Ground Water Monitoring&Remediation

Trends in Ground Water Nitrate Contamination in the Phoenix, Arizona Region

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Abstract

A 60-year database of nitrate concentrations from more than 200 wells in and around Phoenix, Arizona, was studied using geographic information system tools. This information, augmented with land-use maps, ground water levels, and well construction details, was used to examine ground water nitrate concentrations and changes in concentration and their relationship to land use, land-use changes, and changes in hydrologic conditions (as indicated by ground water level changes). Spatially integrated data for the 1960s to 1990s time period suggest slow and subtle temporal changes in the nitrate mass and overall average nitrate concentration in the aquifer, with increases associated with increased nitrogen fertilizer application rates. More recently, declines in nitrate concentration have been observed, which are thought to reflect the combination of declining ground water levels and transitions from agricultural to urban land use in many areas. Temporal trends in ground water nitrate concentration are presented for wells under desert, agricultural, and urban land use and desert-to-agriculture, desert-to-urban, and agriculture-to-urban land-use change scenarios.

Introduction

Nitrate (NO_3) contamination of aquifers is common in agricultural regions in the United States (Bouwer 1990), Europe (van Egmond et al. 2001), and China (Xu and Zhu 2001). Impacts also are observed in urbanized areas with high densities of septic tanks (Lowe et al. 2000). Many studies have been conducted on nitrate cycling in agricultural systems (e.g., Dowdell 1982; Barry et al. 1993; Hall and Risser 1993; Jordan et al. 1994; Hadas et al. 1999; Nkotagu 1996) and on modeling ground water nitrate concentrations (e.g., Tim and Jolly 1994; Geleta et al. 1994; Geng et al. 1996; Follett 1995). As noted by Drake and Bauder (2005), however, few studies have examined broad spatial and temporal patterns at the regional scale. Hallberg (1986) performed surveys in Iowa and Nebraska and identified some trends and mechanisms of ground water nitrogen change. More recently, Drake and Bauder (2005) used geographic information system (GIS) tools to study the relationship between urban development/land-use changes and spatial and temporal trends in nitrate concentrations over a 32-year period in Helena, Montana.

This study is most similar in focus to the Drake and Bauder (2005) study. In this case, a 60-year database of nitrate concentrations (expressed as nitrate-nitrogen, NO_3-N) from more than 200 wells in and around Phoenix, Arizona, was studied using GIS tools. This information, augmented with land-use maps, ground water levels, and well construction details, was used to examine ground water nitrate concentration changes and their relationship to land use, land-use changes, and changes in ground water levels.

The Phoenix metropolitan area has experienced significant population growth and accompanying land-use change. For example, the population roughly doubled from about 1 to 2 million residents between 1970 and 1990, and the Phoenix metropolitan area is currently one of the highest growth areas in the United States (historical census data are available at http://www.census.gov). Much of this growth has resulted in the urbanization of previously agricultural land and some desert areas. Humans have substantially altered the ground water system. Prior to 1980, there was substantial overdraft, resulting in rapidly declining ground water levels. Ground water levels have generally increased since passage of the 1980 Ground Water Act, which promotes long-term sustainability of the ground water system (Corkhill et al. 1993). The ground water system has also become impaired with salts and nitrate (Baker et al. 2004). Understanding spatial and temporal trends in ground water nitrate concentrations is therefore important to understanding long-term sustainability of the water resources of the region. To address this issue, a historical data analysis was performed. A mathematical subsurface transport model was then developed to predict future spatial and temporal changes in ground water nitrate concentrations with projected land-use changes in the Phoenix, Arizona area. The results of the historical analysis are reported here, and the mathematical model is discussed in Xu (2002).

This work builds on the studies by Wolf (1994), who hypothesized that ground water nitrate impacts were associated with cropland and dairies, and by Baker et al. (2001). Baker et al. developed a nitrogen mass balance model for the Phoenix ecosystem, leading to the conclusion that 15% to 20% of all ecosystem nitrogen inputs accumulated within the system in the mid-1990s and that nitrogen accumulation was occurring in the vadose zone and aquifers.

