Agricultural Nitrate 

and

Impacts On Water Quality In Ontario

OWMRSC WORKSHOP: 92

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OBJECTIVES

The purpose of the workshop is to explore current concerns regarding nitrate and associated compounds used in agriculture, to examine their effects on water quality conditions and measures required to minimize impacts. The intent is to share information and to make recommendations relevant to water management research and services in the agricultural industry.

The workshop will provide a forum for engineers, scientists, farmers, suppliers, managers and operators involved in the field of research, development, application and management of nitrate in agriculture.
PREFACE

Recent public awareness on the environment created concern on humans health impacts from potential increases of nitrate concentration in streams and groundwater. Farmers and agricultural-food producers have used nitrogen fertilizers and farm manures over the years to increase farm productions for the benefit of humans. Even though the applications of these natural and artificial fertilizers have been done efficiently, the findings of researchers on the fate and movement of nitrate from soils, and surveillance trend monitoring have shown increasing trends of nitrate in water bodies at strategic locations in Ontario. As a means to address this concern the Ontario Water Management Research and Services Committee (O.W.M.R.S.C.) organized a workshop entitled "Agricultural - Nitrate and Impacts on Water Quality in Ontario"

The purpose of the workshop was to explore current concerns regarding nitrate and associated compounds used in agriculture. To evaluate nitrate effects on water quality conditions and necessary measures to minimize their impacts. The general intent was to share information and to make recommendations relevant to water management research and services in the agricultural industry. The workshop provided a forum for engineers, scientists, university researchers, farmers, suppliers, managers and operators involved in the field of development research, application and management of nitrate in agriculture.

There were 75 registered participants representing federal and provincial agencies, universities, private industries, farmers and grower organizations. Sixteen invited speakers presented papers that addressed the fate and transport of nitrate, nitrogen fertilizer production and uses, understanding nitrate interactions within surface and groundwater, modelling nitrate movements, management and policy issues. Included in this proceedings are the transcripts of twelve speakers.

Dr. L. A. Logan
Chairman, O.W.M.R.S.C.
NITRATE LOSSES IN TILE WATER FROM CONVENTIONAL AND CONSERVATION TILLAGE PLOTS

D.J. McKenney¹, C.F. Drury² and W.I. Findlay²

ABSTRACT

In a three year study, 1989 to 1991 nitrate concentration and total volume of tile runoff were measured from conventional (moldboard plow) tillage (CT), no-till (NT), ridge tillage (RT) all planted in continuous corn, and Kentucky bluegrass (BG) treatments. A total of 178.6 kg N ha⁻¹ was applied to all treatments over the growing season. Total tile water flow was greatest from the CT treatments (1.7 ML ha⁻¹), approximately equal from RT and NT treatments (1.3 ML ha⁻¹) and least from BG (1.1 ML ha⁻¹). Eleven runoff events were recorded in 1989 and sixteen in 1990. Nitrate concentrations in tile water from CT, RT and NT treatments were greater than the maximum recommended safe limit of 10 mg N L⁻¹ in 79% of runoff events with flow weighted concentrations between 12 and 17 mg N L⁻¹ in 1989 and 1990. From BG, flow weighted NO₃ concentrations were only 1.1 and 2.7 mg N L⁻¹ in 1989 and 1990 respectively. The total average NO₃ lost over the two years in tile water was about 23 kg N ha⁻¹ from CT treatments, 17 kg N ha⁻¹ from RT treatments, 16 kg N ha⁻¹ from NT treatments and 2.3 kg N ha⁻¹ from BG treatments. In 1991 considerably lower volumes of water flowed through the tiles and lower overall nitrate losses were observed as a result of a serious drought.

INTRODUCTION

Over the past three years as part of the joint Federal-Provincial Soil and Water Environmental Enhancement Program (SWEEP) we have been studying the effects of several tillage methods on nitrogen transformations in Brookston clay loam, and effects on nitrogen losses through surface runoff and tile drainage.

Nitrate is a very soluble ion and is relatively mobile in soil. If not used by the crop or denitrified NO₃ can eventually be lost by runoff or leaching and contaminate groundwater or adjoining water systems. Although NO₃ itself is not very toxic, high levels of NO₃ in natural water systems is a serious environmental concern. It can contribute to eutrophication of lakes and streams. It can be readily converted to NO₂ which has been implicated in two major health problems, the "blue-baby" syndrome or methaemoglobinemia and stomach cancer. For those reasons the World Health Organization set a limit of 50 mg NO₃ L⁻¹ (i.e. 11 mg NO₃-N L⁻¹) and the Environmental Protection Agency (USA) set a limit of 10⁻³-N L⁻¹ as the maximum recommended safe levels in water supplies for human consumption.

Kladivko et al., (1991) found that NO₃ concentrations in tile water under a corn crop fertilized with 285 kg N ha⁻¹ were typically greater than 10 mg N L⁻¹ and within a range of 20-30 mg N L⁻¹ with total N losses between 1870 kg N ha⁻¹ depending on tile drain spacing. Baker and Johnson (1981) found that the concentration of NO₃ in tile water and the amount of NO₃ lost was proportional to the amount of N-fertilizer applied to the soil. In particular, N fertilization rates of corn between

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-1-
90-100 kg N ha⁻¹ resulted in flow weighted NO⁻³ concentrations of 20 mg N L⁻¹ in tile water with a total loss of 27 kg N ha⁻¹. N-fertilizer rates between 240-250 kg N ha⁻¹ resulted in flow weighted NO⁻³ concentrations of 40 mg N L⁻¹ and total N losses of 48 kg N ha⁻¹. Similarly Prunty and Montgomery (1991) found that flow weighted NO⁻³ concentrations from lysimeters were 8.6 mg N L⁻¹ for a low N management system (95 kg N ha⁻¹ added fertilizer) and 12.3 mg N L⁻¹ for a high N management (145 kg N ha⁻¹) corn production system. In many other corn production systems the concentration of NO⁻³ in tile runoff exceeded the 10 mg N L⁻¹ limit for potable water (Bergstrom and Johansson, 1991). Hubbard et al., (1984), and Baker and Johnson, (1981) found increases in NO⁻³ concentration in tile runoff appeared within 2-3 months after N was applied and persisted up to 3 years following the last application of N fertilizer. Rainfall and irrigation also contribute to the extent and timing of NO⁻³ loss. Bergstrom and Johansson (1991) found that soil texture influenced NO⁻³ leaching losses. Yearly nitrogen losses from lysimeters seeded with spring barley and fertilized with 100 kg N ha⁻¹ were 65 kg N ha⁻¹ for sandy textured soils, 25-40 kg N ha⁻¹ for loamy textured soils and 20 kg N ha⁻¹ or less for clay textured soils.

Conservation tillage systems affect water movement in soil systems. Although conservation tillage systems decreased NO⁻³ losses in surface runoff they may have a deleterious effect on subsurface NO⁻³ concentration. Angle et al., (1984) suggested that most losses of NO⁻³ from a conventional and no-till watershed was a result of flow to subsurface groundwater and not to surface runoff.

Since NO⁻³ contamination of drainage tile water may be a serious problem in intensive corn production systems (Kladivko et al., 1991; Baker and Johnson, 1981; and Prunty and Montgomery, 1991), it is important to evaluate nutrient transport under varying tillage conditions. Therefore the objective of this study was to determine if conservation tillage systems (ridge and no-till) affected the volume, concentration and total amount of NO⁻³ lost in tile runoff as compared to conventional tillage (moldboard) practices.

**MATERIALS AND METHODS**

The plots were established in 1983 at the Totten farm, Agriculture Canada (Harrow). Ammonium, NO⁻³, and NO₂ analysis of tile water was conducted in 1989, 1990 and 1991. Four treatments, replicated twice, consist of continuous corn planted in 75 cm rows at 67,000 seeds ha⁻¹ with conventional (moldboard) tillage (CT) in the fall with spring discing; continuous corn with ridge tillage reformed mid-season each year with a Hiniker soil ridger (RT); continuous corn with no-till (NT) and a Kentucky bluegrass plot (BG) in order to control weeds atrazine (1.8 kg ha⁻¹) and metolachlor (2.64 kg ha⁻¹) were applied pre-emergence. All plots received 132 kg ha⁻¹ 8-32-16 at planting, and another 168 kg N ha⁻¹ ammonium nitrate fertilizer, applied with a brush applicator when the corn was at the 6 leaf stage. The Kentucky BG plot also received 132 kg ha⁻¹ of 8-32-16 in the fall. Precipitation was measured at the site with a Belfort rain gauge.

The plots were 84m by 12m and centred over a tile drain at 0.6m depth. The tiles emptied into an enclosed metal-lined pit. The tile discharge was directed through a proportional logarithmic weir and stage height was indicated on a Stevens water level recorder. The tile flow rate for each event was manually monitored which verified the weir calibration. Tile discharge volume was determined from the area of the hydrograph by integration using Simpson’s Rule algorithm.

Tile water samples were manually collected and stored at 4°C for analysis of NH⁺₄, NO⁻³ and NO₂ during each runoff event. The samples were filtered through 0.45 µm millipore filters and the filtrates were analyzed on a TRAACS 800 autoanalyzer (Bran + Luebbe, Elmsford, New York) for NH⁺₄ using the Berthelot reaction and for NO⁻³ plus NO₂ using the cadmium reduction method (Tel and Heseltine, 1990). Nitrite concentration was determined without the column and Na, content was calculated by difference.
**RESULTS**

The average yearly rainfall for the Woodslee experimental station (Totten farm) is 819 mm (30 year average). In 1989 the rainfall was 709 mm and in 1990 the rainfall was 872 mm and distributed fairly evenly throughout each year (Fig. 1). However in 1991 there was a drought with only 561 mm of rainfall with only 210 mm during the growing season. Hence crop yields were considerably lower than average in 1991.

Tile water samples were collected from each plot in each runoff event over the three years. In 1989 and 1990 several samples were collected during each runoff event for each plot to ensure that the initial, peak and end of each runoff event was sampled. Although NO$_3^-$ content varied between runoff events the variability within an event was relatively small. Similarly Baker and Johnson (1981) found that NO$_3^-$ concentrations in tile drainage did not change appreciably during a flow event but large changes could occur between events. Due to budget constraints in 1991 only one sample from each plot per event was collected for analysis.

There were eleven runoff events in 1989 and 16 runoff events in 1990. The volume of tile drainage water was greatest between January and March of 1990 (Fig. 2). In general, more water flowed through the tiles under the CT plots than under all other treatments. In only one event, February 1990, the BG treatment had a greater volume of water flowing through the tiles as compared to the corn tillage treatments. In several instances the water flowed out of the tiles in only some of the plots (Fig. 2). The cumulative effect of variations in antecedent moisture contents, evapotranspiration (i.e. resulting from differential growth rates of the crop) and hydraulic conductivity (resulting from tillage operations) between the treatments probably accounted for the difference in rate and frequency of tile runoff. The total volume of tile water was greatest under the CT treatment and least under the BG treatment in both 1989 and 1990 (Fig. 3). No differences in tile water loss between the RT and NT treatments were apparent.

Nitrate concentration in tile water under all the corn treatments was greater than 10 mg N L$^{-1}$ in 79% of all runoff events in contrast to the BG treatment where only 11% of all runoff events had Na, concentrations greater than 10 mg N L$^{-1}$, (Fig. 4). There was less than 3 mg N L$^{-1}$ in 88% of the runoff events for the BG treatment in 1989 and 1990. The maximum NO$_3^-$ concentration in tile runoff water occurred in July 1989 during a runoff event 19-20 days following N application. The CT treatment resulted in 53.4 mg N L$^{-1}$ NO$_3^-$ in the tile water whereas the RT, NT and BG treatments had 37.1, 34.9, and 4.0 mg N L$^{-1}$ respectively in the same event. In 1989, the average NO$_3^-$ concentration over all runoff events was greatest with the CT treatment (17.2 mg N L$^{-1}$), and least with the BG treatment (1.1 mg N L$^{-1}$), Fig. 5). The average NO$_3^-$ concentration for the RT and NT plots was 15.6 and 14.9 mg N L$^{-1}$, respectively. In 1990 all corn tillage treatments resulted in similar NO$_3^-$ concentrations, 12.9, 11.9 and 11.6 mg N L$^{-1}$ for the CT, RT and NT treatments with the BG treatment at 2.7 mg N L$^{-1}$ (Fig. 5). Ammonium leaching in tile water was very low with less than 0.25 kg N ha$^{-1}$ lost from all treatments in both 1989 and 1990 (data not shown).

Considerable differences in the total amount of NO$_3^-$ lost in each event resulted from the combined effect of NO$_3^-$ concentration in the tile water and volume of water lost per event (Fig. 6). The CT treatment had the greatest NO$_3^-$ loss and there was comparatively negligible losses of NO$_3^-$ with the BG treatment. In the July 1989 runoff event 8.6 kg N ha$^{-1}$ NO$_3^-$ was lost from the CT plots, about half of this amount lost from the RT and NT treatments, and only 0.7 kg N ha$^{-1}$ lost from the BG treatment. In 1990, although there were no runoff events which resulted in NO$_3^-$ losses as great as those which occurred in July 1989, the total NO$_3^-$ loss in tile water was greater due to the larger number of runoff events over the year. On average 23 kg N ha$^{-1}$ was lost through tile runoff from the CT treatment, about 17 kg N ha$^{-1}$ from the RT treatment, 16 kg N ha$^{-1}$ from the NT treatment whereas only 2.3 kg N ha$^{-1}$ was lost from the BG treatment in 1989 and 1990 (Fig. 7). The
average NO\textsubscript{3} losses from the corn tillage treatments in 1989 and 1990 represented between 9 to 13% of the N fertilizer added. Nitrate losses through surface runoff (data not shown) were considerably less (0.01 to 6.1 kg N ha\textsuperscript{-1}) than through tile discharge in 1989 and 1990. Loss of NH\textsuperscript{+}, in tile water was less than 0.25 kg N ha\textsuperscript{-1} from all treatments in both 1989 and 1990.

We considered the results from 1991 separately as a result of the drought. Furthermore, tile water flow data was not obtained in 1991 for the RT treatment due to instrumentation problems. Compared to 1989 and 1990, the concentration of NO\textsubscript{3} in tile water was greater with all treatments in 1991 as a result of the poor corn growth and nitrogen uptake in this dry year. There was 0.22 (±0.07), 0.41 (±0.03) and 0.02 (±<0.01) ML ha\textsuperscript{-1} of tile runoff from the CT, NT and BG treatments, respectively. The greatest flow weighted average NO\textsubscript{3} concentration in tile water over the year was 26.4 for the CT and 28.5 mg N L\textsuperscript{-1} for NT treatments whereas under the BG treatment the flow weighted NO\textsubscript{3} concentration was 14 mg N L\textsuperscript{-1}. Only 7.4 (±4.2), 11.1 (±2.0) and 0.3 (±0.2) kg N ha\textsuperscript{-1} was lost from the CT, NT and BG treatments in 1991 as a result of the low volume of tile water which occurred in 1991.

**DISCUSSION**

The primary loss of nitrogen from the corn production system was through tile runoff and in most cases NO\textsubscript{3} concentrations were well above the acceptable limit of 10 mg N L\textsuperscript{-1} under all corn production tillage systems. Under conservation tillage the losses were somewhat lower as there were lower volumes of water lost and lower concentrations of NO\textsubscript{3} in tile runoff. These results agree with those of Bolton et al., (1970) who also found that the volume of water which flowed through the same soil type was the predominant factor responsible for nitrogen loss. In addition we noted that increased yield and N-uptake in grain resulting from the conservation tillage systems reduced the amount of NO\textsubscript{3} available for leaching (data not shown). Even though conservation tillage improved yields and N uptake and contributed to lower leaching losses, the concentrations of NO\textsubscript{3} in the runoff water was usually well above the 10 mg N L\textsuperscript{-1} limit.

The lowest losses of NO\textsubscript{3} in tile water occurred under bluegrass sod and the Na, concentration in these waters was usually below 10 mg N L\textsuperscript{-1}. Bolton et al. (1970) also found the greatest losses of N under continuous corn with fall plowing and the least under continuous bluegrass sod. Hubbard et al. (1984) found that NO\textsubscript{3} concentration in tile runoff after an extensive drought was much greater than in years when the rainfall was about average. Increased NO\textsubscript{3} concentrations in tile drainage may be due in part to lower corn yields and crop uptake (Kladivko et al., 1991). The results from this study also showed that NO\textsubscript{3} concentration in tile water was considerably higher after the severe drought in 1991 which resulted in decreased yields and N uptake.

Ammonium loss in tile runoff was very low for all treatments in all years in this study. Similarly Kladivko et al. (1991) found a very low (0.5 kg N ha\textsuperscript{-1}) annual NH\textsubscript{4}\textsuperscript{+} loss in subsurface drainage for an intensive corn production system receiving 285 kg N ha\textsuperscript{-1} yr\textsuperscript{-1}. Prunty and Montgomery, (1991) also found NH\textsubscript{4}\textsuperscript{+} concentrations in runoff water under corn production systems were less than 0.1 mg N L\textsuperscript{-1} from lysimeters.

Although conservation tillage (RT and NT) corn production systems decreased the concentration of NO\textsubscript{3} in tile runoff and total nitrogen losses, these concentrations were still above the 10 mg N L\textsuperscript{-1} limit. The addition of management practices such as the use of cover crops, intercrops or controlled drainage systems may further reduce NO\textsubscript{3} concentration in tile water.

**ACKNOWLEDGEMENTS**

We are grateful to P.S. Hencher and N. Taylor for their capable technical assistance. We also wish to extend acknowledgements to the Soil and Environmental Enhancement Program (SWEEP) for financial support.
REFERENCES


Figure 1: Daily precipitation from 1989-1991 at the Woodslee experimental site.

Figure 2: Tile water losses for each runoff event in 1989-1990 for the conventional tillage (CT), ridge tillage (RT), no-till (NT) and Kentucky bluegrass (BG) treatments.

Figure 3: Cumulative tile water losses for the conventional tillage (CT), ridge tillage (RT), no-till (NT) and Kentucky bluegrass (BG) treatment in 1989 and 1990.

Figure 4: Nitrate concentration in tile runoff for each event in 1989-1990 for the conventional tillage (CT), ridge tillage (RT), no-till (NT) and Kentucky bluegrass (BG) treatments.
Figure 5: Flow weighted mean $\text{NO}_3^-$ concentration in tile water for the conventional tillage (CT), ridge tillage (RT), no-till (NT) and Kentucky bluegrass (BG) treatments in 1989 and 1990.

Figure 6: Nitrate loss in tile runoff for each event in 1989 and 1990 for the conventional tillage (CT), ridge tillage (RT). No-till (NT) and Kentucky bluegrass (BG) treatments.

Figure 7: Cumulative $\text{NO}_3^-$ loss in tile runoff for the conventional tillage (CT), ridge tillage (RT), no-till (NT) and Kentucky bluegrass (BG) treatments in 1989 and 1990.
SALES AND TRENDS OF NITRATE FERTILIZERS IN ONTARIO

T. Sawyer

ABSTRACT
Custom Application relates fertilizer application distribution as follows:

25% of grain corn acres - 28% - applies by custom applicator
- ppi, pre, crop protection chemical

15% of grain corn acres - dry fertilizer applied by custom applicator
- impregnation

The crop residue follows a pattern of:
- increasing, by 4 to 5% per year

The crop protection herbicides application is:
- more post emergent products in this decade

The planting speed by farmers follows:
- growers plant 20% of their single crop acres in one day

Trends and Nitrogen Use
The following is a brief summary on nitrogen usage:
- crop yields will continue to increase
- more side dressing - economics, efficiency, environmental impacts
- more precise fertilizer placement - crop residue increasing
- more split application - economics, efficiency, environmental impacts
- livestock manure & sewage sludge will be spread on more farmed acres
- nitrogen soil test will be the norm in the future
- crop plans will be more formalized
- more legumes/forages in the rotation may be paramount.

Sales Statistics:

I. Volume of Nitrogen Tonne/active

- 1966 68,673
- 1971 130,433
- 1976 161,364
- 1981 205,600
- 1985 237,409
- 1986 213,916
- 1991 174,860

II. Nitrogen Fertilizer Use by Crop in %

- Corn (grain) 50
- Winter Wheat 14
- Barley 6
- Mix Grain 6
- Corn (Silage) 5
- Vegetables 4
- Others 15

Nitrogen Products Used in Ontario by %

- Urea 44
- Solutions 18
- NH₃ 13
- Ammonium Nitrate 10
- Diammonium Phosphate 8
- Monoammonium Phosphate 5
- Ammonium Sulphate 2
- Liquid Starters 1

Total 101

1 The Fertilizer Institute of Ontario, Cambridge, Ontario
THE ROLE OF FERTILIZER IN SUSTAINABLE AGRICULTURAL AND FOOD PRODUCTION

H. Neutens

ABSTRACT

I very much appreciate the invitation to participate in your 1992 conference and workshop. I am somewhat surprised at the invitation because, I am not a scientist or researcher, but, I am an individual deeply interested in the betterment of agriculture in Ontario. I am a fertilizer and pesticide supplier that tries to provide my customers with the best service possible, such as; quality product, professional application, professional advice including the transfer of new technology and the importance of all facets of sustainable agricultural practices.

I am very proud of our industry and the key role we have played and will continue to play in crop production across Canada. The industry, in endorsing the concept of sustainable agriculture, has committed to the following objectives:

- To integrate the principles of economic crop production with environmental protection.
- To create public confidence that farmers use fertilizers responsibly.
- To provide planners and policy-makers with a sound understanding of the role of fertilizer in sustainable systems of crop production.

The following presentation will give you some insight into the great importance of the responsibility we take as a stakeholder in the whole concept of sustainable agriculture.

INTRODUCTION

This presentation has been produced by the Environmental Affairs Committee of the Canadian Fertilizer Institute and is based on agricultural science.

CFI Publications

The Canadian fertilizer industry has also written a position paper on The Role of Fertilizers in Sustainable Agriculture and Food Production and have received very favourable comment world-wide and across Canada. Published pocket size pamphlet "What are you Eating?" to provide information to consumers; and

Production of an educational teaching aid for the classroom - Food for Plants, Plants for Food - available from TFIO or CFI;

Preparation of a scientific report to the Science Council of Canada, which has received world-wide recognition, and have contributed to the International Fertilizer Association report at the Earth Summit Conference in Brazil, June 1992.

Definition of Sustainable Agriculture

The Canadian Fertilizer Institute fully supports Agriculture Canada's definition of sustainable agriculture: "agriculture systems that are economically viable and meet society's needs for safe nutritious food while conserving or enhancing Canadian's natural resources and the quality of the environment for future generations". We truly believe that this should be supported by society in general.

The Bruntland Commission stated that we need economic prosperity to provide the goods and wealth of society, but that we must not maintain our lifestyle at the expense of our children.

1 Canadian Fertilizer Institute
"A balance between economics, social and environmental factors must be achieved", Alvrim Lazar, Director General, Bureau for Environmental Sustainability, Agriculture Canada.

The Agenda we will follow is:

- Impact of Mineral Fertilizer
- Nutrient Sources
- Fertilizer and Water Quality
- Fertilizer and Soil Quality
- Fertilizer and Air Quality
- Food Quantity and Quality
- Strategies for Sustainability
- Summary

THE FOOD WE EAT

We will query ourselves by asking two questions:

Question # 1 what percent of personal disposable income does an average Canadian spend on Food and Non Alcoholic drink at home?
Answer 10.1%

Question # 2 How does Canada compare with other Countries?
Answer U.S.A. 10%
European E.C. 14%
Russia 28%
China 48%
India 53%

Canadian agriculture produces an abundance of safe nutritious food at an economical price to the Canadian consumer --- very favourable when compared to other nations.

Reference: - CFI Position Paper - Page 1
(a) Bruntland Commission,
(b) Agriculture Canada Statement
(c) philosophy of scientific fertilizer use -What are you eating?

The Canadian consumer has no historical experience with famine and therefore tends to take Canada's bountiful supply of safe, economical foods for granted.

IMPACTS OF MINERAL FERTILIZER

By examining the impacts from a soil depletion perspective we acknowledged:

The food production for centuries has been fragile and uncertain.

For centuries man found fertile soil, farmed it until the soil was exhausted and moved to another fertile soil area.

This problem still exists in most third world countries today - e.g. Africa.

Historically, agriculture has not been sustainable. Surprisingly, even during the early 1900's in North America, there was a decline in yields because of declining soil fertility, a depletion of soil reserves of nutrients.

Up until the 1940's corn yields remained at 25-30 bushels per acre, wheat yields remained at 10-20 bushels per acre.

Many believe that sustainability means farming within a "closed cycle". The theory is that plant nutrients pass through the food chain and then are returned to the soil without the need for external supplies. The truth is that plant nutrients are lost to both air an water plus exported from the farm as food. "Mineral fertilizers" replace these "lost" nutrients to the soil.

T. Malthis' 17th century forecast that the growth in the world's population would outstrip its food supplies has not occurred. Plant and animal breeding and other efficiencies have all been important factors. However, the provision of adequate crop nutrients has been an essential part of this increase in productivity.

Reference: - Statistics Canada
Conference Board of Canada CFI Position Paper - Page 2; The Fertilizer Handbook - Page 4, 5, 44; Chapter 1, Page 12; Soil Fertility and Fertilizers - Chapter 1, Page 6-8
Yield Trends

With regards to crop yield trends, it was observed that in the 20th century - from the 1940's - that a virtual "explosion" took place in crop yields in North America.

