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Mixed Land Use Aquifer Protection from Nitrate and Sodium Contamination

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Abstract

Regression analysis is applied to estimate loadings of nitrate and sodium from various land uses in mixed land use aquifers in Massachusetts. The model is then used to illustrate how well-intended local groundwater protection policies which fail to recognize land use substitution and cross pollutant effects may be misdirected.

Key words: Groundwater, Pollution, Nitrates, Sodium, GIS

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Groundwater contamination is a complex problem in part because of the large number of potential pollutants and sources, as well as difficulty in predicting the movement of materials underground. Moreover, groundwater pollution often originates from diffuse or "nonpoint" sources. For the policy maker this creates special difficulties in identifying polluters and imposing regulatory solutions.

In many parts of the United States, a particular problem in formulating groundwater protection policies is the prevalence of mixed land-use aquifers, in which agricultural, residential, commercial, industrial and other land uses exist in close proximity. This type of setting, sometimes described as the "rural-urban interface", characterizes the periphery of many U.S. cities. It is particularly prevalent throughout the northeast. For the state or local policy maker charged with groundwater protection, such a situation makes it difficult to prioritize the risks associated with diverse human land use activities, and to arrive at an efficient protection strategy.

This paper presents an empirical method which may be useful in identifying the predominant land uses contributing to specific pollutants within a given region. The approach is statistical in nature, being based on a study of broad correlations between observed water quality variables and land use patterns over a wide geographical area. The tools employed are a Geographic Information System (GIS) and multivariate regression analysis. To illustrate the approach, we examine the contributions of various land uses to both nitrate and sodium in Massachusetts public groundwater wells.

Sodium and Nitrate Studies

The sources of nitrates in groundwater have been studied widely, and high on most lists are chemical fertilizer use, manure handling, animal feeding activities, cesspools, septic tanks, and sewage pipelines. Agricultural, commercial, industrial and residential land uses have been clearly identified as among the human activities responsible for cases of nitrate contamination. Katz and others (1980) found that sewered areas had lower nitrate concentrations in shallow groundwater than did places using cesspools or septic systems. Porter (1980) reported human waste and turf fertilization as the top two principal sources of nitrate in groundwater in Nassau County, Long Island.

Flipse and others (1984) reported Long Island findings that most nitrogen in groundwater came from cultivation, and that nitrate concentrations increased despite replacement of septic systems with sewer service. They also identified acid precipitation as a significant source of nitrogen.

On the Delaware Coastal Plain, Roberts (1979) found that confined animal feeding areas had the highest contribution to nitrate contamination, followed by septic tanks, natural and chemical fertilizers, and medium to high density residential areas. In the same region, Ritter and Chirnside (1984) found average nitrate concentrations highest near poultry houses and septic tanks.

Grady and Weaver (1988) and Grady (1989) studied Connecticut watersheds for sources of several chemicals. They noted that unsewered areas generally have lower housing densities. The lowest median concentrations of nitrate plus nitrite were found down-gradient from agricultural land uses, and the highest were found down-gradient from residential areas. Agricultural, residential, commercial and industrial uses all had higher nitrogen concentrations than undeveloped land.

Persky (1986) found that housing density accounted for almost two-thirds of the variation in nitrate levels on Cape Cod, where most communities engage in waste disposal on site. Noss (1988) examined 64 aquifers throughout Massachusetts and found that agriculture, non-sewered residential land and presence of landfills best explained variations in nitrate concentration.

Sources of sodium have also been studied in some detail. Most authors list saltwater intrusion or brackish water upconing, the use of deicing salts, and septic tanks and cesspools. Agricultural, residential, commercial and industrial land uses and well operations have been identified as among the human activities responsible for high sodium concentrations, although empirical studies indicate that greater precision is needed in defining these relations.