Study Area and Data Sources

The study area shown in Figure 1 is the 1000-km^2 Salt River Project (SRP) water service area. It encompasses the Phoenix metropolitan area and much of the surrounding agricultural land. As a semiarid area, the study area receives about 18-cm mean annual precipitation (Corkhill et al. 1993). In the past five decades, Phoenix has been experiencing rapid urbanization. In 1955, 72% of the current SRP service area was agricultural land. By 1995, agricultural land was only 21% of the total SRP area, while urban land accounted for 72%, as shown in Table 1 (Knowles-Yánez et al. 1999).

Historically, ground water has been an important source of water supply for the metropolitan Phoenix area. From the 1980s to late 1990s, the annual ground water pumpage has averaged about 10^9 m^3/year (1 million acre-feet) (Corkhill et al. 1993).

As shown in Corkhill et al. (1993), the regional aquifer consists of three alluvial layers. The upper layer has been substantially dewatered in many areas, and a long-term vertical flow regime has been established (Corkhill et al. 1993). The ground water flow system is complex and heavily influenced by seasonal pumping of ground water, but overall, the ground water flow is to the west across the study area. Reference is made occasionally to east and west basins (i.e., basins northeast and southwest of the Salt River, respectively), but the flow system is continuous.

Table 1
Land-Use Change in the Phoenix Metropolitan Area (Knowles-Yánez et al. 1999)

Time	Agricultural (km^2)	Urban (km^2)	Desert (km^2)
1912	722	19	302
1934	989	47	7
1955	748	293	2
1975	580	407	56
1995	217	751	37

The major inflows to the ground water system are irrigation recharge, recharge of water transported from the Colorado River, and through the typically dry bed of the Salt River when overflow water is released from upstream reservoirs. In general, the depth to ground water in this area slightly decreased from the 1960s to 1990s (Figure 2).

Ground water chemistry data were obtained from the SRP and the USGS. Most of the chemistry data for this study came from SRP: 4689 nitrate measurements from 249 SRP wells are available from the 1930s through the late 1990s. Eighty percent of SRP production wells are screened through two or three alluvial layers, with about 60% screened across the upper and middle layers. At SRP, nitrate was analyzed by the cadmium reduction method in the early 1980s and by ion chromatography from the mid-1980s onward. There is no record of analytical techniques used prior to the 1980s, but it is likely that cadmium reduction was used at least back to the 1970s. SRP developed a quality assurance program in the mid-1980s and has maintained this program to the present. As shown in Table 2, fewer wells were sampled prior to 1970. In more recent years, wells were typically sampled four times per year. Annual averages for each well were used in statistical analyses.

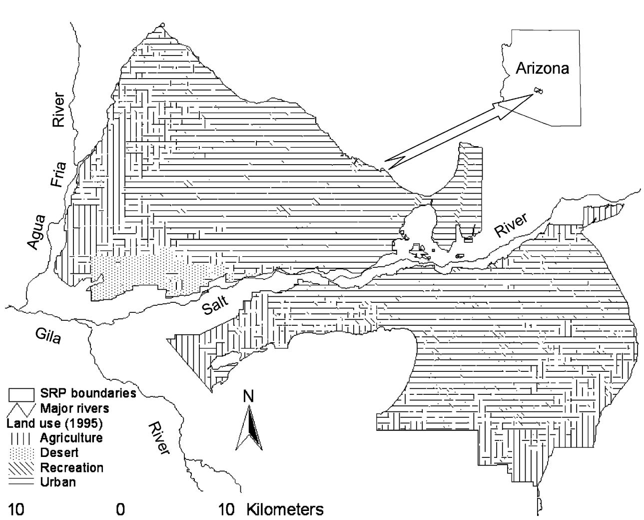

Figure 1. Phoenix metropolitan study area—SRP water service area.

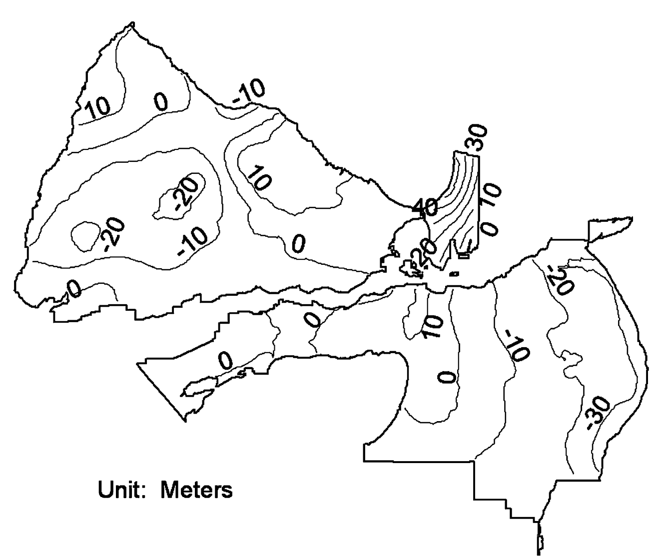

Figure 2. Change in depth to ground water from 1967 to 1997.

