depth of Cut (ft)	Cost of Drainage ($)
	1	80	1.68	1.68	1,132	0.88	10	5.0	3,170
	2	80	1.68	3.36	561	0.05	20	9.5	5,272
	*	-	-	3.36	1,570	0.10	18	5.0	14,028[a]
	3	40	0.84	4.20	817	0.05	22	5.0	6,234
	4	40	0.84	5.04	880	0.05	24	5.0	7,586
	6	160	3.36	3.36	2,200	0.23	16	9.0	13,552
	7	160	3.36	6.72	3,017	0.05	26	10.5	31,085
A	8	40	0.84	7.56	1,131	0.05	28	5.0	12,667
	9	40	0.84	8.40	754	0.05	30	5.0	8,731
	5	80	1.68	1.68	1.697	0.05	16	5.0	8,757
	10[b]	80	1.68	16.80	3,294	0.29	26	5.0	33,668
	11	160	3.36	3.36	1,320	0.05	16	5.0	6,811
	[c]	-	-	29.16	6,034	0.19	30	5.0	69,874
	[d]	960	-	29.16	5,000	-	-	-	76,800
	TOTAL	960	-	-	-	-	-	-	298,235

Avg. cost per acre = $310.7

[a]Extra cost of $5000 is added for the pumping station between well no. 2 and 3.

[b]Tile drainage system designed for the total flow of 16.80 cfs.

[c]Tile drainage system designed for 960 acres.

[d]Cost of relief main system is calculated at the rate of $80/acre.

Table VII-1. Continued

Group	Well Number	Area (acres)	Vol. of Drainage (cfs)	Total Drainage (cfs)	Length (ft)	Slope (%)	Dia. of Tile (inch)	Avg. Depth of Cut (ft)	Cost of Drainage ($)
	12	80	1.68	1.68	2,891	0.17	14	5.0	11,449
	13	80	1.68	3.36	629	0.05	20	5.0	4,428
	14	80	1.68	5.04	2,389	0.05	24	10.5	29,041
	15	80	1.68	6.72	1,006	0.05	26	10.0	13,364
	16	80	1.68	8.40	2,002	0.05	28	5.0	23,183
	17	40	0.84	0.84	377	0.05	12	5.0	1,372
	18	40	0.84	1.68	503	0.05	16	5.0	2,596
B	19	40	0.84	2.52	503	0.05	18	5.0	2,892
	20	40	0.84	3.36	1,383	0.72	12	5.0	5,034
	21[e]	60	1.26	13.02	1,623	0.05	32	10.0	31,288
	[f]	-	-	13.02	1,509	0.05	32	5.0	37,635
	22	80	1.68	1.68	2,011	0.05	16	10.5	13,142
	23[g]	160	3.36	3.36	3,520	0.05	20	5.0	24,781
	24	160	3.36	21.42	4,526	0.05	38	12.0	126,192
	TOTAL	1,020							326,397

Avg. cost per acre - $320/acre

[e]Tile drainage system designed for total flow of 13.02 cfs.

[f]An extra cost of $15,000 was added for the pumping station.

[g]Tile drainage system was designed, though open ditch may have been more economical.

Table VII-1. Continued

Group	Well Number	Area (acres)	Vol. of Drainage (cfs)	Total Drainage (cfs)	Length (ft)	Slope (%)	Dia. of Tile (inch)	Avg. Depth of Cut (ft)	Cost of Drainage ($)
	25	80	1.68	1.68	1,509	0.33	12	5.0	5,493
C	26	80	1.68	3.36	4,777	0.05	20	9.0	42,919
	27	160	3.36	6.72	5,027	0.05	26	10.0	76,459
	TOTAL	320							124,871
				Avg. cost per acre = $121.3					
D	28	160	3.36	3.36	4,903	0.40	14	5.0	
				Avg. cost per acre = $121.3					

Table VII-2. Cost of drainage in Humboldt County.

Group	Well Number	Area (acres)	Vol. of Drainage (cfs)	Total Drainage (cfs)	Length (ft)	Slope (%)	Dia. of Tile (inch)	Avg. Depth of cut (ft)	Cost of* Drainage ($)	Cost/ Acre
A	1	80	1.68	1.68	1,509	0.13	14	12.0	15,016	206.7
	2	80	1.68	3.36	2,263	0.13	16	5.0	18,077	
	3	160	3.36	6.72	3,520	0.28	18	5.0	33,040	
B	4	80	1.68	1.68	1,508	0.20	12	5.0	11,889	146.2
	5	80	1.68	3.36	2,000	0.25	16	10.0	19,220	
	6	160	3.36	6.72	500	0.40	18	5.0	15,675	
C	7	80	1.68	1.68	750	0.27	12	5.0	9,130	114.1
D	8	160	3.36	3.36	2,640	0.10	18	15.0	34,580	216.1
E	9	80	1.68	1.68	377	0.80	10	5.0	7,456	93.2
F	10	160	3.36	3.36	1,250	1.1	12	5.0	17,350	172.2
	11	160	3.36	6.72	1,383	0.1	22	5.0	23,352	
	12	80	1.68	8.40	2,263	0.1	24	9.0	28,170	
G	13	160	3.36	3.36	400	1.6	10	5.0	13,920	87.0
H	14	160	3.36	3.36	750	1.6	10	5.0	14,900	93.1
I	15	160	3.36	3.36	1,000	2.2	10	5.0	15,600	97.5
J	17	80	1.68	1.68	380	3.2	8	5.0	7,103	88.8
K	18	80	1.68	1.68	500	2.5	8	5.0	7,325	91.6
L	19	80	1.68	1.68	2,000	0.9	8	5.0	12,000	150.0
M	20	160	3.36	3.36	1,000	0.60	14	5	16,760	300.4
	21	160	3.36	6.72	5,530	0.04	26	14	72,006	
	22	80	1.68	8.40	2,000	0.10	26	10	27,364	
	23	80	1.68	10.08	2,000	0.05	30	10	33,976	
	24	160	3.36	13.44	2,125	0.10	30	10	42,164	
N	25	160	3.36	3.36	1,509	0.5	14	5	18,776	106.4
	26	160	3.36	6.72	625	1.07	14	5	15,275	

*This includes the cost of relief main ($80/acre).

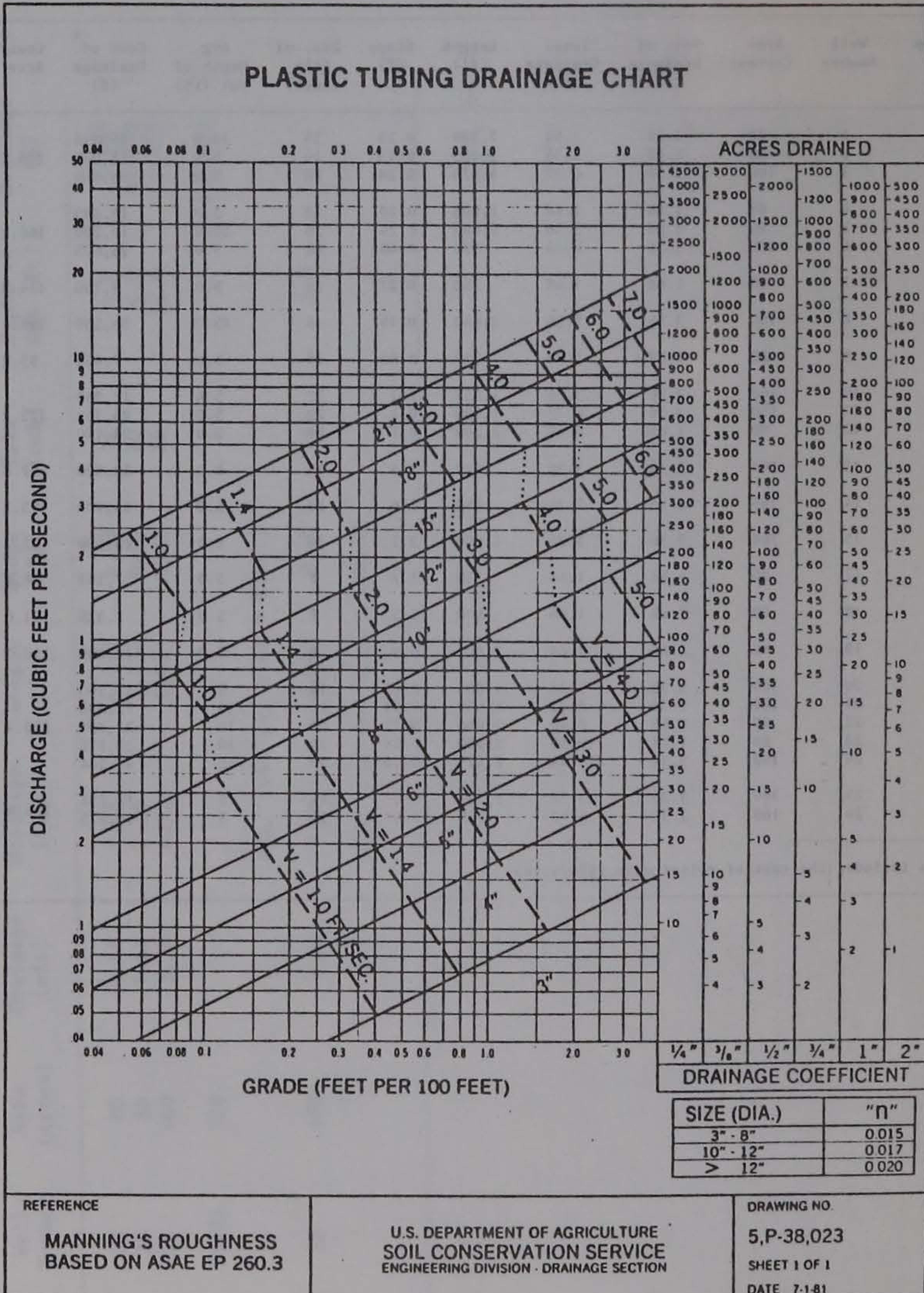

Figure VII-5. Drainage chart for plastic drainage tubing. The space between the diagonal lines indicates the range of tubing capacity for the size shown. V = velocity in feet per second.