In his study of Connecticut watersheds, Grady (1989) found that urbanization results in elevated chloride concentrations. Agricultural and undeveloped land displayed similar concentrations of sodium and chloride, while residential, commercial and industrial land uses all exhibited increased concentrations. Septic tank densities and roads were seen as key factors responsible for these increased concentrations.

In his Massachusetts study, Noss (1988) identified sodium concentration as being a function of open land and wetlands, nonsewered residential lands, and industrial land. On Cape Cod, Persky (1986) found that correlations between housing density and sodium concentrations were not statistically significant. His findings suggested that wind borne ocean spray, road salting and saltwater contamination all had overwhelming influences. For example, pumping wells which are near ocean water beyond recharge capacity depletes the freshwater aquifer and causes saltwater to penetrate.

For noncoastal areas of Massachusetts, road salting is clearly the major cause of increased sodium levels in groundwater. Therefore, the amount of roadway in a given area and proximity of wells to the ocean in coastal areas should be leading factors for sodium in groundwater.

Approach

This paper attempts to shed light on the special problems of mixed land-use aquifers by presenting a technique for evaluating the relative contamination potentials of various land uses within a region's recharge areas. The approach employs a Geographic Information System (GIS) to identify and enumerate land uses within the designated recharge areas of 188 public water supplies for which water quality data (nitrate and sodium levels) are available. Multiple regression analysis is then applied to each contaminant to study the correlations between the contaminant level and the various land uses in the recharge area. The regression model isolates the effect of each land use. The regression coefficients represent the average tendency of each land use to elevate the level of the contaminant in question. They are thus similar in nature to average loading coefficients, though sometimes different in scale.

To illustrate the approach, we first develop a multiple regression model representing nitrate contamination in Massachusetts public water supplies. The model does not attempt to capture site-specific hydrogeological detail, but rather looks at empirical correlations between water quality and human land uses for a broad cross-section of public wells. Nitrate is an appropriate choice for this type of model, because several types of activities within a mixed land-use aquifer are believed capable of contributing to elevated nitrate levels. These include agricultural fertilizer and manure handling, septic systems, and lawn and garden fertilizers applied to both residential and public lands. Where nitrate is a potential problem, policy makers would like to have some idea which of these land uses are likely to be the more important contributors.

Table 1 provides descriptive statistics for the data used in the two regression models. The dependent variables, NO_3 and Na, are measured as concentrations in parts per million. The remaining are explanatory variables which denote land uses, represented as the area of each land use divided by total land area in the recharge cell. Thus, these exogenous variables are expressed as percentages of recharge areas and, under the assumption of constant recharge per acre, the fraction represents the percentage of recharge emanating from each land use. The minimum, maximum and median values are shown. The median is provided rather than the mean because some of the variables are not normally distributed. Also shown is the fraction of the 188 recharge areas that did not contain the particular land use.

Specifically, the nitrate equation to be estimated is:

$$(1)\quad NO_{3i} = \beta_0 + \beta_1\,ACWP_i + \beta_2\,AP_i + \beta_3\,RPUO_i + \beta_4\,R0R1_i + \beta_5\,R2SEW_i + \beta_6\,R2NOSEW_i + \beta_7\,R3SEW_i + \beta_8\,R3NOSEW_i + \beta_9\,UC_i + \beta_{10}\,UI_i + \beta_{11}\,UW_i + U_i.$$

The dependent variable is nitrate concentrations in municipal well i in parts per million. Alternate specifications are possible, and several were tested using another data set. The present formulation was selected because of those preliminary results and more conceptual considerations suggested by Noss (1988).

A second regression analysis examines sodium, another leading groundwater contaminant in the northeast region. Leading contributors of sodium are believed to be road salting, and deicing of residential and commercial areas.