Table 2
Number of Wells in Study Area with Ground Water Level and Nitrate Concentration Measurements by Decade

Time	Ground Water Level	Nitrate Measurements
1930s	4	9
1940s	53	57
1950s	53	144
1960s	96	212
1970s	220	223
1980s	223	235
1990s	194	243

Data used in this study were obtained from several sources. Historical land use and census data were downloaded from the Central Arizona—Phoenix Long-Term Ecological Research (CAPLTER) database. Digitized land-use maps were available for 1912, 1934, 1955, 1975, and 1995 (Knowles-Yánez et al. 1999). Land use was categorized into four generic types: agriculture, urban, desert, and recreation. In the land-use map of 1995, agricultural lands are recognized as crops, citrus, and stockyards.

Digital elevation models were downloaded from the USGS Web site (http://www.usgs.gov). Ground water level data (depth below ground surface) were obtained from the USGS. Well locations from the SRP and USGS, originally in decimal degree coordinate, were projected into the Universal Transverse Mercator (UTM) system with ArcView Projector (distributed by Environmental Systems Research Institute Inc., Redlands, CA) to match other maps used in this study. Well pumping data for 3099 wells from 1984 to 1997 were obtained from the Arizona Department of Water Resources. This represents nearly all of the ground water pumping in the valley.

Methods

To obtain an overall pattern of ground water nitrate concentrations, ArcView was used to interpolate the point nitrate measurements. The interpolation method used for this study is kriging. By employing ordinary kriging, ArcView generated the interpolated surface as well as the surface of interpolation variance. Interpolations of NO_3-N concentration values and ground water levels were done separately for the East and West basins. Kriging outputs (NO_3-N concentration and ground water level) were used to estimate the total mass of NO_3-N in the aquifer. Bedrock elevations were obtained from Corell and Corkhill (1994).

Since the same cell size was used in all kriging maps, the NO_3-N mass beneath a given area was calculated by:

$$M = D^2 \sum_{i=1}^{n} C_i B_i \phi_i \tag{1}$$

where M is the total ground water NO_3-N mass beneath a given area (mass), D is the grid cell size (length), C_i is the concentration in cell i (mass/volume), B_i is the aquifer thickness in cell i (length), and ϕ_i is the soil porosity in cell i (volume pores/volume soil). The weighted NO_3-N concentration beneath a given area was calculated by:

$$\overline{C} = \frac{\sum_{i=1}^{n} C_i B_i \phi_i}{\sum_{i=1}^{n} B_i \phi_i} \tag{2}$$

where $\overline{C}$ is the average concentration for all cells of a given area (mass/volume). There are two dominant transient land-use histories in the Phoenix area: (1) desert land developed for agricultural use and (2) agricultural land redeveloped for urban use. ArcView was used to identify land use associated with a particular well in each time interval.

Average cropland fertilization rates for Arizona cropland were obtained from Doerge et al. (1991) for the period 1938 to 1988. Jarvis (1993) and Wolf (1994) hypothesized that stockyards may be major sources of ground water nitrogen. To evaluate this hypothesis, all wells located within 300 m of the land use "stockyard" in the SRP area were identified. Wolf (1994) indicated that most of the dairies found in the eastern part of the SRP area have been at their present locations for more than 20 years; however, historical records supporting that statement were not available for this study. In this work, the comparison between stockyard wells and other agricultural wells was conducted based on the 1995 land-use map as that was the only map with stockyard locations.

Time of travel through the vadose zone was estimated from irrigation rate, irrigation efficiency, soil porosity, porous water saturation, and the long-term average infiltration velocity:

$$V = Q \times (1 - f)/(\phi \times s) \tag{3}$$

where V is the average linear infiltration velocity (length/time), Q is the irrigation rate (volume/area time), f is the irrigation efficiency (dimensionless), ϕ is the soil porosity (volume pores/volume soil), s is the porous water saturation (volume water/volume pores), and:

$$T = H/V \tag{4}$$

where T is the travel time (time) and H is the distance between ground water and ground surface (length).

