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ABSTRACT

This paper analyses the costs and benefits of controlling groundwater pollution from nitrogen fertilizer. The Village of Hensall, where nitrate concentrations have been observed above 10 mg/l in recent years, was selected as the study site. The CREAMS simulation model was used to estimate the effect of reducing nitrogen fertilizer on nitrate leaching and consequently on nitrate groundwater pollution. Estimates of the value of groundwater were obtained from the literature and used to calculate the off-farm cost of groundwater contamination. This procedure resulted in a wide range of values for the benefits of reducing nitrate pollution. Estimated annual benefits of improved groundwater quality ranged from less than $1000.00 to more than $30,000.00 for the village. The off-farm benefits of nitrate groundwater pollution abatement outweigh the cost of using bottled water and in this case the on-farm cost of reducing nitrogen fertilizer. Placing a tax on nitrogen fertilizer would reduce the level of nitrogen applications, but the farm cost of compliance to a nitrogen tax policy is substantially higher than the compliance cost under a regulatory policy.

INTRODUCTION

Groundwater pollution has been identified as an important environmental issue in Canada (Neufeld, 1987). In 1981, 6.2 million Canadians, or 25 percent of the population, used groundwater as their major water supply. In Ontario, approximately two million people, or 23 percent of the population, use groundwater (Conservation Council, 1987). Evidence of nitrate groundwater pollution above 10 mg/l nitrate nitrogen[1] continues to mount (Gillham, 1978; Gillham, 1988; Agriculture Canada, 1992). A recent study (Agriculture Canada, 1992) sampled approximately 1,300 domestic farm wells in Ontario for agricultural groundwater pollution. The study found that 13% of all wells tested had nitrate-N concentrations above the provincial drinking water standard.

Nitrogen fertilizer used in agriculture has been linked to groundwater pollution (U.S.D.A., 1987). Nitrogen found in groundwater originates from many sources but agriculture is considered to be a major source (U.S.D.A., 1987; Conservation Council, 1987). The main adverse effect of nitrate groundwater pollution are its consequences for human health. High concentrations of nitrate in drinking water can cause methaemoglobinaemia and form carcinogenic N-nitroso compounds (Fraser and Chilvers, 1981).

The level of nitrate concentration in groundwater is influenced by factors such as climate, soil, land use and agricultural practices. Computer simulation models can take into account these factors and estimate the rate of nitrate leaching. The CREAMS[2] model is used in this study to simulate the impact of changing farm nitrogen applications on the rate of nitrate leaching and consequently on the concentration of nitrate in groundwater. Published estimates of the value of groundwater and

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1 Nitrate polluted drinking water at concentrations above 10 mg/l nitrate nitrogen are considered to represent a health hazard to children (Health and Welfare Canada, 1989; U.S. Environmental Protection Agency, 1986).

2 The CREAMS (Chemical, Run-off, and Erosion from Agricultural Management Systems) model was developed by the USDA to simulate the effect of different agricultural management practices on water pollution.

human health risks from exposure to nitrate in drinking water are used to calculate the off-farm cost of nitrate groundwater pollution. We also compare farmers' compliance costs under a nitrogen tax to a simple regulatory policy.

THE HENSALL CASE STUDY

The Village of Hensall is located 65 kilometres north of the City of London in southwestern Ontario (Figure 1). It was selected as a case study because of its well documented nitrate pollution problem. The necessary data to run the CREAMS model and to perform a cost-benefit analysis is available from previous studies. Hensall's pollution problem has been documented by the Ministry of the Environment (1983) and Gartner Lee Limited (1988). The King Street well, which was the main groundwater source until 1982, exhibited levels of nitrate pollution above 10 mg/l nitrate nitrogen (Figure 2). Corn prices increased up to 1983 and then declined (Ontario Ministry of Agriculture and Food, 1989). This may account for the initial increase in nitrate concentration in the King Street well and the decline after 1983. After 1982, the York Street well became the main water source and King Street well was used as a secondary source. The York Street well has also started to exhibit higher levels of nitrate pollution (Figure 2). As a further precaution the Public Utilities Commission drilled a deeper well in 1984 and also informed consumers of the high levels of nitrate in their drinking water.

MEASURING THE COSTS OF NITRATE POLLUTION

Three approaches to estimation of the cost of human health risks have been used in the literature. The simplest approach treats the value of a human life as the present value of lifetime average earnings. The second approach uses income differentials among occupations considered to involve different levels of mortality risk. If a wage premium is observed for more risky occupations,

Figure 1. Farms East of the Village of Hensall and Their Level of Nitrogen Application (kg/ha), in 1987 (Gartner Lee Limited, 1988).