The sodium equation is expressed as:

$$(2) \qquad NA_i = \beta_0 + \beta_1 CR_i + \beta_2 ACWP_i + \beta_3 AP_i + \beta_4 R0R1_i + \beta_5 R2_i + \beta_6 R3_i + \beta_7 UC_i + \beta_8 UI_i + \beta_9 UW_i + U_i.$$

The dependent variable is sodium concentration in municipal well i in parts per million, and as before explanatory variables denote land uses. With the exception of roads (CR), which are expressed in linear miles because areal representations were not possible from the data set, these exogenous variables are again expressed as percentages of each recharge area. Because the major residential source of sodium is deicing activities, it is not conceptually useful in this case to distinguish sewered and nonsewered residential areas, as was done in the nitrate model.

The United States Geological Survey (USGS) provided data on well location and annual water withdrawals for public water supplies in Massachusetts. The Massachusetts Department of Environmental Protection (DEP) supplied data on nitrate and sodium levels in municipal wells, and the Massachusetts Executive Office of Environmental Affairs (EOEA) provided GIS coverages. The water quality database obtained from DEP consisted of Safe Drinking Water Act contaminant data from 1971-1987. The most recent year of complete data (1986) was used for the estimation. Computer maps of land use information, half mile radius well buffers around public supply wells, and political boundaries were supplied by EOEA. Land use areas within the well buffers were extracted from these data.

Official studies defining the zone of contribution ("zone 2") have not been completed for most public groundwater supply wells in Massachusetts. In the interim, the state has designated the areas within a half mile radius of public water supply wells as the "interim protection areas" or well buffers, and these were used in our study. It is an interesting sidebar that when we performed the same analysis using data for those zone 2s which have been completed, the half mile buffer designation showed greater explanatory power than the formally defined zones of contribution. Thus for present purposes, if not for policy formulation, the interim buffer definition proved satisfactory.

Results

The least squares estimation of the nitrate model (1) is summarized in Table 2. Three coefficients were statistically significant: intensive agricultural land use (ACWP), unsewered medium density residential (R2NOSEW), and unsewered low density residential land use (R3NOSEW). These three variables had been expected to be the primary sources of nitrates on the basis of prior studies.

Based on these estimated coefficients, unsewered medium density residential land use appears to have on average the greatest impact on nitrate concentration. For each one percent increase in this land use in the buffer, one might expect an increase of 0.048 milligrams per liter of nitrate to be added to the recharge system. Intensive agricultural land use seems to have an impact of about half that magnitude, and unsewered low density land use somewhat less than half. The regression explained about 20 percent of the variation in nitrate levels in municipal wells throughout the state, and was statistically significant.

The summary of the estimated sodium model appears in Table 3. Three coefficients were again statistically significant: length of major roads (CR), medium density residential (R2) and commercial land use (UC). These three were expected to be the primary sources of sodium. The substantial difference in magnitude among the coefficients of CR and the others arises because CR is not measured as a percentage of land area, but rather in miles. Its coefficient therefore represents the estimated impact on sodium concentration measured at the well per mile of major roads within the buffer. Thus the addition of a mile of roads within the buffer would be estimated to increase sodium levels by about 2.7 milligrams per liter.

In contrast, our regression suggests that each one percent increase in commercial land use (UC), converted from a nonloading use, might increase the sodium levels in the aquifer by about 0.76 milligrams per liter, and each percent increase in medium density residential land use (R2) might cause an increase in groundwater sodium levels on the order of 0.30 milligrams per liter. Thus an increase in these two land uses by ten percent of the total land area might raise the sodium concentration by 7.6 and 3.0 parts per million, respectively. Clearly the coefficients for these two land uses must be interpreted differently from the coefficient for miles of major roads (CR) because of the differences in units. But the impacts are rather comparable--the ten percent increase in medium density land use, for example, represents a doubling of this area from the mean situation. An increase of one mile of major roads also approximately doubles the mean length of road. It would be reasonable then to compare the 3.0 parts per million for residential land use with the 2.7 for a "comparable" increase in roads, and to conclude that these two land uses have similar impacts on sodium. Finally, the regression model explained nearly 30 percent of the variation in sodium levels in municipal wells throughout the state, and was statistically significant.