The travel time through the vadose zone varies with soil characteristics, depth to the aquifer, and irrigation practices. Corell and Corkhill (1994) indicated that the upper unit of the valley basin deposit is mainly sand, silt, and gravel. For all calculations, a porosity of 0.44 (Rawls et al. 1983) and a water saturation of 15% (Environmental Systems and Technologies Inc. 1995) were assumed.

Results

Major Spatial and Temporal Patterns

A set of well-specific ground water NO_3-N concentration maps (Figure 3) were created for four representative

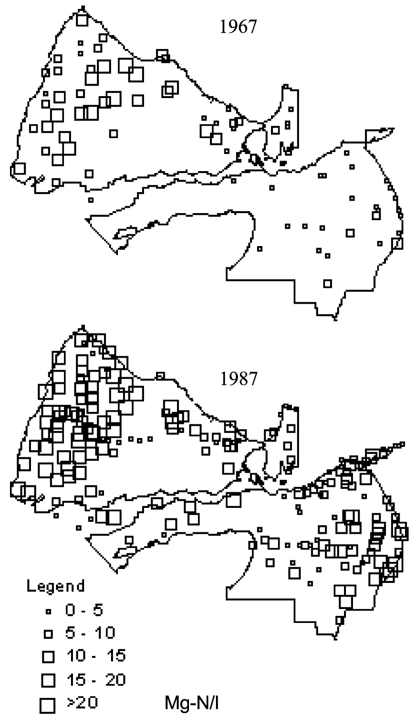

Figure 3. Spatial patterns of ground water NO_3-N (mg-N/L) concentrations in the Phoenix metropolitan area.

years (1967, 1974, 1987, and 1998) with relatively large number of sampled wells. Wells were mapped based on their annual average NO_3-N concentration. More wells were sampled in recent years, and this is reflected on the maps. The maps show slightly increasing NO_3-N concentration trends from the 1960s to 1980s and no discernible trend from the 1980s to 1990s. NO_3-N concentrations are generally higher in the northwestern and southeastern regions of the study area, and these areas correspond to agricultural land use. Concentrations are generally lower in the more urbanized central region, which also has a lower well density.

Krigged NO_3-N concentrations (Figure 4) reveal similar pattern. Spatially integrated regional average NO_3-N ground water concentrations and NO_3-N mass in ground water reveal the overall trends (Figure 5). The total ground water NO_3-N mass and regional average concentration declined from 1950 to the 1960s, they remained relatively stable through the 1970s, and then they increased until the mid-1990s. In the mid-1990s, the trend appears to reverse, suggesting another decline in NO_3-N, and this is discussed in more detail subsequently. Accumulation of nitrate from 1980 to the mid-1990s is supported by an overall ecosystem N balance, which reveals accumulation of N in the Central Arizona—Phoenix ecosystem (Baker et al. 2001).

Background Nitrate Concentrations in Desert Areas

A database search was conducted for wells located hydraulically upgradient of areas impacted by agricultural development or urbanization. Only one SRP well located in a noncultivated desert area with historical data prior to 1960 was identified. This well is located in the