This recent "phenomenon" of crop yield increases coincided with increase fertilizer use.

Similar dramatic increases have taken place in other parts of the world, i.e. - wheat in England and rice in South East Asia.

It is estimated that 40% of the yield increase in Canada and 25% of yield increases world-wide are due to mineral fertilizers.

Fertilizer use increased in response to the needs for:
A. expanded crop production
B. more food
C. world trade
D. a more affluent society

Corn yields in the U.S.A. in 1900 were 25 bushels per acre and remained below 40 bushels per acre until the 1940's.

Wheat yields in England from 800 AD to 1600 AD were 10 to 15 bushels per acre. We may acknowledge that FERTILIZERS ARE THE DRIVING FORCE.

Corn Yields in Ontario

We note that the yields decrease between 1922 and 1941 (evidence of soil depletion); Yield started increasing in the 1940's.

A 48% increase from 1946 to 1955 - 42 to 62 bushels per acre was observed, also,
A 66% increase from 1956 to 1989 - 62 to 103 bushels per acre.
Total 145% increase in 43 years (1946-1989) - 42 to 102 bushels per acre.
Average yield increase of over 3.3% per year.
Yield has increased an average of 1.4 bushels per year.
This phenomenal increase is due to fertilizer use, improved seed varieties and improved farming practices.

Wheat Yields in Western Canada

We note yield decreases in the 1920's and 30's due to soil depletion, not just the drought.

From the 1940's forward there was a gradual yield increase of about +2% per year due to increased mineral fertilizer use.
Dramatic swings in yields are caused by weather variations — dry weather (drought conditions) created decreased yields.

We note that from the 1970's forward there has been a reduction in the dramatic swings in crop yields because increased fertilizer use enhanced water use efficiency, i.e. more bushels of crop produced per inch of available water; also new production practices improved conservation of moisture.

The 1990's average yield in the west was 32.9 bushels per acre.

FERTILIZER SOURCES

Raw materials for mineral fertilizers occur naturally in our environment - they come directly from nature.

Reference: - The Fertilizer Handbook - Chapter 1, Page 1-3 Soil Fertility and Fertilizers - Chapter 1, Page 5, 7-10 CFI Position Paper - Page 2, 9 Soil Fertility and Fertilizers - Page 5-17; Page 145-146 Ontario Ministry of Agriculture and Food Publication 20 Statistics Canada
Industry takes these "natural elements" of nitrogen, phosphorus, potassium and sulphur and turns them into a form which is:

1. readily available to the plant
2. easily transported in the plant (water soluble)
3. safe for plants to use as plant food
4. more controllable and predictable for crop initialization and response.

Since ancient times, man has tried to improve yields by adding other substances to the soil. In Roman times calcium carbonate - fish and bone wastes - were used.

Mineral fertilizers were introduced to Canada in the 1920's and became widely accepted around 1943/44.

**Nitrogen**

Air is about 78% nitrogen by volume, but is in a form that is unavailable for most plants to utilize.

Hydrogen is stripped from natural gas and combined with atmospheric (N₂) to produce anhydrous ammonia (NH₃), the basic building block for all nitrogen fertilizers.

The Canadian fertilizer industry uses approximately 9% of Canada's "domestic" natural gas consumption, this is less than 5% of Canada's total natural gas production.

Residential and commercial heating use approximately 42%. Other industry uses approximately 49% of domestic natural gas consumption.

Approximately 4% of the world's natural gas production is currently wasted as "flare gas", which is enough to manufacture all the nitrogen fertilizer needed to satisfy the world's demand. At the present time natural gas is our most efficient method of producing nitrogen fertilizers - transferring nitrogen from the air to plants; essentially - AGRICULTURAL FOOD PRODUCTION IS BASICALLY NITROGEN DRIVEN.

**Phosphorus**

Phosphate comes from natural phosphate rock ore deposits in the earth. These deposits are mainly compressed fossils and bones laid down in the sea beds millions of years ago.

Phosphate rock is very insoluble and "essentially non-available" for plant use. Strong acids (H₂SO₄ + HCl + H₃PO₄) are reacted with phosphate rock to produce soluble phosphate.

Phosphate is generally "very immobile" in the soil (because of low solubility in soil water). therefore most of its environmental problems are related to soil erosion.

Phosphate is an essential nutrient for both plants and animals. The industry is environmentally responsible.

Also, mined area is being put back into grass and trees, water bodies can then be used for recreational facilities, etcetera. Typically, phosphate mines are restored back to a natural state after the rock is removed.

**Potassium**

Potash comes from natural deposits in the earth. Refining processes small amounts of clay and removes impurities including sodium chloride (table salt) which are "detrimental" to crop production and soil fertility.

Canada's potash ore reserves in Saskatchewan and New Brunswick are some of the world's richest reserves and currently supply 25% of the world's plant nutrient potash consumption.

Not subject to leaching (minimal). Therefore moves into water system via soil erosion. Also, potassium increases Nitrogen uptake into the plant.

**Sulphur**

Sulphur is a natural element, mined as frasch sulphur, derived from petroleum refining, sour gas well and off-gasses as sulphur dioxide (SO₂) byproduct from mineral ore smelting. All sulphur in Canada is derived from these last 3 sources.

The major end use for Canada's sulphur, which represents 40% of world trade in sulphur, is in the production of phosphate fertilizers world wide. In addition, sulphur is often referred to as the 4th plant nutrient. Further, sulphur is an essential element in chlorophyll formation for plant growth.

**ALTERNATIVE NUTRIENT SOURCES**

All are excellent sources of plant nutrients.

Nutrients "regardless of source" (including mineral fertilizer) occur naturally in the environment.

Nutrients from all sources require precise application for efficient use and minimal environmental risk.

At the present time it is difficult to control the amount or the timing of N release from animal manures, forage legumes, or other types of organic wastes. This difficulty could actually increase the environmental risk with their use compared to mineral fertilizers.

**Animal Manure**

Manures are an excellent source of nutrients (soil fertility) which contribute to soil organic matter and soil structure. They offer an opportunity to "recycle on farm nutrients", also,

They provide approximately 1,000,000 metric tonnes of nutrients per year:

- **N** 235,000 m.t.
- **P₂O₅** 150,000 m.t.
- **K₂O** 500,000 m.t.

This amounts to approximately "one third of total nutrients" applied in Canada per year. Manures must be balanced with mineral fertilizers to provide the correct quantity of each individual nutrient required for crop production.

The "nutrient release" from manures (because of mineralization) can be unpredictable thus possibly increasing the environmental risk.

Improper storage handling facilities create an environmental risk. Only 400 manure samples were tested last year for nutrient content in Ontario; therefore, questionable application rates due to variable nutrient content and improper application create environmental risk.

Over application of manures can, under certain circumstances, reach tile lines within 20 minutes. It is a bad philosophy by some agriculturalists to treat manure as a waste and not as a resource.

The use of an organic nitrogen source for crop production requires an equal or greater management intensity than does commercial nitrogen to attain efficient utilization. M.S.U.

One of the greatest challenges facing agriculture today is more "Effective Manure Handling and Utilization Systems". We need more research on the nutrient value of properly stored manure.

Farmers need research on timing of manure application related to soil compaction; also, we need to encourage more education on manure as a resource.

**Sewage Sludge**

The sewage sludge is a good source of nitrogen and phosphate. Human sludge, on the other hand, can supply only a small percentage of total crop production nutrient needs.

Reference:  
- CFI Position Paper - Page 3
- The Fertilizer Handbook - Page 47, 48, 49; Page 96-98
- Soil Fertility and Fertilizers - Chapter 10, Page 419-433; Page 433-440, - Chapter 3, Page 62-66; Page 67-72
- Page 75-78, - Chapter 6, Page 189-239, - Chapter 7, Page 249-285, Chapter 8, Page 292-328

-13-
It may carry toxic chemicals, pathogenic organisms and/or heavy metals which under improper treatment and application could be transmitted to human food sources. They also create higher environmental risks to the soil and groundwater.

Sewage sludge can often only be utilized 1 year in 4 or 5 due to heavy metals. Present disposal methods include application on agricultural land, incineration, or dumping in landfill sites. It will become more significant in the future, possibly just to alleviate other environmental concerns (urban pressure). Good husbandry is to apply sewage sludge when the crops can best utilize it. In addition, environmental regulations and social stigma will be limiting factors in its use for food production.

**Green Manure**

Benefits of legumes and cover crops are well researched and documented. The greatest benefit is that it increases organic matter and improves soil structure. Further, the nitrogen fixation capabilities of legumes help reduce nitrogen needs in subsequent crops, but is unpredictable.

Accurate recommendations for "supplemental" N to meet crop yield goals are difficult to determine because the N release rate and amount from legumes are not well understood. This has "economic limits" in cash crop farming because of its "limited" resale market in non-livestock areas. In today's tight economic farm situation farmers cannot afford to lose the cash flow from $\frac{1}{4}$ to $\frac{1}{3}$ of their annual income for crop rotation purposes.

Research has shown that continuous legume cropping can increase the environmental risk of nitrogen contamination in groundwater (Rothamstead, England). Forages are heavy consumers of phosphate and potash, making applications of non-nitrogen fertilizers necessary.

**What are you Eating? - Page 2**

Consequently, legumes can place a high demand on water reserves in the soil, reducing their benefit in dry or semi-arid regions such as the Canadian prairies.

Nitrogen fixation and N release capabilities of some cover crops are questionable:

- **Oilseed Radish** - excellent absorption, but very early release. Therefore, of no use resulting in groundwater N.
- **Harry Vetch** - good absorption but leaks into sub-soil.
- **Rye** - leaking problems - N release too slow to benefit crop.
- **Red Clover** - probably the best at present time - also leaching problems.

**FERTILIZER AND WATER QUALITY**

A proper and more precise application of any nutrient source reduces the risk of water contamination.

The proper amount, timing and placement of the nitrogen source is essential for optimum plant utilization.

Nitrogen exists in nature, through decomposition of organic matter. This is a natural phenomenon and we must take advantage of this source of nitrogen.

Regardless of source, nitrogen may be found in ground and surface water because nature is not a leak-proof system.

Reference: [CFI Position Paper, Page 12](#)

[The Fertilizer Handbook, Soil Fertility and Fertilizers - Chapter 14, Page 656-662](#)
Nitrogen release from mineral fertilizer is predictable, the rate is well researched; nitrogen soil test in Western Canada has been used successfully for over 20 years.

Nitrogen soil test, just prior to planting time, is under study in Eastern Canada for corn production. This research is partially funded by the fertilizer industry in co-operation with the Ontario Ministry of Agriculture and Food.

More innovative methods of application, e.g. split application, deep placement etc. are under study for more efficient use - both economically and environmentally.

For example:

- 2" of rain on sandy soils can move nitrates as deep as 3 to 4 feet which could cause N deficiency in early corn growth. Limits of Nitrate Nitrogen in ground water
  - Ontario - 10 ppm
  - Germany - 40 ppm
  - Netherlands - 50 ppm

More research to develop better communication is needed to improve the farmers' use of Best Management Practices for nitrogen.

**Soil Nitrogen Cycle**

Mother nature is, overall a reliable supplier of nitrogen and can supply over twice as much nitrogen for crops as does mineral fertilizer nitrogen.

- 25% - biologically fixed by grain and forage crops
- 36% - atmosphere, animal waste, crop residue, rainfall
- 29% - mineral fertilizer

Organic matter mineralization and N fixation by living organisms are the major sources of available N in most soils. According to estimates, only 1% to 4% of the N in organic matter may become available each year. Mineralization of organic N in soils does not proceed at constant rate, but varies according to soil temperature and other factors.

One source of nitrogen is no more natural than the other, chemically they are the same and behave identically in nature.

The availability of soluble nitrogen (ammonium and nitrates) for the food production and its susceptibility of losses to the environment depend on the Nitrogen cycle and steps within it.

All N sources, both organic and inorganic, have the potential to generate nitrate, which can leach into ground water. Changing to low-input systems to diminish the leaching nitrates has little scientific background/basis.

All fertilizer does is to increase the amount of ammonium (NH₄) and nitrate (NO₃) available to the plant to produce the maximum economic yield with minimal impact on the environment.

Nitrogen is one of the key elements of life. From the atmosphere it moves into land and water, undergoes biological and chemical transformations becomes a part of living and non-living matter and eventually returns to the atmosphere. In this cycle, nitrogen from the air we breathe today might come a constituent of protein we consume tomorrow.

"Mother Nature" is a power to be reckoned with, and she will continue to show us "WHO'S THE BOSS". High nitrate levels exist deep in the soil where agriculture has never existed, e.g. - MOJAVE DESERT.

All nitrogen sources, when applied to the soil, are gradually transformed by soil organisms to inorganic ammonium (+) and nitrate (-) ions. Plant roots take up these two nitrogen forms almost exclusively. Nitrate is the nitrogen form found in groundwater because ammonium is held by soil particles and is not subject to leaching.

We may acknowledge that: NITROGEN IS ONE OF LIFE'S VITAL ELEMENTS.

**Nitrate Nitrogen (NO₃-N)**

The results of field tests at the University of Guelph, 1979 suggested that plants need a certain quantity of nitrogen in order to develop a productive crop.
Nitrogen in soil comes from the decomposition of soil organic nitrogen, crop residues and manures. The amount added by mineral fertilizer only fills the nitrogen shortfall for optimum economic crop production.

Surveys of nitrate-nitrogen discharge into tile lines indicates that nitrogen recommendations should be closely followed.

Nitrogen applied at the recommended rate is much more efficient economically and environmentally than when applied below or above the rate required.

Results of fields 1 to 4 showed that nitrate loss is similar to natural ecosystems, application rate and recommendation rate very closely matched with crop requirements. Note that the highest recommendation - field 3 - has the lowest leakage - 4 Kg/ha. High rates of application do not place the environment at risk if the application is agronomically (scientifically) sound.

Results of fields 5 & 6 showed leakage; more nitrogen was applied than was recommended.

The fertilizer industry, both financially and morally, supports the research programs in Ontario to develop a NITROGEN SOIL TEST for making sustainable recommendations. A nitrogen soil test for Western Canada has been used for many years.

**SOIL EROSION**

**Soil Erosion - Water**

Field erosion by water is a very serious matter cross Canada. Water will erode productive soil and plant nutrients either attached to soil particles or in solution. This creates environmental concerns and an economic loss.

More education/communication of proper tillage and crop residue management practices are needed. In addition, construction sites and urban development also contribute to this problem.

Surface water contamination moves into wells and them becomes groundwater contamination.

**Soil Erosion - Wind**

Excessive tillage is a practice that must be discontinued; it incorporates air into the soil, this increases oxidation, i.e. breakdown of the soil organic matter. Also, it breaks the soil up into smaller lumps which present more surface area to wind and water - therefore more subject to wind and water - therefore more subject to their impact (rule of physics-surface area vs mass).

Summer fallowing practices in the West have been a leading contributor to this problem.

Wind erosion results in the loss of soil organic matter, soil structure and plant nutrients.

Our land resource requires careful management, it is estimated that less than 9% of Canada's land mass is suitable for cultivation.

CFI totally supports education to improve land management.

Increased tillage in summer fallowing reduces the amount of organic matter and increases the loss of nitrogen through oxidation, to the air, thus creating an environment risk and an economic loss.

**Erosion Control**

CFI actively supports "conservation tillage" practices designed to promote a viable sustainable agriculture.

A minimum of 30% crop residue coverage on the soil surface is the target level to minimize water and wind soil erosion.
Reduced tillage maintains soil structure and lessens the threat of soil erosion.

Proper fertilizer usage enhances crop production and plant growth which in turn ensures abundant residue for viable conservation practices.

Compare yourself to sleeping in a bed and when the weather is cold you cover yourself with a blanket for protection. Fields also should be covered by a residue for protection.

Research has shown that mineral fertilizers will increase the production of organic residue and enhance the soils productivity.

FERTILIZER AND SOIL QUALITY

The soil quality is the ability of the soil to sustain, accept, store and recycle nutrients and water. Soil consists of 50% solids including organic matter, 25% water solution and 25% air (atmosphere). Solids supply physical support for the plant plus hold nutrients.

Water solutions are the primary carrying agent to move nutrients into the plant.

Air (atmosphere) supplies oxygen and gaseous nitrogen for nutrient utilization by the plants.

Organic matter is the stamina of the soil and a source of nutrients. It is important for the soil's nutrient and water holding capacity. Organic matter improves soil structure. Organic matter is created by returning large volumes of crops residue and livestock manures to the

Soil Structure

Soil texture is the percent of sand, silt, and clay that make up the soil. This does not change under normal tillage practices.

Soil structure is the physical component of the soil. We measure structure by bulk density; good soil structure promotes root and water penetration.

Nutrient Balance

Mining the soil negatively affects soil structure by reducing crop production and plant residue returned to the soil (organic matter). That in turn increases the potential for soil degradation and erosion.

Many studies show when adequate amounts of P & K are supplies, corn yields are highest, and the most efficient use of N results. Historically, fertilizer application in Western Canada has not kept pace with nutrient removal because of the export of crops off the farm. Since 1985, agriculture in Eastern Canada has been mining soil nutrients as well.

Fertilizer is a powerful "soil conservation tool", providing: a) increased crop production; b) increased residue; and organic matter and c) increased water holding capacity.

Through nutrient balance and soil fertility practices; the result is the "reduction" of: a) wind erosion; b) water erosion and c) nutrient losses in groundwater.

We may conclude that MINERAL FERTILIZER HELPS BUILD SOILS.

Microbial Activities in Soil

The bacteria are an essential part of the soil system.

Reference: The Fertilizer Handbook - Pages 24-27, 36-41
Soil Fertility and Fertilizers - Chapter 14, Pages 637, 648-654;
CFI Position Paper - Page 10

Reference: Soil and Fertility and Fertilizers Organic Matter - Chapter 14, Page 634, 635 Soil Structure - Page 557, 584, 585
The production soils contain a variety of microorganisms; the greater the bacteria population, the more efficient is the soil system (within limits of supply and demand).

Soil microbial population increases when organic matter is added. Through residue decomposition nutrient are released.

**Fertilizer use supports, stimulates and increases soil microbial populations and activity.**

This could encourage or RESULT IN A MORE PRODUCTIVE SOIL.

**Fertilizer and Air Quality**

Carbon is the basic building block of all organic compounds, for example carbohydrates, lipids and proteins.

Plants get their carbon by absorbing carbon dioxide from the air. With water from the soil and energy from sunlight, carbon dioxide is converted through a chemical reaction called photosynthesis, into glucose and energy - e.g. head of wheat, cob of corn, etc. This process also releases oxygen.

Photosynthesis has many positive balancing effects on the environment - from consuming carbon dioxide to producing oxygen. A 100 bushel crop of corn will utilize 6.7 tons of carbon dioxide and will generate 4-6 tons of oxygen, basically, plant growth is a cleaner of the air.

**Fertilizer helps "drive" this whole system of "photosynthesis".**

Photosynthesis is the first reaction of life, oxygen consuming animals (animal kingdom) break down the glucose, etc., through respiration and decomposition of food back into carbon dioxide, water and energy for reuse by plants (plant kingdom). This linkage of photosynthesis (plant kingdom) and respiration (animal kingdom) is a major part of the carbon cycle.

**FOOD QUANTITY AND QUALITY**

In 1900, 1 farm fed 3 urban people. In 1990, 1 commercial farm feeds over 300 people. By the year 2000, it is estimated that 20% of the farmers will feed 80% of the people.

The QUANTITY OF FOOD produced contributes to the quality of our diets. Canada rates amongst the highest in FOOD QUALITY in the world. Our effective food production system maintains a constant, low cost food supply and a high quality life style. One of our major health problems is not food quality, but "OBESITY" (**say with kindness**).

Fertilizer can improve quality by:
- improving nutrient balance in food (foods grown on a deficient soil can be deficient in these nutrients),
- increasing protein levels in wheat and potassium levels in bananas and kiwi fruit,
- reducing diseases in fruit and vegetable crop (healthy, well fed crops can better resist diseases), and
- improving livestock rations.

If world population increases as rapidly as experts predict, will additional food come from "new land" or "existing land". It will have to come primarily from "existing productive land".

If current per capita food consumption *stayed constant*, population growth will require that annual food production will have to expand by 70% over the next 37 years; however, if diets improved among the poor and undernourished -- world food demand by the year 2025 would have to more than double the 1988 harvest. Please note the following:

<table>
<thead>
<tr>
<th>Year</th>
<th>Acre Major Crop Production per Person</th>
</tr>
</thead>
<tbody>
<tr>
<td>1980</td>
<td>.43</td>
</tr>
<tr>
<td>1990</td>
<td>.37</td>
</tr>
<tr>
<td>2000</td>
<td>.32</td>
</tr>
<tr>
<td>2010</td>
<td>less than .30</td>
</tr>
</tbody>
</table>

Reference: - CFI Position Paper - Pages 5, 6
The Fertilizer Handbook
Soil Fertility and Fertilizers - Pages 176-178, 548; Chapter 2, Page 36-38
Therefore a "sustainable agricultural system" cannot just maintain its production capacity but must increase it to be useful to society.

We may compare this with toxic substances:

**TOXIC RESIDUE** - media uses 100 ppb instead of 0.1 ppm because it sounds worse (more sensational).

1 P.P.M. = 1 grain of salt in a large swimming pool  
1 P.P.B. = 1 cent in $10 million  
4 cm. in distance around the world  
1 grain of wheat in 2,000 bu.

**LIFE EXPECTANCY** - average for male and female

<table>
<thead>
<tr>
<th>Year</th>
<th>Male</th>
<th>Female</th>
</tr>
</thead>
<tbody>
<tr>
<td>1920</td>
<td>58</td>
<td>58</td>
</tr>
<tr>
<td>1940</td>
<td>63</td>
<td>63</td>
</tr>
<tr>
<td>1960</td>
<td>70</td>
<td>70</td>
</tr>
<tr>
<td>1980</td>
<td>75</td>
<td>75</td>
</tr>
</tbody>
</table>

Due to improved diets (quality and quantity of food) and improved medicines.

**STRATEGIES FOR SUSTAINABILITY AND VIABILITY**

We plan to take several directives:

- Field Specific Fertilizer recommendation
- Precise Fertilizer application
- Conservation Farming
- Crop Rotation
- Communication - extension - education
- Research and development

We have a plan and this is what we are doing.

To become more efficient, both economically and environmentally we must take into account regional sensitivity and knowledge of local agro-environmental realities:

- soil type  
- crop rotation  
- credit nutrients from all sources  
- production trends  
- environmental impact on water and soil quality

We must develop a FARM NUTRIENT BUDGET (IMPORTS AND EXPORTS AT THE FARM).

**D.R.I.S.** - Diagnosis and Recommendations Integrated Systems
- indicates most limiting nutrient
- allows one to classify yield factors in order of limiting importance.

**SYSTEMS AGRICULTURE:** Precision Fertility Management
- involves an understanding of how various agronomic variables interact with each other to impact overall production efficiency and environmental sustainability.

Hence, WE MUST BECOME EVEN MORE SCIENTIFIC AND SITE SPECIFIC IN MAKING CROP RECOMMENDATIONS.

**Precise Fertilizer Application**

Precise fertilizer application techniques must be more precise in rate, placement and timing.

Application equipment is becoming more sophisticated, and our ability to measure soil parameters is becoming more precise.

Split application of N; the careful selection of rates, placement, sources and timing makes it possible, with mineral fertilizers, to supply nutrients at rates close to optimum levels to achieve economical and environmental efficiency.

New equipment of the future, grid sampling, on the go mixing, on the go N calibration, etc.

**REFERENCE:**  
  Soil Fertility and Fertilizers - Chapter 15, Page 695
Conservation Practices

Key areas to consider using:

Conservation practices by farmers are responsible stewards of the soil. We must conserve this national resource, protect the soil from water and wind erosion and build soil quality. Farmers are environmentally conscious. They know the environmental factors of their land better than most, and they incorporate this understanding into the use of their land.

Crop Rotation

- improves soil structure
- improves soil quality
- improves soil organic matter,
- is essential for ECONOMIC STABILITY and is environmentally stable.

Crop rotations - without added fertilizer, help maintain organic matter and N levels in the soil, but do not maintain high productivity levels (Morrow Plots - University of Illinois, 1876).

A combination of crop rotation and fertilizer produces the highest yields and maintains the highest soil organic matter levels. (Morrow Plots).

Communication, Extension and Education

Farmers

Technology transfer must be improved, the communication gap can be solved by extension.

World knowledge must be tested, adapted and adopted or rejected faster and faster.

Farm suppliers must assume more responsibility in technology transfer and extension, this is part of the shared responsibility of the agro-industry's stakeholders.

Dealers must become more scientific in this leadership role.

Research data must be turned into user friendly information more practical for adoption.

Sustainable technology must be accessible to farmers.

Consumers

We must communicate with and educate consumers so they can better understand the realities of food production in Canada.

Research and Development

Research and development is the strength of the agriculture industry.

Canadian agriculture is based on science and we must continue to promote scientific answers in the key areas.

The adoption of new research will not involve radical changes as some extremists advocate, but rather the continued improvement of production agriculture.

Competitiveness and value added are key components of our research for the 90's.

The fertilizer industry is committed to production agriculture with minimal impact on the environment.

The real erosion facing agriculture in the 90's may not be the erosion of our top soil, but the erosion of technology and extension to support it.