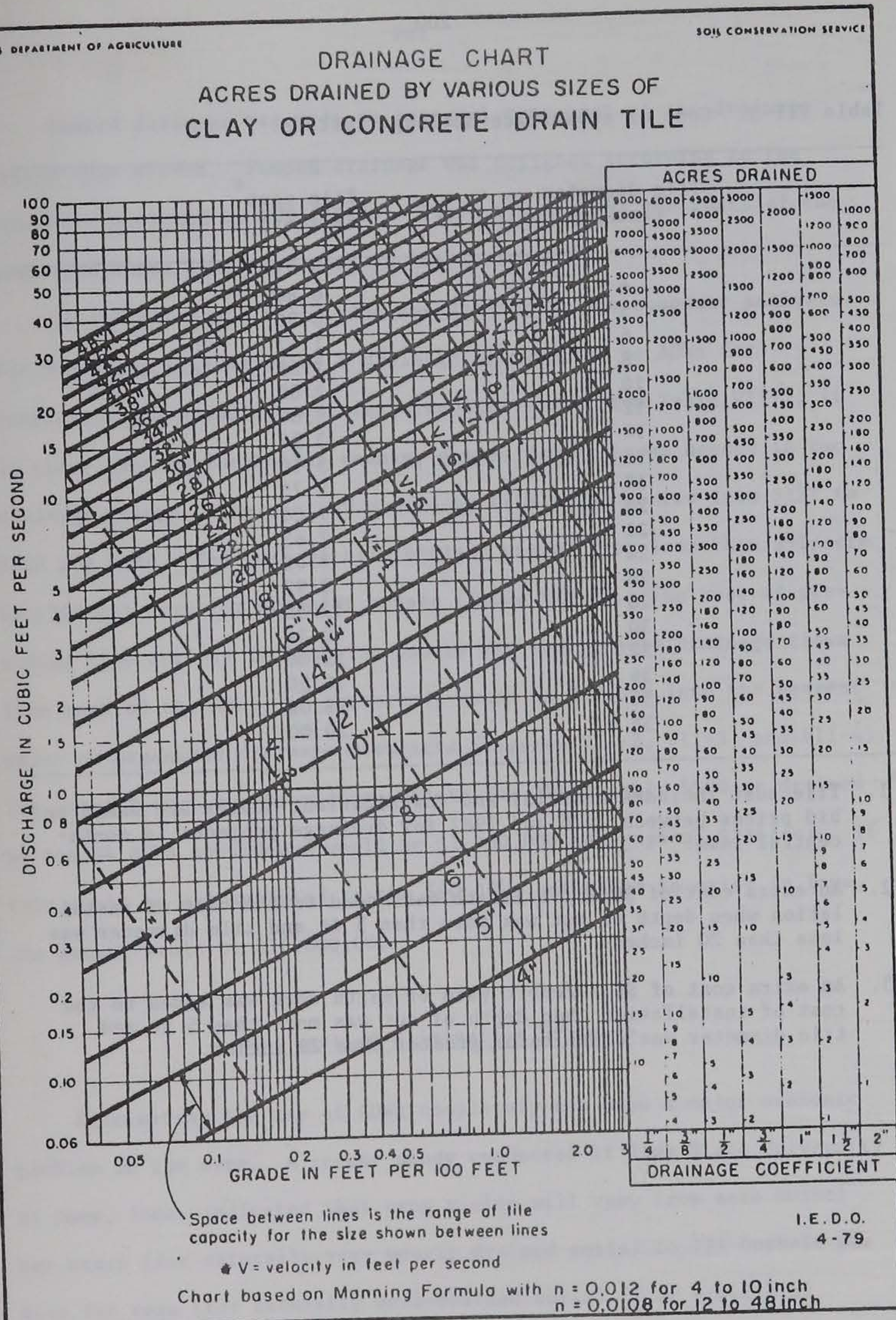

Figure VII-6. Drainage chart for clay and concrete drain tile.

Table VII-3. Cost of subsurface drainage systems.

Tile diameter (inches)	Tile cost* ($/foot)
4	0.80
5	0.95
6	1.47
8	1.85
10	2.80
12	3.64
14	3.96
16	5.16
18	5.75
20	7.04
22	7.63
24	8.62
26	9.92
28	11.58
30	13.14
32	15.00
34	16.50
36	18.00
38	20.00

*1. Tile cost includes material and installation. Costs are based on bid prices between 1976 and 1982 for drainage projects in north-central Iowa.

2. An extra cost of $0.25/ft length was added to the cost of installation when depth of cut was more than 5 ft and tile diameter was less than 20 inches.

3. An extra cost of $0.70/cubic yard of earth work was added to the cost of installation when depth of cut was more than 5 ft and tile diameter was equal to or greater than 20 inches.

Pumped drainage was made an integral component of the drainage system when needed. Pumped drainage was designed according to the procedure outlined by SCS (1983). Table VII-4 gives the costs of the pumps and sumps that were used in the analysis.

Tables VII-1 and VII-2 give the details of the economic analysis for providing tile mains as a drainage alternative to ADWs for Pocahontas and Humboldt counties, respectively. From Table VII-1, it is clear that in Pocahontas county, pumped drainage was needed in two drainage groups. The cost of providing tile mains varied from $121 to $320 per acre. The cost of tile outlets varied from less than $90/acre to $300 per acre for Humboldt county (Table VII-2). The cost of providing tile outlets in Humboldt county was low because drainage flows from most of the ADWs, if permitted, could be drained into the nearest mains of organized drainage districts (Figures VII-2, VII-3, and VII-4). The overall cost for providing outlets for the roughly 5500 ac drained by the 54 ADWs considered would be $1,300,000 (average of $236/ac). If this could be extrapolated to the 690 ADWs estimated to exist in Iowa, the amount would be $16,600,000.

B. Cost of Eliminating Drainage

Eliminating the use of ADWs completely can pose a major economic problem in the area. A recent study conducted at Iowa State University at Ames, Iowa, indicated that crop yields will vary from zero bushel per acres (for naturally very poorly drained soils) to 121 bushels per acre for corn (for naturally well-drained soils) if no artificial

Table VII-4. Pricing guide for drainage pumps.

Acres to be drained	Pump model	Avg. capacity U.S. G.P.M.	Motor H.P.	Minimum storage gal.	Price for 10' sump	10' sump with storage tank or screened inlet	Extra/Ft. pump	Extra/Ft. sump	Extra for larger motor	Dresser type coupling
5-20	SE151	290	1½	550	$ 1460.00	48" 1465.00	N/A	40.00	N/A	N/A
0-50	PL-6	600	2	1000	$ 2050.00	60" 1890.00	65.00	60.00	50.00	65.00
0-125	PL-8 PL-8G PL-8D	1100	3 11 gas 4 diesel	1200	$ 2475.00 $ 2475.00 $ 3090.00	60" 1890.00	72.00	60.00	170.00 N/A N/A	70.00
0-175	PL-10 PL-10pto	1600	5 540/1000 ppm pto	1800	$ 2925.00 $ 3050.00 10' vert	60" 1890.00	85.00 120.00/vert. ft.	60.00	190.00 N/A	90.00
50-250	PL-12	2400	7½	2400	$ 3275.00	72" 2250.00	100.00	77.00	240.00	105.00
40-300	PL-14	3100	10	3000	$ 3975.00	72" 2250.00	108.00	77.00	P.O.A.	160.00
00-400	PL-16T PL-16S	4500	15 3ph 15 1ph with con-verter	3600	$ 6750.00 $ 8350.00	84" 3100.00	125.00	105.00	P.O.A.	175.00
PL-16pto			540/1000 ppm pto		$ 6850.00 10 vert.	ditch bank mount	175.00/vert.ft.		N/A	
00-600	PL-20T PL-20S	5800	20 3ph 20 1ph with con-verter	4200	$ 9145.00 $11380.00	84" 3100.00	135.00	105.00	P.O.A. P.O.A.	240.00

drainage is provided in the areas of the Upper Des Moines River basin (Kanwar et al., 1983). Table VII-5 summarizes some of the results of this study.

Based on field surveys, it is evident that farmers in the area drained by ADWs currently have drainage systems with a drainage coefficient $\geq$ 3/8 inch. This means, as shown in Table VII-5, that their average corn yields range from 116 to 129 bu/ac, and soybean yields range from 42 to 47 bu/ac depending on the drainage level, with the higher yields at the 1/2 inch drainage coefficient. As is also shown, before any drainage, or if the ADWs were closed, corn and soybean yields would average only 70 and 26 bu/ac, respectively. Therefore, farmers would lose 46 to 59 bu/ac of corn and 16 to 21 bu/ac of soybeans, and at \$3/bu for corn and \$8/bu for soybeans, the average annual loss would be at least \$128/ac. In addition to the loss of crop yields, other problems such as equipment miring, delayed planting and harvesting, inefficient use of fertilizers, and extreme variability in crop yields would add further economic burdens on farmers of the area.

C. Other Options

1. Local Land Treatment

The possibility exists that some control could be exercised over the land within the close proximity of an ADW or surface inlet (for instance, in a radius of 100 m) in an attempt to reduce pollutant transport to it. Limiting chemical application to that land or limiting its use to production of grass are two suggestions.

Table VII-5. Estimated average corn and soybean yields (bu/acre) in the Upper Des Moines River Basin.*

Drainage Level	Low Success Drainage Districts ∼ 95% of the area (drained <¼" d.c.)		High Success Drainage Districts ∼ 5% of the area (drained > 3/8" d.c.)		Average for the Basin	
	Corn bu/acre	Soybean bu/acre	Corn bu/acre	Soybean bu/acre	Corn bu/acre	Soybean bu/acre
Before any drainage	70	26	70	26	70	26
Present drainage	91	33	115	41	92	33
Drainage at 3/8" d.c.	116	42	118	43	116	42
Drainage at 1/2" d.c.	129	47	129	47	129	47

*Yield averages are based on the assumption of high management levels and good weather. For a year when drainage needs will be maximum (a heavy rainfall year), corn and soybean yields would be around 37 and 13 bu/acre respectively for the category "Before any drainage."

For those chemicals of concern transported mainly with water (namely, most pesticides and NO_3-N), reducing applications near an ADW with a surface inlet would reduce chemical loadings roughly in proportion to the land area controlled. For example, if an ADW drained 80 acres and no pesticide was allowed to be applied within 100 m of the ADW (7.8 acres), pesticide loading would be reduced by about 10%. In the case of an ADW that had no surface inlets, which resulted in ponded areas where water was forced to flow through the soil to subsurface drains, limiting chemical application on the ponded area could reduce the increased loading otherwise expected from that area because of more water movement. However, because of a natural wetness problem in that area, in many years it may not receive chemical applications anyway.

For bacteria, for those chemicals that adsorb to soil or sediment, and for sediment itself, grass strips around a surface inlet could reduce pollutant transport, but only if flow and/or pondage depths are less than the grass height. The use of vegetative filters has been considered for feedlot runoff treatment (Vanderholm and Dickey, 1980), and could also slow runoff in a field situation allowing for some sediment and bacteria deposition and adsorption of chemicals to depositing sediment or in-place soil. However, if these pollutants are of concern, a much better method of control would be slowing the flow to or completely closing the surface inlets (discussed later in this section).