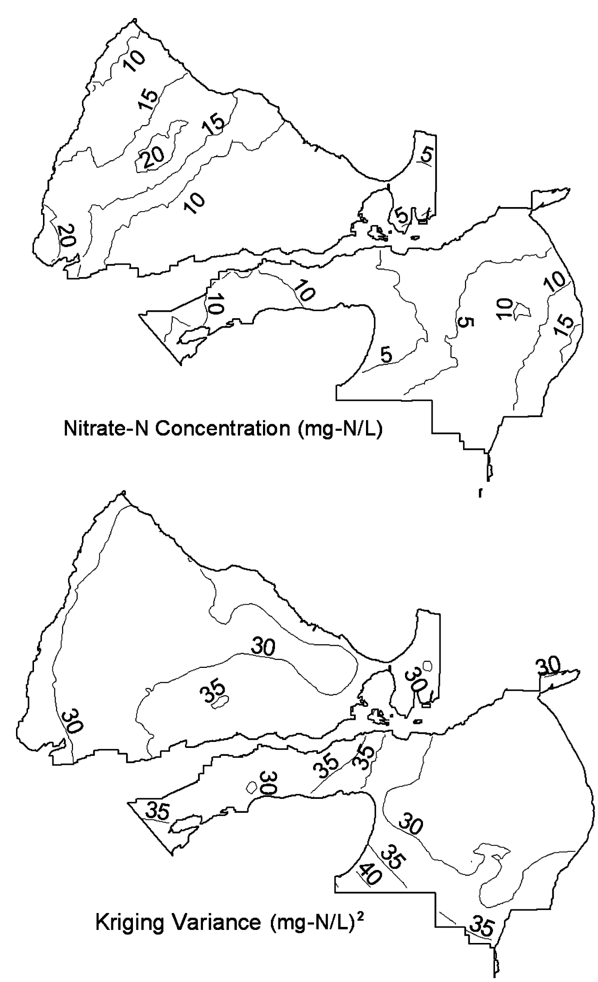

Figure 4. Krigged NO_3-N concentration map for 1996 and the variance map.

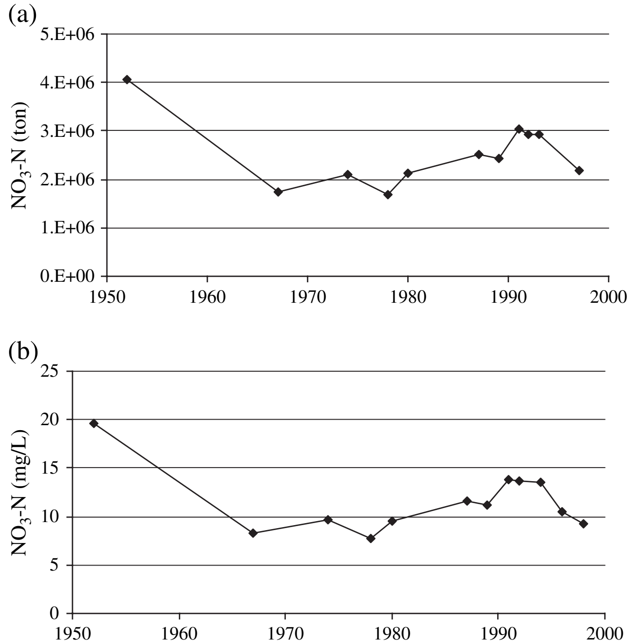

Figure 5. Trends of (a) kriged total ground water NO_3-N mass and (b) depth-weighted average NO_3-N concentrations beneath the Phoenix metropolitan area.

northeastern part of the study area. Historical records of this well show that the NO_3-N concentration was always less than 5 mg/L and always less than 1 mg/L before the early 1980s. There are six wells located immediately outside the SRP area that have remained in desert land-use areas since the 1970s. The average NO_3-N concentrations for these wells were 3.6 mg/L (1970s), 4.3 mg/L (1980s), and 3.8 mg/L (1990s). In summary, NO_3-N concentrations in wells located in the undisturbed desert are low and exhibit no clear upward or downward trend.

Effect of Agricultural and Urban Land Development

The historical land-use maps were overlaid to identify long-term (1912 to 1995) urban areas. NO_3-N concentrations within those areas were interpolated for selected years (those with more than 50 measurements) using kriging.

Figure 6 shows that average NO_3-N concentrations for both agricultural and urban lands were around 20 mg/L in the 1950s. Then, both dropped below 10 mg/L in the mid-1960s. From the mid-1960s forward, the average NO_3-N concentrations under agricultural lands were generally 5 to 6 mg/L greater than the average concentrations under urban lands. The peak average NO_3-N concentration for urban areas in the 1990s was 12.1 mg/L, whereas under agricultural land use the peak average NO_3-N concentration was 17.9 mg/L. For the 1990s, the difference between agricultural and urban land uses is significant at the 0.1 level (t-test). NO_3-N concentrations in wells located in long-term (from 1955 on) urban and agricultural land uses were elevated by at least two to three times relative to the background ground water NO_3-N concentration in desert area wells.

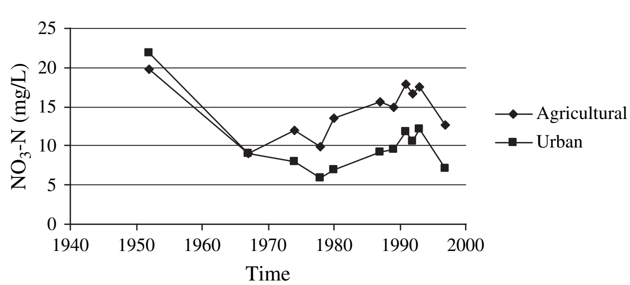

Figure 6. Weighted NO_3-N concentrations beneath agricultural and urban lands in the Phoenix metropolitan area.