Farmers are innovators, their expertise should be sought in consultation on the direction of the research and development area.

A viable agriculture requires a strong sustainable effort in science and technology.

What form of nitrogen we apply and maintain in the soil can play a significant role in its use. Studies have shown that plants respond to - an enhanced supply of ammonium nitrogen (mixture of \( NO_3 \) & \( NH_4 \)). A nitrification inhibitor is used so the roots can "see" an enhanced supply of ammonium - inhibitor slows down the conversion to nitrate N. Research will help dealers find the right N combination to maximize N use for the customer.
Eventually corn hybrids and varieties will respond to "enhanced" N on a scale of 1 to 10.

We cannot go back to the past in agriculture, any more than we can go back to the past in medicine, space transportation or any other branch of science and culture.

SUMMARY

Agriculture in Canada

Farm cash receipts show this is a $22 billion business in Canada.

Food/Beverage/Restaurant Industry

Is one of Canada's largest industries: 20% of our population work in this industry - this strong industry allows other people to work at other productive jobs.

Agricultural Trade

Over $4 billion trade surplus
Fertilizer trade surplus is a further $2 billion.

Food in Canada

Economical - 10.1% of average Canadian's income.

High quantity; high quality.

Health of Canadians

Our children's development is based on excellent nutrition.

Food Production

Farming is based on science.

Fertilizer

Plays an essential part in the science of food production.

We conclude that: AGRICULTURE TODAY IS SUSTAINABLE AND WILL, IN THE FUTURE, MAINTAIN A PROPER BALANCE WITH THE ENVIRONMENT.

CONCLUSION

Sustainable agriculture systems are those that are capable of maintaining their productivity and usefulness to society indefinitely. These systems must be resource conserving, environmentally sound, socially supportive and commercially competitive. These systems must be derived region by region, farm by farm, crop by crop and field by field. Fertilizer will play a major role in these systems.

CONCERNS:

GROUNDWATER CONTAMINATION - N-("Industry gets the blame")

- tracing N in wells because nitrate occurs naturally and originates from many sources and which source is responsible?
- is well location the problem? feed lots, septic tanks, fertilizer handling sites etc.
- some studies have shown that N levels averages less when the well site was closer to production fields than the homestead.

N- TEST

- strong supporter but question (at present time) its validity for making accurate "reliable" economic N recommendations.

QUESTIONS:

- reliability of sampling techniques and timing?
- freezing or air drying of samples?
- accuracy evaluating residual N after plowdown legumes, small grains, etc.?
- consistent accuracy of test before and after planting, had variances in recommendations as high as 100 lb. per acre?
- 40 bu./acre difference in yields from "O" recommended site and normal N recommendation - was N test accurate?
- works well after manure application, but again the rate of mineralization creates questionable supplemental N recommendations?
- conditions in Western Canada for N test are much different from Eastern Canada
rainfall, temperature, etc.
- can we isolate N only or should we take into consideration the P & K levels to get a true picture of N efficiency and leaching capabilities (nutrient interaction).
- fertilizers applied to growing crops are not the major cause of nitrate pollution - the way we farm.
- do we use the N test alone to dictate N requirements and throw out the N recommended rates as established by O.S.M.R.C. over the past 20 years of research and experience?
- do we fully appreciate the value of extra N in the soil to help build organic matter or are we going to deplete the soil N levels resulting in loss of O.M. and soil structure? (e.g. Western Canada - summer fallowing released N, depleted O.M., thus poor soil structure resulting in soil erosion).

WOULD YOU AT THE PRESENT TIME, AS A SUPPLIER, MAKE AN N RECOMMENDATION BASED STRICTLY ON THE N TEST?

REMEMBER: YOUR REPUTATION AND COMPANY SUCCESS MAY BE AT STAKE!!

************

WE MUST CONTINUE THE RESEARCH BUT PROCEED WITH CAUTION FOR ITS WIDESPREAD USE; REMEMBER WE ARE PLAYING WITH SOMEONE'S ECONOMIC VIABILITY - IT'S SCARY!!!

************

In short, the foundation of current food production is being challenged and threatened by technological constraints, social doubts, poor scientific education, and philosophical differences. Any mandatory limit on fertilizer use will place a ceiling on yield levels and curtail farmer’s management flexibility and use of B.M.P. for specific cropping systems. At the same time, it is the responsibility of the farmer and his suppliers to see that fertilizer is applied accurately and at rates consistent with M.E. yield goals and environmental sustainability.

The fertilizer industry is a mature industry and knows its responsibility in food production not only in Ontario but as a stakeholder on a global basis.
PERFORMANCE OF RIESLING GRAPES AND THE LEACHING OF NITRATE ON A CLAY LOAM SOIL

R.A. Cline

ABSTRACT

Grape cultivars grown for wine production in the Niagara Peninsula of Ontario have changed drastically over the past 10 years. The trend has been away from hardy labrusca types to more cold sensitive vinifera cultivars and hybrids. Nitrogen (N) management becomes very important because of the need for high quality fruit and the possibility of winter injury to such vines. In 1991 an experiment was initiated to study the effects of time and rate of N fertilizer application on the performance of mature Riesling vines. Also of concern is the loss of N to ground water and the effects of times and rates of application on this loss.

Ammonium nitrate, the common source of N grapes, was applied once on April 4, 26, May 14 or 31 at a total of 34 or 68 kg N/ha and compared to a 0 N treatment. The soil, a Jeddo clay loam, is poorly drained, but drainage is improved by small stones throughout the profile area and by tile drainage.

The soil was sampled at 5-cm intervals to 30 cm, and 15-cm intervals from 30 to 60 cm in mid-June, July, August and October. Samples were analyzed for nitrate using the specific ion electrode method. Yield, pruning weight and petiole analysis data were collected to determine vine response.

Soil nitrate concentrations in June reflected very well the rates of N fertilizer application. Differences were greatest at the surface, but did show to the 60 cm depth. In June the nitrate concentration at 0-5 cm was largest with the last date of application, and decreased with earlier times of application. Nitrate levels decreased with depth. Nitrate levels were lower in the 0-5 cm samples in July and August than in June. The pattern of nitrate concentrations no doubt will be affected by rainfall. In 1991 no growth differences between N treatments were measured. Yield of Riesling was largest for the current general recommended practice of applying 34 kg N/ha in mid May.

INTRODUCTION

In the Niagara Peninsula of Ontario there has been a dramatic change in grape cultivars since the introduction of the Free Trade Agreement. Labrusca and Hybrid cultivars have been replaced by Vinifera or hybrid cultivars on better soil and climate locations. Nitrogen (N) requirements of these cultivars have not been determined under Ontario conditions. Since these cultivars are sensitive to winter injury, excessive N could result in serious damage to vines from low winter temperatures and reduced production. Clore and Brummond (1) have reported that nutritional imbalances, including excessive N, can lead to more serious freezing injury in grapes. Wample et al (3) suggest that the relationship between high N and reduced grapevine hardiness is widely accepted but not conclusively proven because of wide variability to genetic, environmental and cultural factors. Diseases such as Botrytis are more difficult to control with these cultivars and may be more serious when N levels are high (2). Excessive N fertilization could also lead to elevated nitrate levels in ground water, which is of increasing concern from an environmental standpoint. An experiment was initiated in 1991 to study the effects of N fertilizer rate and time of application on the growth, yield, nutrient uptake and fruit quality of Riesling grapes. The effect of N treatments on nitrate concentrations at various soil depths in the soil profile and ground water were also determined.
MATERIALS AND METHODS

The experiment was conducted in a mature Riesling vineyard located adjacent to Lake Ontario, between Vineland and Beamsville. The soil, a Jeddo clay loam, is classed as poorly drained. The vineyard had been tile drained and stones in the soil profile aided water movement. Every other row of the vineyard was cultivated and a cover crop of oil seed radish and ryegrass was seeded in early August. Alternate rows were planted to permanent grass. Treatments were applied to the cultivated rows using two rates of N as ammonium nitrate (34 or 68 kg total N/ha) on April 4, 26, May 14 or 31 in 1991.

Soil samples were taken 4 times beginning in June and repeated in mid July, August and November. Samples were collected at 5-cm intervals to a depth of 30 cm and then at 15-cm intervals to a depth of 60 cm. Samples were analyzed for nitrate using a nitrate electrode and a coming Model 12 research pH meter. Five-cm diameter plastic pipes were placed in augured holes to collect drainage water at 1.25 m depth for nitrate analyses.

Leaf samples were collected opposite bunches in June, and leaf area, leaf nitrate and total leaf N were determined. Leaf petiole samples taken from leaves opposite bunches in late August were analyzed for total N by the Kjeldahl procedure and for K, Ca, Mg, Mn, Fe and Zn using ashing and atomic emission or absorption procedures. Yield, fruit sugar, acid and pH were measured. Pruning weight was used as a measure of growth.

The experiment used a randomized block design with date of application as the main plot, and rate of N was assigned at random within each block. Plot size was 8 vines, with records taken from the centre 6 vines. There were 6 replications of each treatment. Duncan's multiple range test was used to determine significance after analysis of variance.

RESULTS

Although total rainfall in the 1991 growing season was near the long term average, the rainfall in May and June was only 65% of normal at Vineland. This no doubt influenced the effects of treatments, rate of movement of nitrate, and soil nitrate content at various depths.

The soil nitrate concentration in June 1991 reflected the rate of N fertilizer application (Fig. 1A). Average soil nitrate at the 0-5 cm depth for the 68 kg N/ha application rate was double that for the 34 kg N/ha rate. Nitrate concentrations decreased with depth. Differences in soil nitrate concentration between N rates also decreased with depth, but were consistently higher with the higher rate of N application at all depths. The samples from check plots had the lowest nitrate concentration at all depths.

Nitrate concentrations in the 0-5 and 510 cm samples were much lower in July than June with both N fertilizer rates (Fig. 1B). Since nitrate concentrations did not increase at depths greater than 10 cm, the N was very likely taken up and utilized by the vine rather than lost through leaching. Similarly, August soil nitrate concentrations reflected rate of N fertilizer but did not indicate leaching loss, since concentrations did not increase at lower depths of the soil profile (Fig. 1C). Soil nitrate concentrations tended to increase with time in control plots, indicating a release of organic nitrogen by microbial activity as the soil warmed.

The effects of time of N fertilizer application on soil nitrate concentrations are shown for the 68 kg/ha rate of N in Fig. 2. Trends were similar for the 34 kg/ha of N. Soil nitrate concentration at the 0-5 cm depth tended to be highest in June when N fertilizer was applied April 26, and lowest with the April 4 application (Fig. 2A). By August, nitrate had moved down from the top 10 cm when N fertilizer was applied in April, too be higher at all depths greater than 15 cm than with May fertilizer application (Fig. 2C). However, except for the samples taken at 0-5 cm in June, there were no large or consistent differences in soil nitrate with the different times of fertilizer application.
The effect of sampling time on soil nitrate concentrations is better shown by Fig. 3. Nitrate concentrations at the 0-5 cm depth decreased with time after fertilizer applications. This was most evident at the 68 kg/ha N rate (Fig. 3A). At 15-20 cm this trend was less pronounced (Fig. 3B) and did not occur at the 45-60 cm depth (Fig. 3C). In contrast, the soil nitrate concentration tended to increase with time with the check treatment, particularly at the 45-60 cm depth (Fig. 3C).

In June, all N treatments except 34 kg N applied May 14 increased leaf blade fresh weight compared to the control (Table 1). Leaf nitrate also tended to be higher when N fertilizer was applied. However, leaf weight or nitrate content did not differ consistently among treatments. Total leaf blade N in June also reflected time and rate of N fertilizer (Table 2). All times of application of 68 kg N resulted in higher total leaf N than the check. The April 4 application resulted in higher N than other times. The differences between rate of application were not significant. In August there were no significant differences in petiole N. The concentration of other nutrients in leaves and petioles was not affected by N fertilizer treatments. Magnesium concentrations were lower and potassium higher than expected. (Data not shown).

Yields in 1991 (Table 3) tended to be larger with 34 kg/ha rate of N than with 68 kg N/ha or the check treatment. The largest yield was with the mid-May time of application. Earlier or later applications were not as effective. Late May application of 68 kg N/ha depressed yields. Mid-May application of 34 kg N/ha corresponds very closely to the rate and time of application currently recommended and practised by many growers in Niagara.

Fruit sugar was not affected by fertilizer treatments other than a reduction of sugar by the low rate of N April 26 compared to the check and some of the other treatments (Table 3). Growth of vines as measured by pruning weights as not significantly different for the different times or rates of N fertilizer. (Data not shown).

**CONCLUSIONS**

There are no indications in this study that current N fertilizer recommendations for grapes need to be changed for the Riesling cultivar in regard to time of application of N fertilizer. Although soil nitrate concentrations were higher for the higher rate of N applied, there was no indication of excessive leaching loss of nitrate from the treatments in this experiment. Soil nitrate measurements were useful to monitor nitrate movement. Climatic factors, particularly rainfall, could influence these results, which are for only one year. The experiment will continue.

**BIBLIOGRAPHY**


Table 1.  The effect of time and rate of N application on leaf weight and nitrate concentration in June 1991.

<table>
<thead>
<tr>
<th>Time of application</th>
<th>Leaf fresh wt/g</th>
<th>Leaf Nitrate (ppm)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>N rate (Kg/ha)</td>
<td>N rate (Kg/ha)</td>
</tr>
<tr>
<td>Leaf fresh wt/g</td>
<td></td>
<td></td>
</tr>
<tr>
<td>34</td>
<td>68</td>
<td></td>
</tr>
<tr>
<td>April 4</td>
<td>2.10b&lt;sup&gt;1&lt;/sup&gt;</td>
<td>64.3ab</td>
</tr>
<tr>
<td>April 26</td>
<td>2.27c</td>
<td>73.7b</td>
</tr>
<tr>
<td>May 14</td>
<td>2.13b</td>
<td>74.2b</td>
</tr>
<tr>
<td>May 31</td>
<td>2.11b</td>
<td>73.5b</td>
</tr>
<tr>
<td>Control (Non)</td>
<td>2.03a</td>
<td>62.2a</td>
</tr>
</tbody>
</table>

<sup>1</sup> means followed by the same letter were not significantly different, P = 0.05

Table 2.  The effect of time and rate of N application on June 7 leaf blade N and August 26 leaf petiole N.

<table>
<thead>
<tr>
<th>Time of application</th>
<th>Leaf Blade % dry wt.</th>
<th>Leaf Petiole % dry wt.</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>N rate (Kg/ha)</td>
<td>N rate (Kg/ha)</td>
</tr>
<tr>
<td></td>
<td>34</td>
<td>68</td>
</tr>
<tr>
<td>34</td>
<td>68</td>
<td></td>
</tr>
<tr>
<td>April 4</td>
<td>3.50ab&lt;sup&gt;1&lt;/sup&gt;</td>
<td>0.78a</td>
</tr>
<tr>
<td>April 26</td>
<td>3.51ab</td>
<td>0.82a</td>
</tr>
<tr>
<td>May 14</td>
<td>3.55a</td>
<td>0.76a</td>
</tr>
<tr>
<td>May 31</td>
<td>3.57a</td>
<td>0.80a</td>
</tr>
<tr>
<td>Control (Non)</td>
<td>3.45b</td>
<td>0.78a</td>
</tr>
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</table>

<sup>1</sup> means followed by the same letter were not significantly different, P = 0.05

Table 3.  The effect of time and rate of N application on yield and fruit sugar content of Riesling grapes in 1991.

<table>
<thead>
<tr>
<th>Time of application</th>
<th>Yield (kg/Vine)</th>
<th>Fruit Sugar (°Brix)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>N rate (Kg/ha)</td>
<td>N rate (Kg/ha)</td>
</tr>
<tr>
<td>34</td>
<td>68</td>
<td>34</td>
</tr>
<tr>
<td>April 4</td>
<td>7.3ab&lt;sup&gt;1&lt;/sup&gt;</td>
<td>17.0ab</td>
</tr>
<tr>
<td>April 26</td>
<td>7.1b</td>
<td>16.6b</td>
</tr>
<tr>
<td>May 14</td>
<td>7.8a</td>
<td>17.2a</td>
</tr>
<tr>
<td>May 31</td>
<td>7.3ab</td>
<td>17.3a</td>
</tr>
<tr>
<td>Control (Non)</td>
<td>7.0b</td>
<td>17.3a</td>
</tr>
</tbody>
</table>

<sup>1</sup> means followed by the same letter were not significantly different, P = 0.05
FIGURE 1: The Effect Of Rate Of N Application On Soil Nitrate Content At Various Soil Depths.
FIGURE 2: The Effect Of Time Of Application Of 68 Kg N/ha On Soil Nitrate Content At Various Soil Depths.
FIGURE 3: The Effect Of Time Of Sampling On Soil Nitrate Content At Three Soil Depths.
THE LAND APPLICATION OF LIQUID SWINE MANURE AND ITS EFFECTS ON TILE DRAIN WATER QUALITY

M.E. Foran and D. Dean

ABSTRACT

Research conducted by the ABCA in 1989 and 1990 revealed that land application of liquid manure under normal farm practices degraded tile drain water quality. This contamination is of concern as it may result in downstream beach closures due to elevated bacterial concentrations and excessive algae growth. Farm operators and custom applicators involved in this study were asked to spread their liquid manure as they would normally and the investigators monitored changes in bacteria and nutrients in the soil and tile drain water. In eight out of 12 spreading events monitored, manure components travelled rapidly through the soil column and reached tile drains. It appeared as though flow in the tile drain was required for contamination to occur at the time of manure application and an acceptable rate of application may be dependent on whether or not there was tile flow.

In 1991 research focused on ways of reducing the impact of liquid manure application on tile drain water and groundwater quality. Spreading on recently tilled land versus untilled land and spreading using an irrigation gun versus an injector tanker were investigated. Two liquid manure spreading events for each of the alternatives was carried out. Tilling the land prior to manure application decreased the amount of bacteria and nutrient loading to the tile drain water. The injection method of application had a higher loading in the tile drain water compared to the irrigation method.

INTRODUCTION

Agricultural drains in Ontario are often characterized by high bacterial, nitrogen and phosphorus concentrations. Studies by the Ausable Bayfield Conservation Authority (Balint 1985) and the Ontario Ministry of the Environment (OME 1984) established that in some cases land application of liquid manure causes bacterial and chemical contamination of field tile drains. That contamination may result in beach closures due to elevated bacterial concentrations and excessive algae growth.

Evans and Owens (1972) found a 30-900 fold increase in fecal bacteria concentrations in the tile effluent from a sandy clay loam pasture field within two hours of spreading liquid manure. A one time observation by Patterson et al. (1974) found the water from 1.5 m deep drains in a plowed field to be cloudy and foul smelling within 30 minutes after a 5 mm (50,000 L/ha) application of liquid manure.

A study by Aubertin (1971) showed that macropores can conduct large volumes of water very quickly. Quisenberry and Phillips (1978) concluded that in some soils with strong structure and rapid water addition nearly all water flows through the soil macropores. Macropores need not extend to the soil surface for this to occur, however, flow rates are reduced. Beven and Gennann (1982) suggested that the presence of macropores at the soil surface always increases the infiltration rates because additional surfaces are made available for infiltration into the matrix at depth. Smith et al. (1985) stated that macropore flow significantly reduces the soils ability to retain bacteria and viruses and increases

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1 Ausable Bayfield Conservation Authority
the potential of groundwater contamination. Almost half of all water borne diseases are caused by contaminated groundwater (Gerba and Bitton, 1984).

This research project consisted of two separate objectives over the three year period. The objective of the first two years of research (1989 and 1990) was to determine the impact of liquid manure application on the tile drain water quality under normal farming practices as well as to determine the factors influencing the impact. The objective of the third year of research (1991) was to determine ways of reducing the impact of liquid manure application on tile drain water quality. The irrigation and the injection methods of liquid manure application were compared as was applying liquid manure to recently tilled land versus unfilled land. The project was conducted at a field scale and involved the spreading of liquid manure on farmland then monitoring the changes in the tile drain water.

1989 AND 1990 RESEARCH

Methodology

Twelve manure spreading events were monitored at seven sites in Huron and Perth Counties, Ontario. Farm operators and custom applicators were asked to spread the liquid manure as they would normally and investigators monitored changes in the field tile drain waters. Observation chambers were installed at the sites to facilitate water sampling from the field tiles. Field tiles were located approximately 90 cm below the soil surface. The drain tile spacing at most of the field sites was systematic at 15 metre intervals. The irrigation method of spreading was used for 11 of the manure applications, while a tanker spreader with manure broadcast on the ground was used for the remaining one.

Water and manure sampling and analysis

Weekly grab samples of water from the tile drains were taken for one month prior to the scheduled manure application. More intensive sampling was done on the day of manure application. Daily water samples were taken for approximately one week immediately following spreading. Tile drain discharge measurements were taken at time of sampling. This was accomplished by measuring the amount of time it took to fill a container of known volume.

Samples of the liquid manure were taken from the manure storage prior to manure application. Samples were also taken from collection pans placed in the area of manure application. A minimum of three bacterial and two chemical samples were taken from the manure storages and collection pans so average concentrations could be calculated. Concentrations of bacteria and nutrients in the collection pans confirmed whether or not changes in concentrations occur as a result of pressurization during application.

Bacterial parameters analyzed in the water and manure included: fecal coliform, fecal streptococcus and Escherichia coli. The biotracer, nalidixic acid resistant Escherichia coli (EC(NA)) was added to the irrigation pipe prior to pressurization and to the tanker in order that their movement could be traced through the soil column and into the receiving water. The tracer organism was originally isolated from the environment by G. Palmateer, Ontario Ministry of Environment Southwestern Region (OMESWR). The EC(NA) is non-entero-pathogenic, is easily recovered from water and soil and is not commonly present in the natural environment. The tracer bacteria was injected in concentrations that were approximately equal to the concentrations of the indigenous bacteria found in the liquid waste. Water and manure samples were stored on ice immediately and analysis was performed within 24 hours.

The chemical parameters analyzed in the tile drain water and manure included; biochemical oxygen demand, suspended solids, free ammonia, total kjeldahl nitrogen, nitrate, nitrite, total phosphorus, dissolved phosphorus, pH, chloride, conductivity and potassium.

The microbiological and chemical analysis was performed at the OME-SWR. Laboratory methods used by the microbiology and chemistry departments are those outlined by the Handbook of Analytical Methods of Environmental Samples.
(OME 1984b). In addition the microbiology department uses the Standard Methods for the Examination of Water and Wastewater (APHA 1985).

**Soil sampling and analysis**

Soil samples for bacterial analysis were taken a few days prior to and within three hours following manure application. Samples were taken using a sterilized coring device. Soil samples were taken at 1-10 cm, 30-35 cm, and 70-75 cm depths in the soil column. Soil samples were collected in sterile glass jars, stored on ice, and analyzed within 24 hours. The same bacterial parameters described for the manure and water are examined in the soil samples. Again, samples were analyzed at the OME-SWR. The multiple tube fermentation technique was used to determine the bacterial concentrations in the soil (APHA 1985). Results of the examination of replicate tubes and dilutions were reported in terms of the Most Probable Number (MPN).

**RESULTS AND DISCUSSION**

Table 1 gives a brief summary of the manure application events including rate of manure application, field conditions and changes in bacteria and ammonia concentrations in the water.

Eight of the 12 manure spreading events resulted in water quality degradation within 20 minutes to 6 hours following the initiation of manure application. Tile flow increased at the Event 9 and 10 sites in response to the liquid manure application. For two of the spreading events (Event 2 and 4) that did not result in water quality degradation, there was no tile flow at the time of manure application. For one other spreading event (Event 7) when there was no significant contamination, the ground had been tilled just prior to manure application. This may have impeded movement of the manure components by shearing the macro pores at the soil surface. For the final spreading event (Event 6) for which there was no significant contamination, the soil structure was platy. This perhaps impeded the downward movement of the manure components. A 15 mm rainfall the day following manure application, however, resulted in further tile drain contamination by the manure components.

For each spreading event, soil bacterial levels including the biotracers were measured at three depths in the soil before and after manure application. Previous research on bacterial movement through soil columns, including studies by Gerba et al. (1975), Smith et al. (1972), and McCoy (1969), indicated that soil was an efficient filter for bacteria. Figure 1 shows the concentration of fecal coliform at various depths in the soil, before and after spreading, expressed as # of bacteria per 100 g of soil. This profile is typical of the other spreading events monitored, with the exception of the event where the soil surface had been tilled just prior to manure application. For that particular event fewer bacteria were detected at the lower depths in the soil column.

**CONCLUSIONS**

This study has shown that when farm liquid waste was applied to land the waste can rapidly penetrate the soil and contaminate the field tile water. The presence of macropores in the soil allowed for rapid manure infiltration and penetration to depth in the soil. Flow in the tile at the time of the manure application was required for contamination to occur. An acceptable rate of application was difficult to determine.