Overall, local land treatment could reduce pollutant transport to ADWs to some degree, but it could be assumed that the net return from the land that was receiving no chemicals or that was in grass would be near zero. Therefore, land treatment does not seem to have the potential of some other options for larger, more cost-effective reductions.

2. Chemical Management

Better nitrogen management in row-crop production could be used to reduce NO_3-N leaching and transport to the aquifer through ADWs. Lower rates, better application(s) timing, and possibly use of additives to prevent nitrification of ammonia should decrease leaching losses. Modeling using the CREAMS model showed that decreasing the N application rate from 150 kg/ha to 75 kg/ha on continuous corn would reduce the NO_3-N concentration in total drainage (27 mm surface runoff, 89 mm subsurface flow) from 17.6 to 9.3 mg/L; using three well-timed applications rather than one pre-plant application of a total of 150 kg/ha would reduce concentrations from 17.6 to 11.0 mg/L; and a combination of reduced rate, to 75 kg/ha, plus multiple applications, was predicted to reduce concentrations to 5.9 mg/L.

Reduction in N application rate from 150 to 75 kg/ha (from 134 to 67 lb/ac) would result in a corn yield reduction. Depending on relative prices of nitrogen and corn, cost to the farmer would vary. Interpolation of data in Table VII-6, taken from Voss et al. (1975) for a soil and climate representing much of north-central Iowa, shows that a yield reduction of 12 bu/ac would occur, causing a net loss of \$26/ac (with corn at \$3/bu and N at \$0.15/lb, roughly current prices).

If three applications were made instead of one, the additional cost of two extra trips would total about \$6/ac (Edwards, 1982). However, there would be some benefit, currently unquantifiable, in improved N use efficiency. Using a nitrification inhibitor would also cost about \$6/ac, with both environmental and economic benefits unquantifiable at this time.

Table VII-6. The effect of fertilizer N on yields, total costs of corn production and net returns.

Item	N, lb/ac								
	0	20	40	60	80	100	120	140	160
Yield (bu/ac)	77	93	106	118	125	130	133	132	130
Production* cost ($)	186	195	198	204	207	213	216	219	222
Net+ return ($)	45	84	120	150	168	177	183	177	168

*Considering extra cost for handling more grain at higher yields and N at 15¢/lb.

+Considering corn at $3/bu.

3. Structural Modification of ADWs

It is apparent from the field data collected in this project that those wells receiving both surface and subsurface drainage water had higher concentrations of sediment, bacteria and pesticides. However, surface drainage had lower levels of nitrate nitrogen. One management alternative that could be implemented would be to modify the ADWs to eliminate all direct surface runoff from entering the well. Surface runoff enters the well through inlets directly into the subsurface tile and/or through drainage into the well casing itself. At several field locations, inlets in road ditches were found to be connected to the subsurface tile. By eliminating these inlets, longer periods of wet soil conditions will result, thereby reducing the ability to do field work (trafficability) as well as reducing the crop yield.

It was not possible to estimate the cost of such structural modifications since the costs are so site-specific. If this alternative is selected, all surface inlets will have to be identified and eliminated, thus requiring the drainage water to filter through the soil into the subsurface drain tile. In some cases, this will require additional surface drainage in order to maintain crop productivity. Costs for alternative surface drainage systems have been estimated to be about \$236/ac (from \$90 to \$700/ac). The casings of many of the ADWs will need to be raised above the maximum ponding level in order to present direct entry of the water into the well. In addition, steps to prevent the water from moving directly down between the casing and the soil will be needed. Berms of compacted soil around the extended casings should prevent this short circuiting. In many of the ADWs observed in this study, the casings would require some reconstruction in order to be extended above the maximum ponding level. The cost of these modifications could be from less than \$100 to greater than \$1000 per ADW. When considered on a per acre drained basis, the estimated cost could be from \$1/ac to more than \$10/ac on the average.

Eliminating direct surface drainage will help reduce sediment loads, bacteria and pesticide transport into the ground water; however, forcing the water to percolate through the soil will tend to increase the NO_3-N transport. Thus, this alternative is not a solution in itself, but must be combined with chemical and fertilizer management in order to be most effective.

As discussed earlier in Section I relative to Fig. I-5, the use of settling basins within the cisterns of ADWs would not be effective in trapping sediment and sediment-borne pollutants. In addition, they would require considerable maintenance to keep them clean.

D. References

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VIII. ACKNOWLEDGMENT

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Special tribute goes to Taun Novak, who ably served as project officer for the study reported here, and whose untimely death saddened all those who knew her.

XI. APPENDIX

Table A-1. Concentration data for well no. 1.

DRAINAGE WELL NO. 1

	MICROBIOLOGICAL				CHEMICAL						SOLIDS		PESTICIDES					
DATE	TC	FC	FS	PH	NH4-N	NO3-N	PO4-P	CL	CA	FE	SS	TS	ATRAZINE	BLADEX	LASSO	DIELDRIN	SENCOR	BANVIL
					---MG/L---						---MG/L---		---µG/L---					
2/24/81	[a]												0.0	0.0	0.0	0.0	0.0	0.0
5/24/81	500					3.6		5.0			5360		0.0	80.00	0.70	0.028	0.0	0.90
6/24/81	>[b]			6.6		26.0		2.5	39		5360	5540	0.0	16.00	0.15	0.006	0.0	0.0
8/25/81						3.6		5.0			1372		0.0	1.20	0.0	0.0	0.0	0.0
8/26/81				7.0		10.0		9.5	58		4	296	0.0	0.13	0.0	0.0	0.0	0.0
2/22/82					0.49	1.5	0.137	1.0	13	0.16	180	258						
3/19/82					0.28	6.7	0.039	8.5	28	0.44	1121	1248						
4/20/82	>	20	90	7.8	0.03	12.0	0.080	8.0	60	0.15	1	320						
5/18/82	90000	330	2600	7.2	0.01	13.0	0.050	12.0	58	0.09	2	310	0.0	0.0	0.25	0.0	0.0	0.0
5/25/82	2000	40	110	7.2	0.02	13.0	0.020	12.0	62	0.03	2	230	0.0	0.0	0.0	0.0	0.0	0.0
6/ 1/82	6800	30	230	7.1	0.04	14.0	0.060	12.0	61	0.05	1	370	0.12	0.0	0.10	0.0	0.0	0.0
7/15/82						15.9												

[a]Blank indicates no analysis made.

[b]> indicates too numerous to count.

Table A-2. Concentration data for well no. 2.

DRAINAGE WELL NO. 2

DATE	MICROBIOLOGICAL TC	FC	FS	PH	CHEMICAL NH4-N	NO3-N	PO4-P	CL	CA	FE	SOLIDS SS	TS	PESTICIDES ATRAZINE	BLADEX	LASSO	DIELDRIN	SENCOR	BANVIL
					-----	-----	MG/L	-----	-----	-----	---MG/L	---	-----	-----	µG/L	-----	-----	-----
5/ 4/81	[a]			8.0		3.0		120.0	82		24	586	0.0	0.70	0.57	0.007	0.0	0.0
5/24/81						20.0		36.0			100		0.0	0.0	2.80	0.009	0.0	6.10
5/27/81				7.7		18.0		30.0	110		1	512	0.0	0.0	0.80	0.005	0.0	0.0
6/ 3/81				7.7		10.0		28.0	120		2	562						
6/10/81				7.6		10.0		26.0	120		1	560	0.0	0.0	0.04	0.0	0.0	0.0
6/16/81				7.7		11.0		28.0	110		64	613	0.0	0.0	0.07	0.0	0.0	0.0
6/24/81	510			8.2		9.1		30.0	99		1	580	0.0	0.0	0.0	0.0	0.0	0.25
6/28/81				7.4		15.0		24.0	110		1	580	0.0	0.0	0.06	0.0	0.0	0.0
7/ 7/81				7.5		15.0		22.0	120		8	625	0.0	0.0	0.0	0.0	0.0	0.0
7/28/81				7.3		11.0		47.0	120		1	678	0.0	0.0	0.0	0.0	0.0	0.0
8/ 3/81				7.4		11.0		40.0	130		1	659	0.0	0.0	0.0	0.0	0.0	0.0
8/26/81				7.3		7.6		66.0	140		1	678	0.0	0.12	0.0	0.0	0.0	0.0
9/ 1/81				7.5		10.0		46.0	140		2	636	0.0	0.0	0.0	0.0	0.0	0.0
9/ 8/81				7.9		2.3		78.0	140		1	710	0.0	0.30	0.0	0.0	0.0	0.0
10/ 4/81				7.5		13.0		36.0	14		8	642	0.0	0.0	0.0	0.0	0.0	0.0
10/14/81				8.2		14.0		32.0	140		3	619	0.0	0.0	0.0	0.0	0.0	0.0
10/20/81	>[b]	< 10	< 10	7.9	0.01	15.0	0.060	36.0	140	0.01	0	695						
10/28/81	71		9	7.5	0.01	13.0	0.050	38.0	140	0.04	5	636	0.0	0.0	0.0	0.0	0.0	0.0
11/ 3/81	30	< 10	< 10	7.4	0.01	13.0	0.030	35.0	150	0.04	12	350	0.0	0.0	0.0	0.0	0.0	0.0
11/11/81	9	< 10	< 10	7.2	0.01	15.0	0.040	33.0	150	0.02	3	667	0.0	0.0	0.0	0.0	0.0	0.0
2/22/82					1.12	8.8	1.882	38.2	60	0.12	126	375						
3/19/82					0.75	10.7	0.629	35.8	76	0.21	159	548						
3/30/82[c]	>	< 10	50	7.4	0.01	13.0	0.140	34.0	110	0.13	11	606	0.0	0.0	0.0	0.0	0.0	0.0
4/ 5/82	< 10	< 10	< 10	7.5	0.04	15.0	0.050	41.0	120	0.04	6	598						
4/13/82	3700	< 10	10	7.3	0.06	13.0	0.060	39.0	110	0.03	5	460						
4/20/82	>	< 10	< 10	8.0	0.01	16.0	0.060	40.0	120	0.04	1	590						
4/27/82	100	< 10	< 10	7.4	0.06	18.0	0.080	42.0	120	0.04	8	600	0.0	0.0	0.0	0.0	0.0	0.0
5/ 4/82	< 10	< 10	< 10	7.5	0.03	17.0	0.060	34.0	120	0.06	1	630	0.0	0.0	0.0	0.0	0.0	0.0
5/ 7/82	600	< 10	40	7.8	0.04	9.3	0.040	40.0	130	0.07	2	690	0.0	1.00	0.0	0.0	0.0	0.0
5/11/82	1000	40	< 10	7.2	0.01	18.0	0.050	44.0	130	0.01	2	620	0.0	0.0	0.0	0.0	0.0	0.0
5/18/82	1200	50	320	7.3	0.04	21.0	0.060	42.0	130	0.01	2	640						
5/25/82	200	10	30	7.2	0.01	22.0	0.030	42.0	130	0.03	1	620	0.0	0.45	0.0	0.0	0.0	0.0
5/27/82[d]	6000	250		7.5	0.33	17.0	0.510	28.0	96	1.50	70	560	0.18	7.40	2.00	0.0	0.0	0.0
6/ 1/82	220	80	< 10	7.1	0.01	24.0	0.050	40.0	120	0.03	1	620	0.11	0.91	0.0	0.0	0.0	0.0
6/ 8/82	140	< 10	10	7.3	0.04	24.0	0.060	40.0	120	0.01	1	660	0.0	0.65	0.0	0.0	0.0	0.0
6/15/82	>	< 10	< 10	7.3	0.01	25.0	0.030	38.0	130	0.02	1	390	0.10	0.54	0.0	0.0	0.0	0.0
6/22/82	< 10	< 10	< 10	7.2	0.11	25.0	0.010	36.0	130	0.03	1	570	0.0	0.49	0.0	0.0	0.0	0.0
6/29/82	>	10	< 10	7.1	0.02	22.0	0.050	38.0	130	0.01	6	680	0.0	0.34	0.0	0.0	0.0	0.0
7/15/82						22.4												

[a]Blank indicates no analysis made.