Nitrate Contamination as a Function of Ground Water Depth

This analysis was limited to wells with paired ground water depth and NO_3-N measurements located under areas that had been in continuous agriculture since the 1950s with no nearby dairies. The best year for this analysis was 1991, which had the greatest number of directly paired observations ($n = 15$). Most of these wells have NO_3-N concentrations greater than 10 mg/L. For these 15 wells, there is a strong negative correlation (Pearson coefficient $= -0.619$) between the NO_3-N concentration and depth to ground water level (i.e., higher nitrate concentrations tend to associate with shallower ground water levels; Figure 7). One possible explanation is that the greater travel time associated with a greater depth to ground water increases the lag between increased fertilizer application rate and increased NO_3-N concentration in the aquifer. Alternatively, longer vadose zone travel times may provide more opportunity for nitrogen loss (through transformation).

Effects of Long-Term Agricultural Activities

The six SRP wells shown in Figure 8 had long agricultural land-use histories and relatively stable ground water levels (within $\pm$ 3 m from the 1960s to 1990s). All were in agricultural land use by 1935, and all but one (#190) underwent conversion from agricultural land to urban land after 1975. The ground water NO_3-N concentration trend from the 1950s through the 1970s was slightly upward in four of these wells. Well #213 was variable, and well #101 had the lowest NO_3-N concentrations—usually lower than 5 mg/L before 1980. By the 1980s, the NO_3-N concentrations fluctuated for all wells; yet, a collectively upward trend can be

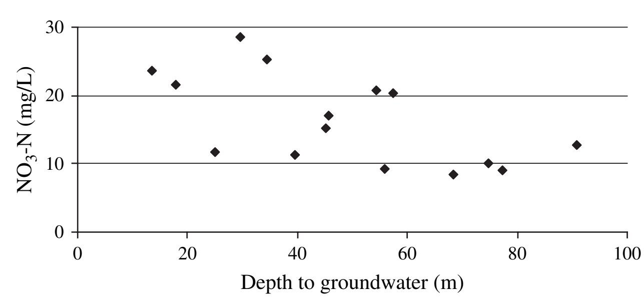

Figure 7. Correlation between ground water NO_3-N concentration and ground water level.

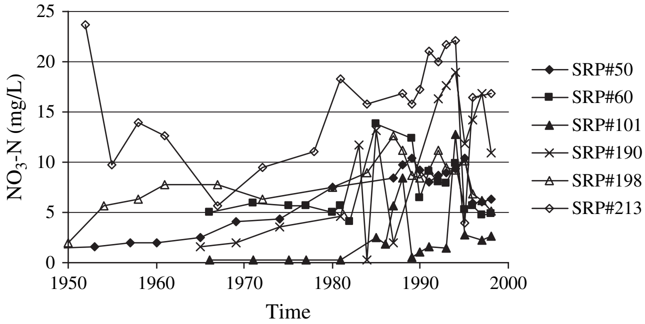

Figure 8. Ground water NO_3-N in wells with long agricultural histories.

observed from the 1970s to late 1980s. While fluctuation also occurred in the 1990s, the clustered data show a decreasing trend from the mid to late 1990s.

Effect of Dairies: Comparison of Dairy Impacted Sites with Other Agricultural Sites

There are many dairies located in the region and they have, on the average, about 1000 cows each. The production of manure in these concentrated operations is a major potential source of ground water contamination. Among all 19 continuous agricultural wells, seven wells were located near (within 300 m) land use identified as "stockyard" in 1995. Most of these stockyards were dairies (there are no current beef stockyards in the region). For the 1990s, the average NO_3-N concentration was 20.9 mg/L (standard deviation = 7.6 mg/L) for wells located near dairies and 15.4 mg/L (standard deviation = 6.7 mg/L) for agricultural wells not near dairies. The difference is significant at the 0.1 level (t-test). The difference also increases over time (Figure 9). This divergence may simply reflect the effect of travel time or perhaps intensification of the dairy industry over time (a trend toward more concentrated operations).