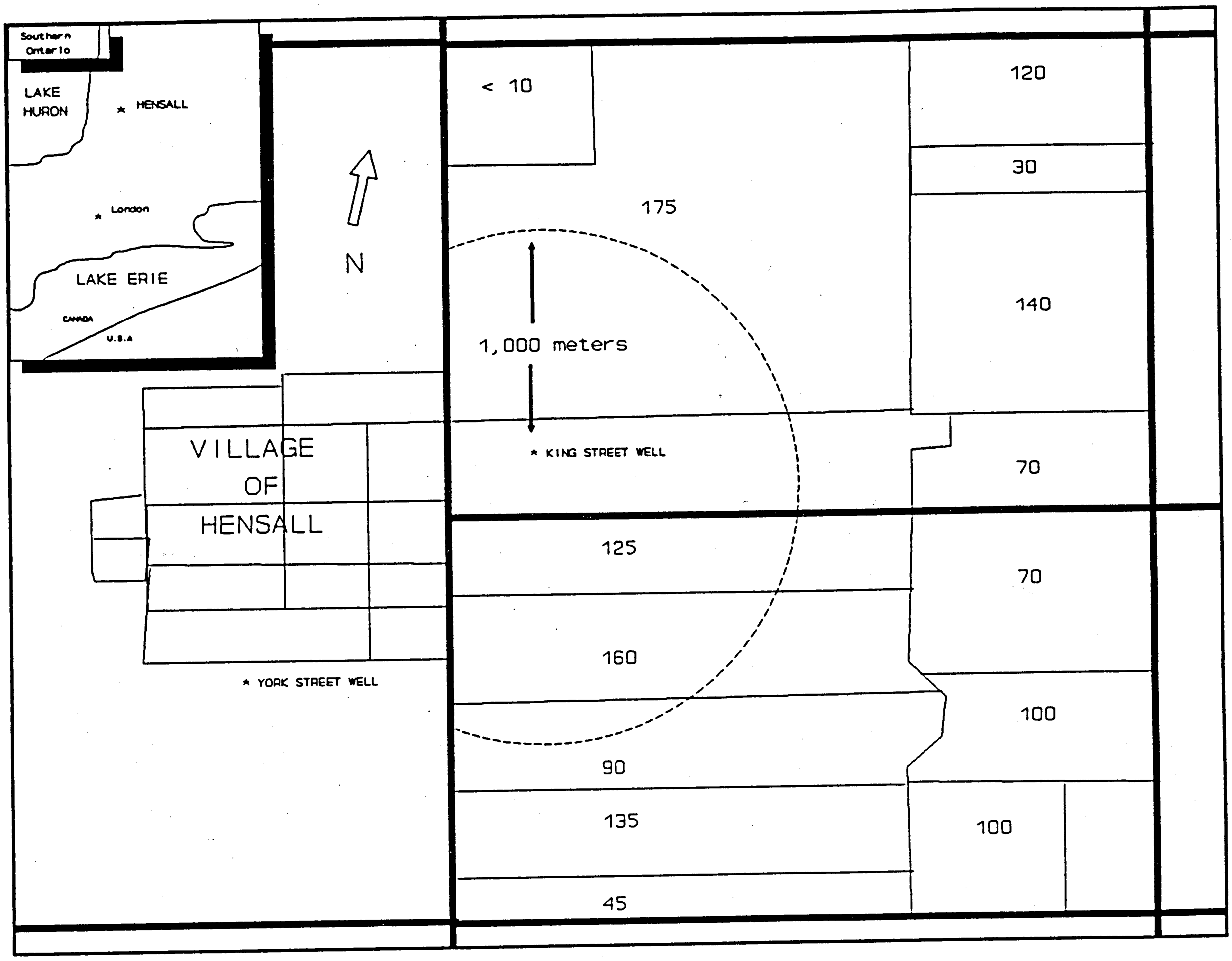
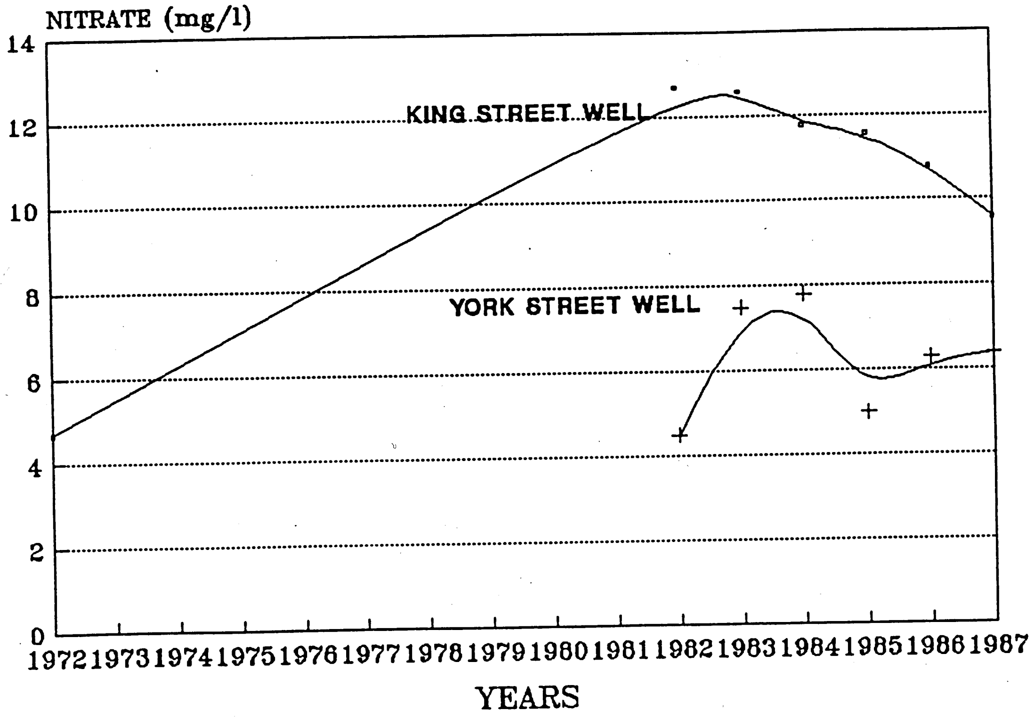


Figure 2. Historical Water Quality Trends in the Hensall Municipality Wells.