[b]> indicates too numerous to count.

[c]Chlordane detected at 1.8 µg/L.

[d]Furadan detected at 0.6 µg/L.

Table A-3. Concentration data for well no. 6.

DRAINAGE WELL NO. 6

DATE	MICROBIOLOGICAL TC	FC	FS	PH	CHEMICAL NH4-N	NO3-N	PO4-P	CL	CA	FE	SOLIDS SS	TS	PESTICIDES ATRAZINE	BLADEX	LASSO	DIELDRIN	SENCOR	BANVIL
					-------	-------	MG/L	-------	-------	-------	---MG/L---		-------	-------	-------	µG/L	-------	-------
4/ 6/81	[a]					1.7		49.0			1		0.0	0.0	0.0	0.0	0.0	0.0
5/24/81						18.0		15.0				2260	0.0	7.50	55.00	0.011	0.20	12.00
5/27/81				7.7		17.0		32.0	100		1	470	0.0	0.24	0.44	0.010	0.41	0.0
6/16/81				7.8		17.0		36.0	110		1	536	0.0	0.10	0.27	0.0	0.0	2.60
6/24/81	>[b]			7.3		12.0		20.0	76		196	716	0.0	2.30	1.10	0.007	0.40	0.25
6/28/81				7.5		16.0		31.0	99		2	560	0.0	0.06	0.18	0.0	0.20	0.0
2/22/82					1.18	2.7	0.985	9.5	18	0.12	101	176						
3/19/82					3.78	6.0	1.992	23.0	53	0.64	1222	1465						
3/30/82[c]	16	< 10	50	7.5	0.03	15.0	0.110	38.0	110	0.04	9	636	0.0	0.0	0.0	0.0	0.0	0.0
4/ 5/82	>	4700	7600	7.6	0.19	17.0	0.150	40.0	120	0.14	5	602						
4/13/82	40000	22000	23000	7.4	0.53	15.0	0.390	41.0	120	0.16	20	560						
4/20/82	>	2500	600	7.5	0.16	18.0	0.220	39.0	120	0.08	1	580						
4/27/82	<100	30	< 10	7.5	0.13	21.0	0.140	40.0	120	0.03	1	590	0.0	0.0	0.0	0.0	0.0	0.0
5/ 4/82	< 10	10	< 10	7.6	0.14	21.0	0.110	41.0	140	0.04	1	610	0.0	0.0	0.0	0.0	0.0	0.0
5/ 7/82	80000	44000	20000	7.9	0.33	22.0	0.340	42.0	130	0.30	12	690	0.0	0.0	0.0	0.0	0.0	0.0
5/11/82	500	40	90	7.3	0.11	23.0	0.090	39.0	130	0.01	4	610	0.0	0.0	0.0	0.0	0.0	0.0
5/18/82	3000	10	180	7.3	0.01	26.0	0.090	40.0	130	0.01	1	620	0.0	0.0	0.0	0.0	0.0	0.0
5/25/82	300	20	40	7.3	0.01	27.0	0.040	42.0	120	0.04	1	620	0.0	0.0	0.0	0.0	0.0	0.0
5/27/82	2260000	180000		7.1	0.19	22.0	0.810	36.0	91	2.60	140	640	0.11	0.07	0.13	0.0	0.0	0.0
6/ 1/82	400	140	70	7.2	0.02	30.0	0.120	42.0	120	0.03	1	580	0.0	0.0	0.0	0.0	0.0	0.0
6/ 8/82	1000	10	20	7.4	0.01	30.0	0.110	42.0	130	0.01	1	710	0.0	0.0	0.0	0.0	0.0	0.0
6/15/82	>	< 10	60	7.4	0.01	34.0	0.090	42.0	130	0.05	1	650	0.0	0.0	0.0	0.0	0.0	0.0
6/22/82	30	20	30	7.4	0.12	31.0	0.090	42.0	130	0.05	1	610	0.0	0.0	0.0	0.0	0.0	0.0
6/29/82	140	10	< 10	7.4	0.03	29.0	0.120	42.0	130	0.01	5	750	0.0	0.0	0.0	0.0	0.0	0.0
7/15/82						31.1												

[a]Blank indicates no analysis made.

[b]> indicates too numerous to count.

[c]Chlordane detected at 0.7 µg/L.

Table A-4. Concentration data for well no. 7.

DRAINAGE WELL NO. 7

DATE	MICROBIOLOGICAL TC	FC	FS	PH	CHEMICAL NH4-N	NO3-N	PO4-P	CL	CA	FE	SOLIDS SS	TS	PESTICIDES ATRAZINE	BLADEX	LASSO	DIELDRIN	SENCOR	BANVIL
					---MG/L---						---MG/L---		---µG/L---					
2/24/81	a												0.0	0.0	0.0	0.0	0.0	0.0
5/24/81						14.0		25.0				28	0.0	5.60	0.10	0.007	0.0	1.80
5/27/81				7.8		16.0		20.0	85		1	398	0.0	0.83	0.0	0.016	0.0	0.0
6/16/81				7.4		16.0		22.0	88		1	412	0.0	0.14	0.04	0.0	0.0	0.18
7/ 2/81				7.2		14.0		19.0	82		2	440	0.0	1.10	0.0	0.0	0.0	0.0
2/22/82					0.26	6.3	1.715	22.6	36	0.11	121	258						
3/19/82					0.16	8.5	0.293	26.6	65	0.15	130	483						
3/30/82	>[b]	< 10	< 10	7.7	0.01	10.0	0.090	24.0	84	0.04	9	458	0.0	0.0	0.0	0.0	0.0	0.0
4/ 5/82	< 10	< 10	< 10	7.6	0.05	14.0	0.080	26.0	88	0.05	5	398						
4/13/82	400	< 10	< 10	7.7	0.04	14.0	0.110	26.0	110	0.04	2	380						
4/20/82	>	< 10	< 10	7.7	0.01	17.0	0.110	29.0	97	0.04	1	490						
4/27/82	<100	< 10	< 10	7.7	0.02	19.0	0.050	31.0	100	0.03	16	480	0.0	0.0	0.0	0.0	0.0	0.0
5/ 4/82	< 10	< 10	< 10	7.8	0.04	19.0	0.040	31.0	110	0.04	1	480	0.0	0.0	0.0	0.0	0.0	0.0
5/11/82	110	< 10	< 10	7.6	0.03	20.0	0.040	31.0	110	0.01	1	490	0.0	0.0	0.0	0.0	0.0	0.0
5/18/82	90	10	< 10	7.5	0.01	21.0	0.040	34.0	110	0.01	1	540	0.0	0.0	0.0	0.0	0.0	0.0
5/25/82	< 10	< 10	< 10	7.5	0.10	23.0	0.020	36.0	110	0.04	2	500	0.0	0.0	0.0	0.0	0.0	0.0
5/27/82[c]	4000	2000		7.4	0.01	20.0	0.220	31.0	85	1.70	22	470	0.50	0.66	1.20	0.0	0.0	0.0
6/ 1/82	850	< 10	< 10	7.4	0.02	24.0	0.040	35.0	110	0.05	1	520	0.0	0.0	0.0	0.0	0.0	0.0
6/ 8/82	20	< 10	< 10	7.5	0.02	24.0	0.040	36.0	110	0.01	1	580	0.0	0.0	0.0	0.0	0.0	0.0
6/15/82	< 10	< 10	10	7.4	0.01	25.0	0.100	34.0	110	0.05	1	520	0.0	0.0	0.0	0.0	0.0	0.0
6/22/82	< 10	< 10	< 10	7.4	0.09	26.0	0.020	37.0	110	0.04	1	530	0.0	0.0	0.0	0.0	0.0	0.0
6/29/82	< 10	< 10	< 10	7.3	0.13	26.0	0.050	38.0	120	0.01	6	610	0.0	0.0	0.0	0.0	0.0	0.0
7/15/82						17.8												

[a] Blank indicates no analysis made.

[b] > indicates too numerous to count.

[c] 2,4-D detected at 0.4 µg/L.

Table A-5. Concentration data for Sheldon Park well.