**1991 RESEARCH**

**Methodology**

The manure application trials were conducted on a four hectare field of imperfectly drained Perth clay loam in Stephen Township, Huron County. This systematically tilled field, with 15 metre tile spacings, had been in a corn/bean/wheat rotation under conventional tillage for the past 10 years. Three observation chambers, which facilitate taking water samples and discharge measurements, were installed. Four sampling wells varying in depth from 2.2 to 4.1 metres, were also installed in order to monitor groundwater quality.
The water sampling and soil sampling procedure and analysis was similar to the 1989-1990 research except that in 1991 shallow groundwater was being sampled and analyzed also. Weekly grab samples of water from the wells were taken for one month prior to the scheduled manure application. Daily samples were taken for approximately one week immediately following manure application. A sterilized copper tube is lowered down into the well in order to obtain the samples. The water depth in the wells was measured using a water sensor at the time of sampling. From May 23 to July 15, 1991 samples for bacterial analysis were taken. Upon review of the data, samples for chemical analysis began on July 18, 1991. The chemical parameters analyzed included dissolved organic carbon, ammonia, nitrate, nitrite, dissolved phosphorus, chloride, pH, and conductivity.

Results and Discussion

Table 2 gives a brief summary of the manure application events including rate of application, method of application, bacterial and chemical loading and comments on changes in water quality.

Figure 2 shows the soil EC(NA) concentrations after the manure application for the injected and the irrigated trials for Event 3. The concentration decreased from the surface to the 70 - 75 cm depth for both methods of application. The irrigated trial has a considerably lower concentration at the 70 -75 cm depth compared to the injected trial. Manure was injected below the soil layer with the higher organic matter content and in an area of least disturbed macropores thus there was less opportunity for filtering to occur.

Figure 3 shows the soil EC(NA) concentrations after the manure application for the chisel plowed trial and the untilled trial for Event 4. The surface concentrations were similar for both trials. A much greater decrease was seen at the lower depths on the chisel plowed trial compared to the untilled trial. It appeared that tilling the soil surface prior to manure application disrupted the macropore system present, slowing the infiltration rate and allowing for more filtering of the manure components to take place.

In late November when tile drains resumed flow, water samples showed an increase in nitrate concentrations compared to earlier levels recorded (10 mg/L). Levels increased up to 40 mg/L. Possible sources leading to this increase were manure applications, breakdown of crop residue, and fertilizer application (W. Cook, personal communication). No increase in bacteria, phosphorus or potassium was noted in the tile drain water.

Both nitrite and chloride concentrations increased in the well water in late November. Earlier levels (<0.1 mg/L) increased up to 50 mg/L in the shallower well (3.3 m). In the deeper wells (4.1 m), nitrate concentrations increased up to 10 mg/L.

CONCLUSIONS

From this year of research preliminary recommendations to avoid or lessen field tile drain water contamination from liquid manure application are as follows:

1) apply manure when tile drain flow is absent or very low
2) till soil prior to manure application
3) apply manure using irrigation method as opposed to injection method
4) apply manure at low rates (ie. 40,000 L/ha); use split applications as opposed to one large application
5) if inject manure avoid injecting directly over tile line
REFERENCES


<table>
<thead>
<tr>
<th>Event #</th>
<th>Date</th>
<th>Soil Type</th>
<th>Crop Rotation</th>
<th>Ground Cover and % Residue†</th>
<th>Tile Flow</th>
<th>Rate of Appl'n L/ha</th>
<th>Bacterial Increase and Ammonia Increase, Respectively</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>May '89</td>
<td>Perth cl</td>
<td>corn/bean/wheat fall plowed, 6%</td>
<td>Yes</td>
<td>76,500</td>
<td>- 3400 fold within 2 hours - 1800 fold</td>
<td></td>
</tr>
<tr>
<td>2</td>
<td>Aug '89</td>
<td>Perth cl</td>
<td>corn/bean/wheat wheat res., 95%</td>
<td>No</td>
<td>140,700</td>
<td>- small when flow resumed - 1600 fold within 4 hours - 1000 fold</td>
<td></td>
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<tr>
<td>3</td>
<td>Sept '89</td>
<td>Perth cl</td>
<td>corn/barley/hay corn silage res., 57%Yes</td>
<td>93,000</td>
<td>- 1600 fold within 2 hours - 1000 fold</td>
<td></td>
<td></td>
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<td>4</td>
<td>Oct'89</td>
<td>Brookston cl</td>
<td>soybean/corn soybean res., 76%</td>
<td>No</td>
<td>56,000</td>
<td>- small when flow resumed - 1000 fold within 6 hours - 18,000 fold</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>Nov '89</td>
<td>Wauseon sl</td>
<td>soybean/corn soybean res., 78%</td>
<td>Yes</td>
<td>54,000</td>
<td>- 18,000 fold</td>
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<tr>
<td>6</td>
<td>May '90</td>
<td>Listowel sil</td>
<td>soybean/corn fall plowed, 6%</td>
<td>Yes</td>
<td>36,000</td>
<td>- 200 fold within 5 hours - 67 fold</td>
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<tr>
<td>7</td>
<td>Jun'90</td>
<td>Perth cl</td>
<td>corn/bean/wheat seed bed, 1%</td>
<td>Yes, low</td>
<td>111,000</td>
<td>- small within 3 hours. - 30 fold within 20 minutes - 15 fold</td>
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<tr>
<td>8</td>
<td>Jul'90</td>
<td>Perth cl</td>
<td>hay/corn/grain hay, 93%</td>
<td>Yes, high</td>
<td>100,000</td>
<td>- 45,000 fold within 2.5 hours - 5140 fold</td>
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<tr>
<td>9</td>
<td>Aug '90</td>
<td>Perth cl</td>
<td>corn/bean/wheat wheat res., 94%</td>
<td>Yes, low</td>
<td>106,000</td>
<td>- 30,000 fold within 2.5 hours - 9080 fold</td>
<td></td>
</tr>
<tr>
<td>10</td>
<td>Sept '90</td>
<td>Listowel sil</td>
<td>corn/bean/wheat wheat res., 94%</td>
<td>Yes, low</td>
<td>44,300</td>
<td>- 725,000 fold within 2 hours - 10,500 fold</td>
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<tr>
<td>11</td>
<td>Sept '90</td>
<td>Huron cl</td>
<td>corn/bean/wheat wheat res, 96%</td>
<td>Yes, low</td>
<td>72,000</td>
<td>- 10,500 fold</td>
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<td>12</td>
<td>Nov '90</td>
<td>Perth cl</td>
<td>corn/bean/wheat fall plowed, 3%</td>
<td>Yes</td>
<td>159,000</td>
<td>- 290 fold within 2 hours</td>
<td></td>
</tr>
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</table>

† PERCENT RESIDUE COVER - A 100 foot rope with markings every foot is laid diagonally to the direction of cultivation. The percent residue is determined by counting the number of times that residue is present at the foot markings. The minimum residue size is 1½" by ¾".
Table 2. Summary of Manure Application Events for 1991

<table>
<thead>
<tr>
<th>Event Date</th>
<th>Trial</th>
<th>Tile Flow Rate</th>
<th>Application Method</th>
<th>Application Rate L/ha</th>
<th>% Surface Residue</th>
<th>Surface Loading bacti/ha/day</th>
<th>TKN g/ha/day</th>
<th>EC(NA) TKN</th>
<th>TP g/ha/day</th>
<th>K g/ha/day</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 May'91</td>
<td>0.0032 irrigation</td>
<td>101,000</td>
<td>7</td>
<td>cultivated</td>
<td>$2.4 \times 10^8$</td>
<td>4.9</td>
<td>0.5</td>
<td>1.9</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>2 May'91</td>
<td>0.0032 irrigation</td>
<td>101,000</td>
<td>10</td>
<td>fall plowed</td>
<td>$6.7 \times 10^9$</td>
<td>67</td>
<td>12.0</td>
<td>39.5</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 June'91</td>
<td>0.0026 injection</td>
<td>81,700</td>
<td>20</td>
<td>scuffled</td>
<td>$9.1 \times 10^9$</td>
<td>119</td>
<td>12.0</td>
<td>130.0</td>
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<tr>
<td>2 June'91</td>
<td>0.0015 irrigation</td>
<td>121,000</td>
<td>24</td>
<td>scuffled</td>
<td>$9.9 \times 10^7$</td>
<td>57</td>
<td>4.9</td>
<td>48.0</td>
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<tr>
<td>1 Oct'91</td>
<td>injection</td>
<td>82,150</td>
<td>30</td>
<td>corn stubble</td>
<td>----</td>
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<td>2 Oct'91</td>
<td>irrigation</td>
<td>93,400</td>
<td>26</td>
<td>corn stubble</td>
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<td>1 Nov'91</td>
<td>irrigation</td>
<td>67,700</td>
<td>10</td>
<td>chisel plow</td>
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<td>2 Nov'91</td>
<td>irrigation</td>
<td>67,700</td>
<td>25</td>
<td>corn stubble</td>
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* TKN - Total Kjeldahl Nitrogen, P - Total Phosphorus, K - Potassium

- EC(NA) contamination within 2 hr.
- TKN, P conc. increase within 1.5 hr.
- 4x increase in tile discharge
- EC(NA) contamination within t hr.
- TKN, P, K conc increase within 1 hr.
- 7x increase in tile discharge
- EC(NA) contamination within 15 min
- TKN, P, K conc increase within 1 hr.
- 20x increase in tile discharge
- EC(NA) present in well within 5 hr.
- EC(NA) contamination within 30 min
- TKN, P, K conc increase within 1 hr.
- 20x increase in tile discharge
- EC(NA) present in well within 5 hr.
- no flow before spreading, and no contamination resulted
- FS present in well within 2 days
- no flow before spreading, and no contamination resulted
- well was dry
- no flow before spreading, and no contamination resulted
- well was dry
- well was dry
- no flow before spreading, and no contamination resulted
- well was dry
- no flow before spreading, and no contamination resulted
- well was dry

-36-
Figure 1: Soil Bacteria Concentration, Event #2 — August 1989

Figure 2: Soil EC(NA) Concentrations, October 7, 1991
Figure 3: Soil EC(NA) Concentrations, November 5, 1991
ABSTRACT

Agricultural practices have the potential to significantly impact groundwater quality especially where nitrogen is broadly applied (non-point source) to permeable soils. The largest impacts are associated with row crop fertilization and the spreading of animal wastes over farm land. Although a quick-fix solution to non-point source(s) of nitrate contamination is not at hand, a number of options are available to improve existing conditions. These may include the following: utilizing surface water supplies; detailed hydrogeological investigations involving well field optimization, deeper drilling to uncontaminated aquifers, and/or groundwater exploration; groundwater treatment; and, improved land use management (nitrate reduction) including timing fertilization to coincide with crop nutrient requirements, the use of nitrification inhibitors and/or scavenger crops.

In response to elevated nitrate occurrences in their groundwater supplies, the Paris Public Utilities Commission is developing a groundwater management program that will combine hydrogeological initiatives and, ultimately, improved land use management. The goal of the hydrogeological work is to secure existing groundwater supplies. However, to protect future groundwater quality it is recognized that land management practices that reduce nitrate loading to the groundwater will also need to be implemented.

INTRODUCTION

Nitrates are one of the most frequently identified contaminants in groundwater. This problem is a result of the abundant natural and anthropogenic sources of nitrogen that, through a series of reactions, can be converted into the mobile and highly soluble nitrate species.

Nitrates are associated with point (focused) sources of pollution such as septic systems, feed lots, and manure storage tanks as well as non-point (dispersed) sources derived from agricultural fertilizers and manure spreading. Of the various nitrogen (N) species, nitrate (NO₃⁻) and nitrite (NO₂⁻) represent the greatest risk to human health. Ontario Drinking Water Objectives (ODWO) require that nitrate does not exceed 10 mg/L (as N) in drinking water supplies.

With the exception of denitrification (the microbial reduction of nitrate to nitrogen gas), nitrates are unlikely to be attenuated in the subsurface. Although there is evidence that denitrification may occur under anaerobic conditions in a shallow water table setting (<3 metres below ground surface), a source of labile organic matter appears to be a primary control (Gillham, 1988). In many hydrogeological settings, however, these conditions do not exist and the process of denitrification may not control nitrate leaching. Therefore, in agricultural areas where the excess application of nitrogen-based fertilizers occurs, the potential exists for non-point source nitrate contamination of groundwater. This appears to be the case in Paris, Ontario.
Although a quick-fix solution to the non-point source(s) of nitrate contamination is not at hand, a number of options are available to improve existing conditions. Part A of this paper provides a brief review of these alternatives; Part B is a case study of the nitrate problem in Paris, Ontario. Part B includes a summary of the hydrogeological investigation and an outline of the groundwater management strategy that has been developed by Golder Associates Ltd. and Lotowater Ltd., in conjunction with the Paris Public Utilities Commission (PUC).

PART A: SOLUTIONS/ALTERNATIVES TO NITRATE IMPACTED GROUNDWATER SUPPLIES

The solution to groundwater supplies that have been impacted by nitrates is at first glance straightforward: eliminate all excess nitrogen inputs to the groundwater. Unfortunately, implementation of programs that would guarantee reduced nitrate loadings are more complicated. For instance, questions arise to how the program should be administered (the use of regulations versus taxes or incentives) and by whom (individuals versus government). Nevertheless, the success of novel programs such as PINS (Partners in Nitrogen Studies - developed through the University of Guelph) suggests that the farming community is willing to modify fertilization practices, practices that could over time reduce nitrate leaching and improve groundwater quality.

Typically, communities that are faced with elevated nitrates in their groundwater supply can improve existing water quality problems through the implementation of one or more "alternatives". These are briefly discussed below. It should be pointed out that these alternatives are not necessarily presented in any sequential order nor are they mutually exclusive of one another. The alternative(s) best suited to a particular situation would, of course, depend on local conditions.

Alternative #1 - Utilize Another Water Supply

Where the groundwater resource has been significantly impacted by nitrates, the municipal supply could be supplemented or replaced by surface water. However, in the decision to utilize surface water, several interrelated factors must be considered.

First, the reliability of surface water to meet existing and future demands must be clearly demonstrated. In addition, proposed water abstraction rates should not have an adverse impact on baseflow (the groundwater contribution to stream flow). Baseflow is important in terms of preserving both aquatic life and assimilating effluent discharges such as treated municipal waste. Baseflow in turn influences the second determinant to utilize surface water, water quality. Although water treatment would certainly be required, the level of treatment, and therefore cost, involved would depend on natural water quality and potential contaminant threats upstream.

Third, the costs to build and operate a surface water intake and treatment facility are often much higher than for an equivalent groundwater pumping operation. For instance, the Mannheim water intake system in Kitchener, Ontario is presently designed to treat 16 million imperial gallons per day (IGPD) from the Grand River and will cost some $96 million. Assuming that a system capable of delivering 1 million gallons per day (about the average daily consumption in Paris) would be proportionally less, then the cost would be about $6 million. The annual interest on this capital would more than cover annual costs associated with groundwater pumping, monitoring, and additional groundwater exploration. Although actual costs could be somewhat different than outlined in this simplistic example, surface water is generally much more expensive than groundwater.

While surface water may be necessary at some point in the future, other groundwater alternatives should be reviewed first.

Alternative #2 - Detailed Hydrogeological Investigations
Detailed hydrogeological investigations are warranted if groundwater has been utilized in the past and if additional good quality groundwater is likely available. This work could involve well field optimization, deepening of existing wells, and/or groundwater exploration.

Well field optimization entails developing a pumping strategy that maximizes existing water quality. Through detailed pumping tests and water quality analyses, a pumping schedule may be arrived at that maintains nitrate levels below ODWO.

Deepening of existing pumping wells may be feasible if an aquifer exists that has not been contaminated by nitrates. However, depending on local geologic conditions, groundwater quality may decline with depth particularly with respect to increased concentrations of the aesthetic parameters sulphate, iron, and total dissolved solids.

Groundwater exploration may be practical if the nitrate problem is limited in areal extent. Nevertheless, depending on the location of the new wells, water quality could also be impacted in the future unless nitrate loadings are reduced.

**Alternative #3 - Groundwater Treatment**

Depending on the success or viability of other alternatives, groundwater treatment of nitrates may be required at some stage. Traditionally, the treatment process has been carried out in two steps. The first step involves the denitrification of nitrate in an anaerobic reactor; the second step removes the suspended solids and biodegradable materials leaving the reactor.

In the Netherlands, recent improvements in groundwater treatment technology has eliminated the secondary treatment process (Van der Hoek and Klapwyk, 1987). Despite these improvements, groundwater treatment remains costly.

**Alternative #4 - Improving Land Use Management**

While alternatives #1 - #3 may provide a short term solution to elevated nitrate levels in groundwater supplies, in the long term, water quality could further deteriorate unless existing land use practices are improved. These improvements could include the following:

- reducing the amount of fertilizer applied annually;
- timing fertilizer applications to coincide to when nutrient requirements are greatest;
- using nitrification inhibitors to reduce the rate of conversion of ammonium to nitrate; and,
- growing "scavenger" cover crops to absorb residual nitrate in the soil horizon.

Normally, the amount of fertilizer applied to a crop is determined by the farmer based on previous yields and fertilizer costs. This approach, however, does not take into consideration the residual nitrogen in the soil (ie. the amount of nitrogen in the soil prior to fertilization). In an effort to determine residual nitrogen levels and appropriate application rates, the Nitrogen Soil Test was developed in 1991 under the PINS Program. Preliminary data from this program indicated that in one example, the nitrogen supplying capacity of the soil exceeded 460 kg/ha (Schleiauf et al, 1992 - In Workshop Proceedings). As the nitrogen requirements for corn are about 120-150 kg/ha, it is apparent that in this case, additional fertilization was not required. By determining "environmentally safe" application rates (ie. annual application rates equivalent to annual nutrient requirements), nitrate leaching below the root zone and impacts on groundwater quality can be significantly reduced.

The improper timing of fertilization can have a similarly adverse impact on groundwater quality. For example, if fertilization does not coincide to when nutrient requirements are highest, such as in the early spring, the potential exist for any excess applied nitrogen to be leached to the groundwater.
Similarly in the fall when nutrient requirements are lower, excess fertilization can also result in nitrogen leaching. By synchronizing fertilizer applications with maximum nutrient requirements, nitrogen leaching can be reduced and the impacts on groundwater quality diminished.

Nitrification inhibitors reduce the rate of nitrogen conversion from the ammonium (NH$_4^+$) form to nitrate (NO$_3^-$). By maintaining nitrogen in the ammonium form, it is retained by the cation exchange complex of the soil (Peterson and Frye, 1989). These compounds therefore reduce nitrate leaching except where the cation exchange capacity of the soil is very low. Studies have indicated that, under certain conditions, the use of nitrification inhibitors may actually increase yields in crops such as corn (Peterson and Frye, 1989).

Scavenger cover crops offer the potential to reduce nitrate loadings to groundwater by absorbing residual nitrate in the root zone. Both alfalfa and rye appear to be particularly beneficial in this regard. In addition, cover crops may help reduce soil erosion.

Some of the alternatives outlined above have been incorporated into a nitrate management strategy at Paris, Ontario. This is discussed in the following section.

PART B: CASE STUDY - PARIS, ONTARIO

The Town of Paris, Ontario is currently experiencing above background nitrate levels in groundwater at the Telfer and Gilbert municipal well fields (Figure 1). To provide an assessment of the extent of the water quality problems, Golder Associates Ltd. (Golder) was retained by the Paris Public Utilities Commission (Paris PUC) to undertake a hydrogeological investigation at and adjacent to these well fields. The scope of work consisted of:

- a water well inventory and sampling of private wells;
- a review of background data to determine the distribution (both vertical and horizontal) of the aquifer and aquitard units along with spatial and temporal water quality trends; and, based on these results,
- the development of a groundwater management strategy.

Regional Setting

Study Area and Physiography

The Town of Paris is located in the County of Brant at the confluence of the Grand and Nith Rivers (Figure 1). As shown in Figure 1 the study area is approximately 30 square kilometres. It extends to about 4 kilometres north of the Town of Paris and is bounded by Ayr Road in the west and the Grand River in the east.

Over much of the area farming has traditionally been the primary land use activity. In the past, many farming operations included livestock; within the last ten to fifteen years, corn production has grown while the number of dairy and pig farms have reduced dramatically.

Total relief in the study area is about 70 metres with the elevation decreasing from around 300 metres (above sea level) in the north, to about 230 metres in the vicinity of the Grand River. The landscape is gently sloping to rolling with local relief generally less than 10 metres.

The most prominent physiographic features include the Paris and Tillsonburg Moraines. The Paris Moraine lies south of the Town of Paris, beyond the study area. The Tillsonburg Moraine is an extensive ridge southwest of Paris, but is more subdued in the study area where it forms the broad upland district west of Highway 24A. East of Highway 24A, glaciofluvial deposits form the surficial material over a broad tract of land east of Charlie Creek to the Grand River. The Tillsonburg Moraine and, to a greater extent, the outwash deposits east of the moraine, are important features not only for aggregate
potential, but because much of the infiltration to the groundwater flow system likely occurs there.

The limited number (2) of streams and creeks in the study area suggest that runoff is minimal and infiltration is high. Given the coarse texture of the surficial sediments, infiltration is estimated to be in the range of 0.15 metres - 0.25 metres/year.

**Water Resources**

Water supplies for the Town of Paris and surrounding farmsteads are derived from two sources. The first is from the shallow water table aquifer. This is used mostly for domestic purposes; however, it also forms the municipal supply at the Gilbert collector system (Figure 1). The second source is from the deep aquifer system. This includes the deep overburden and bedrock aquifers which are utilized to a small extent for domestic purposes, but mostly as the municipal supply (Telfer well field).

The Paris PUC currently operates the Telfer and Gilbert well fields (Figure 1). The Telfer well field includes five pumping wells (P31, P32, P33, P34, P35), although only some of these are presently in operation. Wells P32 and P35 are completed in the bedrock aquifer whereas wells P31, P33, and P34 are completed in the overburden aquifer directly overlying bedrock.

Some constraints in water quality exist, particularly with regard to elevated levels of nitrate and sulphate. Nitrate is generally in the range of 5 mg/L to 10 mg/L (expressed as N) or over 50% of the ODWO of 10 mg/L (as N). Sulphate is widely variable and in the range of 50 mg/L to 700 mg/L. The ODWO is 500 mg/L. Water quality is discussed in greater detail in the next section. No constraints exist with respect to water supply.

The Gilbert well field includes a shallow collector system consisting of well points and tiles installed at depths of less than about 8 metres below ground surface. It also includes two high capacity wells (P28 and P29) constructed in bedrock.

In terms of water quality, few constraints exist at the Gilbert well field although in the collector system, nitrate is at about 5 mg/L. From the standpoint of water supply, the Gilbert collector system is dependent on regular precipitation and its capacity is therefore diminished over extended dry periods. Testing of municipal wells P28 and P29 suggests large groundwater abstractions are possible from the bedrock aquifer (Lotowater, 1991). However, the long term capacity of these wells and potential groundwater level interference has yet to be determined.

**Water Balance**

A water balance is one means of determining the volume of water entering a groundwater flow system and it is often used to provide a rough estimate of the recharge area to a pumping well or well field. A water balance was made to estimate the recharge area to the Gilbert and Telfer well fields.

A water balance performed by Hydrology Consultants (1975) for the Gilbert collector system at normal pumping rates (between 600 IGPM and 700 IGPM) estimated a recharge area of about 5 square kilometres assuming an infiltration rate of 0.30 metres/year. This value represents the upper range of infiltration measured in southern Ontario but is consistent with the coarse texture of the outwash deposits and the limited surface drainage (and runoff) in the area. If the actual infiltration rate is less than above, a greater recharge area would be required to sustain groundwater withdrawals.

Recharge to the deep aquifer system is also accomplished through surface infiltration. As the Telfer well field consists of similar coarse materials that extend to bedrock at most locations, an infiltration rate in the range of 0.15 metres/year to 0.20 metres/year could be reasonably expected. Therefore, pumping from the Telfer well field at normal rates (about 250 IGPM) would require a recharge area of about 3 square kilometres. This calculation does not take into consideration water derived from bedrock.
Hence, the recharge area for the Telfer well field is likely somewhat less than estimated above for the given pumping rate. At higher pumping rates (ie. >250 IGPM) this area would be larger.

Similar calculations for the deep wells (P28 and P29) at the Gilbert well field suggest that a recharge area of about 12 square kilometres could contribute to groundwater flow. This is comparable with pumping tests at the Gilbert well field that show the limit of drawdown to be over 2 kilometres (Lotowater, 1991). The recharge area is larger than at the Telfer well field as a pumping rate of 650 IGPM was used and because the presence of fine grained materials overlying the aquifer would reduce the infiltration rate to the deep aquifer system.

The water balance calculations demonstrate that recharge to the Telfer and Gilbert well fields includes an area up to several kilometres or more to the north and, therefore, water quality is influenced to a large extent by the agricultural practises noted earlier.

**Nitrate Sources and Estimate of Loading Rates**

There are several nitrate sources that have the potential to impact groundwater quality in the Paris area. These include point sources such as septic systems, feedlots, manure storage piles, and non-point sources such as from row crop fertilization over broad tracts of land.

Agricultural land use practices dominate north of the Telfer and Gilbert well fields. Within the study area, the density of individual farm houses to arable land has been estimated from historical airphotos and 1:10,000 base maps to be less than one farm house per 1.3 square kilometres. While point sources of nitrate exist from septic systems and to a lesser extent from feedlots and manure storage piles, the predominant nitrate source in the study area is derived from nitrogen-based fertilizer.

It is possible to estimate the nitrate loading rate in areas where crop fertilization is practised. Generally, one-quarter of the nitrogen applied as fertilizer moves as nitrate to the groundwater (Keeney, 1982). Therefore, for a typical nitrogen fertilizer application of 120 kg/ha/year, approximately 30 kg/ha/year would be leached to groundwater. If the infiltration rate is 0.15 metre/year, the nitrate concentration of the infiltrating water would be 20 mg/L.