Source: Ministry of the Environment, 1983

Gartner Lee Limited, 1988

this value can be used to calculate the value that workers place on incremental changes in mortality risk. This can be extrapolated to an estimate of the value of life. Contingent valuation approaches use questionnaires or interviews to elicit willingness to pay to obtain a stipulated reduction in mortality risk or, alternatively, an improvement in environmental quality. The costs of contamination of groundwater by nitrate were calculated as the product of estimates of the value of a human life and the probability of mortality from consuming drinking water containing nitrate in concentrations in excess of current provincial standards.

The medical literature has not yet determined the exact correlation between nitrate ingestion and the risk of methaemoglobinaemia. Nevertheless, studies such as Super et al. (1981) provide an indication of the possible health risk. In the Super et al. study one infant out of 486 in a village in South Africa suffered methaemoglobinaemia. The infant population had been drinking well water that contained nitrate concentrations of 56 mg/l as nitrogen. The mortality rate for infants with methaemoglobinaemia has been 8% worldwide. If any deaths occurred in the Village of Hensall it would be a result of negligence since the symptoms of methaemoglobinaemia are easily recognizable by a physician, the treatment is simple and consumers in Hensall are aware of the pollution problem. However, assuming a mortality rate of 8% and a risk of 1/486 of contracting the illness, the risk of death for infants from exposure to nitrate polluted water in the range of 10 to 20 mg/l, would not exceed 0.08 deaths out of 486 or 16.5 deaths out of 100,000. This estimate of the risk of mortality is likely high for North America since no deaths due to methaemoglobinaemia have been reported since 1949 (Agriculture Canada, 1992). This risk level would give a worst case scenario. The population of the Village of Hensall was 1,155 in 1986 (Canadian Almanac and Directory, 1991). There were 14.6 live births per 1,000 population in Ontario in that year so one would expect around 17 births per year in the Village of Hensall. This estimate of mortality risk is used in conjunction with calculations of the present value of lifetime earnings to estimate annual health costs for the

Table 1: Lifetime Earnings Calculations and Estimated Annual Mortality Costs*

Annual Income	Discount Rate 2% Present Value	2% Mortality Costs	5% Present Value	5% Mortality Costs	10% Present Value	10% Mortality Costs
$25,000.00	$747,308.00	$2096.00	$447,001.00	$1254.00	$246,882.00	$693.00
$50,000.00	$1,494,616.00	$4192.00	$894,003.00	$2508.00	$493,764.00	$1385.00
$75,000.00	$2,241,924.00	$6289.00	$1,341,005.00	$3762.00	$740,646.00	$2078.00

* Present value calculated at birth for the assumed annual income earned from age 20 to age 65, inclusive, cost of health risk calculated as 17 x $\frac{16.5}{100,000}$ x Present Value.

Hensall situation (Table 1). Depending on the discount rate and annual income assumed, the costs of nitrate contamination of groundwater obtained using this approach range from \$693.00 to \$6289.00 per year.

Wage risk studies examine willingness to accept compensation for exposure to risk associated with an occupation. It derives values from actual rather than proposed or expected behaviour and is therefore a market based approach. A summary of estimates of the value of life obtained from occupational risk studies is presented by OECD (1989). The estimated value of life ranged from \$260,000 to \$11 million, with a mean of \$3 million. Using this mean value of \$3 million, then the health costs of nitrate water pollution above 10 mg/l in the Village of Hensall will be \$8,415[3] per year. The annual health costs would be \$729.00 and \$30,855.00 respectively for estimates of the value of a life of \$260,000 and \$11 million.

Using a Contingent Valuation approach, Hanley (1989) concluded that individuals in the United Kingdom were willing to pay £12.97 (\$25.92 Cdn) per person, per year, for a guarantee that water supplies meet nitrate standards. For the 1,155 individuals in the Village of Hensall that would total approximately Cdn \$29,938 per year. The estimated cost of nitrate pollution tends to be higher using the contingent valuation approach than it is with the other two approaches. This may be the result of several factors. First, individuals perception of the risk from nitrate pollution may be higher than the value obtained from epidemiological studies. In contingent valuation, it is the level perceived by individuals that matters. Second, individuals willingness to pay for a reduction in risk from nitrate pollution may be higher than what people are normally willing to pay for the same reduction in risk caused by another factor such as a car accident. This may be a result of a distinction between so-called voluntary and involuntary risks. Third, the uncertainty surrounding the health effects of nitrate pollution may cause individuals to be concerned about their own health, not just that

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3 (17x16.5/100,000)x\$3 million

of their children.