SHELDON PARK WELL

DATE	MICROBIOLOGICAL TC	FC	FS	PH	CHEMICAL NH4-N (MG/L)	NO3-N	PO4-P	CL	CA	FE	SOLIDS SS (MG/L)	TS	PESTICIDES ATRAZINE (µG/L)	BLADEX	LASSO	DIELDRIN	SENCOR	BANVIL
4/ 9/81	< 1	[a]		7.4		0.1		2.0	84		2	388	0.0	0.0	0.0	0.0	0.0	0.0
4/22/81	< 10			7.4		0.1		1.5	84		1	370	0.0	0.0	0.0	0.0	0.0	0.0
4/27/81	< 1			7.3		0.1		2.0	83		15	366	0.0	0.0	0.0	0.0	0.0	0.0
5/ 5/81	< 1			7.4		0.1		2.0	93		1	388	0.0	0.0	0.0	0.0	0.0	0.0
5/13/81	< 1			7.3		0.1		1.5	87		1	382	0.0	0.0	0.0	0.0	0.0	0.0
5/20/81	< 10			7.4		0.1		1.5	92		3	362	0.0	0.0	0.0	0.0	0.0	0.0
5/27/81	24			7.4		0.1		5.5	84		1	352	0.0	0.0	2.70	0.0	0.15	0.0
6/ 3/81	32			7.5		0.2		2.5	87		1	346						
6/10/81	1			7.3		0.1		3.0	85		1	360	0.0	0.0	0.07	0.0	0.0	0.0
6/16/81	1			7.4		0.1		2.0	77		1	367						
6/24/81	32			7.3		0.1		3.0	85		1	402						
6/28/81	7			7.5		2.9		8.0	76		1	380	0.0	0.0	0.06	0.0	0.02	0.0
7/ 7/81	15			7.2		0.1		1.5	82		7	380	0.0	0.0	0.0	0.0	0.0	0.0
7/15/81	2			7.4		0.1		2.0	84		1	370	0.0	0.0	0.0	0.0	0.0	0.0
7/22/81	7			7.4		0.1		3.0	88		1	379	0.0	0.0	0.0	0.0	0.0	0.0
7/28/81	3			7.4		0.1		2.0	85		1	390	0.0	0.0	0.0	0.0	0.0	0.0
8/ 3/81	3			7.4		0.1		1.0	85		1	373	0.0	0.0	0.0	0.0	0.0	0.0
8/11/81	1			7.4		0.1		1.0	96		3	380	0.0	0.0	0.0	0.0	0.0	0.0
8/19/81	25			7.5		0.1		3.0	82		1	374	0.0	0.0	0.0	0.0	0.0	0.0
8/26/81	34			7.3		0.1		2.5	86		2	354	0.0	0.0	0.0	0.0	0.0	0.0
9/ 1/81	54			7.4		0.2		2.0	86		1	354	0.0	0.0	0.0	0.0	0.0	0.0
9/ 8/81	1			7.3		0.1		2.5	82		5	368	0.0	0.0	0.0	0.0	0.0	0.0
9/16/81	2			7.5		0.1		2.0	74		6	366	0.0	0.0	0.0	0.0	0.0	0.0
9/23/81	0			7.4		0.1		2.5	80		6	346	0.0	0.0	0.0	0.0	0.0	0.0
9/29/81	< 1			7.3		0.2		2.0	83		13	332	0.0	0.0	0.0	0.0	0.0	0.0
10/ 4/81	< 1			7.4		0.1		2.5	84		1	336	0.0	0.0	0.0	0.0	0.0	0.0
10/14/81	< 1			7.4		0.1		2.5	86		1	357	0.0	0.0	0.0	0.0	0.0	0.0
10/20/81	12	< 10	< 10	7.3	0.01	0.1	0.120	2.5	84	0.14	1	342						
10/28/81	1		< 10	7.2	0.02	0.1	0.050	2.5	84	0.49	5	358	0.0	0.0	0.0	0.0	0.0	0.0
11/ 3/81	< 1	< 10	< 10	7.3	0.05	0.1	0.190	3.0	87	0.28	14	650	0.0	0.0	0.0	0.0	0.0	0.0
11/11/81	< 1	< 10	< 10	6.8	0.01	0.1	0.050	2.0	87	0.11	1	369	0.0	0.0	0.0	0.0	0.0	0.0
4/20/82	>	< 10	< 10	7.9	0.01	0.2	0.040	3.0	53	0.69	18	290						
4/27/82	<100	10	< 10	7.5	0.02	4.3	0.520	24.0	74	6.50	1	350	0.0	0.0	0.0	0.0	0.0	0.0
5/ 4/82	< 10	< 10	< 10	7.5	0.05	6.2	0.070	39.0	83	0.14	1	350	0.0	0.0	0.0	0.0	0.0	0.0
5/11/82	10	< 10	20	7.3	0.11	8.3	0.270	33.0	93	0.05	1	430	0.0	0.0	0.0	0.0	0.0	0.0
5/18/82				7.8	0.01	8.2	0.050	34.0	92	0.04	1	440	0.0	0.0	0.0	0.0	0.0	0.0
5/25/82	120	20	30	7.4	0.01	11.0	0.050	33.0	86	0.04	1	360						
6/ 1/82	50	< 10	20	7.3	0.01	5.7	0.120	22.0	86	0.06	1	380	0.11	0.0	0.0	0.0	0.0	0.0
6/ 8/82	< 10	< 10	10	7.3	0.03	4.8	0.060	18.0	86	0.01	1	410	0.0	0.0	0.0	0.0	0.0	0.0
6/15/82	>[b]	20	10	7.4	0.01	5.1	0.080	19.0	87	0.04	1	620	0.0	0.0	0.0	0.0	0.0	0.0
6/22/82	60	10	10	7.4	0.10	3.5	0.080	13.0	85	0.03	1	370	0.0	0.0	0.0	0.0	0.0	0.0
6/29/82	10	< 10	< 10	7.3	0.04	1.4	0.070	10.0	91	0.01	3	420	0.0	0.0	0.0	0.0	0.0	0.0

[a]Blank indicates no analysis made.

[b] > indicates too numerous to count.

Table A-6. Days of the month when precipitation exceeded 10 mm.

1981	
April	3 (21 mm), 12 (12)
May	3 (20), 4 (16), 23 (64)
June	9 (20), 23 (40), 24 (10), 29 (12)
July	19 (14), 20 (13), 22 (14), 25 (53), 27 (12)
August	1 (13), 25 (52), 27 (10), 28 (20)
September	24 (13)
October	3 (28), 17 (10)
1982	
April	2 (11), 15 (66), 16 (17)
May	5 (26), 6 (13), 12 (30), 13 (11), 15 (15), 17 (12), 21 (17), 26 (28)
June	6 (10)

Table A-7. Concentration data for two farm water supply wells near ADWs no. 6 and 7.

Well	Date	NO_3-N	Fe	Hardness	Pesticides
		------------mg/L	----------------	----------------	
Near ADW 6	12/16/80	4.4	<0.1	390	none detected
	7/1/81	5.5	0.06	425	none detected
Near ADW 7	12/16/80	2.9	<0.1	430	none detected
	7/1/81	2.9	0.09	395	none detected

Table A-8. Concentration data for farm water supply wells (area 1).

WELL INFORMATION
AREA 1

WELL NO.	YEAR DRILLED	WELL DEPTH M	DEPTH TO WATER M	DIA. CM	YIELD L/MIN	GROUTED	NO3-N FIRST SAMPLING (7/19-8/12/82) MG/L	NO3-N SECOND SAMPLING (1/9-1/13/83) MG/L	NO3-N THIRD SAMPLING (7/18-7/29/83) MG/L	NO3-N AVG. MG/L	BACTERIA (2ND SAMP.) FECAL COLI #/100ML	BACTERIA (2ND SAMP.) FECAL STREP #/100ML	BACTERIA (2ND SAMP.) TOTAL COLI #/100ML
39A	1920	27.4	10.7	15	68	NO	0.0	0.0	0.1	0.0	0	1	0
37	1981	42.4	18.3	91	45	YES	0.0	0.0	1.2	0.4	0	30	0
36	1920	36.6	30.5	15		NO	3.4	3.8	4.6	3.9	5	5	47
35							16.0	12.9	12.3	13.7	0	4	0
32	1949	24.4	13.7	17	492	NO	4.4	4.5	4.6	4.5	0	0	2
33	1966	60.4	39.0	12		NC	5.2	6.5	6.9	6.2	0	2	0
34	1949	49.7	17.7	15		YES	4.1	5.3	3.4	4.3	0	2	0
138	1950	35.6	30.5	15	34	NO	0.0	0.0	0.0	0.0	0	98	0
137A	1940	33.5	19.8	25	68	NO	0.2	12.6	14.1	9.0	0	179	0
1376		17.7	1.8	30	56		0.0	0.0		0.0	0	4	1
38	1959	33.5		15	83	NO	0.0	0.0	0.2	0.1	0	0	0
70	1977	121.9	36.6	20	37	NO	2.3	0.2	2.7	1.7	0	93	0
73							0.1	0.2	4.0	1.4	0	>200	0
69							9.8	8.9	5.4	8.0	0	97	0
47	1955	44.8	41.8	25	30	NO	24.0	29.3	6.6	20.0	2	9	18
48	1920	61.0	30.5	15		NO	9.8	9.0	4.7	7.8	0	1	5
175								1.6		1.6	11	82	11
30							0.6	0.5	0.5	0.5	0	0	0
29	1950	79.2	45.7	15		NO	12.6	11.9	18.2	14.2	0	2	0
31	1930	67.1	18.3	15	132	NO	8.6	6.4	9.3	8.1	0	7	2
136	1910	39.6	9.1	15		NO	0.0	0.0	0.5	0.2	0	1	92
64							0.3	0.1	0.9	0.4	0	31	0
74	1936	9.1				NO	0.1	22.7	35.5	19.4	0	2	0
42	1928	30.5				NO	0.0	0.0	1.2	0.4	0	1	0
43		42.7	27.4	15		NO	5.8	2.7	5.2	4.6	0	1	1
68							9.6	6.6	5.6	7.3	6	>200	200
67							5.4	0.0	9.5	5.0	0	3	0
51	1976	39.6					8.2	9.6	6.4	8.1	1	5	30
49	1920	57.9	7.6	15		NO	0.3	1.0	0.5	0.6	0	0	0
45	1920	36.6	9.1	15		NO	10.8	9.0	16.4	12.1	0	0	0
46	1920	32.0	6.1	15	56	NO	33.5		33.9	33.7			
27	1940						12.1	11.8	19.2	14.4	0	14	0
28	1954	65.4	41.1	12	60	NO		9.9	10.7	10.3	0	1	0
24				10			0.0	1.2	13.5	4.9	0	2	0
63	1900	44.2	13.7	15		NO	12.2	13.9	12.6	12.9	0	5	0
25	1960	91.4	76.2	15		YES	9.4	11.5	15.8	12.2	0	1	0
40	1890	12.2	6.1	45		NO	27.4	33.8	58.0	39.7	0	24	2
41							0.0	0.1	0.3	0.1	0	5	0
161	1922	10.4		12		NO	65.6	61.3	60.8	62.6	0	86	0
44	1900	19.3	11.5	60		NO	8.5		4.8	6.6			
66	1900	10.7	4.6	35		NO	93.6		38.5	66.0			
50	1917	18.3	13.7	15		YES	14.9	8.3	6.7	10.0	0	7	4
75	1941	18.3	4.3	15		YES	42.2	37.7	4.5	28.1	>200	0	>200
65	1937	24.4	19.8	15		NO	0.3	0.0	0.0	0.1	0	3	0
72	1940	45.7	19.8	15		NO	2.1	1.2	2.5	1.9	0	116	0
71	1920	38.1	18.3	15		NO	11.4	11.4	11.9	11.6	0	98	1
26							31.4	33.2	33.3	32.6	0	3	0

Table A-9. Concentration data for farm water supply wells (area 2).