Ground Water Nitrate Trends during Land-Use Transitions

Wells under Transition: Desert to Agriculture

The four wells shown in Figure 10a are located in areas that were desert prior to 1955, were first farmed between 1955 and 1975, and have since remained in

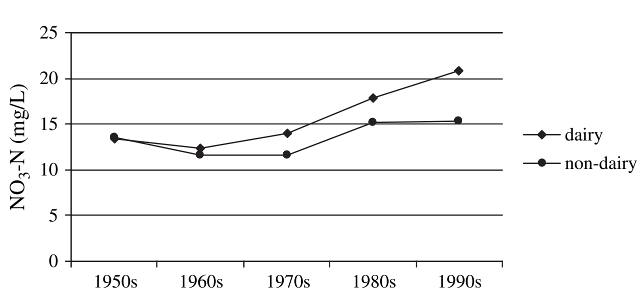

Figure 9. Historical comparison of ground water NO_3-N in wells located near dairies and in agricultural land but not near dairies.

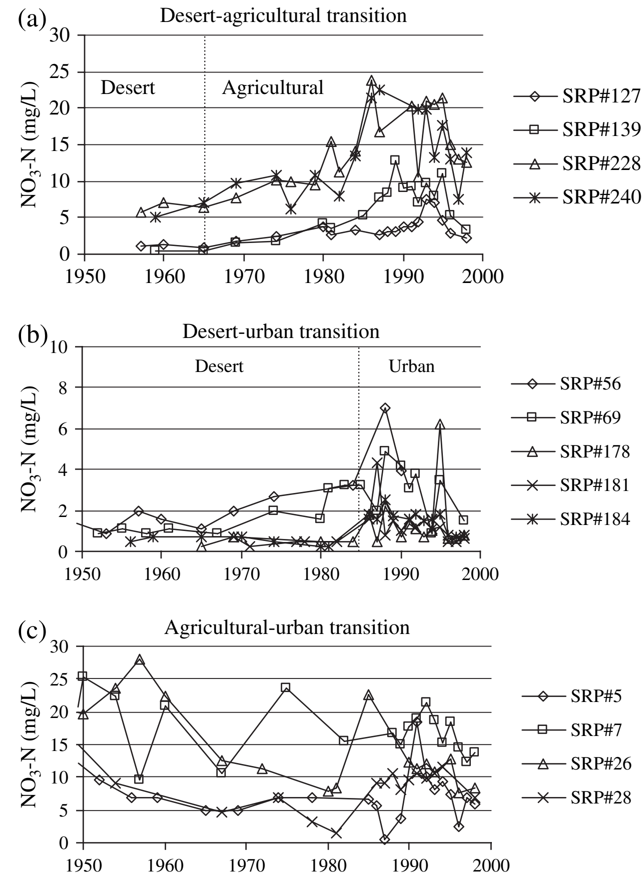

Figure 10. Effects on nitrate concentration during land-use transitions.

farmland. In all cases, NO_3-N levels increased after agricultural development. The increases were most pronounced for wells #228 and #240, which had water tables about 10 m below the surface and were less pronounced for wells #127 and #139, with water tables 60 to 80 m below the surface. The upward trend ceased between the late 1980s and early 1990s and was followed by a downward trend during middle to late 1990s.

Wells under Transition: Desert to Urban

The five wells shown in Figure 10b are located in areas that remained desert until 1975 and were then urbanized between 1975 and 1995. NO_3-N concentrations increased, but maximum concentrations were never greater than 8 mg/L and were generally less than 4 mg/L.

Wells under Transition: Agricultural to Urban

The four wells shown in Figure 10c are located on land that was urbanized from agricultural use in the period 1934 to 1955 and remained urbanized land use through 1995. All wells show decreasing trends during the 1950s and 1960s, increasing trends during the 1980s, and decreasing trends again in the 1990s.