Apart from the health value of an aquifer, economists have estimated bequest values[4] and option values.[5] Edwards (1988) reported that individuals households in Cape Cod, Massachusetts, would be willing to pay $1,623 annually to protect an aquifer from uncertain future nitrate contamination. This study attempted to only measure option value and bequest value. In Ontario, the average number of persons per household is 2.8 (Statistics Canada, 1987). Consequently based on Edwards (1988) study, the bequest and option value of nitrate safe groundwater would total around $669,487.49[6] per year in the Village of Hensall.

The large discrepancy between health value and bequest and option values brings into doubt the validity of Edward's results. The main concern of nitrate groundwater pollution is its health effect and if individuals are only willing to pay $25.92 per year to reduce the present risk of nitrate ingestion, it is questionable that they would be willing to pay $579.64 per year to preserve an aquifer safe from nitrate pollution for future use and future generations. The high cost of nitrate pollution calculated in Edwards' study may be a result of several factors. First, there may have been double counting if individuals included their health costs with the bequest and option value. It may be very difficult for an individual to separate the bequest and option value from the health cost in a contingent valuation study. Second, this issue was discussed widely by the news media at the time of the survey and created the impression that nitrate pollution is of greater risk than it really is. Third, Edwards' study may have any one or several of the problems facing the contingent valuation studies, such as strategic biases, design bias and operational bias.

In summary, based on Hanley (1989) and Edwards (1988), the total annual cost of nitrate

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4 Individuals desire to preserve the aquifer for the benefits of future generations.

5 The desire to preserve the option of a clean aquifer for future use.

6 [$1,623*(1,155 individuals in the Village of Hensall/1.8 individuals per household)]

groundwater pollution above 10 mg/l in the Village of Hensall would be between $30,000 and $700,000 per year depending on whether bequest and option costs are included. Based on the lifetime earnings and wage-risk approaches, the annual costs ranged from $693.00 to $30,855.00.

These results create a quandary for the cost-benefit analyst. Depending on the technique employed, estimates of the cost of groundwater contamination can vary by a factor of 100. As we indicated above, we believe that there are reasons to question the values of life derived from the Contingent Valuation approach.[7] Similarly, the $260,000.00 value reported in one of the studies in the OECD survey seems low relative to the present value of lifetime earnings in Table 1. For a real discount rate of 5% and an estimated annual income of $50,000.00, the health costs reported in Table 1 amount to $2508.00. The mean values of health costs based on the OECD survey of wage-risk studies would be $8415.00 annually. We will use these values in our comparison of the costs and benefits of abatement.

ENVIRONMENTAL ANALYSIS

The CREAMS model (version 1.8) was used to predict the impact of changes in agricultural practices. The CREAMS model was chosen for the following reasons. First, CREAMS was specifically developed to analyze the effect of changes in agricultural production practices on chemical pollutants such as nitrogen. Second, CREAMS has been widely used in similar studies. Third, CREAMS is well supported. Environmental simulation models, such as CREAMS, are able to better predict relative changes in pollution levels than absolute values (Barfield et al., 1989). For this study, however, absolute pollution levels are already known. Therefore the main concern of this study is in predicting the relative effect that changes in agricultural practices will have on nitrate pollution

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7 See Fox (1992) and Cambridge Economics Inc. (1992) for a more in-depth assessment of Contingent Valuation.

levels.

Modelling

CREAMS was used to estimate the level of nitrogen in the surface run-off and the level of nitrate leached out of the rootzone. Based on interviews with Dr. Rudra and Dr. Dickinson[8], it was assumed that changes in nitrate leaching affect groundwater quality proportionally. For example, a 10% reduction in nitrate leaching will result in approximately a 10% reduction in nitrate groundwater pollution. Since the pollution levels in the King Street well were between 10-12 mg/l nitrate nitrogen, approximately a 16.67%[9] reduction in nitrate leaching is needed to reduce the pollution to a safe level of 10 mg/l nitrate nitrogen. The CREAMS model was used to identify agricultural practices that would result in a 16.67% reduction in nitrate leaching. Corn was assumed to be the only crop grown in the study site. Approximately 90% of the acreage within a 1,000 meter cone around the King Street well were planted to corn in 1987.

It was concluded that the major source of nitrate in the King Street well was unused residual nitrogenous fertilizer leaching from the soil (Ministry of the Environment, 1983; Gartner Lee Limited, 1988). Natural nitrate concentration in groundwater is commonly around 1.0 mg/l (Ministry of the Environment, 1983). Other sources of pollution, such as fertilizer and manure storage, feedlot run-off and septic tanks, were examined and not considered a major hazard to the groundwater in this area (Ministry of the Environment, 1983). Furthermore most of the households have sewer connections to the sewage treatment plant and would not be considered a threat.

Based on the field data and calculations, the Ministry of the Environment (1983) found that

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8 R.P. Rudra, Associate Professor; W.T. Dickinson, Professor, School of Engineering, University of Guelph, Guelph, Ontario, Canada.