WELL INFORMATION
AREA 2

							NO3-N				BACTERIA(2ND SAMP.)		
WELL NO.	YEAR DRILLED	WELL DEPTH M	DEPTH TO WATER M	DIA. CM	YIELD L/MIN	GROUTED	FIRST SAMPLING (7/19-8/12/82) MG/L	SECOND SAMPLING (1/9-1/13/83) MG/L	THIRD SAMPLING (7/18-7/29/83) MG/L	AVG. MG/L	FECAL COLI #/100ML	FECAL STREP #/100ML	TOTAL COLI #/100ML
52	1950						14.5	15.1	17.9	15.8	0	40	3
53	1930	15.2	7.6	15	18	NO	11.7	10.8	14.7	12.4	0	38	50
14	1940						4.8	4.9	9.4	6.4	0	44	0
15							0.8	1.6	3.7	2.0	0	2	1
139	1955	88.4	61.0	12		NO	1.5	1.4	1.2	1.4	0	1	0
17	1964	67.4	48.8	12	30	NO	3.6	3.0	2.2	2.9	0	7	1
173								3.0	4.8	3.9	0	4	0
76	1910	18.3		10		NO	2.7	6.0	2.1	3.6	0	160	0
20	1958	29.0		12	37	NO	2.5	5.9	5.3	4.9	0	21	0
23	1920	45.7	30.5	15	41	NO	3.3	0.9	15.4	6.5	0	81	2
21	1945	91.4	70.1	15	52	NO	0.7	5.2	6.9	4.3	0	1	0
22	1982	65.5	36.6	20	113	YES	5.8	17.1	17.0	13.3	0	45	2
140	1935	22.9	15.2			NO	0.0	0.0	0.2	0.1	0	93	0
157	1931	66.8	45.7	20	56	NO	32.3	52.3	31.5	38.7	0	8	1
152	1935	35.6	24.4	15		YES	0.0	10.1	0.1	3.4	3	9	3
79	1940	39.6	6.1	12		NO	0.0	0.0	0.0	0.0	0	1	0
158	1940	30.5		15		NO	19.1	23.6	25.4	22.7	1	3	23
54	1972	65.2	24.4	15		YES	16.6		16.1	16.4			
16	1900	16.5	7.6	12	60	NO	10.2	11.4	10.5	10.7	0	48	1
18	1982	64.0	51.8	12		YES	1.4	1.9	4.8	2.7	0	5	0
89	1967	67.1	62.5	15	68	NO	4.8	7.6	7.5	6.6	0	33	0
88	1947	46.6		12		NO	3.1	3.8	1.2	2.7	0	15	0
19	1980	79.2	61.0	17	181	YES	6.2	3.0	6.1	5.1	0	1	0
171								0.0	0.2	0.1	0	6	0
77	1950	83.8	6.1	15	34	YES	0.2	0.2	0.1	0.2	0	4	0
153	1976	28.0	17.7	12	45	NO	10.4		14.0	12.2			
78							0.3	0.3	0.3	0.3	0	5	0
81	1905	18.3	7.6	15		NO	0.0	0.0	0.0	0.0	0	9	0
82							0.0	0.0	0.5	0.2	0	22	0
142	1905	30.2	12.2	12		NO	0.0		0.0	0.0			
143	1980	50.3		10		YES	0.7	0.8	1.5	1.0	0	>200	0
55	1966	36.5		20		NO	26.0	29.2	32.0	29.1	0	9	0
58	1950	38.7	18.3	15		NO	69.6	76.2	53.9	66.6	0	3	2
4A	1939	30.5	9.1	12		YES	46.8	18.8	34.4	33.3	0	31	0
57	1880	36.6	13.7	20		NO	18.8	20.3	25.9	21.7	0	7	0
59	1915	30.5	24.4	15		NO	33.8	35.4	39.9	36.4	0	6	0
87	1910	30.5	25.9	15		NO	1.4	0.5	0.8	0.9	0	3	0
156	1931	9.1					4.5			4.5			
141			13.7				0.0	0.0	0.3	0.1	0	2	0
86	1979	92.7	61.0	17	37	YES	0.1	0.0	0.2	0.1	0	8	0
134	1958	18.3	15.2	15		NO	0.1	4.1	0.0	1.4	0	1	0
154A	1957	30.8		12	75	NO	0.0		0.2	0.1			
80	1981	0.0	0.0	17		NO	0.0	0.0	0.2	0.1	0	0	0
84	1920	21.9	12.2	10	22	YES	0.0	0.0	0.2	0.1	0	8	0
13	1940	152.4	115.8	20		NO	0.0	0.0	0.2	0.1	0	2	0
2	1964	25.9	7.9	12		NO	3.1	3.0	4.8	3.6	0	80	0
5	1956	21.3	6.1	12		NO	26.3	13.0	13.3	17.5	0	43	0
6	1928	13.7	3.0	15	37	NO	11.0	14.5	18.6	14.7	0	7	0
56	1945						13.9	19.6	10.6	14.7	0	>200	0
60	1950	38.7	2.4	20		YES	5.4	5.0	7.6	6.0	0	3	0
8	1940	30.5	19.8	15		NO	59.4	82.1	63.7	68.4	0	6	0

Table A-9. (continued).

WELL INFORMATION
AREA 2

WELL NO.	YEAR DRILLED	WELL DEPTH M	DEPTH TO WATER M	DIA. CM	YIELD L/MIN	GROUTED	NO3-N FIRST SAMPLING (7/19-8/12/82) MG/L	NO3-N SECOND SAMPLING (1/9-1/13/83) MG/L	NO3-N THIRD SAMPLING (7/18-7/29/83) MG/L	NO3-N AVG. MG/L	BACTERIA (2ND SAMP.) FECAL COLI #/100ML	BACTERIA (2ND SAMP.) FECAL STREP #/100ML	BACTERIA (2ND SAMP.) TOTAL COLI #/100ML
85		3.5	3.4	30	11		4.1	8.1	46.6	19.6	0	5	0
135	1938						0.1	0.0	0.0	0.0	0	1	0
11	1976	41.1		12		YES	0.0	0.0	0.6	0.0	0	1	0
174								10.5	2.9	6.7	0	86	0
12	1944	24.4		15	3	NO	0.1	0.0	2.5	0.9	0	5	26
83	1974	32.9	21.3	12	45	NO	0.0	0.0	0.4	0.1	0	23	0
9	1927					NO	0.0	0.0	0.1	0.0	0	11	2
1	1940	25.9	4.6	12	34	YES	8.6	5.5	7.8	7.1	0	12	0
3		32.0	18.9	15		YES	9.7	11.1	10.8	10.5	0	2	0
7	1972	54.9	33.5	15	71	YES	0.0	0.0	0.2	0.1	0	8	0
61	1980	14.6	9.1	15		YES	3.9	25.4	3.2	10.8	0	17	120
170								0.0	0.0	0.0	0	3	0
155	1972	85.3	80.8	10	45	NO	0.1	0.0	0.4	0.2	0	4	0
10	1976	52.4	2.4	15	30	YES	0.0	0.0	0.4	0.1	0	5	0

Table A-10. Concentration data for farm water supply wells (area 3).

WELL INFORMATION
AREA 3

							NO3-N				BACTERIA (2ND SAMP.)		
WELL NO.	YEAR DRILLED	WELL DEPTH M	DEPTH TO WATER M	DIA. CM	YIELD L/MIN	GROUTED	FIRST SAMPLING (7/19-8/12/82) MG/L	SECOND SAMPLING (1/9-1/13/83) MG/L	THIRD SAMPLING (7/18-7/29/83) MG/L	AVG. MG/L	FECAL COLI #/100ML	FECAL STREP #/100ML	TOTAL COLI #/100ML
126	1900	18.3		15		NO	18.1	19.0	24.0	20.4	0	0	0
124							0.9	0.9	0.9	0.9	0	4	0
165	1902	30.5	24.4	15	37	YES	14.8	1.3	9.9	8.7	0	26	0
95		27.4	21.3	10		YES	0.1	0.0	0.4	0.2	0	1	0
93	1945	30.5	18.3	12			0.0	0.0	0.3	0.1	0	2	0
94	1954	36.6	21.3	15	94	NO	0.0	0.0	0.3	0.1	1	3	0
90		30.5					0.0	0.0	0.4	0.1	0	7	0
92	1980	43.6	13.7	12		YES	0.0	0.0	0.2	0.1	0	23	0
166								0.0	0.5	0.2	0	1	0
127	1905			15		NO	15.9	11.6	19.4	15.6	28	38	0
125	1942	13.7		15		NO	12.9	13.3	2.9	9.7	0	64	0
149	1956	39.0		12		YES	9.9	0.0	5.5	5.1	0	5	1
160	1940	27.4	15.2	15		NO	0.0	0.0	0.2	0.1	0	6	0
122	1915	24.4	7.6	15	94	NO	0.0	0.0	0.3	0.1	0	13	0
147	1930	30.5	13.7	15	30	YES	0.0	0.0	0.4	0.1	0	2	0
123							0.0	0.0	0.3	0.1	0	3	0
121		39.3		15		NO	0.0	0.0	0.3	0.1	0	1	0
164	1964						0.0	0.0	0.6	0.2	0	3	0
96	1920	61.0	6.1	15	49	NO	0.0	0.0	0.6	0.2	0	62	0
91							0.0	0.0	0.5	0.2	0	1	2
97	1915	59.4	25.9	15		NO	0.4	0.1	1.0	0.5	0	2	0
128							0.0	0.0	0.2	0.1	0	59	0
129		19.3					0.0	0.0	0.4	0.1	0	2	1
145							0.0	0.3	0.4	0.2	0	77	0
131	1980	24.4	14.6	10	45	YES	0.0	0.0	0.5	0.2	0	0	0
119	1956	39.3	10.7	15		NO	0.0	0.0	0.9	0.3	0	3	0
120	1957	67.1	22.9				0.0	0.0	0.1	0.0	1	1	3
159	1920	30.5				NO	0.0	0.0	0.0	0.0	0	3	0
118							0.0	0.0	0.5	0.2	0	4	0
99	1905	24.4	18.3	15		NO	0.0	0.0	1.6	0.5	0	2	0
108							0.0			0.0			
98							0.0	0.0	0.8	0.3	0	10	0
130							0.0	0.1	0.1	0.1	0	0	0
150							0.0	1.7	0.3	0.7	0	1	0
163	1953	41.1		10		NO	28.3	32.0	19.9	26.7	0	60	0
132	1915	15.2	6.7	15		NO	0.0	0.0	0.2	0.1	0	0	0
133	1952	41.1	10.7	15	22	YES	28.6	28.3	30.1	29.0	0	1	1
103							0.2	0.1	0.2	0.2	0	24	0
117	1900	22.9	5.5	15		NO	0.0	0.0	0.2	0.1	0	6	0
104	1963	62.2	51.8	15	56	NO	0.0	0.0	2.0	0.7	0	1	0
115							0.1	0.0	0.5	0.2	0	1	0
105							0.0	0.0	0.8	0.3	0	1	0
106							0.0	0.0	1.6	0.5	0	2	0
114	1920	36.6	25.6	15		NO	0.2	0.0	0.6	0.3	0	3	0
113	1935	38.1	25.9	12		NO	0.2	0.0	0.5	0.2	0	5	0
107	1905	61.0	45.7	12		NO	0.0	0.0	0.4	0.1	0	2	0
112	1954	51.8	41.1	10		NO	0.1	0.0	0.6	0.2	0	3	0
109	1963	30.5	15.2	20		NO	0.0	0.0	0.8	0.3	0	3	0
151	1950	30.5		15		NO	1.5	0.0	3.3	1.6	0	>200	0
146							39.5	43.6	47.5	43.5	0	1	0
100							0.0	0.1	0.2	0.1	0	0	1

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Table A-10. (continued).