As the infiltration moves downward through the vadose zone (the unsaturated zone above the water table) and reaches the water table, some dilution will occur. This would reduce the resulting nitrate concentration in the groundwater. However, if nitrate loading continues at the same rate as above, over time, nitrate in the groundwater could approach values in the range calculated above.

**Existing Conditions**

**Hydrogeology**

The geologic materials encountered in the study area consist of an alternating sequence of coarse grained glaciofluvial deposits and fine grained silty clay till units. Whereas the coarse grained materials were deposited in moving water and are generally quite well sorted, the till units were deposited by moving ice and are consequently less well sorted and more variable in texture in the horizontal and vertical directions. These properties are important because they influence groundwater flow patterns.

To simplify the discussion of groundwater flow in the following section, this complex arrangement of geologic materials have been grouped into hydrostratigraphic units based on hydraulic characteristics. The coarse grained, permeable sediments form aquifers and the fine grained, impermeable sediments form aquitards. The hydrogeology can be characterized as a multi-aquifer system. In general, the following hydrostratigraphic units are identified.

- an Upper Aquifer, consisting of the shallow sands and gravels in the overburden,
• an Upper Aquitard, consisting of clay to silty clay till,

• a Lower Aquifer, consisting of the deep sands and gravels in the overburden and bedrock, and

• a Lower Aquitard, consisting of clay to silty clay till; locally this unit separates the deep sands and gravels from bedrock.

These units are illustrated in hydrogeologic cross sections A-A' and B-B' (Figure 2a and 2b) and are described below.

**Upper Aquifer**

The Upper Aquifer is extensive over much of the area and is comprised of outwash sand and gravel deposits. These deposits attain a thickness of up to 20 metres with the greatest accumulations occurring north and east of the Gilbert well field within a former meltwater channel of the Grand River.

This aquifer supplies the Gilbert collector system and numerous private wells. It is also the source of baseflow to Gilbert and Charlie Creek.

North and west of the Gilbert well field, a discontinuous clay unit, generally less than 5 metres thick, is associated with the shallow overburden aquifer (Figure 2a and 2b).

**Upper Aquitard**

The Upper Aquitard consists of clay and silty clay till and separates the Shallow and Deep Aquifers (Figure 2a and 2b). It is relatively continuous across the study area; however, it thins markedly from over 10 metres at the Gilbert well field to several metres or less both east and west of the well field (Figure 2). Excluding P35, this unit is not encountered at the Telfer well field but appears to thicken north of the Gilbert well field (Figure 2a and 2b).

The Upper Aquitard is particularly important in terms of minimizing water quality impacts arising from land use activities. Although the texture is variable, its low permeability restricts the downward migration of groundwater. This unit will reduce, but not eliminate, the rate (flux) shallow groundwater moves into the deeper flow system. It is significant that in the vicinity of P31, P32, P33 and P34, the entire overburden sequence consists of coarse grained deposits. Therefore, little natural protection from nitrate leaching in the shallow subsurface is afforded the groundwater supplies.

Where the Upper Aquitard is thin or absent, surface infiltration will reach the Lower Aquifer in a relatively short period of time. For example, conservatively estimating an infiltration rate of 0.15 metres/year, and an effective porosity of 0.15 in the vadose zone, a downward velocity of 1 metre/year is determined. At P31 where the Upper Aquitard is absent and the water table is at a depth of about 20 metres, infiltration could reach the water table and impact water quality in 20 years.

**Lower Aquifer**

The Lower Aquifer is continuous along both hydrogeologic cross sections (Figure 2a and 2b). It consists of the deep outwash sands and gravels in the overburden and bedrock. This grouping is made since in most locations the hydraulic response of the deep sand and gravel aquifer cannot be distinguished from the bedrock aquifer.

As shown in Figure 2a and 2b, the Telfer and possibly Gilbert well fields appear to be located in bedrock depressions or valleys as first proposed by Mayhew (1964). The importance of this feature is that it may, to a certain extent, modify groundwater flow and influence water quality.

The deep sands and gravels attain a thickness of 5 metres to 10 metres or more with the greatest thickness encountered in the vicinity of the Telfer well field (Figure 2a and 2b). The transmissivity of the Lower Aquifer is reportedly in the range of 350 m²/day to 550 m²/day with a storativity in the range of 1.0 x 10⁻⁵ to 5.0 x 10⁻⁴.
(Hydrology Consultants, 1975; Lotowater, 1991). In some locations, such as at the Telfer well field, the deep sands and gravels directly overlie bedrock; at the other locations, such as at the Gilbert well field and further north, it is separated from the bedrock by the Lower Aquitard.

**Lower Aquitard**

The Lower Aquitard consists of clay to silty clay till and is relatively continuous in the northern half of the study area but is discontinuous in the southern portion (Figure 2a and 2b).

As discussed for the Upper Aquitard, where present, this unit also provides a level of natural protection to the deep groundwater supplies. However, between the Gilbert and Telfer well fields the unit is thin or absent (Figure 2a and 2b). North of the Gilbert well field, the Lower Aquitard appears to thicken and may attain a thickness of over 10 metres (Figure 2a and 2b). Therefore, in these areas, the unit may limit nitrate leaching downward from the shallow subsurface.

**Groundwater Flow**

Groundwater flow can generally be described in terms of a shallow and a deep groundwater flow system. Each of these are discussed below.

**Shallow Groundwater Flow System**

The water table within the shallow overburden is encountered at depths ranging from several metres at the Gilbert well field to over 15 metres at the Telfer well field. The relatively deep water table across the study area is attributed to the high permeability of the surficial sediments and elevated topography.

The water table typically follows topography and lateral groundwater flow is towards the south and southeast in the direction of the Grand and Nith Rivers (Figure 3). Recharge to the water table likely occurs over much of the study area but particularly where coarse grained outwash deposits form the surficial material. Local groundwater discharge areas occur where streams such as Gilbert and Charlie Creek intersect the water table; these act as a drain to shallow groundwater flow (Figure 3).

As the elevation of the water table is generally greater than the hydraulic head in the deep groundwater flow system, there is also a downward component of flow from the water table to the deeper flow system (Figure 2a and 2b). In these locations, vertical hydraulic gradients are in the range of 0.1 to 0.4.

**Deep Groundwater Flow System**

The deep groundwater flow system includes the deep overburden and bedrock aquifers. Recharge occurs as downward leakage from the shallow groundwater flow system and as groundwater inflow from north of the study area. Lateral groundwater flow is generally in a south to southeasterly direction towards the Grand and Nith Rivers (Figure 4). The regional hydraulic gradient is in the range of 0.01.

The hydraulic head in the deep groundwater flow system is usually between 2 metres to 10 metres below the water table elevation (Figure 2a and 2b). Therefore, the potential exists for groundwater to move downward from the shallow flow system to the deep flow system. This condition is noteworthy because it signifies that if water quality is impacted in the shallow flow system, over time, it will likely effect water quality in the deep flow system.

In contrast to the above condition, the local hydrogeology appears to be more complicated in the vicinity of the Gilbert well field and upward hydraulic gradients exist in the deep overburden aquifer (Figure 2a and 2b). The static groundwater level elevation is more than 1 metre above grade and is over 7 metres greater than the hydraulic head in the bedrock aquifer.
Thus, under existing conditions, near the Gilbert well field the potential does not exist for shallow groundwater to impact water quality in the deep flow system.

Nevertheless, this condition is local, and less than 0.7 kilometres west of the Gilbert well field, hydraulic gradients are downward through the entire multi-aquifer sequence, however, the hydraulic head in the deep overburden aquifer at TW6/89 is still greater than in the bedrock aquifer (Figure 2a and 2b). This vertical gradient is likely attributable to the upper bedrock surface behaving as an underdrain. This is consistent with the very high permeabilities reported during drilling (T. Lotimer, personal communication).

It is possible that long term pumping from P28 and P29 at the Gilbert well field could depressurize the deep overburden aquifer and reverse the hydraulic gradient. In this case, the potential would exist for shallow groundwater to impact deep groundwater supplies at the Gilbert well field. It is not known, however, if pumping from these wells would be sufficient to cause a gradient reversal.

### Water Quality

Groundwater quality 2 to 3 kilometres upgradient (north and northwest) of the Telfer and Gilbert well fields was assessed from the compilation of existing data and from the sampling of fourteen private wells.

In general, the results indicated that water quality in the shallow groundwater flow system is characterized as calcium bicarbonate type with nitrate in the range of 1 to 10 mg/L and sulphate in the range of 30 to 120 mg/L. It is of interest to note that with the exception of one private well, nitrate impacts were not detected in the remaining private wells. However, these results are not considered to necessarily be representative of the shallow groundwater flow system: water quality in shallow domestic wells is often influenced by local factors such as pumping rates and the condition of the seal or the casing around the well. Therefore, it is probable that additional monitoring wells, designed specifically for representative water quality sampling, would be required to adequately characterize the extent of nitrate impacts in the shallow groundwater flow system.

Water quality in the deep groundwater flow system is very hard and also characterized as calcium bicarbonate type. Nitrate is less than 1 mg/L and sulphate is variable between 30 to 1560 mg/L.

### Telfer Well Field

Water quality sampling began at the Telfer well field in 1965, when pumping wells P31, P33 and P34 were constructed. However, between 1965 and 1987, water quality was analyzed infrequently and over this period, the database is incomplete. Since 1987, water quality has been monitored more regularly. All water quality data is from wells completed in the deep groundwater flow system; shallow water table observation wells do not exist at the Telfer well field.

The variations in water quality (nitrate, sulphate, and chloride) at the Telfer well field between 1987 and 1991 in P31 and P32 are presented in Figure 5. Several important trends are apparent. First, as noted above, nitrate has increased from between 1 mg/L - 5 mg/L in both pumping wells since 1987 and is presently in the range of 5 mg/L - 12 mg/L. Second, sulphate, and to a lesser extent chloride, increased between 1987 and 1990 in P31 and P32; then decreased markedly. This trend appears to be inversely related to nitrate; that is, as nitrate increases, sulphate decreases and vice versa. Third, trends in chloride closely approximate that of sulphate and it is suggested that chloride is derived from the Salina Formation.

From these observations, it is probable that the water quality trends in P31, P32 bear a strong relationship with pumping rates. As pumping increases, hydraulic gradients steepen and the potential exists for additional shallow groundwater to be drawn towards the pumping well. As previously noted, the shallow groundwater is characterized as being high in nitrates and low in sulphates. Therefore, at higher pumping rates, the
"blend" of water pumped from the municipal wells could be lower in sulphates although higher in nitrates.

Insufficient data is available to determine the length of time involved between variations in pumping rates and the resultant water quality. Similarly, it is not possible to predict future water quality.

However, in the short term, by pumping regularly at lower rates, it may be possible to improve or, at a minimum, maintain existing water quality by limiting the component of shallow groundwater pumped from P 31, P32 and P34. To better establish preferred groundwater pumping rates, additional observation wells are required to monitor groundwater quality.

Unless nitrate loadings are reduced, in the long term water quality will degrade as shallow groundwater, high in nitrates, infiltrates the deep groundwater flow system.

**Gilbert Well Field**

Although the Gilbert well field has been in operation for over 50 years, historical data are limited with respect to nitrate, sulphate, and chloride concentrations.

The existing data indicates the shallow groundwater flow system supplying the Gilbert spring collectors has been impacted by nitrates. Nitrate has generally remained in the range of 4 mg/L to 7 mg/L. Sulphate is usually below 50 mg/L. Chloride is in the range of 30 mg/L to 40 mg/L. A gradual increase in chloride is observed; however, the reason(s) for this are not known. It is likely that chemical de-icing compounds are contributing to the elevated levels of chloride although this has not been substantiated.

Less data are available for the deep groundwater flow system. Both wells P28 and P29 were completed in bedrock in 1990, but to date have not been used pending receipt of the Certificate of Approval and the Permit to take Water. Nitrate is below 0.5 mg/L and sulphate is less than 80 mg/L. Chloride is below 20 mg/L.

**Nitrate Management Strategy**

The nitrate management strategy developed for the Town of Paris includes both short and long term objectives. The short term objective is to secure existing groundwater supplies. The long term objective is to adopt land management practices to control nitrate leaching to the groundwater flow system and thereby protect, and possibly improve, groundwater quality. These are each described below.

**Short Term**

The delivery of an adequate supply of potable water is one of the primary functions of the Paris PUC. Therefore, it is essential that all available water supplies at the Telfer and Gilbert well fields be utilized to the fullest extent possible. This shall be accomplished through hydrogeological initiatives involving aquifer testing at the Telfer well field, the installation of multi-level observations wells, and the formation of an integrated groundwater monitoring program.

Aquifer testing at the Telfer well field shall be carried out to develop a pumping schedule that maintains nitrate levels below Provincial Objectives. This program of well field optimization shall involve groundwater quality monitoring at various pumping rates and durations. Based on these results, an optimal well field operations plan shall be formulated.

The installation of multi-level observation wells upgradient of the Gilbert and Telfer well fields shall provide a better assessment of the vertical distribution of nitrate throughout the multi-aquifer sequence. As well, the geologic data will provide a better indication of the degree of natural protection afforded to the deep groundwater flow system and shall identify where it is most susceptible to nitrate leaching.
The development of an integrated monitoring program shall provide for the regular determination of water quality and water levels in the aquifer system. By establishing a monitoring network that includes a number of observation wells throughout the study area, a more thorough evaluation of the extent of the nitrate problem is possible. In the future, these installations will provide a means of assessing the effectiveness of alternative land management practises.

**Long Term**

To substantially improve existing groundwater quality and to protect future supplies, it is recognized that land management practises that reduce nitrate loadings to the groundwater will need to be implemented. It is expected that the groundwater nitrate management strategy being developed by the Paris PUC shall include land management practises discussed in Part A. This may include: reducing the amount of fertilizer applied; improving the timing of application; the use of nitrification inhibitors and; the use of scavenger cover crops.

The success that these measures will have on improving water quality shall depend largely on the extent to which they are implemented. To this end, it is important that the nature and extent of the problem and the options available to the farming community be clearly articulated. If programs such as PINS, albeit designed to determine optimal nitrogen application rates, are any indication, their appears to be strong support from farmers. However, as pointed out by Follett and Walker (1989), implementation is beyond the scope of private decision-making and some form of government involvement is required.

In many agricultural settings we are only now beginning to detect nitrate impacts in groundwater supplies. Although a number of options are available to improve existing conditions, it is critical that consensus be reached between governments and the farming community towards establishing land use practices that reduce nitrate leaching to the groundwater. Many of these measures could in fact improve crop productivity through more efficient nitrogen management. However, unless these are adopted the nitrate problem in groundwater will likely worsen.

**ACKNOWLEDGEMENTS**

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**REFERENCES**


Figure 1: Study Area
Figure 2a: Hydrogeologic Cross Section A-A’
Figure 2b: Hydrogeologic Cross Section B-B'
Figure 3: Water Table Elevation
Figure 4: Regional Groundwater Elevation
Figure 5: Dissolved Nitrate, Sulphate and Chloride Concentrations
ABSTRACT

The Provincial Water quality Monitoring Network (PWQMN) was established in 1964 to monitor water quality data at strategic geographic locations throughout Ontario. The objectives of the program are: to determine water quality trends through continued surveillance; to ensure provincial water quality objectives are met for various uses; and, to provide data for specific MOE programs and studies. Currently there are about 700 active sites.

Total nitrate is one of about 20 core parameters that have been sampled as part of the PWQMN. An analysis of PWQMN data for the period 1988 to 1991, shows that median total nitrate concentrations of <0.5 mg/L predominate in surface waters of the Canadian Shield and areas of south central Ontario. Higher median total nitrate concentrations are evident in areas of intense urban and agricultural land use activity. Median total nitrate concentrations ranging from 4 to 10 mg/L are common in surface waters of south western Ontario.

SPANS Geographic Information System (GIS) was used to generate maps showing median total nitrate concentrations for the period 1988 to 1991 and for the period 1973 to 1976. The maps were compared showing changes in median total nitrate concentrations over a 15 year period. In-house software for robust graphical time series analysis was also used to investigate total nitrate trends. Evidence suggests that total nitrate trends have increased at many surface water quality sites in southern Ontario.

INTRODUCTION

The intent of this report is to provide a brief overview of the Ministry of Environment's (MOE) Provincial Water Quality Monitoring Network (PWQMN); to indicate total nitrate levels presently observed in surface waters of southern Ontario on the basis of the data collected as part of the PWQMN; and, to identify trends in total nitrate concentrations that have occurred over the years since the inception of the program in 1964.

The PWQMN was put in place to collect surface water quality data at strategic geographic location throughout Ontario to monitor the effects of point sources of pollution from municipal and industrial discharges and from non-point sources such as urban and agricultural land uses. The principle objectives of the PWQMN are to determine water quality trends through continued surveillance; to ensure that Provincial Water Quality Objectives are met for various uses; and, to provide data for specific MOE programs and studies.

Figure 1 shows the locations of PWQMN sites that have been sampled since 1964. Approximately 1800 sites have been monitored since the beginning of the program. Presently there are about 700 active PWQMN sites. Figure 1 shows that the majority of the sites in the PWQMN are distributed throughout southern Ontario. This is in keeping with the higher population density and the greater intensity of urban and agricultural land use activity that is common to southern Ontario, thus leading to a higher potential for surface water quality problems.

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Conservation authorities in southern Ontario play a major role in the PWQMN field program. They provide the transportation and staff to collect the PWQMN samples within their jurisdiction in exchange for the water quality data provided by MOE. Samples are also collected by MOE regional staff and some contract samplers.

Samples are collected once per month at most sites and are delivered or shipped to MOE laboratories in London, Toronto and Thunder Bay. The samples are analyzed for a core group of about twenty-five parameters that include physical/chemical parameters such as pH, conductivity, suspended solids and chlorides; heavy metals such as copper, lead and zinc; nutrients such as phosphorus and nitrogen; microbiology such as fecal coliform; and some toxic parameters.

**PRESENT LEVELS OF TOTAL NITRATES IN SOUTHERN ONTARIO RIVERS**

Total nitrates data collected as part of the PWQMN was used to determine the current levels of total nitrates in rivers and streams throughout Ontario and to investigate trends. The data base for this analysis consists of observations for total nitrate directly, or observations for the nitrite component and the nitrate component which were summed to provide a measure of total nitrates.

To provide a snapshot of current levels of total nitrates the data for the four year period from 1988 to 1991 inclusive, was extracted from the data base. Summary statistics including minimum, maximum, median, first and third quartiles, and ninth decile, were computed for each site. This four year window of data was considered to be suitable for providing sufficient observations upon which to calculate summary statistics and yet short enough to reduce any bias in the statistics that may occur as a result of trends in the data.

The median or middle value for the total nitrate concentration observed at each site was plotted at the geographic location of the PWQMN station using the SPANS Geographic Information System (GIS). The contouring module within the SPANS GIS was then used to translate the points of median total nitrate concentration into a classified map showing areas that have the same median total nitrate concentration. The result of this exercise for the period 1988 to 1991 is shown in Figure 2.

Figure 2 shows areas of central and south central Ontario which are predominated by the land features of the Canadian Shield, have surface waters with median total nitrate concentrations of less than 0.5mg/L. As one progresses towards the extreme tip of south eastern Ontario and towards south western Ontario, surface waters are in a transition from lower to higher concentrations of total nitrates. This transition also corresponds to the transition from a lower to higher population density as well as a lower to higher urban and agricultural land use activity. Surface waters in a large portion of south western Ontario show median total nitrate concentrations in the range from 4 to 10mg/L.

**TRENDS IN TOTAL NITRATES CONCENTRATIONS**

The same procedure of data analysis and SPANS GIS mapping was also applied to a four year block of PWQMN data for the period 1973 to 1976. The result of the mapping exercise is shown in Figure 3. In comparing Figure 2 and Figure 3, one is able to see changes in nitrate concentrations in surface waters that have occurred over a fifteen year period.

Figure 3 shows more areas of central and south central Ontario, eastern Ontario and the Bruce Peninsula area as having surface waters with median total nitrate concentrations of less than 0.5mg/L. The same transition from lower to higher concentrations of nitrates as observed in Figure 2 is also evident in Figure 3, in particular, as one progresses towards south western Ontario. However, Figure 3, which represents the 1973 to 1976 block of data, shows that south western Ontario is predominated by surface waters with median total nitrate concentrations in the range from 1.5 to 4mg/L.
rather than the 4 to 10mg/L range as observed in figure 2. Only a small area in south western Ontario is shown as having surface waters with median total nitrate concentrations in the range from 4 to 10mg/L as evident in figure 3.

The comparison of Figures 2 and 3 suggests that total nitrate concentrations in surface waters in some areas of the province, with specific reference to south western Ontario, have increased from 1973 through to 1991.

Trend Plots

The data collected as part of the Provincial Water Quality Monitoring Network is typically a discrete time series of irregular spaced observations and is characterized by high natural variability, non-normal sample populations, extreme values or outliers and seasonal variation. Because of these characteristics, the application of conventional statistics for data analysis is somewhat limited. The Ministry of Environment however, has developed software capable of identifying trends in water quality data using other statistical methods and graphical procedures.

Figures 4 through 14 show trends of total nitrate concentrations at a select number of PWQMN site throughout Ontario. These graphs have been adjusted such that changes resulting from seasonality have been removed from the trend plot. Given the time and effort to prepare the data series and to run the trend program, only trends at some PWQMN stations representing a variety of land use features across Ontario are presented.

Figure 4 shows the trend plot at a site on the Teeswater River in the upper reaches of the Saugeen watershed which discharges to Lake Huron. The trend in land use activity in the area is generally towards increasing agricultural practice. The trend plot shows a statistically significant positive increase in total nitrates since 1975. The mean annual median total nitrate concentration is still somewhat low at 1.4mg/L.

Figure 5 shows a site on the Maitland River which also discharges to Lake Huron. The plot shows a significant positive increase in the total nitrate concentration over time with a mean annual median concentration of 2.4mg/L. This watershed is predominated by intense agricultural land use activity.

The Bayfield River which discharges to Lake Huron is a watershed also characterized by intense agricultural activity. Figure 6 show there has not been significant changes in total nitrate concentrations at this site since about 1979. The mean annual median concentration is 4.8mg/L.

Figure 7 shows the trend plot as at PWQMN site south of Innerkip on the Thames River. The Thames River discharges to Lake St. Clair and is also a predominantly agricultural watershed. The trend plot again shows a statistically significant positive increase in total nitrates since 1967. The mean annual median concentration is 3.1 mg/L.

The trend plot in Figure 8 for the site on Big Creek which discharges to Lake Erie shows a dramatic positive increase in total nitrates since 1972. The mean annual median is 1.7 mg/L.

Figure 9 shows the trend plot for the Grand River at Blair. The upstream watershed area from this site is predominated by both urban and agricultural land use activity. Several sewage treatment plants also discharge to the Grand River. The trend plot shows a significant positive increase in total nitrate concentrations over the years with a mean annual median concentration of 2.3 mg/L.

Figure 10 shows the trend plot for the site at the mouth of the Don River discharging to Lake Ontario. The river flows through the heart of metropolitan Toronto and also receives a sewage treatment plant discharge. The trend plot shows a significant positive increase in total nitrate concentration with a mean annual median of 1.9 mg/L.

The site on the Rouge River, further east of the Don River watershed, shows a statistically significant decrease in nitrate concentration as
shown in Figure 11. This down trend is likely brought on by the removal a sewage treatment plant discharge to the watercourse in 1981. The mean annual median total nitrate concentration at the site is 1.2 mg/L.

Shelter Valley Brook which discharges to the eastern end of Lake Ontario is a relatively pristine watershed. The trend plot, shown in Figure 12, however, does show a significant positive increase in the total nitrate trend likely due to increasing agricultural and urban developments in the watershed. The mean annual median at the site is 0.7 mg/L.

Figure 13 shows the trend plot for the PWQMN site at the outlet of Cameron Lake near Fenelon Falls. This site is in the Kawartha Lakes region and is popular area for tourists, cottagers and others seeking outdoor recreation. The plot indicates that there is no statistically significant change in the total nitrate levels. The mean annual median nitrate concentration is 0.06 mg/L.

The final trend plot, Figure 14, is at a site in the South Nation watershed in south eastern Ontario. This watershed is also predominated by agricultural activity. The plot shows a significant positive increase in total nitrate concentrations with a mean annual median of 0.64 mg/L.

Additional time and effort is required to investigate total nitrate trends at other PWQMN sites. This preliminary work does suggest that total nitrate concentrations in surface waters are generally increasing across southern Ontario.

## SUMMARY

In summary, the Provincial Water Quality Monitoring Network has generated a substantial data base of surface water quality data since 1964. The value of this resource is substantial and will be further highlighted as GIS technology and trend software is employed in the analysis and presentation of the data.

For example, the SPANS GIS was used to provide a map of median total nitrate levels in surface water of southern Ontario for the period 1988 to 1991. This map shows that median total nitrate concentrations of <0.5 mg/L predominate in surface waters of the Canadian Shield and areas of south central Ontario. Higher median total nitrate concentrations are evident in areas of intense urban and agricultural land use activity. Median total nitrate concentrations ranging from 4 to 10 mg/L are common in surface waters of south western Ontario.

A similar map was also prepared using data for the period 1973 to 1976. A quick and easy visual comparison of the maps shows there have been increases in median total nitrate concentrations over a fifteen year period in some areas of southern Ontario.