9 ((12 mg/1-10 mg/1) (12 mg/1)*100

potential over-fertilization was occurring in the immediate area of the King Street well and in the area up gradient from the well. The ministry suggested that in order to reduce pollution levels farmers should be solicited to reduce their nitrogen application levels. These conclusions were further supported by Gartner Lee Limited (1988). Furthermore, it was recommended that actions to reduce pollution should be kept within a 1,000 m cone around the well (Gartner Lee Limited, 1988). Targeting fields within this area substantially reduces the cost of pollution abatement.

CREAMS simulations were used to explore the effects of crop production practices, such as the use of a winter cover crop, chisel plowing and contour plowing, as well as reductions in application rates for nitrogen fertilizer on nitrate concentration in groundwater. The base practice was growing corn using conventional tillage practices, that is mouldboard plow, and applying 147 kg/ha of nitrogen fertilizer in June. The base practice was derived from field data and information provided by the Ministry of the Environment (1983) and Gartner Lee Limited (1988). We also simulated the effect of using a winter cover crop with residue management, since this was expected to reduce nitrate leaching (Ministry of the Environment, 1983).

The parameters for the CREAMS input files were mainly derived from field data provided by Karrow (1977), Ministry of the Environment (1983) and Gartner Lee Limited (1988). Further data corresponding to the characteristics of the study area were selected from the CREAMS manual and tables and from Rudra et al. (1985). The rainfall data file consists of daily rainfall data from the Brucefield weather station, close to the Village of Hensall, for the years 1986 to 1988. The temperature data for the hydrology parameter file were also obtained from the Brucefield weather station. The solar radiation data was from London, Ontario (Environment Canada, 1985). The hydrology parameter file requires weather, soil and crop data. The crop parameters, describing agricultural practices such as winter cover factor, were varied to test the relative effect of different agricultural practices on pollution levels. The other hydrology parameters remain constant. The erosion parameter file requires data on soil, topography, agricultural practices and physical characteristics of water related to overland and channel flow. Once again the crop parameters were varied to test the relative effect of agricultural practices on pollution levels.

The nutrient component requires data on soil and in this case on nitrogen fertilizer practices. The crop parameters, such as nitrogen application were varied to test the relative effects of nitrogen management practices on nitrogen run-off and nitrate leaching.

MODEL RESULTS

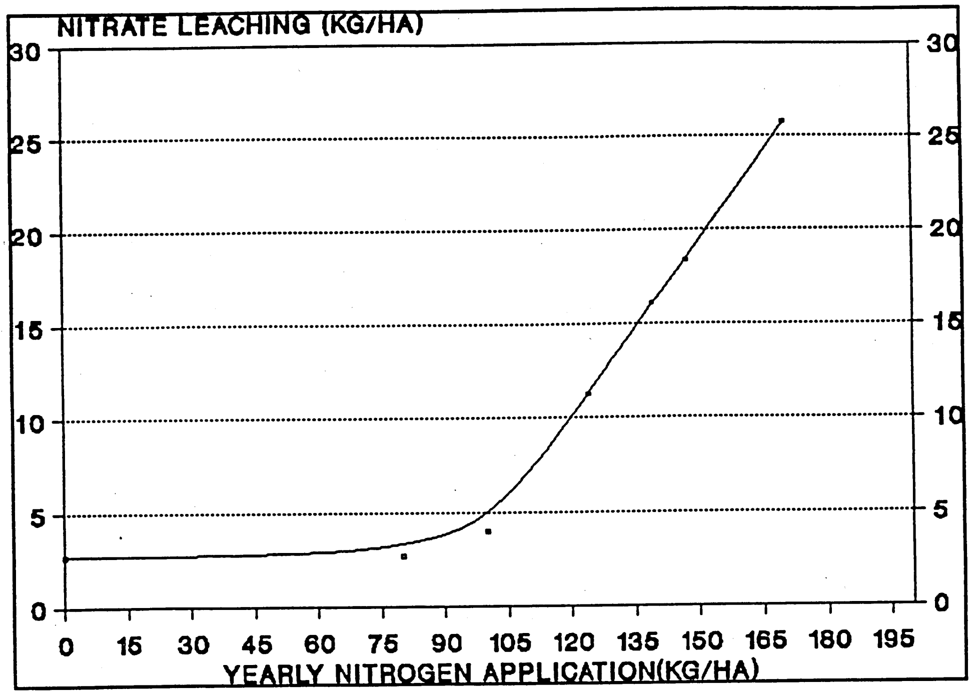
The results of simulations representing changes in agricultural practices and levels of fertilizer use are presented in Table 2 and Figure 3. According to CREAMS estimates, under the base run practice, the average yearly nitrate leaching was 18.35 kg/ha (the concentration of nitrate in the water leaching into the ground was 15.15 mg/l) between 1986 and 1988. Contour tillage and conservation tillage practices using a chisel plow significantly reduced the levels of nitrogen loss in surface runoff (Table 2), however, these practices showed a slight increase in nitrate leaching. Winter cover crop and residue management resulted in an overall 1.12% reduction in nitrate leaching. These results are consistent with Crowder and Young (1988) who concluded that soil conservation practices reduce surface water pollution, but can increase nitrate leaching. These results also indicate that soil conservation practices are ineffective in protecting groundwater quality. An alternative would be to reduce nitrogen fertilizer applications.