WELL INFORMATION
AREA 3

WELL NO.	YEAR DRILLED	WELL DEPTH M	DEPTH TO WATER M	DIA. CM	YIELD L/MIN	GROUTED	NO3-N FIRST SAMPLING (7/19-8/12/82) MG/L	NO3-N SECOND SAMPLING (1/9-1/13/83) MG/L	NO3-N THIRD SAMPLING (7/18-7/29/83) MG/L	NO3-N AVG. MG/L	BACTERIA (2ND SAMP.) FECAL COLI #/100ML	BACTERIA (2ND SAMP.) FECAL STREP #/100ML	BACTERIA (2ND SAMP.) TOTAL COLI #/100ML
144	1950	39.6	7.6	12		NO	0.0	0.0	0.1	0.0	0	26	45
102	1972	76.2	9.8	15		NO	0.0	0.0	0.2	0.1	> 80	11	> 80
101	1925	23.2	15.2	15		NO	0.0	0.0	0.1	0.0	0	3	0
116	1967	38.1		15			0.0	0.0	0.9	0.3	0	>200	0
111							0.1	0.0	0.5	0.2	0	60	0
110		30.5	15.2	12		NO	0.0	0.0	1.0	0.3	1	>200	0

XII. ADDENDUM (Pesticide Data: Water Supply Wells)

Tables AD-1, AD-2, and AD-3 present the results of pesticide analyses for 165 samples taken from water supply wells in July, 1983 (third sampling for NO_3-N)in areas 1, 2, and 3 as discussed in Section V. As shown in summary Tables AD-4 and AD-5, overall 44% (73 of 165) of the water supply wells sampled contained at least one herbicide at the detectable level of 0.1 µg/L or above; however, of the 87 detectable levels measured, all but 7 were below 1.0 µg/L.

Lasso (or alachlor) was detected in 25% of the water samples analyzed, with a maximum concentration of 19.6 µg/L measured. The next highest level measured was 2.3 µg/L, and the average concentration in samples in which alachlor was detected (excluding the 19.6 value) was 0.3 µg/L. Atrazine was detected in 24% of the water samples analyzed, with a maximum concentration of 17.2 µg/L measured. The next highest level measured was 2.8 µg/L and the average concentration in samples in which atrazine was detected (excluding the 17.2 value) was 0.3 µg/L. The herbicides cyanazine, metribuzin, and propachlor were detected in only three, two, and two samples, respectively, and all at values less than or equal to 0.5 µg/L.

Table AD-6 presents the results of a special sampling in August of 1984 of select wells that provided the basis for a comparison of the results from the University Hygienic Laboratory at Iowa City (where the samples of drainage to ADWs were analyzed) with results from the ISU Agricultural Engineering Laboratory at Ames (where the water supply well samples were analyzed). The wells sampled were chosen based on the desire that some herbicide be present to be detected and the results of the July 1983 sampling. The high value (17.2 μg/L) for atrazine in well no. 25 in July 1983 was again measured in August 1984, with good agreement between the two laboratories. These high values for this particular well may possibly result from local contamination. At the time of the August sampling a container of pesticide adjuvant was observed setting on the well platform, a line of colored material was visible on the gravel near the well, where apparently a sprayer was tested, and there was a faint odor in the air resembling the smell of pesticide formulations. An attempt was made to also resample well no. 14, with its high alachlor concentration, but the well owner was not at home. It is interesting to note, as measured by the ISU lab, that all eight of the wells sampled in 1984 had detectable levels of atrazine (six did in 1983), and none of the wells had detectable levels of alachlor (while three did in 1983).

In a preliminary sampling of only 38 randomly selected water supply wells from the three areas (during the second sampling for NO_3-N, January 1983), no alachlor, metribuzin, or propachlor above detectable levels (0.1 μg/L) was found in any of the samples. Of the 21 samples analyzed for atrazine, 11 contained detectable amounts (maximum, 0.3 μg/L; average, 0.2 μg/L). Of these 11 wells, five were determined to have measurable levels of atrazine six months later, whereas nine of the ten not contaminated in January were still not found to be contaminated in July.

A comparison of herbicide concentrations in water supply well samples in Tables AD-1 through AD-3 with those given in Table II-2 and

Tables A-1 through A-5 for drainage to ADWs shows that, when detected, concentrations of herbicides are generally in the same low (< 1.0 μg/L) range, when surface runoff is excluded. However, it is not clear what the dominant source is for the low levels found in water supply well samples.

As shown in Table AD-4 and Figure AD-1, the locations of the wells that had samples with detectable levels of herbicides were not limited to areas that had ADWs in the vicinity, although the highest percentage of wells with contaminated water (56%) did come from area 1 with the most ADWs, and the lowest percentage (38%) was for the area with no ADWs. As is also shown in Table AD-4 and Figure AD-1, depth-to-bedrock was not a dominant factor either, with, overall, slightly more wells in areas where depth-to-bedrock exceeded 15 m being contaminated than those in areas where depth-to-bedrock was less than 15 m.

Figure AD-1 suggests that the contaminated wells are grouped to some degree. For example, in area 1, most of the contaminated wells are in a diagonal from northwest to southeast, with most of the uncontaminated wells near the border of the sampling area; the same might be said for area 2. For area 3, most of the contaminated wells are in the eastern half of the sampling area and more remote from the ADWs. The loose grouping in area 1 would indicate that ADWs are a probable source of some of the herbicide contamination; but in area 3, the grouping would indicate that neither ADWs nor shallow soils within (or close to) the sampled area are responsible for the contaminated groundwater (recharge water from surface drainage to one of the quarries in the county might explain the contamination in area 3, but considerably more work would have to be done to actually determine the source of contamination.)

While the NO_3-N data strongly indicated that drainage to ADWs was deteriorating the quality of groundwater at least locally, Table AD-4 and Figure AD-1 show there is little or no correlation between herbicide contamination and high NO_3-N levels. In two of the three areas, and overall, average NO_3-N concentrations for herbicide contaminated samples were less than for samples with no detectable herbicides.

The comparison of results between the University Hygienic Laboratory (UHL) and the Agricultural Engineering Laboratory (ISU) shown in Table AD-6 shows reasonable agreement for these low levels, when identical water samples were analyzed by both laboratories. It should be noted that two of eight samples (no. 45 and 55) in addition to atrazine were found to contain dyfonate (or fonofos) at 0.2 μg/L by the UHL; fonofos was not quantified by the ISU laboratory. Unfortunately, for the sake of comparison, none of the August 1984 samples contained detectable amounts of alachlor. However, a comparison of four extracts saved from the July 1983 sampling were also made, and the results are shown in Table AD-7. In some cases, agreement was quite good; for others, unexplainably, it was not so good; but at these low levels these results are not uncommon when different laboratories' results are compared.

In summary, overall the results of sampling 165 water supply wells for herbicides do not provide a clear-cut answer relative to the impact of ADWs on groundwater quality. While groundwater contamination in the area of ADWs was observed, the level and extent of herbicide contamination was not related to NO_3-N contamination, which is believed to be locally related to drainage to ADWs, and similar herbicide contamination was observed in an area remote from ADWs.

Table AD-1. Herbicide concentration data for farm water supply wells (area 1).

Well No.	Depth-to-Bedrock x equals > 15 m	Atrazine	Bladex (cyanazine)	Lasso (alachlor) µg/L	Sencor (metribuzin)	Ramrod (propachlor)	NO_3-N mg/L
39	x	*		0.1			0.1†
37	x						
36	x			0.1			4.6
35	x						
32	x						
33	x						
34	x	0.1					3.4
138	x			0.1			0.0
137A	x	0.1					14.1
137B	x	NS**	NS	NS	NS	NS	NS
38	x						
70	x			0.2			2.7
73	x			0.1			4.0
69	x						
47	x	0.2					6.6
48	x	0.2					4.7
175	x	NS	NS	NS	NS	NS	NS
30	x						
29	x						
31	x	0.1		0.1			9.3
136	x						
64	x	0.1					0.9
74	x						
42	x			0.4			
43	x	0.2					5.2
68	x			0.1			5.6
67	x	0.1		0.3			5.0
51	x						
49	x						
45		0.3		0.5			16.4
46		0.3					33.9
27				0.3			19.2
28	x						
24				0.1			13.5
63	x			0.1			12.6
25	x	17.2					15.8
40	x						
41	x			0.1			0.3
161	x						
44	x			0.1			4.8
66	x						
50	x	0.3		0.2			6.7
75							
65							
72							
71				0.3			11.9
26	x						

*A blank means < 0.1 µg/L.

†NO_3-N concentration from July 1983 sampling (from Table A-8).

**NS represents no sample.

Table AD-2. Herbicide concentration data for farm water supply wells (area 2).