The development of the trend program capable of robust graphical time series analysis with specific application to PWQMN data now allows a way of clearly identifying changes or trends in water quality data over time. This program was a applied to a cross section of PWQMN sites across southern Ontario. Further work with this program is required but preliminary findings suggest that total nitrate concentrations have increased at a number of sites.

## REFERENCES


Figure 1: Location of Provincial Water Quality Monitoring Network Sites
Figure 2: Median total Nitrate Concentration in the surface water from 1973 to 1976
Figure 3: Median Total Nitrate Concentration in the Surface Water from 1988 - 1991
TRENDS IN DE-SEASONALIZED DATA SERIES
Total Nitrates (mg/L)
TEESWATER RIVER (0.39) AT CTY RD. 1

Run #2: 1 FAR OUTLIERS DELETED
Run date: MAR 17, 1992
pgm = TRX

— Annual median
— Mean annual median
= 1.406

Maximum trend 1988 = 1.878
Minimum trend 1975 = 0.648
Trend range = 1.232

Observations = 183
Series begins AUG 10 1975
Series ends NOV 18 1991
Iterations = 5
Minimum Window - \( \frac{1}{6} \) YEAR
Minimum N Per Window = 13
TREND TEST ON ANNUAL T—W MEANS:
Spearman RHO = 0.868
p(RHO) = 0.000 dol = 15
% Variation due to trend = 57.2

Figure 4

TRENDS IN DE-SEASONALIZED DATA SERIES
Total Nitrate (mg/L)
MAITLAND RIVER (004) EAST OF WROXETER

Run # 2: 5 FAR OUTLIERS DELETED
Run date = MAR 17, 1992
pgm = TRX

— Annual median
— Mean annual median
= 2.391

Maximum trend 1989 = 3.690
Minimum trend 1973 = 1.241
Trend range = 2.439

Observations = 226
Series begins JAN 10 1972
Series ends DEC 2 1991
Iterations = 5
Minimum Window = \( \frac{1}{6} \) YEAR
Minimum N Per Window = 13
TREND TEST ON ANNUAL T—W MEANS:
Spearman RHO = 0.901
p(RHO) = 0.000 dol = 18
% Variation due to trend = 48.7

Figure 5
TRENDS IN DE-SEASONALIZED DATA SERIES
Total Nitrate, (mg/L)
BAYFIELD RIVER (008) NORTH OF VARNA

Run # 2: 3 FAR OUTLIERS DELETED
Run date: MAR 17, 1992
pgm = TRX

— Annual median
--- Mean annual median
= 4.798
Maximum trend 1990 = 7.393
Minimum trend 1978 = 3.189
Trend range = 4.224
Observations = 189
Series begins AUG 26 1975
Series ends NOV 12 1991
Iterations = 5
Maximum Window = ½ YEAR
Minimum N Per Window = 13
TREND TEST ON ANNUAL T-W MEANS:
Spearman RHO = 0.245
p(RHO) = 0.343 dot = 15
% Variation due to trend = 18.2

Figure 6

TRENDS IN DE-SEASONALIZED DATA SERIES
Total Nitrates (mg/L)
THAMES RIVER (018) 1ST RD. SOUTH OF INNERKIP

Run # 2: 4 FAR OUTLIERS DELETED
Run date = MAR 18, 1992
pgm = TRX

— Annual median
--- Mean annual median
= 3.107
Maximum trend 1981 = 4.798
Minimum trend 1987 = 0.820
Trend range = 3.975
Observations = 238
Series begins APR 17 1987
Series ends FEB 17 1988
Iterations = 5
Maximum Window = ½ YEAR
Minimum N Per Window = 13
TREND TEST ON ANNUAL T-W MEANS:
Spearman RHO = 0.800
p(RHO) = 0.000 dot = 20
% Variation due to trend = 26.5

Figure 7
TRENDS IN DE-SEASONALIZED DATA SERIES
Total Nitrate, (mg/L)
BIG CREEK (003) AT NORFOLK CT. RD. 42

Run # 2: 5 FAR OUTLIERS DELETED
Run date: MAR 17, 1992
pgm = TRX

— Annual median
— Mean annual median
= 1.660

Maximum trend 1990 = 2383
Minimum trend 1972 = 0.981
Trend range = 1.40.3
Observations = 210
Series begins MAR 1 1972
Series ends AUG 13 1991
Iterations = 5
Minimum Window = % YEAR
Minimum N Per Window = 13

TREND TEST ON ANNUAL T—W MEANS:
Spearman RHO = 0.949
p(RHO) = 0.000 dot = 18
% Variation due to trend = 45.0

Figure 8

TRENDS IN DE-SEASONALIZED DATA SERIES
Total Nitrates (mg/L)
GRAND RIVER (012) AT BLAIR BRIDGE

Run # 2: 4 FAR OUTLIERS DELETED
Run date: MAR 18, 1992
pgm = TRX

— Annual median
— Mean annual median
= 2.281

Maximum trend 1981 = 3247
Minimum trend 1968 = 1.017
Trend range = 2.229
Observations = 359
Series begins JUL 19 1965
Series ends DEC 18 1991
Iterations = 5
Maximum Window = % YEAR
Minimum N Per Window = 13

TREND TEST ON ANNUAL T—W MEANS:
Spearman RHO = 0.848
p(RHO) = 0.000 dot = 25
% Variation due to trend = 43.1

Figure 9
TRENDS IN DE-SEASONALIZED DATA SERIES
Total Nitrates (mg/L)
DON RIVER (001) AT LAKESHORE ROAD

Run #2: 9 FAR OUTLIERS DELETED
Run date: MAR 17, 1992
pgm = TRX

— Annual median
— Mean annual median
= 1.894

Maximum trend 1982 = 3.502
Minimum trend 1967 = 0.487
Trend range = 3.015
Observations = 440
Series begins MAR 7 1966
Series ends OCT 16 1991
Iterations = 5
Minimum Window = ½ YEAR
Minimum N Per Window = 13
TREND TEST ON ANNUAL T—W MEANS.
Spearman RHO = 0.813
p(RHO) = 0.000 dot = 24
% Variation due to trend = 39.1

Figure 10

TRENDS IN DE-SEASONALIZED DATA SERIES
Total Nitrates (mg/L)
ROUGE RIVER (007) HWY. 2 W. OF ROUGEHILL

Run #2: 1 FAR OUTLIERS DELETED
Run date: MAR 17, 1992
pgm = TRX

— Annual median
— Mean annual median
= 1.185

Maximum trend 1979 = 2.034 Minimum trend 1988 = 0.828
Trend range = 1.205
Observations = 211
Series begins APR 6 1972
Series ends SEP 4 1991
Iterations = 5
Minimum Window = ½ YEAR
Minimum N Per Window = 13
TREND TEST ON ANNUAL T—W MEANS.
Spearman RHO = 0.448
p(RHO) = 0.048 dot = 18
% Variation due to trend = 19.6

Figure 11

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**TRENDS IN DE-SEASONALIZED DATA SERIES**
Total Nitrates (mg/L)
**SHELTER VALLEY BROOK (001) S. OF GRAFTON**

Run 32: 3 FAR OUTLIERS DELETED
Run date: MAR 17, 1992
pgm = TRX

---

Annual median
Mean annual median
= 0.669

Maximum trend 1987 = 0.885
Minimum trend 1973 = 0.448
Trend range = 0.438
Observations = 231
Series begins APR 27 1971
Series ends DEC 9 1991
Iterations = 5
Minimum Window = ½ YEAR
Minimum N Per Window = 13
TREND TEST ON ANNUAL T—W MEANS:
Spearman RHO = 0.922
p(RHO) = 0.000 dot = 19
% variation due to trend = 85.2

**Figure 12**

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**TRENDS IN DE-SEASONALIZED DATA SERIES**
Total Nitrates (mg/L)
**CAMERON LAKE OUTLET (023) HWY. 35 FENELON FALLS**

Run #2: 11 FAR OUTLIERS DELETED
Run date: MAR 17, 1992
pgm = TRX

---

Annual median
Mean annual median
= 0.062

Maximum trend 1985 = 0.094
Minimum trend 1983 = 0.050
Trend range = 0.044
Observations = 190
Series begins MAR 28 1968
Series ends DEC 7 1987
Iterations = 5
Maximum Window = ½ YEAR
Minimum N Per Window = 13
TREND TEST ON ANNUAL T—W MEANS:
Spearman RHO = 0.360
p(RHO) = 0.109 dot = 18
% variation due to trend = 12.1

**Figure 13**
TRENDS IN DE—SEASONALIZED DATA SERIES
Total Nitrates (mg/L)
SOUTH NATION RIVER (020) HWY. 17 PLANTAGENET

Run # 2: 8 FAR OUTLIERS DELETED
Run date: MAR 17, 1992
pgm = TRX

— Annual median
— Mean annual median
   = 0.635

Maximum trend 1988 = 1.785
Minimum trend 1968 = 0.200
Trend range = 1.588
Observations = 168
Series begins DEC 8 1966
Series ends DEC 3 1991
Iterations = 5
Minimum Window = ½ YEAR
Minimum N Per Window = 13
TREND TEST ON ANNUAL T—W MEANS:
Spearman RHO = 0.808
p(RHO) = 0.000 dot = 19
% Variation due to trend = 53.6

Figure 14
EVALUATION OF AN INTEGRATED SOIL, CROP AND WATER MANAGEMENT SYSTEM TO ABATE NITRATE LOSS

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ABSTRACT

The corn management practices, incorporating annual ryegrass intercrop, conservation tillage and water table management were monitored for their effect on nitrate loss. Surface runoff and subsurface drainage are channelled to a central collection building where flow volumes are recorded by datalogger and transmitted daily to a computer via modem. Samples of surface runoff and subsurface drainage are also collected with autosamplers for nitrate analyses.

The results of this study to date demonstrate the effectiveness and responsiveness of the water table control treatments in reducing the volume of subsurface runoff, the concentration of nitrate in subsurface drainage water and therefore the total nitrate loss. Although the tillage treatments were only implemented in November 1991, there already appears to be a tillage response to water movement and nitrate loss through subsurface drainage. The combined effect of tillage, intercropping and water table control systems on water quality should be apparent in the next 2-3 years. During this time we will be able to determine the average nitrate concentration in subsurface drainage water per year, total volume of water lost, total nitrate loss per year and the nitrogen balance for each of the soil, crop and water management systems.

INTRODUCTION

Soil and crop management practices have changed dramatically within the Great Lakes basin during the last decades. Livestock-forage based farming has been replaced with monoculture cash-cropping and the accompanying increased use of fertilizers, pesticides, and large machinery. However, these changes have resulted in soil compaction and structural deterioration and, therefore, increased surface runoff, erosion, and losses in productivity. Application of increased chemical inputs in response to losses in productivity, in conjunction with increased erosion and runoff has resulted in contamination of surface and ground water by nutrients and pesticides.

The development and implementation of an integrated soil, crop, and water management system shows promise as a means of managing agricultural chemicals within the rooting zone and maintaining soil structure while sustaining crop production. Current research indicates that conservation tillage alone will not reduce pesticide (Sauer and Daniel, 1987) or nutrient pollution of runoff water (McKenney and Drury, 1992) which eventually discharges into the Great Lakes. However, an intercrop would stimulate soil microbial activity and increase biomass through enhanced carbon and nitrogen supply (Drury et al. 1991). Further, the combination of N fertilization and water table control would provide favourable conditions for enhanced microbial activity. Water table management may therefore enhance nitrogen utilization by the crop and microorganisms, thereby, improving the quality of discharge water. Some studies suggest most pollutants are discharged during the first stages of runoff, thus retention of early runoff (Jury et al. 1985) events may also significantly reduce field losses of nutrients (Gillam et al., 1979). Improving soil structure and managing nutrient uptake and release through intercropping, cover crops, and enhancement of root uptake efficiency may also significantly reduce runoff for nutrients and pesticides (Smith, 1982).

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The objective of this paper is to describe the establishment of a field experiment to evaluate the effects of an integrated soil, crop and water management system on nitrate loss.

Site Characteristics

The experimental site for this study is located at Eugene F. Whelan experimental farm (Agriculture Canada, Woodslee, Ontario). The dominant soil series is Brookston clay loam, a poorly drained lacustrine soil (Typic Argiaquoll). The soil at the experimental field has a 30 cm deep dark brown, clay loam Ap horizon with approximately 2.5% organic matter. The B horizon has a clayey texture and extends to a depth of 1.5 m.

Experimental Design

The experiment was initiated in the Spring of 1991 and consists of four crop/tillage and two water table management treatments in a randomized complete block design with two replicates. The crop/tillage treatments are conventional tillage (Fall moldboard plow) with and without annual ryegrass intercrop and conservation tillage (Soil saver) with and without annual ryegrass intercrop. Water table management treatments were drainage only and controlled drainage/subirrigation. All experimental plots received the same pesticide and fertilizer application.

Field Layout and Installation

The layout of the experimental field is shown in Fig. 1. It consists of sixteen plots each 15 m wide by 67 m long with an area of 1005 m². Each plot contains two 104 mm diameter subsurface drains. Drains are installed at 7.5 m spacing and 0.6 m depth in the West-East direction. Experimental plots are isolated by: (1) Double layer 4 mil thick plastic barrier from the surface to a depth of 1.2 m to prevent leakage and subsurface interaction between the adjacent treatments; (2) A 7.5 m wide by 67 m long buffer area with a single drain to prevent cross contamination between plots; and (3) Surface ridges surrounding each plot to contain surface runoff.

Water table in the irrigated plots is controlled with water level control structures (Fig. 2). These structures are built such that, when the bottom drain valve is closed the water rises to desired level in the structure creating a pressure head which forces the water into the subsurface drains for subirrigation. When the bottom valve is opened water drains freely from the plots. The water level in these structures can be maintained at a given height by means of a float valve during irrigation. An overflow pipe permits drainage to proceed when the water table in the centre of the plot rises above the pre-set level. These structures are used for subirrigation during the growing season and controlled drainage during fall, winter and spring. The source of water is an irrigation pond located at the North-West corner of the experimental field. Irrigation water is pumped and conveyed to the water table control structures via an underground 50 mm diameter polyethylene pipe. Water meters located at the control structures record the total volume of irrigation water delivered to each plot.

Subsurface drains from each individual plot are intercepted at the lower border of each plot and rerouted to a central instrumentation building at the North-East corner of the experimental field via 104 mm corrugated non-perforated drain pipes. Each plot has a 0.5 m diameter surface catch basin at its lower boundary to collect the surface runoff. The surface catch basins are also connected to the central instrumentation building through underground non-perforated drain pipes. The 6 m by 8 m instrumentation building is equipped with a electrical circuit breaker panel, heater, fan, telephone line, data acquisition facilities and a backup generator to provide power to the system when electrical failures occur (Fig. 3).

Agronomy

Corn (Pioneer 3573) was planted at a rate of 65,000 seed/ha on June 10, 1991 with a Kinze 4 row planter. Fertilizer (8-32-16) was banded beside the seed at a rate of 132 kg/ha. Urea (46-0-0) was applied with a brush applicator.
(250 kg/ha) on July 4. The quantity of added N was reduced by 55 kg N/ha (i.e. from 170 to 115 kg N/ha) to allow for the mineralization of organic N from the decomposition of alfalfa (the previous crop). Atrazine at 1.1, metribuzin at 0.5 and metolachlor at 1.68 kg/ha was applied pre-emergence to the treatments on June 12, 1991. The herbicides were applied in a 37.5 cm band over the corn row in 370 L water/ha at 210 kPa with 8004 flat fan nozzles.

Surface and Subsurface Flow Measurements

Surface and subsurface drainage water from the sixteen experimental plots, delivered to the instrumentation building are collected in 32 polyethylene sumps (500 mm diameter by 750 mm deep; Fig. 4). Each sump is equipped with an electrical, float activated effluent pump. Surface runoff and subsurface drainage from each individual plot flowing into the respective sumps are pumped through water meters to an outlet drain. Each water meter has the capability of recording drainage volumes mechanically as well as sending analog and digital pulse signals. A multi-channel datalogger utilizes the analog signal of the water meters to monitor, measure, and store the drainage volumes on a continuous basis. The data stored in the datalogger are automatically transmitted 32 km to an IBM PC computer at the Harrow Research Station (HRS) via modem every 24 hours.

Water Quality Sampling and Frequency

Samples of surface runoff and subsurface drainage water are collected automatically with 32 autosamplers (one for each source) stationed in the instrumentation building (see Fig. 4). Each autosampler contains 24 one-litre bottles. The autosampler logic is activated by digital signals from the water meter. For our application, the autosamplers are programmed to take a 1000 ml sample for every 500 litres of surface and every 1000 litres of subsurface drainage discharged through the water meter during the growing season whereas every 2500 liters will be sampled for both surface and subsurface discharge during the rest of the year. Sampling quantity and frequency may be adjusted as required by simply altering the program on the autosamplers. At the end of each runoff event the collected samples are transferred to HRS and stored in a cold room at 4°C for subsequent nitrate analyses.

Field Measurements

Water table elevation

Water table elevations in each plot are monitored by eleven 25.4 mm dia. perforated PVC pipes, wrapped in filter material. Nine pipes are installed midway and adjacent to the subsurface drain in three rows. The other two pipes are installed at the centre of the plot midway between the drains and the plastic barrier on each side. Water table depth is monitored every second day during the growing season and on a weekly basis during the off season. The water table monitoring operation will be automated in the near future.

Shallow groundwater quality

A grid of piezometers are installed at the centre of each plot. They will be used to measure hydraulic head gradients and to obtain water samples for nitrate analysis. Each group consists of six piezometers, placed at respective depths of 40, 60, 80, 100, 140 and 180 cm. They are made of 25.4 mm dia. PVC pipes, sealed at the bottom and perforated with one row of 6.4 mm diameter holes approximately 20 mm from the bottom. The water level with respect to top of the piezometer is recorded, then using a hand pump all of the water is pumped out. A water sample is then taken 24 hours after water has seeped into the piezometer. Soil water samples from piezometers are collected on a weekly basis.

Soil samples

Surface soil samples 0-30 and 30-60 cm will be taken from each plot within 1 week of planting corn and 0 and 3 days, 1,2,4,8,12,16 and 20 weeks after N is applied. Soil samples (20 cm depth increments) will also be taken in the spring and fall to 1 m depth. These samples are transferred to HRS and stored at 4°C for chemical analysis.
Climatic measurement

An automatic weather station at the experimental field collects and transfers detailed meteorological data to HRS via modem. These data include maximum and minimum temperature, solar radiation, rainfall intensity and amount, wind speed and direction, relative humidity, and soil temperature.

Yields

Corn is planted in mid-May and harvested in mid-October. For each treatment, plant growth parameters and yield are measured. Plant tissue and grain sample are also analyzed for nitrogen content.

Laboratory Measurements

Analysis of the water and soil samples collected from the experimental plots is conducted at the Harrow Research Station. These samples are analyzed for nitrate concentration using TRAACS 800 autoanalyzer technology. Results from these chemical analyses in conjunction with flow data will help to quantify the amount of nitrate losses under the different treatments.

RESULTS AND DISCUSSION

The drought conditions in 1991 growing season affected crop establishment, growth and levels of nitrate concentration in water. Movement of nitrate through surface runoff and subsurface drainage was low from cropping practices because no major runoff events occurred until late in the fall.

Nitrate concentration was determined for each tile water sample collected in each runoff event. In some events the nitrate concentration varied with sample number, however with frequent sampling, nitrate concentration curves were readily obtained. For example, in the Nov. 26 runoff event the nitrate concentration of the first sample collected from the water table control treatment with fall moldboard plow tillage (W-MP) was 20 ppm which increased to a maximum of 75.1 ppm with the fourth sample and then decreased to 29.8 ppm with the tenth and final sample collected (Fig. 5). Since the total nitrate loss for each experimental plot is calculated by summing the products of the nitrate concentration and water volume we were able to accurately determine total nitrate loss for each experimental plot in each runoff event.

Water table control systems were implemented on July 14, 1991 and tillage treatments were initiated on November 5, 1991. Therefore runoff events which occurred after November 5, 1991 would begin to reflect the combined effect of tillage and the water table control system. From October 29, 1991 to December 6, 1991 the water table control system was opened to allow for free drainage of excess water which may have otherwise interfered with fall harvesting and tillage operations. The total volume of water which flowed through the tiles in the two runoff events (November 6, 1991 and November 26, 1991) during this period were greater with the water table control treatments than with the drainage plots (Fig. 6).

The nitrate concentration in tile water ranged from 8 to 75 ppm over the seven runoff events between November 6, 1991 and March 2, 1992 (Fig. 7). The nitrate concentration in tile runoff events over the winter 1991-1992 exceeded 10 ppm in 96% of the events. Water table control reduced the nitrate concentration in tile runoff water by 32%. This was due in part by the greater nitrogen uptake and corn yield with the water table control treatment in 1991 which resulted in lower residual nitrate levels in the soil.

Total nitrate loss per treatment per event ranged from 0.42 to 13.2 kg N ha⁻¹ (Fig. 8). Since there can be 15 or more runoff events occurring in a given year, it is easily realized that the total nitrogen loss can be both economically and environmentally deleterious. Total nitrate lost through subsurface drainage was greater in the drainage plots than the water table control treatments in all runoff events except November 6, 1991. The November 6
runoff event was the one immediately following the opening of the water table control system in which the volume of water with the water table control treatments was over five times that of the drainage treatments. Therefore, it was not surprising that there was more nitrate lost in this runoff event despite the fact that the concentration of nitrate in tile water was lower in the water table control treatments.

Subirrigation was initiated on July 14 and ended on September 4. During this period a total of 706 m³ irrigation water was delivered to eight water table control plots. It was desired to maintain the water table at an average depth of 60 cm below the soil surface at midspacing between the drains. However, on July 14, the water table had already dropped to a depth of 120 cm and our attempt to raise the water table to the desired depth due to low hydraulic conductivity of the soil (4 cm/day) and high evapotranspiration demand was not successful. Nevertheless, the average water table depth in the water table control plots remained higher than in the drainage plots during the growing season.

Figure 9a illustrates the water table changes during the growing seasons in plot #1 which is typical of the other plots. As it was expected the water table rose to 40-50 cm above the drain, however, at midspacing the water table remained 120 cm below the soil surface. The effect of this water table depth variation on the corn yield is depicted in Fig. 9b. Area on top of the drain (higher water table) produced more yield than the area at the midspacing between the drains. Corn yield was harvested on Oct 23 and 24. Water table control plots produced consistently higher average yield (9.8%) than that of the drainage plots (Fig. 10).

The results of this study to date demonstrate the effectiveness and responsiveness of the water table control treatments in reducing the volume of subsurface runoff, the concentration of nitrate in tile water and therefore the total nitrate loss. Although the tillage treatments were only implemented in November, there already appears to be a tillage response to water movement and nitrate loss through subsurface drainage. The combined effect of tillage, intercropping and water table control systems on water quality should be apparent in the next 2-3 years. During this time we will be able to determine the average nitrate concentration in tile water per year, total volume of water lost, total nitrate loss per year and the nitrogen balance for each of the soil crop and water management systems.

ACKNOWLEDGEMENTS

This research has been supported by grants from the Preservation Fund, Great Lakes Water Quality Action Plan.

REFERENCES


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Figure 1: Experimental plot layout showing instrumentation building, drainage system and underground conveyance drain pipes
Figure 2: Schematic diagram of the water table control structure
Figure 3: Plan layout of the instrumentation building
Figure 4: Profile diagram of one flow measurement and water sampling unit
Figure 5: Nitrate concentration in tile water samples taken during the November 26, 1991 runoff event for the drainage moldboard plow (D-MP), drainage-soil saver (D-SS), water table control moldboard plow (W-MP) and water table control soil saver (W-SS) treatments.

Figure 6: Volume of water leaving the tiles between November 6, 1991 and March 2, 1992 runoff events for the drainage moldboard plow (D-MP), drainage-soil saver (D-SS), water table control soil saver (W-SS) treatments.
**Figure 7:** Average nitrate concentration in tile water samples between November 6, 1991 and March 2, 1992 runoff events for the drainage moldboard plow (D-MP), drainage-soil saver (D-SS), water table control moldboard plow (W-MP) and water table control soil saver (S-SS) treatments.

**Figure 8:** Total nitrate lost through tile water samples between November 6, 1991 and March 2, 1992 runoff events for the drainage moldboard plow (D-MP), drainage-soil-saver (D-SS), water table control moldboard plow (W-MP) and water table control soil saver (W-SS) treatments.
Fig. 9a: Typical water table fluctuation during 1991 growing season (Plot #1)

Fig. 9b: Effect of water table depth variation on the corn yield.
Figure 10: Corn yield for the drainage moldboard plow (D-MP), drainage-soil saver (D-SS), water table control moldboard plow (W-MP) and water table control soil saver (W-SS) treatments.
NITROGEN MANAGEMENT - A BUDGETARY APPROACH

Michael J Goss¹

ABSTRACT

One method for predicting nitrate losses from agriculture to groundwater is to calculate the nitrogen balance for the whole farm, taking account of animals and crops. The resultant N-budget can be formulated so that a positive balance indicates the amount of N potentially available for leaching. This amount together with the average annual through-drainage is then used to predict maximum nitrate-N concentrations moving to groundwater from the farm. Calculation of an N-budget can be simplified by assuming that soil organic matter content, and consequently soil N content, remain constant on a yearly basis for monoculture systems or over the course of a rotation when a sequence of crops are grown. The N-budget for one cycle of the farming system, either one year or the length of a crop rotation can indicate the long term potential of a given farming system to cause nitrate-N contamination of groundwater.