CREAMS simulation results exhibit a positive correlation between nitrogen application level and nitrate leaching (Figure 3). This effect would translate into a proportional reduction in nitrate groundwater pollution if nitrogen application levels were reduced. A reduction in average nitrogen applications from 147 kg/ha/yr to 140 kg/ha/yr would result on average in a 16.67% reduction in nitrate leaching, thus reducing nitrate groundwater pollution to the safety standard or below.

Table 2. Effectiveness of agricultural practices in reducing nitrate pollution as predicted by CREAMS.

Agricultural Practice	Nitrogen in Surface runoff (kg/ha)	Nitrate Leached Out of The Rootzone (kg/ha)	% Change in Nitrogen in Surface Runoff	Change in Nitrate Leached Out of The Rootzone
Base Practice	34.96	55.05	---	---
Winter Cover Crop/Residue Management	30.93	54.43	-11.53%	-1.12%
Conservation Tillage (reduced tillage)	31.61	55.00	-9.58%	-0.1%
Contour/ Conservation Tillage	28.44	55.19	-18.65%	0.25%

Figure 3. CREAMS simulation results showing the yearly average effect of reducing nitrogen applications on the total level of nitrate leaching (top figure) and on the concentration in the water leaching (bottom figure).



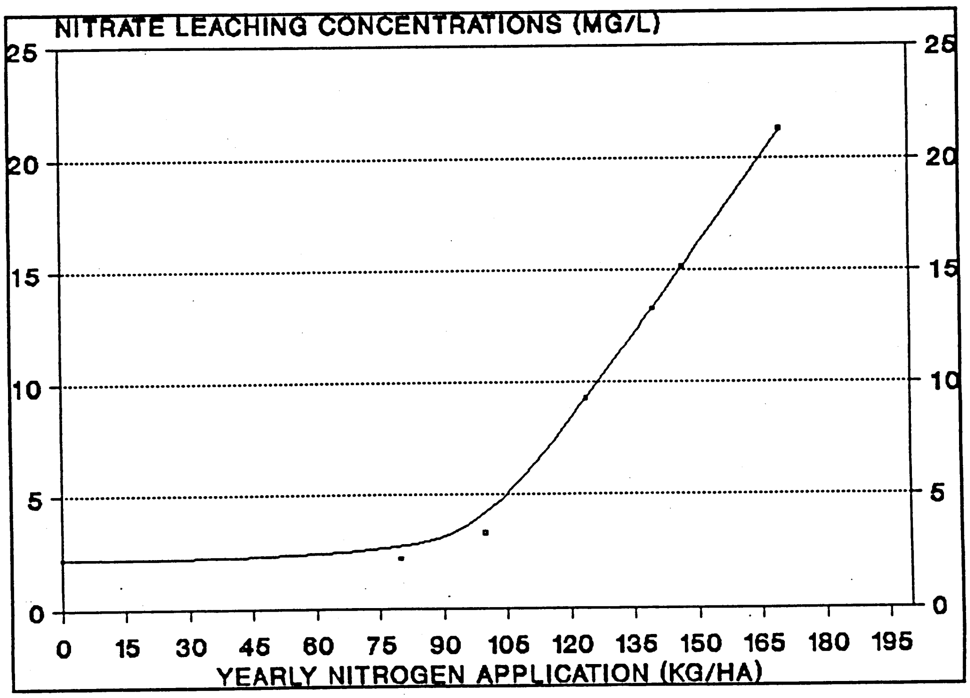


Figure 3 suggests that reducing nitrogen application below 75-90 kg/ha will have no effect on nitrate leaching. This may be caused by the fact that up to a certain point the crop will absorb most if not all the nitrogen fertilizer and consequently there will be minimal nitrate leaching. Reductions of nitrate fertilizer below this threshold will have no effect on nitrate pollution since at that level nitrogen fertilizer is not causing nitrate pollution.

COSTS OF REMEDIATION

In one sense, eliminating the health risk of methaemoglobinaemia is relatively simple since infants can be given human milk or bottled water in the milk mix. In England, infants are fed with bottled water when nitrate pollution levels were above 100 mg/l. This arrangement appears satisfactory in dealing with the nitrate groundwater pollution problem (Fraser and Chilvers, 1981). Bottled water is relatively inexpensive, approximately $0.5 per litre. Considering infants drink around 750 ml/day (Behrman and Kliegman (1987)), the cost of switching from tap water to bottled water is approximately $0.38 per day per infant. If there are seventeen infants and knowing that infants are at risk from methaemoglobinaemia for only the first six months of life, the maximum total cost of such a solution will be around $1,179[10] per year (Table 3). This cost can be reduced by blending bottled water with tap water. By blending 125 ml of bottled water with 625 ml of polluted water (12 mg/l nitrate nitrogen) one can reduce the pollution levels to 10 mg/l. The daily cost of this solution would be $0.06 per child or a yearly cost of $189 for the Village of Hensall.