Discussion

A growing number of numerical computer models have been advanced recently for purposes of predicting possible groundwater contamination episodes from given nonpoint source loadings of chemicals (Khan and Liang, 1989). These models typically take chemical loadings as givens and simulate impacts on groundwater through use of engineering and hydrologic equations. These models hold great promise of utility in policy analysis.

However, loadings are not typically known in a nonpoint source setting, and are clearly functions of land use activities in the proximity of the recharge area. The same contaminant may originate from various sources, and the literature has been rather silent on the subject of the range of loading coefficients pertinent to the alternative land uses. Regression analysis, as applied here, reveals the average separate loading coefficients, and therefore the relative contributions of different land uses to the final level of the contaminant in groundwater. Unlike most models of groundwater contamination, this one is completely empirical. To be sure, some may regard a 20 or 30 percent explanatory power to be rather low. Yet in the absence of detailed site characteristics and more disaggregate land use delineations, it would be surprising to explain a higher proportion of the variability of contaminant levels in wells throughout the state.

The loading-type coefficients estimated here may be of use to builders of numerical models, and these regression coefficients can be instructive in their own right for purposes of policy analysis. For example, recall that in our model land uses are stated as percentages of the recharge area. A one percent increase in a given land use in the recharge area is always matched by a one percent reduction in combined other uses. Therefore, the policy maker must evaluate the land use tradeoff. The contaminant loading from the increased land use (β_j) must be evaluated net of the decreased loadings from any land uses that would be replaced.

In this environment, suppose certain recharge areas are dominated by agricultural land use (ACWP), and state or local officials are concerned about the possibility of nitrate contamination of groundwater from chemical fertilizers. In such a situation, policy makers are sometimes quick to embrace a policy restricting agricultural operations from the recharge area. The results of Table 2 caution wariness, however, for this land use might well be replaced by other land users which also contribute to contamination. For example, if agriculture is replaced by unsewered low density residential use (R3NOSEW), the anticipated reduction in nitrate levels may not materialize, because the estimated contributions of nitrate from these land uses are so similar (0.0232 and 0.0190) and not statistically different. If, instead, agricultural use were replaced by unsewered medium density housing (1/4 to 1/2 acre lots, R2NOSEW), the likelihood is that more harm than good would result as far as nitrates are concerned. In this case, our estimates are that average nitrate loading would, for each percent of land converted, be expected to <u>increase</u> by 0.0251 milligrams per liter (the difference between 0.0483 and 0.0232). This difference is statistically significant.

Why might these sorts of results be important for policy? They are vital in targeting regulations to the major sources of contamination in the region, and also in anticipating consequences of possible land use substitutions. In this context, policy analysis is more complicated than the casual observer, and sometimes the policy maker, may believe. Clearly, partial solutions based on one contaminant and regulation of one land use may result in poor decisions. It is vital to have information on loading rates of all land uses in addressing potential problems from a given pollutant.

But this is not enough. In addition, enlightened policy must also consider interactions among different pollutants when land use patterns change. In Massachusetts, for example, nitrates and sodium have been two of the nonpoint source chemicals of concern for some time. Consider again the naive policy of restricting agricultural land to protect against nitrate pollution. From the estimation in Table 3, we observe that agricultural land uses (ACWP, AP) have an average sodium loading coefficient not different from zero statistically. Thus if, for example, agricultural land were replaced by medium density residential land use, not only might nitrate loadings increase, as suggested above, but also sodium loadings would be (statistically) significantly higher. The policy would have been ill-conceived from the standpoint of both contaminants. If, instead, the land uses replacing the previous agricultural land were commercial, then according to the Table 2 estimations, nitrate loadings might indeed subside, but Table 3 indicates that the policy maker should be prepared to experience a statistically significant and rather substantial elevation of sodium levels in the groundwater in this part of the country where salt is routinely used to de-ice parking lots and walkways.