Nitrogen budgets for some 400 farms are currently being prepared in conjunction with a Farm Groundwater Quality Survey being carried out under the Environmental Sustainability Initiative funded through Agriculture Canada. This enables the predictions of nitrate concentrations in groundwater made from nitrogen balances to be compared with measured values. In this paper some predictions for model cropping systems are presented together with some for commercial farms in Ontario. The limits and potential of the budgetary approach are considered.

INTRODUCTION

In Ontario, intensification of agricultural production has resulted in the development of specialization of enterprises. In consequence there are arable farms with soils having depleted organic matter where only mineral fertilizers are available. Equally there are some animal enterprises producing more manure than can safely be applied to the associated land base. However, within the main geographic regions some mixed farming is also practised, and speciality crops are grown in a wide range of locations. This makes it difficult to evaluate directly the consequences of different management practices on the environment at a regional scale.

One method for predicting regional losses of nitrate from agriculture to groundwater is to calculate the nitrogen balance for a whole farm, taking account of animals and crops. The resultant N-budget can be formulated so that a positive balance indicates the amount of N potentially available for leaching. This amount for typical farming systems could then be combined with hydrological information and climatic data using a Geographic Information System to predict maximum nitrate-N concentrations moving to groundwater from farming in the region.

The basic relationships for the nitrogen budget of a farm can be summarized as:

Nitrogen in inputs = nitrogen in output + change in nitrogen assets

Calculation of an N budget can be simplified by assuming that there is no net change in nitrogen assets. Thus for an arable farm it is assumed that soil organic matter content, and consequently soil N content, remain constant on a yearly basis for monoculture systems or over the course of a rotation when

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a sequence of crops are grown. Similarly for a livestock operation it is assumed that the number of animals and their demography remain constant. The N-budget for one cycle of the farming system, either one year or the length of a crop rotation can then indicate the long term potential of a given farming system to cause nitrate-N contamination of groundwater.

**COMPONENTS OF A SIMPLIFIED NITROGEN BUDGET**

**Nitrogen In Inputs**

The main components of the nitrogen inputs to agricultural systems are summarized in Table 1. They derive from direct purchases from off-farm suppliers, and from natural processes that occur during the growth of crops (e.g. symbiotic nitrogen fixation) or from natural processes that are influenced by anthropogenic activity (e.g. atmospheric deposition).

Fertilizer inputs can be obtained from agricultural statistics on a county basis, or farmer records can be used for a whole farm, or some farmers even record inputs on a field by field basis. Average nitrogen contents of animal manures (solid or liquid) (Fraser, 1985) were used in this study to convert weights or volumes of organic fertilizer applied to fields to an equivalent weight of nitrogen.

The nitrogen content of seeds used in crop production can be determined from average contents published by analysts (e.g. McBride, 1987) together with statistical or farmer data on seeding rates. Feed contents can be determined in the same way.

**Nitrogen In Outputs**

The main components of the nitrogen outputs from agricultural systems are summarized in Table 3. They derive from direct sales of plant and animal materials off-farm, and from gaseous and leaching losses.

Off-sales were calculated from the crude protein content or nitrogen concentration of materials, and the weight of the material.

Gaseous losses were assumed to be zero except for volatilization of ammonia from organic manures used or produced on the farm. Loss from animal manures was estimated to be 39% of total nitrogen (Beauchamp and Burton, 1985). Loss from sewage sludge was assumed to be 5% of total nitrogen.

The budgets were formulated to calculate the excess nitrogen on the farm at the end of a crop cycle. This weight of nitrogen was assumed to be susceptible to leaching. The annual through drainage was calculated from examination of stream discharge over the province. Coote *et al.* (1982) obtained a value of 320 mm per year for the average rate of stream discharge in 10 watersheds in Southern Ontario. The groundwater contribution to stream discharge was estimated to be 51%. Assuming that groundwater discharge and recharge were equal in magnitude, annual through drainage was estimated to be 160 mm.

The nitrogen in animals bought in can be estimated from average weights of cattle, pigs or poultry typically purchased for fattening, breeding or milking, and assuming typical values for protein content.

Natural inputs through symbiotic nitrogen fixation need to be estimated from empirical relationships between plant/crop growth and nitrogen fixed. The relationship used for grain legumes (Table 2) was developed from a literature review including unpublished information from the University of Guelph. The value used for alfalfa-hay (Table 2) was obtained in the same way, except the best regression obtained used the extra yield of hay gained in the presence of the legume compared with the yield of unfertilized grass. An average yield for unfertilized grass of 3.2 t ha\(^{-1}\) (Sheard, 1977) is currently assumed.

The value used for atmospheric deposition was also obtained from the literature. Two components were identified. Wet deposition in precipitation and mist which is estimated at 10.4 kg N ha\(^{-1}\) (Ro *et al.*, 1988). Dry deposition in dust
Table 1. Major components responsible for inputs of nitrogen to agriculture on farms

<table>
<thead>
<tr>
<th>NITROGEN INPUTS</th>
<th>Purchases</th>
<th>Natural inputs</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fertilizer</td>
<td>Mineral</td>
<td>Nitrogen Fixation</td>
</tr>
<tr>
<td>Organic (manure)</td>
<td></td>
<td>Symbiotic</td>
</tr>
<tr>
<td>Seed</td>
<td></td>
<td>Non-symbiotic</td>
</tr>
<tr>
<td>Animals and feed</td>
<td></td>
<td>Atmospheric deposition</td>
</tr>
</tbody>
</table>

Table 2. Regression coefficients for the relationships between yield (t ha⁻¹) and nitrogen fixation (kg N ha⁻¹) obtained by correlation of published results for Ontario.

<table>
<thead>
<tr>
<th>Crop</th>
<th>Regression Coefficient</th>
<th>Constant</th>
<th>Significance level</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soybean</td>
<td>81.1</td>
<td>-98.5</td>
<td>r² = 0.81, p &lt; 0.01</td>
</tr>
<tr>
<td>Alfalfa hay</td>
<td>27.3</td>
<td>+73.4</td>
<td>r² = 0.92, p &lt; 0.01</td>
</tr>
</tbody>
</table>

EXCESS NITROGEN IN DIFFERENT FARMING SYSTEMS

The main assumption made in this simplified budgetary approach was that the farming practices had reached a state of equilibrium. For arable farming, the key factor is the soil organic matter content, and whether it is in a quasi-steady state with the total returning to the same value at the end of each cycle of crop rotation. A simple system that would be expected to be closest to equilibrium, and hence be a good test of the budgetary approach would be a well established orchard. Close agreement was found between the concentration of nitrate-N predicted to be in the groundwater, and the value measured in the well water (Table 4).

Nitrogen budgets for some 400 farms are currently being prepared in conjunction with a Farm Groundwater Quality Survey being carried out under the Environmental Sustainability Initiative funded through Agriculture Canada. This enables the predictions of nitrate concentrations in groundwater made from nitrogen balances to be compared with measured values. In this paper predictions of nitrate nitrogen in groundwater for some model cropping systems and for some commercial farms in Ontario are compared with measured values of water draining from the rooting zone or water from farm wells. The limits and potential of the budgetary approach are considered.

other particulate material plus gaseous absorption estimated at 8 kg N ha⁻¹ (Beauchamp et al., 1978; Dasch and Cadle, 1985; Ro et al., 1988; Moller and Schieferdecker, 1989; Scheider et al., 1979; Sirois and Barrie, 1988).

The fact that organic manures were applied to crops did not necessarily cause a problem in making close predictions (Table 6).
Table 3. Major components responsible for output of nitrogen from farms.

<table>
<thead>
<tr>
<th>NITROGEN IN OUTPUTS</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Off-sales</strong></td>
</tr>
<tr>
<td>Manure</td>
</tr>
<tr>
<td>Grain/plants/plant products</td>
</tr>
<tr>
<td>Animals/animal products</td>
</tr>
</tbody>
</table>

Table 4. Nitrogen budget for a well established orchard.

* Total Inputs-outputs, leaching loss not included.

<table>
<thead>
<tr>
<th>Input</th>
<th>kg N ha⁻¹</th>
<th>Output</th>
<th>kg N ha⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fertilizer</td>
<td>22.4</td>
<td>Plant/Plant products</td>
<td>24.2</td>
</tr>
<tr>
<td>Fixation</td>
<td>5.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Atmospheric deposition</td>
<td>18.4</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td><strong>45.8</strong></td>
<td><strong>24.2</strong></td>
<td></td>
</tr>
<tr>
<td>Imbalance*</td>
<td>21.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Predicted groundwater contamination</td>
<td><strong>13.5 mg L⁻¹</strong></td>
<td>Measured groundwater contamination</td>
<td><strong>13.6 mg L⁻¹</strong></td>
</tr>
</tbody>
</table>

Table 5. Nitrogen budget for a tobacco farm where practices have changed in the last 6 years.

<table>
<thead>
<tr>
<th>Input</th>
<th>kg N ha⁻¹</th>
<th>Output</th>
<th>kg N ha⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Seed</td>
<td>0.6</td>
<td>Plant/Plant products</td>
<td>32.4</td>
</tr>
<tr>
<td>Fertilizers</td>
<td>37.4</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fixation</td>
<td>5.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Atmospheric deposition</td>
<td>18.4</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td><strong>61.4</strong></td>
<td><strong>32.4</strong></td>
<td></td>
</tr>
<tr>
<td>Imbalance*</td>
<td>29.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Predicted groundwater contamination</td>
<td><strong>18.1 mg L⁻¹</strong></td>
<td>Measured groundwater contamination</td>
<td><strong>10.4 mg L⁻¹</strong></td>
</tr>
</tbody>
</table>

* Total Inputs-outputs, leaching loss not included.
Table 6. Nitrogen budget for a tomato and seed corn farm where sewage sludge is applied regularly.

<table>
<thead>
<tr>
<th>Input</th>
<th>kg N ha⁻¹</th>
<th>Output</th>
<th>kg N ha⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Seed</td>
<td>1.0</td>
<td>Plant/Plant products</td>
<td>116.6</td>
</tr>
<tr>
<td>Fertilizers</td>
<td></td>
<td>Gaseous loss</td>
<td>4.2</td>
</tr>
<tr>
<td>Inorganic</td>
<td>34.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sewage sludge</td>
<td>84.2</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fixation</td>
<td>5.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Atmospheric deposition</td>
<td>18.4</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td><strong>1433</strong></td>
<td><strong>120.8</strong></td>
<td></td>
</tr>
<tr>
<td>Imbalance*</td>
<td></td>
<td></td>
<td>22.5</td>
</tr>
<tr>
<td>Predicted groundwater</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>contamination</td>
<td>14.1 mg L⁻¹</td>
<td>Measured groundwater</td>
<td>133 mg L⁻¹</td>
</tr>
<tr>
<td>contamination</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

* Total Inputs-outputs, leaching loss not included.

Table 7. Nitrogen budget for a mixed farm growing barley wheat and corn to feed dairy cattle and hogs.

<table>
<thead>
<tr>
<th>Input</th>
<th>kg N ha⁻¹</th>
<th>Output</th>
<th>kg N ha⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Seed</td>
<td>0.6</td>
<td>Plant/Plant products</td>
<td>19.1</td>
</tr>
<tr>
<td>Feed</td>
<td>3.0</td>
<td>Animals/animal products</td>
<td>6.3</td>
</tr>
<tr>
<td>Fertilizers</td>
<td>51.9</td>
<td>Gaseous loss</td>
<td>12.4</td>
</tr>
<tr>
<td>Livestock</td>
<td>0.6</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fixation</td>
<td>41.8</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Atmospheric deposition</td>
<td>18.4</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td><strong>1163</strong></td>
<td><strong>413</strong></td>
<td></td>
</tr>
<tr>
<td>Imbalance*</td>
<td></td>
<td></td>
<td>75.1</td>
</tr>
<tr>
<td>Predicted groundwater</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>contamination</td>
<td>38.1 mg L⁻¹</td>
<td>Measured groundwater</td>
<td>11.6 mg L⁻¹</td>
</tr>
<tr>
<td>contamination</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

* Total Inputs-outputs, leaching loss not included.
Table 8. Effect of yield (Tomato-soybean rotation) and fertilizer practice (Corn-soybean-wheat rotation) on the nitrogen imbalance.

<table>
<thead>
<tr>
<th>Model system</th>
<th>Fertilizer applied kg N ha(^{-1})</th>
<th>Yield tonnes ha(^{-1})</th>
<th>Imbalance kg N ha(^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tomato-soybean</td>
<td>90.0</td>
<td>45.0</td>
<td>5.6</td>
</tr>
<tr>
<td>Tomato-soybean</td>
<td>90.0</td>
<td>30.0</td>
<td>17.6</td>
</tr>
<tr>
<td>Corn-soybean-wheat</td>
<td>180.0</td>
<td>6.9</td>
<td>35.8</td>
</tr>
<tr>
<td>Corn-soybean-wheat</td>
<td>105.0</td>
<td>6.1</td>
<td>13.6</td>
</tr>
</tbody>
</table>

However, in a number of cases the use of organic manures or cover crops was associated with a prediction of large imbalances, and predicted values of nitrate-N concentration in groundwater were considerably greater than the values measured. This suggests that these practices have not been in use long enough on the farms studied for the soil to have established a new equilibrium. Many livestock farms appear to give large imbalances, which may be due to the fact that these operations are also not in equilibrium (Table 7). If intensification is still in progress then the organic matter content of soil may still be increasing.

The conclusion from the comparisons so far made suggests that the budgetary approach can predict contamination of groundwater for farming systems that are close to equilibrium. They can therefore be used to compare the potential of different farming systems to cause contamination of the environment.

FEATURES OF FARMING PRACTICES THAT HAVE LESS ENVIRONMENTAL IMPACT

Using the budgetary approach a number of model systems have been compared. These assume average yields and recommended rates of nitrogen fertilizer. They show that maintaining good soil conditions for crop growth is important, since for the same inputs yield level determines the nitrogen off take (Table 8). Similarly if the yield obtained is similar then imbalances reflect the fertilizer application (Table 8).

FUTURE RESEARCH NEEDS

The results presented clearly indicate that the use of a simplified budgetary approach has considerable potential for assessing possible environmental contamination from farming systems and individual practices. The lack of a good prediction of nitrate contamination under livestock systems needs closer examination. Similarly there is insufficient information for gaseous losses to be predicted reliably.

LITERATURE CITED


ABSTRACT

The Chalk aquifer of eastern-central England is seriously affected by nitrates leached from agricultural land and, like many similar aquifers, its future as a potable resource must depend considerably on the nature and rates of denitrification. The regional distribution of nitrate supports a viewpoint based on thermodynamic criteria that denitrification is actively occurring and that the problem will be short-lived. More detailed considerations based on major-ion and environmental isotope data, however, indicate that denitrification is not significant and that the apparent lowering of nitrate concentration (from >10 to <2mgL\(^{-1}\) NO\(_3\)-N) in the apparent direction of flow is due primarily to mixing between waters of different origins. Presence of reduced nitrogen species is possible evidence of denitrification in some older waters (>4000 yr. B.P.). It is concluded that denitrification can not be relied upon to reduce elevated concentration of nitrate in modern recharge waters.

INTRODUCTION

The contamination of groundwater by nitrate has been a considerable concern since an association was established between elevated concentrations of nitrate in drinking water and a disease known as methaemoglobinemia (Comley, 1945). The disease is caused by the bacterial reduction of nitrate to nitrite in the intestinal tract. The nitrate enters the bloodstream and combines with the haemoglobin to form methaemoglobin which reduces the capacity of the blood to transport oxygen. The reduction to nitrite occurs primarily in infants because the lower acidity of their gastric juices provides a better environment for nitrate-reducing bacteria.

In its uncomplicated form, methaemoglobinemia is usually easily diagnosed and its treatment understood. However, as a precaution the World Health Organization recommends that the concentration of nitrogen as nitrate (NO\(_3\)-N) in drinking water should not exceed 11.3 mL\(^{-1}\), although occasional levels up to 22.6 mgL\(^{-1}\) might be acceptable. In North America the recommended maximum permissible concentration limit is normally 10 mgL\(^{-1}\) NO\(_3\)-N.

More recent work has suggested that nitrate may also play a role in the production of nitrosamines in the stomach. Nitrosamines are known carcinogens and M.J. Hill et al. (1973) have offered evidence for a possible link between gastric cancer and high nitrate concentrations in ingested water.

In recent years contamination by the leaching of nitrate from agricultural land has become a particular concern (Saffigna and Keeney, 1977; Young and Gray, 1978; A.R. Hill, 1982). Many sources of the nitrate contamination have been proposed, some natural (Kreitler and Jones, 1975), others manmade (Meisinger, 1976). In most cases, however, the sources of the leached nitrate can be associated with either of the following:

1. Chemical fertilizers. Efficiency of use of fertilizer nitrogen by crops and pastures can vary widely but lysimeter isotope studies by Cannel and Burford (1976) have shown that over a 3-yr. period as little as 50% of the applied nitrogen may be recovered as herbage, the remainder accumulating in the soil zone and available for leaching.
Natural soil nitrogen. Most agricultural soils contain between 1500 and 6000 kg ha\(^{-1}\) of organically bound nitrogen in the top 150 mm (Cooke and Williams, 1970). Deep ploughing aerates the soil profile and converts part of this nitrogen to nitrate which then migrates to the water table during the subsequent wet season. The process is endorsed by Young and Gray (1978), who have shown that the nitrate profile, as determined from pore waters in the unsaturated zone of the Chalk beneath a site in southern England, shows peaks which can be correlated historically to the alternating fixation of nitrogen in grass leys and subsequent release by ploughing.

Although the incidence of groundwater contamination by nitrate has increased in recent times, the real extent of the problem must ultimately depend on the chemical behaviour of nitrate in the groundwater environment. From the purely theoretical viewpoint the groundwater nitrate problem should be short-lived. As indicated by the speciation diagram in Fig. 1 (Marsh, 1978), nitrate is not a stable nitrogen species under natural groundwater conditions and at a pH of 7 will tend to reduce to gaseous nitrogen (N\(_2\)), at redox potentials below -700 mV. Significantly, as supported by Dreyer (1982, Fig. 12-1), this means that gaseous nitrogen should be the stable nitrogen species in oxygenated groundwaters where Eh-values are typically in the range 450-750 mV.

Chemically the denitrification reaction can be expressed in the form:

\[
5\text{CH}_2\text{O} + 4\text{NO}_3^- \xrightarrow{\text{W}} 2\text{N}_2(\text{g}) + 5\text{HCO}_3^- + 2\text{H}_2\text{O} \ (1)
\]

where CH\(_2\)O represents organic material. In a very reducing environment (Eh < -200 mV at pH 7), NH\(_4^+\) is the stable nitrogen species and the complete denitrification process can be represented in the form:

\[
2\text{CH}_2\text{O} + \text{NO}_3^- + \text{H}_2\text{O} \xrightarrow{\text{W}} \text{NH}_4^+ + 2\text{HCO}_3^- \ (2)
\]

The reduction reactions are generally slow but are considerably accelerated if catalyzed by mediating bacterial organisms.

Published literature concerning nitrate and the denitrification process is extensive (N.A.S., 1978), but deals almost exclusively with surface water, waste water, soil water and water in the unsaturated zone. With the exception of Gillham and Cherry (1978) and possibly Edmunds and Walton (1983) there is remarkably little evidence of denitrification in groundwater systems. In part this must reflect the complexity of sub-surface chemical processes and the difficulty of identifying the denitrification process. It may also be an indication that denitrification is not a significant process in the aquifer environment.

This paper examines nitrate variation and postulated denitrification processes in the Chalk aquifer of eastern England. The aquifer is seriously affected by nitrates leached from agricultural land and like many similar aquifers its future as a potable resource will depend considerably on the nature and rates of denitrification.

**HYDROGEOLOGY**

The Chalk is a white fine-grained limestone of Late Cretaceous age. In the west (Fig. 2) the Chalk is exposed along a broad upland area known as The Wolds where recharge of a predominantly direct nature occurs (Howard and Lloyd, 1979). In this region groundwater heads locally exceed 100 m above sea level and the thickness of the unsaturated zone varies between 20 and 30 m to the north of Caistor to over 50 m in the more elevated area around Binbrook. Groundwater flow occurs towards the north and northeast where the aquifer is confined beneath a coastal till plain.

Historically the region was sparsely populated and The Wolds supported natural permanent pasture and occasional woodland. The Chalk was developed as an aquifer only locally at this time and most of the recharge re-appeared at overflow springs (blow wells) (Fig. 3) that had developed in the otherwise impermeable till cover. After the World War II virtually all the established grassland along The Wolds was
ploughed for the production of cereal crops. In addition the Chalk aquifer was developed extensively through a series of major production wells (Fig. 3). At present over 160 M liters of good-quality water are pumped from the Chalk daily and natural outflows are rarely observed.

**HYDROLOGICAL APPROACH AND METHOD**

A comprehensive major-ion, minor-ion and environmental isotope study was conducted as part of an in depth multi-disciplinary investigation of saline intrusion and its influence on aquifer management (Evans et al., 1979). The primary objectives of the study were to determine the origins of the saline waters and to develop an understanding of the regional groundwater flow system as a prerequisite to numerical modelling. Significantly the magnitude of the nitrate problem in the area did not become apparent until the later stages of the study. Nevertheless, as demonstrated below, the recognition of groundwater origins made possible by the comprehensive hydrochemical approach adopted proved to be crucial in correctly evaluating the regional distribution of nitrate.

The study was based on an exhaustive groundwater sampling programme involving 450 samples from 346 production wells, observation wells and springs. All samples were analysed for pH, major ions (Ca, Mg, Na, K, HCO₃, SO₄, Cl and NO₃) (Howard, 1985) and minor ions (Br, Sr, F and I) (Howard and Lloyd, 1978; Lloyd et al., 1982) using standard analytical procedures. Ammonium ion data were supplied from an earlier survey. Redox electrode determinations were limited to artesian overflow and submersible pumped samples where atmospheric contamination could be excluded during the field measurements.

To supplement these data, 29 carbon isotope samples were collected using a field precipitation technique described by Lloyd (1981). These samples were analysed for ¹⁴C and ¹³C in the University of Birmingham laboratories and calculated groundwater ages (Lloyd and Howard, 1978) were corrected for rock-water reaction processes, using a method proposed by Wigley (1976). Tritium data were analysed and made available from an earlier unpublished study by the British Geological Survey (formerly Institute of Geological Sciences).

During the final stages of the investigation twenty samples were collected from selected pumped wells for enumeration of nitrate-reducing bacteria. Sampling was carried out using sterile bacteriological techniques and the analysis (MacFaddin, 1980) utilised a peptoneNaNO₃ culture to grow the nitrate-reducing bacteria. Most probable numbers of organisms per 100 ml of sample were determined using a standard multiple-tube method.

**RESULTS AND DISCUSSION**

The saline groundwater origins were delineated primarily on the basis of major- and minor-ion chemical data. Three saline groundwater types were identified (Howard and Lloyd, 1978, 1983), the oldest and most prevalent being associated with a major marine transgression known to have occurred during the Ipswichian (120,000 yr. B.P.). Regional flow distributions were interpreted with additional support from the environmental isotope data. In this part of the study, four chemical water types were resolved (Table I; Figs. 3 and 4) (Howard, 1985) each of which appears to have evolved independently as a function of its own unique aquifer history.

The magnitude and extent of the nitrate problem in the study area is demonstrated by Fig 5 in which the regional distribution of NO₃-N is shown in relation to the major production wells. For ease of reading, the 346 sample concentrations and their sites have been omitted from the figure. The most significant feature of the diagram is the gradual depletion of nitrate from the recharge zone in the west to the outflow sites in the east. Normally this would imply rapid denitrification in the direction of groundwater flow, a model that is supported both by a regional lowering of redox potential in a downflow direction (Fig. 6) and the presence of nitrate-reducing bacteria at all selected sample sites (Fig. 7).
Furthermore, the postulated denitrification process would conform with theoretical stability relationships shown in Fig. 1, whereby nitrogen gas is more stable than nitrate in moderately oxidising waters.

Unfortunately, despite the simplicity and obvious appeal of the denitrification model, the line of reasoning applied is only valid if groundwaters throughout the area have evolved from a single chemical water type. In the study area four, effectively independent water types have been identified and these must be examined individually if the true behaviour of nitrate in the Chalk groundwaters is to be evaluated.

NITRATE GEOCHEMISTRY OF THE FOUR WATER TYPES

Type IV. Most type-IV waters are old saline waters that pre-date deposition of an extensive covering of glacial drift throughout eastern parts of the area. On the basis of geological and hydrochemical evidence (Lloyd and Howard, 1978; Howard and Lloyd, 1983) these waters are associated with a marine transgression during the most recent, Ipswichian interglacial. All type-IV waters display low values of nitrate (< 1 mg L\(^{-1}\) NO\(_3\)-N) which due to their low redox potentials (< 100 mV) could be attributed to nitrate reduction. However, since low nitrate concentrations are characteristic of seawater (Hem, 1970), it is more likely that the concentrations simply reflect initial input levels and are not the result of denitrification processes.

Type III. Type-III waters are old low-salinity waters that replenished the aquifer when sea- and groundwater levels began to recover following the Devensian glacial episode (-15,000 yr B.P.). The waters are preserved in extremely low-flow zones immediately east of the historically active blow well sites (Figs. 3 and 4) and the Louth flow barrier (Figs. 2 and 3).