The use of bottled water does not prevent nitrate groundwater pollution. Consequently this solution does not eliminate the bequest and option costs. An alternative solution is to reduce the pollution at its source, in this case the farm. As identified by the results of the CREAMS runs, nitrate pollution can be diminished by reductions in the level of nitrogen application (Figure 3). In

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10 ($0.38*17 infants*365 days/2)

the King Street well, nitrate pollution levels were around 10-12 mg/l. Consequently a 16.67 reduction in pollution levels would suffice to meet the safety standard of 10 mg/l nitrate nitrogen. A reduction in average nitrogen applications from 147 kg/ha to 140 kg/ha would result, on average, in approximately a 17% reduction in nitrate leaching (Figure 3) and consequently the need of a 16.67% reduction in the pollution level to meet the nitrate safety standard of 10 mg/l.

Reducing nitrogen applications will reduce production costs however there will also be a reduction in corn yields. This decline in corn yields will lower farm revenues and will not be fully compensated by the reduction in farm cost. Consequently any reduction in nitrogen fertilizer from the most profitable level will result in a reduction in farm profits.

The cost of reducing nitrogen applications were estimated as follows. Farmers are assumed to be applying nitrogen at the profit maximizing level. Farmers receive $12 per tonne for corn, which was the average corn price for the last ten years (Ontario Ministry of Agriculture and Food, 1979 to 1989). $0.60 per kilogram is used as the cost of actual nitrogen. The relationship between nitrogen applications and yields is taken from Beauchamp and Sheard (1987), page 75 (equation S70). This equation best represents Hensall since it is for a clay soil with corn following corn. However, since yield data varies from one area to another this equation is only an estimate of how corn yield responds to Nitrogen around Hensall. The source of nitrate pollution is limited to the farms within a 1,000 meter cone, approximately 157.08 ha, west of the King Street well (Gartner Lee Limited, 1988). Based on this information the farm cost of reducing nitrogen fertilizer applications can be expressed as:

$$\text{Cost} = (p_c \Delta y - p_n \Delta n)\text{area} \qquad (1)$$

where: Cost = total farm cost of reducing nitrogen applications

P_c = price of corn ($127/Ton)

Δy = $6430 + 34.054\Delta n - 0.0951(\Delta n)^2$

p_n = price of nitrogen (0.60/kg)

Δn = change in nitrogen application

area = main area contributing to nitrate pollution (157.08 ha)

Using equation (1), a reduction in nitrogen application from 147 kg/ha to 140 kg/ha (Δn=7 kg/ha) will result in a yearly farm cost of $1.81/ha. Consequently based on equation (1) the total cost of reducing nitrogen applications around Hensall from 147 kg/ha to 140 kg/ha will be around $284.31.

The cost of reducing nitrogen applications from 147 kg/ha to 140 kg/ha is low because at the optimal level of nitrogen application a small reduction in nitrogen fertilizer has a very small effect on yield. However the cost of further reductions in nitrogen fertilizer increases at an increasing rate. Changes in corn prices influence the cost of reducing nitrogen application. Furthermore if the sources of nitrate pollution extended more than a 1,000 meter corn or if corn yields were more sensitive to nitrogen fertilizer reductions, the cost of reducing nitrogen applications would increase. Johnson et al. (1991) estimated that a 25% N reduction would cost corn farmers in Oregon (USA) around $200/ha. Using $200/ha the total cost to Hensall would be around $31,000 for a 25% N reduction. However farmers in Hensall only need a 4.76% N reduction to meet groundwater quality standards, consequently the abatement costs in Hensall would be less than Oregon. In any case, the cost of reducing nitrogen applications appears to be about 1/10 of the estimated annual health costs reported earlier.

NITROGEN TAX

Imposing a tax on a polluting input, in this case nitrogen fertilizer, has been suggested as a means of pollution abatement. By increasing the price of nitrogen through a nitrogen tax, farmers will reduce nitrogen application levels if the level of nitrogen application responds to the ratio between the expected corn price and the fertilizer price (Atwood and Johnson, 1990; Huang and

Lantin, 1993). Governments may find such a policy attractive since the revenue derived from the tax could be used to finance the program.

By manipulating equation (1) it was estimated that a nitrogen tax of $0.34/kg is necessary to reduce nitrogen fertilizer applications from 147 kg/ha to 140 kg/ha. The cost to the farmer would be $49.7/ha/yr. The farmers' compliance cost for nitrate pollution abatement would increase from $1.81/ha, with regulatory policy, to $49.70/ha, with nitrogen a tax. A nitrogen tax would be levied on all of the N used by a farmer, whereas a restriction on N use effects only applications above 140 kg/ha.

CONCLUSIONS

Based on our earlier interpretation of the benefits of water quality literature, the value of a reduction in nitrate concentrations to meet provincial drinking water standards would amount to $2500.00 to $8400.00 per year in the Hensall situation. Estimation of health benefits continues to be controversial, but a substantial reduction in the value of these benefits would be necessary to reduce these to below the $284.00 per year in lost farm profit from a restriction on nitrogen use. Reduction of nitrogen application appears to be a less costly means of remediation relative to bottled water unless a blending approach is used. The preceding conclusions were based upon simulation results, the medical literature and the economic literature. There is uncertainty in all these sources of information.