Well No.	Depth-to-Bedrock x equals > 15 m	Atrazine	Bladex (cyanazine)	Lasso (alachlor) µg/L	Sencor (metribuzin)	Ramrod (propachlor)	NO_3-N mg/L
52		0.6	*				17.9†
53							
14			0.3	19.6			9.4
15							
139							
17							
173		1.6					4.8
76		2.8					2.1
20	x						
23	x	0.9					15.4
21	x						
22	x						
140	x						
157	x						
152	x						
79	x						
158							
54		0.2					16.1
16							
18	x						
89	x			0.2			7.5
88	x			0.3			1.2
19	x			0.1			6.1
171	x						
77	x						
153	x	0.1					14.0
78	x						
81							
82							
142	x						
143	x						
55	x	0.2	0.2				32.0
58	x						
4		0.1		0.2			34.4
57							
59							
87	x			0.1			0.8
156	x	NS**	NS	NS	NS	NS	NS
141	x			2.3			0.3
86	x			0.1			0.2
134	x			0.3			0.0
154	x			0.2			0.2
80	x	0.2		0.1			0.2
84							
13	x						
2							
5				0.6			13.3
6							
56							
60							
8		1.3					63.7
85	x	0.2					46.6
135	x			0.3			0.0
11	x	0.2		0.2			0.0
174	x						
12		0.3		0.2			2.5
83				0.1			0.4
9	x						
1		0.1	0.2				7.8
3							
7							
61							
170	x						
155	x						
10	x	0.2					0.4

*A blank means < 0.1 µg/L.
†NO_3-N concentration from July 1983 sampling (from Table A-9).
**NS represents no sample.

Table AD-3. Herbicide concentration data for farm water supply wells (area 3).

Well No.	Depth-to-Bedrock x equals > 15 m	Atrazine	Bladex (cyanazine)	Lasso (alachlor) μg/L	Sencor (metribuzin)	Ramrod (propachlor)	NO_3-N mg/L
126	x						
124	x						
165	x						
95	x						
93	x				0.1		0.3†
94	x						
90	x						0.2
92	x	0.1					
166	x						
127	x						
125	x						
149	x						
160	x						
122	x						
147	x						0.3
123	x	0.2					
121	x			1.4	0.5	0.1	0.6
164	x			0.1			0.6
96	x						
91	x						
97	x						
128	x						
129	x			0.2			0.4
145	x						
131	x						
119	x						0.1
120	x	0.1					
159	x					0.2	0.5
118	x			0.1			1.6
99	x	NS**	NS	NS	NS	NS	NS
108	x						0.8
98	x	0.1					
130	x						
150				0.1			19.9
163							
132	x						
133							
103	x						
117	x						2.0
104	x	0.2					0.2
115	x	0.3					
105	x			0.2			1.6
106	x	0.1					0.6
114	x	0.2					0.5
113	x	0.2					0.4
107	x	0.1					
112	x			0.1			0.8
109	x						
151							
146							
100	x						
144	x						
102	x						
101	x						0.9
116	x	0.1					0.5
111	x	0.2		0.1			1.0
110	x						

*A blank means < 0.1 μg/L.

†NO_3-N concentration from July 1983 sampling (from Table A-10).

**NS represents no sample.

Table AD-4. Effect of area and depth-to-bedrock on herbicide contamination of water supply well samples, and relation to NO_3-N contamination.

Area	Water Samples Containing Herbicide <15 m	>15 m	All	Avg. NO_3-N Concentrations Samples with Herb.	All
	----------------	%	---------------	---------- mg/L	---------
1	62	54	56	8.1	10.8
2	38	46	42	11.0	9.2
3	20	39	38	1.6	3.3
Overall	40	46	44	7.3	7.7

Table AD-5. Contamination of water supply well samples by individual herbicides.

Herbicide	Samples Contam.* %	Maximum	Average	Average (minus maximum)
		µg/L		
Atrazine	24	17.2	0.8	0.3
Bladex (cyanazine)	2	0.3	0.2	0.2
Lasso (alachlor)	25	19.6	0.7	0.3
Sencor (metribuzin)	1	0.5	0.3	0.1
Ramrod (propachlor)	1	0.2	0.2	0.1

*165 samples taken.

Table AD-6. Comparison of results for August 1984 water supply well samples.

Well No.	Lab.*	Atrazine	Bladex (cyanazine)	Lasso (alachlor)	Sencor (metribuzin)
		µg/L			
5	UHL	0.1	<0.1	<0.1	<0.1
	ISU	0.3	<0.1	<0.1	<0.1
	(July, '83)	(<0.1)†	(<0.1)	(0.6)	(<0.1)
6	UHL	<0.1	<0.1	<0.1	<0.1
	ISU	0.2	0.1	<0.1	<0.1
	(July, '83)	(<0.1)	(<0.1)	(<0.1)	(<0.1)
25	UHL	25.0	<0.1	<0.1	<0.1
	ISU	22.0	0.1	<0.1	<0.1
	(July, '83)	(17.2)	(<0.1)	(<0.1)	(<0.1)
45	UHL	0.4	<0.1	<0.1	<0.1
	ISU	0.3	<0.1	<0.1	<0.1
	(July, '83)	(0.3)	(<0.1)	(0.5)	(<0.1)
55	UHL	0.2	<0.1	<0.1	<0.1
	ISU	0.7	0.1	<0.1	<0.1
	(July, '83)	(0.2)	(0.2)	(<0.1)	(<0.1)
76	UHL	0.3	<0.1	<0.1	<0.1
	ISU	0.7	<0.1	<0.1	<0.1
	(July, '83)	(2.8)	(<0.1)	(<0.1)	(<0.1)
106	UHL	<0.1	<0.1	<0.1	<0.1
	ISU	0.4	<0.1	<0.1	<0.1
	(July, '83)	(0.1)	(<0.1)	(0.2)	(<0.1)
115	UHL	<0.1	<0.1	<0.1	<0.1
	ISU	0.5	0.1	<0.1	<0.1
	(July, '83)	(0.3)	(<0.1)	(<0.1)	(<0.1)

*UHL represents the University Hygienic Laboratory; ISU the Agricultural Engineering Laboratory.

†Results for the July 1983 sampling (ISU analyses) are given for comparison and to show the effect of time.

Table AD-7. Comparison of results for July 1983 water supply well samples.

Well No.	Lab.*	Atrazine	Bladex (cyanazine)	Lasso (alachlor)	Sencor (metribuzin)
		----------	---------- μg/L	----------	----------
1	UHL	<0.1	0.2	<0.1	<0.1
	ISU	0.1	0.2	<0.1	<0.1
14	UHL	<0.1	<0.1	1.5	<0.1
	ISU	<0.1	0.3	19.6	<0.1
45	UHL	0.2	<0.1	0.3	<0.1
	ISU	0.3	<0.1	0.5	<0.1
76	UHL	<0.1	<0.1	<0.1	<0.1
	ISU	2.8	<0.1	<0.1	<0.1

*UHL represents the University Hygienic Laboratory (they analyzed for, but did not detect fonofos in any of the extracts); ISU represents the Agricultural Engineering Laboratory.

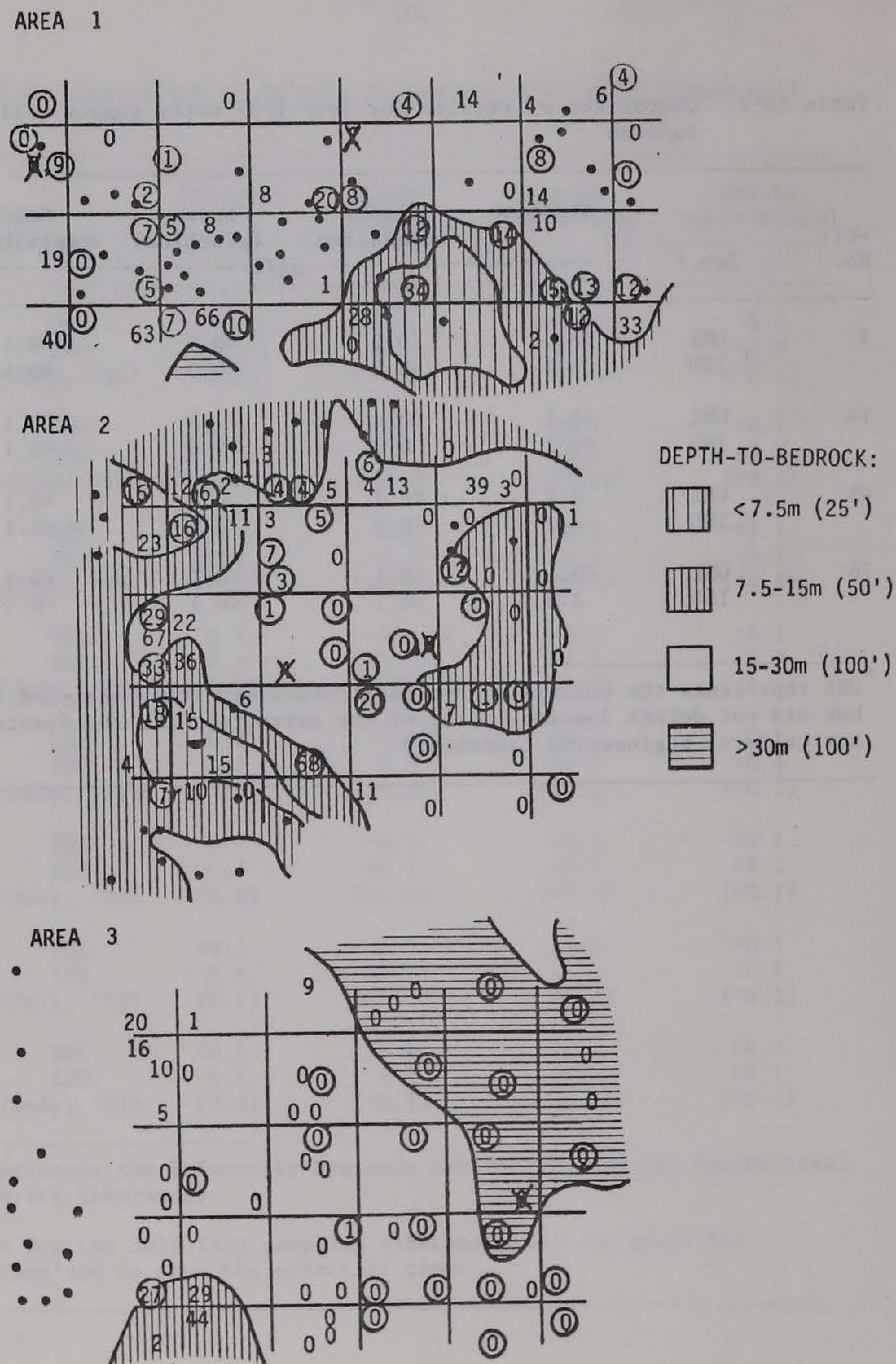

Figure AD-1. Location of ADWs (.'s), average NO_3-N concentrations, and whether herbicides were detected (circled = yes; x = not sampled) in water supply wells located at positions of the NO_3-N values.