The size of the estimated standard errors of several of the estimated coefficients suggests caution in making precise statements about the magnitude or even directions of these particular effects. We have not discussed the implications of the effects of these variables, and suggest that they be viewed in qualitative rather than more precise quantitative terms.

Do certain land uses make a statistically significant difference in regard to groundwater contamination? Our analysis supports the affirmative. And can we distinguish the amount of pollution which comes on average from these significant sources? Again, our results answer in the positive. They reveal that land uses do account for a significant portion of the variation in two nonpoint source groundwater contaminants, nitrates and sodium. We also show that regional policy decisions to protect groundwater from nonpoint source pollutants which ignore land use substitution and cross pollutant effects may well be counterproductive. Estimations using empirical procedures such as those employed here should be undertaken separately for other regions where hydrogeologic and other factors may be quite different from those in the Northeast.

Table 1
Descriptive Statistics

Variable	Definition	Fraction Zero Observations	Median	Minimum	Maximum
NO_3	NO_3 in milligrams per liter	----	0.700	0.100	8.300
NA	Na in milligrams per liter	----	15.000	1.500	100.000
ACWP	Cropland, orchards, nurseries, cranberry bogs[1]	.128	3.563	0.000	41.313
AP	Pasture	.356	0.713	0.000	18.065
RPUO	Golf courses, parks, cemeteries, undeveloped land	.138	1.748	0.000	20.377
RØR1	Residential--multifamily and Less than 1/4 acre lots	.564	0.000	0.000	27.134
R2	Residential--1/4 to 1/2 acre lots	.144	7.809	0.000	59.359
R2SEW	R2 with sewer	.840	0.000	0.000	59.359
R2NOSEW	R2 without sewer	.303	4.714	0.000	47.644
R3	Residential--larger than 1/2 acre lots	.037	7.221	0.000	46.692
R3SEW	R3 with sewer	.851	0.000	0.000	22.777
R3NOSEW	R3 without sewer	.186	6.060	0.000	46.692
UC	Commercial	.399	0.385	0.000	22.726
UI	Industrial	.548	0.000	0.000	27.893
UW	Landfills, sewage lagoons	.739	0.000	0.000	17.189
CR	Length of major roads in miles	.303	0.882	0.000	6.763

[1]The land use variables ACWP through UW are expressed as percentages of the total recharge area, here taken to be the 1/2 mile radius about the well. For example cropland, ACWP, ranged from 0% to 41.313% of the recharge areas studied, with 12.8% of the observations being zero and a median value of 3.563%.

Table 2
Nitrate Model Estimation

Variable	Estimated Coefficient	Standard Error
ACWP	0.0232*	0.0102
AP	-0.0016	0.0243
RPUO	0.0202	0.0186
RØR1	0.0227	0.0146
R2SEW	0.0127	0.0103
R2NOSEW	0.0483*	0.0082
R3SEW	0.0284	0.0272
R3NOSEW	0.0190*	0.0094
UC	-0.0047	0.0277
UI	0.0377	0.0238
UW	-.0509	0.0540
Constant	0.1336	0.2178
n	188	
R^2	0.195	

Table 3
Sodium Model Estimation

Variable	Estimated Coefficient	Standard Error
CR	2.7237*	0.7855
ACWP	-0.0966	0.1129
AP	-0.3883	0.2707
RØR1	0.1649	0.1617
R2	0.3018*	0.0786
R3	0.1519	0.1036
UC	0.7552*	0.3189
UI	0.2627	0.2606
UW	-0.1987	0.6070
Constant	8.9610	2.4343
n	188	
R^2	0.289	

*Denotes large sample statistical significance at the 95% level. While the dependent variables appear not to be distributed normally based on a test of the residuals, the variance of the stochastic error appears constant and therefore the ratios of estimated coefficients to standard errors are distributed asymptotically standard normal. Judge, et al [1985, p. 158].

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