REFERENCES

Agriculture Canada. 1991. Ontario Farm Groundwater Quality Survey. Ontario: Agriculture Canada.

Atwood, J.D. and S.R. Johnson. 1990. The Potential for 'LISA' - Type Nitrogen Use Adjustments in Mainstream U.S. Agriculture, WP 49, Centre for Agricultural and Rural Development, Iowa State University, Ames.

Beauchamp, E.G., P.G. Newdick and R.W. Sheard. 1987. Nitrogen Requirements for Corn in Southern Ontario. Ontario: Department of Land Resource Science, University of Guelph.

Behrman, R.E. and R. Kliegman. 1987. Essentials of Paediatrics. Philadelphia: W.B. Saunders Company.

Cambridge Economics Inc. 1992. Contingent Valuation: A Critical Assessment. Cambridge, Massachusetts.

Canadian Almanac and Directory. 1991. Canadian Almanac and Directory 1991. Toronto: Canadian Almanac and Directory Publishing Company Limited.

Conservation Council. 1987. The Ground Water Pollution Primer. New Brunswick: Environment Canada.

Crowder, B. and C.E. Young. 1988. Managing Farm Nutrients. Washington, D.C.: USDA, Agricultural Economic Report Number 583.

Edwards, S.F. 1988. Option Prices for Ground Water Protection. *Journal of Environmental Economics and Management*. 15:474-487.

Environment Canada. 1985. Solar Radiation Data Analyses for Canada 1967-1976. Ottawa: Ministry of Supply and Services Canada.

Fox, G. 1992. The Pricing of Environmental Goods: A Praxeological Critique of Contingent Valuation. *Cultural Dynamics*. 5(3):245-259.

Fraser, P. and C. Chilvers. 1981. Health Aspects of Nitrate in Drinking Water. *The Science of Total Environment*. 18:103-116.

Gartner Lee Limited. 1988. Investigation of the Nitrate Contamination in the King Street Well at Hensall. Ontario: Ontario Ministry of the Environment.

Gillham, R.W. 1979. Studies of the Agricultural Contribution to Nitrate Enrichment of Groundwater and the Subsequent Nitrate Loading to Surface Water. Ontario: PLUARG Project No. 41122.

Gillham, R.W. 1988. Nitrate Contamination of Groundwater in Southern Ontario: A Hydrogeologic Perspective. Edited by J.A. Stone and L.L. Logan in *Agricultural Chemicals and Water*

Quality in Ontario. Kitchener: Proceeding of a Workshop sponsored by The Ontario Water management Research and Services Committee, November 17-18.

Hanley, N. 1989. Problems in Valuing Environmental Improvements Resulting from Agricultural Change: The Case of Nitrate Pollution. Edited by Alex Dubgaard and A. Hjortshoj Nielsen. *Economic Aspects of Environmental Regulation in Agriculture*. Wissenschaftsverlag: Vauk Kiel KG.

Huang, W. and R.M. Lantin. 1993. A Comparison of Farmers' Compliance Costs to Reduce Excess Nitrogen Fertilizer Use Under Alternative Policy Options. *Review of Agricultural Economics*. 15(1):51-62.

Johnson, S.L., R.M. Adams and P.M. Gregory. 1991. The On-Farm Costs of Reducing Groundwater Pollution, *American Journal of Agricultural Economics Association*. 73:1063-1073.

Karrow, P.F. 1974. Quaternary Geology of the St. Marys Area (Western Half) Southern Ontario. Ontario: Ontario Div. Mines, OFR 5116 and Maps P918 and P956.

Ministry of the Environment. 1983. *Investigation of Nitrate Contamination in the King Street Communal Well in the Village of Hensall Township of Tuckersmith London Road Survey 1, Lot 6 County of Huron*. London: Ontario Ministry of the Environment.

Ontario Ministry of Agriculture and Food. 1988. *Nitrogen Recommendations for Corn*, Ontario: Publication No. 88-112, August.

Ontario Ministry of Agriculture and Food. (1989). *Agriculture Statistics for Ontario*. Ontario: Ontario Ministry of Agriculture and Food, Publication 20, AGDEX 11/540, Years 1979-1989.

Organisation for Economic Co-operation and Development. 1989. *Environmental Policy Benefits: Monetary Valuation*. Paris: OECD.

Rudra, R.P., W.T. Dickinson and G.J. Well. 1985. Application of the CREAMS Model in Southern Ontario Conditions. *American Society of Agricultural Engineers*. 28:1233-1240.

Statistics Canada. 1987. *Census of Canada, 1986 Population and Dwelling Characteristics - Census*. Ottawa: Ministry of Supply and Services Canada, Catalogue 94-111, September.

Super, M.H., D.V. Heese, D. MacKenzie, W.S. Dempster, J. Du Plessis and J.J. Ferreira. 1981. An Epidemiological Study of Well-Water Nitrates in a Group of South West African/Namibian Infants, *Water Research*. 15:1265-1270.

USDA. 1987. *The Magnitude and Cost of Groundwater Contamination From Agricultural Chemicals*, Washington, DC:USDA.