Discussion

Spatial and temporal patterns of nitrate concentrations in the Phoenix area are controlled by several factors. The

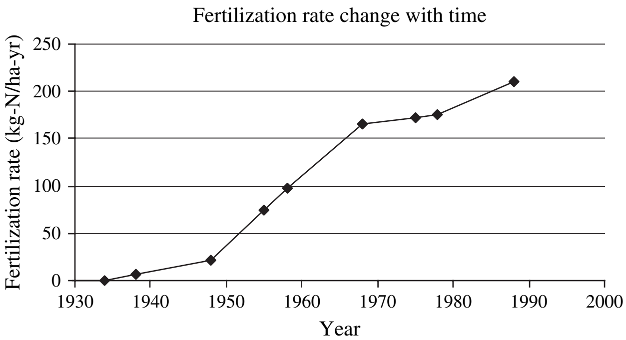

Figure 11. Average NO_3-N fertilization rates for cropland in Arizona (adapted from Doerge et al. 1991).

high nitrate concentrations in the 1930s and 1940s may represent a "first flush" of nitrate from desert soils following initiation of irrigation. Albus and Knighton (1998) and Beck (1983) also found that deteriorated ground water quality might result from accelerated mineralization when converting from dry land to irrigated production. It is possible that when the region was first irrigated, nitrate leached from soils and the vadose zone downward, increasing ground water nitrate concentrations quickly, at least under some circumstances. Leaching would subside eventually, and in the absence of fertilizer inputs, ground water NO_3-N concentrations would be expected to decline.

However, agricultural nitrogen fertilization rates for Arizona cropland increased after World War II, and particularly during the 1960s through the mid-1980s (Figure 11, based on Doerge et al. 1991), paralleling the national trend (National Academy of Science 1989). Using methods outlined in Baker et al. (2001), estimated total areal nitrogen input to agricultural land (total = commercial fertilizer + manure + waste water irrigation + waste water biosolids + atmospheric deposition) decreased by 35% between 1975 and 1995, from 350 to 225 kg/ha/year. Most of the decline was due to decreased application rates for commercial fertilizer.

Travel time through the vadose zone has also changed with time as a result of changes in irrigation rates/efficiency and depth to ground water. Average krigged depth-to-ground water levels and estimated irrigation efficiencies for each time interval were used to estimate travel times using Equation 3 (Table 3). It is recognized that travel times vary spatially, but the spatially averaged travel time is useful for broad comparisons.

The average depth to ground water of the measured SRP wells was only 24 m in the 1940s. Overdraft increased the average depth to ground water to 61 m by the 1960s. As a result of better water management and increased use of surface water sources, water tables have been rising, decreasing the average thickness of the vadose zone to 51 m in the 1990s. The estimated average vadose zone travel times in Table 3 suggest time lags of 2 to 6 years between changes in surface conditions (i.e., nitrogen application and irrigation rates) and nitrate fluxes to ground water.

Finally, changing land use affects ground water NO_3-N concentrations. Farming expanded from 1912, when irrigation water first became available, through the 1970s. Urbanization accelerated during the 1970s, with urban land being developed from conversion of both farmland and desert. Farmland, in particular, has declined in the SRP area. Total nitrogen application rates to urban land (total = septic tanks + waste water irrigation + pet waste + commercial fertilizer) have remained nearly constant, being about 125 kg/ha/year in 1975 and 112 kg/ha/year in 1995 (Baker et al. 2001). Areal nitrogen input for urban land was therefore one-third of that for agricultural land in 1975 and one-half of that for agricultural land in 1995. This is reflected in the lower NO_3-N concentrations observed in long-term urbanized vs. long-term agricultural land (Figure 6) and by the decline in NO_3-N that occurs when agricultural land becomes urbanized.

By 2030, the region is expected to be fully urbanized, with a population of about 6 million and virtually no remaining agriculture. The results of this study, along with nitrogen mass balance estimates (Baker et al. 2001), suggest that over the period of several decades, area average ground water NO_3-N concentrations will likely decline but not to background levels. Numerical model-based projections of future ground water NO_3-N concentrations in the study area are discussed in Xu (2002).

Acknowledgments

This project was supported in part by the SRP and the CAPLTER Program (NSF # DEB9714833).

Table 3
Estimated Travel Times to the Aquifer Based on Spatially Averaged Depths to Ground Water

	1940s	1950s	1960s	1970s	1980s	1990s
Average depth to aquifer (m)	24	44	61	60	56	51
Irrigation volume (m/year)	1.7	1.7	1.7	1.8	1.7	1.8
Irrigation efficiency[1]	0.62	0.62	0.62	0.62	0.62	0.62
Travel time (year)	2.5	4.5	6.2	5.8	5.7	4.9

[1]It is known that the irrigation efficiency improved with time, but no detailed efficiency data are available for this study. The efficiency data in this table are obtained from Corell and Corkhill (1994).

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