Redox potentials of type-III waters are in the range +100 to -50 mV and nitrate concentrations are less than 1 mg L\(^{-1}\) NO\(_3\)-N. These ground waters plot close to the N\(_2\) - NH\(_4\) boundary of the stability diagram in Fig. 1 and ammoniacal nitrogen is detected at selected sample sites (Table II). In the absence of natural organic sources of ammonia in the aquifer, the presence of reduced nitrogen species suggests type-III waters have been subjected to denitrification. However, it is not possible to determine the extent of the denitrification as the initial nitrate concentration must be dependent on The Wolds' vegetation at the time of recharge (12,000 - 4,000 yr B.P.) and is not known. It is therefore possible that, as with type-IV waters, the low levels of nitrate largely reflect low concentrations of nitrate in the initial input waters and that denitrification is not significant.

Type II. Type-II are relatively pure calcium bicarbonate waters with redox potentials in the range 200 - 300 mV. The waters are modern in radiocarbon terms and from tritium evidence pre-date type-I waters described below. Any possibility that type-II waters are derived from type-I waters through chemical evolution (including denitrification) can be discounted as there is no satisfactory process that could explain the apparent depletion in chloride (typically 25 - 50 and 12 - 20 mL\(^{-1}\) in types I and II, respectively).

Type II can be simply interpreted as waters that recharged the Chalk before the postwar transformation of The Wolds to a major cereal-producing area. Nitrate concentrations in these waters occur in the range 2 - 6 mg L\(^{-1}\) NO\(_3\)-N, which accords with the values recorded at northern production wells prior to about 1970 when high nitrate concentrations were first observed (Fig. 8). The range is also consistent with the average value of 4 mg L\(^{-1}\) NO\(_3\)-N determined in lysimeter experiments by Cooke and Williams (1970) for waters leached beneath unploughed Chalk grassland in Central England. There is no evidence to suggest that denitrification has in any way influenced nitrate concentrations in type-II waters.

Type I. Before the 1960's the active flow zone between recharge area and the naturally discharging blow wells was occupied by type-II waters described above (Fig. 4a). In recent years, increased pumping, particularly at inland
production wells has lowered the piezometric surface and most blow wells have ceased to flow. This has resulted in the development of a new active flow zone between recharge area and production wells (Fig. 4b), a zone in which type-II waters are rapidly being displaced by high nitrate type-I waters of very recent origin.

The influx of type-I waters is most marked in the north of the study area where the Chalk's unsaturated zone is relatively thin (20 -30 m) and offers least temporal resistance to migrating recharge waters. In this area, displacement of type-II waters (4 - 6 mgL⁻¹ NO₃-N) by type-I waters resulted in a three-fold increase in nitrate at major pumping stations during the late 1970's (Fig. 8). This is consistent with the model proposed by Young et al. (1976) in which nitrates and other solutes released by ploughing during the 1940's move through the unsaturated zone at an average rate of ≈ 1 m yr⁻¹. Further south, where the thickness of the unsaturated zone tends to be greater, type-II waters still occupy the eastern extent of the present active flow zone.

Nitrate concentrations in type-I waters display considerable variation (between 6 and 18 mgL⁻¹). However, in view of the high redox potentials (> 300 mV), their very recent origin and the apparent chemical stability of older type-II waters, this range is thought to be due to variations in land use, the thickness of the unsaturated zone and groundwater mixing, rather than evidence of active denitrification.

CONCLUDING DISCUSSION

The role of denitrification in the aquifer environment is extremely difficult to evaluate. Nitrate inputs are spatially and temporally variable, flow conditions are transient, and mixing frequently occurs between waters of different origins. In the study described, reasonable control over the origins and histories of the groundwaters is afforded by the comprehensive availability of major-, minor-ion and environmental isotope data. Nevertheless, only general conclusions concerning nitrate behaviour can be drawn and specific questions remain to be answered.

The major conclusion is that denitrification can not be relied upon to reduce elevated nitrate concentrations in modern recharge waters. There is no indication of denitrification in type-II waters, the older and more reducing of the two modern water types. In addition the downgradient reduction of nitrate in type-I waters can be satisfactorily explained by type-I - type-II mixing. Only type-III waters show any evidence of denitrification and these waters are over 4000 yr. old. The study clearly illustrates the risk of interpreting regional nitrate distributions before the origins of the groundwaters and their evolutionary behaviour have been established.

ACKNOWLEDGEMENTS

Carbon isotope analyses were carried out by Mr. R.E.G. Williams in the Radio-carbon Laboratory at the Department of Geological Sciences, University of Birmingham. Tritium isotope data made available by the British Geological Survey, London. The author also gratefully acknowledges the help and support of Dr. J.W. Lloyd, project supervisor, Dr. N. Pacey, project technician, Mr. M. Carey who commented on the manuscript, and the Anglian Water Authority who undertook major-ion analyses and provided permission to publish.

REFERENCES


Table 1. Characteristics of origin of water type I - IV (Howard (1985)).

<table>
<thead>
<tr>
<th>Water Type</th>
<th>Origin</th>
<th>Cl (mg/L)</th>
<th>NO₃ N (mg/L)</th>
<th>Eh (mV)</th>
<th>pH</th>
<th>Tritium Count (T.U.)</th>
<th>Radiocarbon Age (Yr.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>I</td>
<td>Modern recharge water</td>
<td>25 - 50</td>
<td>6 - 18</td>
<td>&gt; 300</td>
<td>6.8 - 7.4</td>
<td>17 - 117</td>
<td>Modern</td>
</tr>
<tr>
<td>II</td>
<td>Recent recharge waters pre-dating type I</td>
<td>12 - 20</td>
<td>2 - 6</td>
<td>200 - 300</td>
<td>7.2 - 7.7</td>
<td>2 - 6</td>
<td>Modern</td>
</tr>
<tr>
<td>III</td>
<td>Early post-glacial recharge water preserved east of active flow zone</td>
<td>15 - 200</td>
<td>&lt; 1</td>
<td>100</td>
<td>7.2 - 8.2</td>
<td>&lt; 2</td>
<td>4,000 - 12,000</td>
</tr>
<tr>
<td>IV</td>
<td>Saline water intruded during lyswichian interglacial (120,000 yr. B.P.)</td>
<td>&gt; 200</td>
<td>&lt; 1</td>
<td>&lt; 100</td>
<td>7.3 - 8.6</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

Table 2. Nitrogen species in type-II water.

<table>
<thead>
<tr>
<th>Site</th>
<th>N NO₃-N (mg/L)</th>
<th>Total Ammonium N NH₃ - NH₄ (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maitby No. 1</td>
<td>0.1</td>
<td>2.48</td>
</tr>
<tr>
<td>Moumoy No. 2</td>
<td>0.1</td>
<td>2.52</td>
</tr>
<tr>
<td>Gayton</td>
<td>0.2</td>
<td>2.64</td>
</tr>
<tr>
<td>Tneddlethorpe</td>
<td>0.2</td>
<td>1.22</td>
</tr>
<tr>
<td>Polar Farm</td>
<td>0.2</td>
<td>0.46</td>
</tr>
<tr>
<td>Newfoundland Farm</td>
<td>0.2</td>
<td>0.51</td>
</tr>
</tbody>
</table>
Figure 1: Location of study area and feature map.
Figure 2: Stability fields of nitrogen species as a function of Eh and pH at 25°C and 1 atm pressure. Total N = 14 mg L⁻¹ (after March, 1978).
Figure 3: Water types in relation to discharge/abstraction sites.
Figure 4: Postulated changes in hydrochemical conditions during the past 50 years.
Figure 5: Relationship between nitrate ion distribution and discharge/abstraction conditions.
**Figure 6:** Distribution of oxidation-reduction potentials.
Figure 7: Numbers of nitrate-reducing bacteria determined at representative sites.
Figure 8: Temporal changes in nitrate concentration in groundwater from major production wells in the north of the study area (Well locations in Figure 2).
MODELLING NITRATE MOVEMENT FOR MANAGEMENT

H.R. Whitely\textsuperscript{1}, W.N. Stammers\textsuperscript{1}, and J. Sakupwanya\textsuperscript{2}

ABSTRACT

Transport of nitrate from agricultural fields to streams through buried-pipe drains at concentrations above the drinking water standard of 10mg/L of nitrate-N is frequently observed in southern Ontario. This movement of nitrate is of concern both as a possible contribution to poor water quality in the stream, and as an economic loss to the land owner. A model is described in this paper that uses a simple mass-response function to represent this solute transport coupled with a transfer function for pipe flow. The model is easy to calibrate and has been used successfully to simulate observed patterns of nitrate movement through a drainage system at the Elora Research Station. The model could be used to prepare long-term records of nitrate flux from an area under different management scenarios. Such records are required, for example, in the application of a Bayesian decision-analysis framework for evaluating strategies for controlling nonpoint nitrate pollution.

INTRODUCTION

The concentration of nitrate nitrogen in surface water and in groundwater is under close scrutiny in many locations including southern Ontario. Nitrate nitrogen in surface water may contribute to high rates of plant (algal) growth and thus create environmental problems. Concentrations of nitrate nitrogen above the limit of 10mg/L set for drinking water renders surface or groundwater unusable as a source of potable water unless expensive treatment is added to remove the excess nitrate nitrogen.

The most prevalent source of excess amounts of nitrogen in water is manufactured fertilizer and manure applied to agricultural land. Management of the amounts and timing of the nitrogen applied is capable of reducing the concentrations of nitrate nitrogen in surface and groundwater receiving overland runoff and seepage from agricultural fields. However, there are costs involved in such management of nitrogen. These costs include the extra equipment, labour, and supervision that may be involved and the reduction in yield that also may result from lower amounts of nitrogen being used for crops.

To choose the best method and level of management for nitrogen use in agriculture in the context of sustaining a healthy environment it would greatly benefit the decision maker to know the relationship between benefits to the environment and costs to agriculture for each management option. With this knowledge it would be easier to select the best option i.e., the one with greatest net benefit.

Optimization and a Bayesian framework for decision analysis are alternate methods to use to select a best option for management. The Bayesian approach has an advantage in situations of uncertainty in knowledge because this approach explicitly accounts for the amount of uncertainty present.

We have investigated the applicability of Bayesian analysis to the problem of management of nitrate nitrogen in agriculture. The basic structure of the decision model is

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\end{footnotesize}
given later in this paper. The decision model requires data on nitrate flux amounts under various alternative scenarios of management of nitrogen. Our main emphasis in this paper is on the utility of a transport model we have developed to satisfy these data requirements.

The problem situation used as an example includes transport of nitrogen from cultivated land to receiving water via a systematic buried-pipe systems of field drains. Data from the Elora Research north of Guelph was used to calibrate the transport model.

**TRANSPORT MODEL**

A lumped linear-system approach was used to represent the transport of nitrate nitrogen from the ground surface through the root zone of the soil to the buried pipe drainage system and on to the receiving water. The concept is similar to the "unit hydrograph" approach used in the generation of flood hydrographs from watersheds. In this case the watershed is the volume of soil contributing flow to the systematic buried pipe drainage system upstream of its outlet.

The transport model was developed in two parts. The first part calculates the flowrates of water at the outlet of the drainage system by convoluting the rate of water arrival at the watertable (assumed known) with a linear impulse function (a set of cascading linear reservoirs) representing the behaviour of the shallow groundwater system and the buried-pipe drain. The order of calculation is shown in Figure 1.

The inputs required for the model are:

1. The seepage rates to the watertable as a function of time. This can be estimated from a hydrological model with adequate representation of infiltrability and soil water movement through the unsaturated zone.
2. The two parameters of the linear-cascade impulse function i.e., n, the number of linear reservoirs in the cascade, and K the time constant of each reservoir. K and n have so far been obtained by calibration using an observed hydrograph of flowrate at the outlet of a buried-pipe drainage system.

The second part of the transport model establishes the concentration of nitrate nitrogen in the water entering the buried-pipe drain. The product of this concentration with the flowrate from the first part of the model, as both vary with time, is the nitrate nitrogen flux into the receiving water. The nitrate concentration is modelled as a transfer process from the immobile portion of soil water to the mobile portion that is moving to the buried pipe. The immobile portion is assumed to have a constant nitrate concentration C_e during a single flow event.

The order of calculation for the second part of the transport model is shown in Fig. 2. The parameters required for the model are:

1. The concentration C_e of nitrate-nitrogen in the immobile soil water that is fixed for each event. The values for C_e for a sequence of events will follow a seasonal pattern and are expected to be highest early in the growing season after fertilizer has been applied and again after harvest when plant uptake of nitrate has stopped while nitrification of organic matter continues.
2. The mass transfer parameter h that has units of time. So far this parameter has been found by calibration with observed concentration of nitrate nitrogen in buried-pipe flow. No time (i.e., seasonal) dependence has been observed in this parameter.

**FIELD OBSERVATIONS**

The transport model has been tested using data from a buried pipe drainage system at the Elora Research station. The station is located about 20 km northwest of Guelph, Ontario. Flowrates and nitrate nitrogen concentrations have been observed for a 42 ha system of
buried-pipe drains since 1986. A map of the watershed is shown in Fig. 3.

The soil overlying the buried-pipe system is principally an imperfectly-drained Concestoga silt loam. The buried pipes are situated about 90 cm below the soil surface. There is a discontinuous sand layer at the level of the pipes and extending at some locations to a depth of 4 m. Water from this sand layer has been sampled for nitrate nitrogen using a set of shallow piezometers.

Below 4 m a tight hard till provides a low-permeability barrier to seepage downward. Sampling of piezometric head at a depth of 15m shown a vertical gradient downward to the underlying dolomite bedrock. Sampling of water also took place from these piezometers near the bedrock at a depth of 15m.

The observed concentrations of nitrate nitrogen in groundwater at varying depths is summarized for the years 1988 through 1990 in Table 1, taken from a report by Paul et al (1991). The concentration of nitrate nitrogen in buried pipe outflow frequently exceeds the standard for drinking water of 10mg/L.

Table 2 from the same source sets out the flux of nitrate nitrogen from the watershed. There is considerable variation from year to year but the amounts leaving the watershed in some years are quite appreciable (expressed in kg/ha).

RESULTS FROM THE TRANSPORT MODEL

The transport model was tested by assessing its ability to match the observed patterns of flowrate and flux of nitrate nitrogen at the outlet of the buried-pipe drainage-system watershed at Elora. Examples of the performance of the flowrate portion of the model, after calibration, during validation runs are shown in Fig. 4a and Fig. 4b taken from Sukupwanya (1991). The model provides a close approximation of the observed pattern of outflow rates.

The match between model and observation for nitrate-nitrogen flux is shown in figures 5a and 5b. Again the correspondence between observed and calculated values is acceptably close. Because of a scarcity of events with detailed nitrate measurement over time only calibrated events are available, i.e., no independent validation runs have been made.

LINK TO THE DECISION MODEL

A possible Bayesian-analysis model for the analysis of control actions on the application of fertilizer-nitrogen has the form:

\[ TSC = C_j + C_i (1 - e_j)R \]

(assessed for \( j = 1,2,...J \))

where

- \( TSC \) is the total social cost per year of control action \( j \), \( C_j \) is the direct cost of action, \( j \) (mostly reduced yield); \( C_i \) is the annual value of the receiving water to society; \( e_j \) is efficiency of action \( j \) in reducing the probability of nitrogen loading causing a loss of the value \( C_i \), \( R \) is the occurrence or non occurrence of a loss of \( C_i \).

The occurrence of violations (losses of the value of receiving water, \( C_i \)), as expressed in \( R \), has a stochastic character because of uncertainty, more particularly uncertainty about the amount of nitrogen that will reach the receiving water in any year. As a result of the stochastic nature of \( R \) the selection of a best action cannot be done simply by selecting the action with the minimum \( TSC \). Instead the decision requires minimization of an expected opportunity loss. This minimization is based on specification of the prior probability distribution of \( R \).

It is common to assume a log-normal distribution for the annual nitrogen loading. With this assumption, and specification of the amount of nitrogen required to produce a violation, knowledge of the mean and variance of the nitrogen loadings are sufficient to allow a prior probability distribution of \( R \) to be prepared.
In general long records of nitrogen fluxes from buried-pipe drainage systems are not available as a source of estimates of the mean and variance of nitrogen loading. The application of the transport model described above to a long period of rainfall and snowmelt records would provide the required set of annual nitrogen outputs from which the mean and variance could be derived.

It should be noted that since snowmelt is not a measured meteorological quantity it would have to be calculated from a snowpack model using precipitation and temperature data as inputs. As stated earlier a infiltration/soil water percolation model would also be needed to prepare the sequence of seepage-to-the-watertable data needed as input to the transport model.

CONCLUSIONS

The transport model that was described here is simple to calibrate and performed well in tests using data from the Elora Research Station. It requires further testing to demonstrate its ability to simulate results from other buried-pipe drainage systems.

If further testing confirms its suitability the model can be used in conjunction with a Bayesian decision-analysis model to evaluate alternate control methods for regulating the application of nitrogen to agricultural land in the presence of environmental damage due to excess nitrogen.

REFERENCES


Table 1: Observed concentrations of nitrate-N in drainage and in groundwater at the research watershed of the Elora Research Station 1988-1990.

<table>
<thead>
<tr>
<th>CATEGORY</th>
<th>YEAR</th>
<th>SEASON</th>
<th>HIGH NITRATE N mg/L</th>
<th>LOW NITRATE N mg/L</th>
<th>AVERAGE NITRATE N mg/L</th>
</tr>
</thead>
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<tr>
<td>TILE 1988</td>
<td>WINTER</td>
<td>12</td>
<td>1</td>
<td>6.8</td>
<td></td>
</tr>
<tr>
<td></td>
<td>SPRING</td>
<td>20</td>
<td>16</td>
<td>11.5</td>
<td></td>
</tr>
<tr>
<td></td>
<td>FALL</td>
<td>17</td>
<td>8</td>
<td>12.1</td>
<td></td>
</tr>
<tr>
<td>1990</td>
<td>WINTER</td>
<td>32</td>
<td>6</td>
<td>12.7</td>
<td></td>
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<tr>
<td></td>
<td>SPRING</td>
<td>35</td>
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<tr>
<td></td>
<td>FALL</td>
<td>32</td>
<td>13</td>
<td>24.6</td>
<td></td>
</tr>
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<td>1991</td>
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Table 2: Summary of nitrate-N output in water from the research watershed at the Elora Research Station 1986-1990.

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<tr>
<th>YEAR</th>
<th>SEASON</th>
<th>NITRATE N DRAIN kg/ha</th>
<th>NITRATE N OVERLAND kg/ha</th>
<th>NITRATE N PERCOL. kg/ha</th>
<th>TOTAL OUTFLOW N kg/ha</th>
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<td>0.3</td>
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Figure 1: Flowchart for subsurface flow submodel.

Figure 2: Flowchart for mass transport submodel.

Figure 3: Map of the research watershed at the Elora Research Station
Figure 4a: Validation for July 19, 1986 event.

Figure 4b: Validation for July 24, 1987 event.
Figure 5a: Calibration for March 22, 1990 NO$_3$-N flux.

Figure 5b: Scattergram for March 22, 1990 NO$_3$-N flux.
ENVIRONMENTAL SIGNIFICANCE OF NITROGEN TO AQUATIC LIFE

N. Bazinet

ABSTRACT

The surface waters of the province are the ultimate receiver of anthropogenic nitrogen either through surface run-off, groundwater percolation or aerial precipitation. Inorganic nitrogen (mostly as nitrate, nitrite and ammonia) is the form that most significantly impacts aquatic ecosystems. These different inorganic forms induce several aquatic environment responses including acute effects (e.g. BOD, mortality, avoidance) and chronic effects (e.g. eutrophication, reproductive failure, methaemoglobinaemia, increases in populations of detrimental species). Inorganic nitrogen transforms readily from one form to another in aquatic ecosystems depending upon redox potential.

A Provincial Water Quality Objective (PWQO) has been developed for ammonia (MOE 1984). A PWQO is designed to protect all aquatic life forms from long-term exposure to contaminants and also to protect against aesthetic effects (e.g. taste, odour, fish tainting, swimming). No PWQOs have been developed for nitrate or nitrite. However, federal guidelines are available for nitrate and nitrite (CCREM 1987). Nitrite is the most toxic form of nitrogen to aquatic life.

Nitrate and nitrite concentrations have been increasing linearly in the lower Great Lakes since the early 1970s. Mean concentrations in the early 1970s were estimated at approximately 100 µg/L (nitrate + nitrite) in Lake Erie and 180 µg/L (nitrate + nitrite) in Lake Ontario (IJC 1987). The linear relationships established in those studies allow extrapolation to 1992 ambient levels which estimate concentrations at about 250 µg/L in Lake Erie and 400 µg/L in Lake Ontario. Concentrations have been increasing in Lake Erie at a rate of about 75 µg/L per decade and in Lake Ontario at a rate of 100 µg/L per decade.

There are four major sources of nitrogen input to aquatic ecosystems:

1) Influent nitrogen from surface waters originates largely from agricultural sources and municipal sewage treatment plants.

2) Precipitation and outfall nitrogen from atmospheric sources to the lower Great Lakes have been estimated at ca 1.0 g N m⁻² yr⁻¹ (Wetzel 1975). Bird droppings also contribute a significant amount of atmospheric nitrogen, although the volume from this component has not been quantified.

3) Phytoplankton in the form of blue-green algae (also known as cyanobacteria) can fix a significant volume of nitrogen from the atmosphere if the blue-greens become dominant (Sevrin-Reyssac and Pletikosic 1990). Their dominance depends upon a nitrogen-phosphorous ratio of less than 5:1 (when inorganic nitrogen is limiting) (Nicholls and Carney 1986).

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1 Contaminants Assessment Scientist, Water Resources Branch, Ontario Ministry of the Environment, Toronto, Ont.
4) Groundwater nitrogen can be a significant source of input since groundwater contamination from nitrates is common in agricultural watersheds. There are few studies quantifying nitrogen input from this source to the Great Lakes.

Nitrogen undergoes a complex series of transformations upon entering an aquatic ecosystem. These conversions depend upon the conditions and properties of the receiver. These transformations are based on one of six major processes:

1) The assimilation of inorganic forms (e.g. ammonia (NH₃), nitrate (NO₃), nitrite (NO₂), and nitrous oxide (N₂O) to organic forms (e.g. amino acids, nucleic acids, proteins) by autotrophic bacteria and higher plants (CCREM 1987).

2) Heterotrophic conversions are complicated and uncommon. These involve the transfer of organic forms of nitrogen from one organism to another.

3) Ammonification involves the breakdown of organic nitrogen to ammonia by bacteria, fungi, zooplankton and fish.

4) Nitrification is the microbial oxidation of ammonia to nitrite and nitrate primarily under aerobic conditions by bacteria. The optimum conditions for this process are a pH of 8 and a temperature of 30°C. The process slows considerably when pH and temperature drop below these values.

5) Denitrification under anaerobic conditions produces nitrous oxide and molecular nitrogen (N₂) by microorganisms. The denitrification process occurs at temperatures between 5 and 60°C and optimally between pH 7-8. As with the nitrification process, the rate is directly temperature dependent.

6) Biological nitrogen fixation involving the conversion of atmospheric nitrogen gas to ammonia and then to organic nitrogen compounds occurs with cyanobacteria (blue-green algae). This process occurs under specific conditions when nitrogen-phosphorous ratios are less than 5:1 and inorganic nitrogen concentrations are limiting (Nicholls and Carney 1986).

When nitrogen enters an aquatic ecosystem, it rapidly converts to any of its several natural forms in a dynamic system dependent upon prevailing water conditions and the nature of the biological community. The group of factors governing the nitrogen cycle include temperature, redox potential, pH, hardness, dissolved organic carbon as well as the floral and faunal composition of the biological community.

**AQUATIC EFFECTS AND CRITERIA**

Inorganic nitrogen compounds induce a range of toxicological effects that can adversely affect all trophic levels in an aquatic ecosystem. A variety of effects have been identified for several species. These are reported from the readily available literature.

A. *Nitrate*

Acute lethal studies for nitrates with invertebrates produced LC₅₀s ranging from 300mg/L for the water flea *Daphnia magna* to 960mg/L for a snail *Lymnaea* sp. (Dowden and Bennett 1965) (Table 1). An immobilization study with *D. magna* yielded an EC₅₀ of 6200mg/L (Anderson 1944). A series of acute lethality studies with fish produced LC₅₀ values ranging from 99mg/L for mosquitofish (*Gambusia affinis*) (Wallen et al. 1957) to 6200 mg/L for channel catfish (*Ictalurus punctatus*) (Colt and Tchobanoglous 1976) (Table 1). Acute effects for fish and invertebrates were within the same range. Only warm water fish have undergone acute toxicity testing.
Chronic toxicity testing with nitrates has been reported for rainbow trout (*Salmo gairdneri*) (mild toxicity at 5-10mg/L) (Kinchloe et al. 1979), cutthroat trout (*Salmo clarki*) and chinook salmon eggs (*Oncorhynchus mykiss*) (mild toxicity at 20mg/L) (Kinchloe et al. 1979), and *D. magna* (96 h LC50 of 24mg/L) (Dowden and Bennett 1965) (Table 1). An acute study with invertebrates is carried out over a period of 48 hours or less while an acute study with vertebrates is carried out over a period of 96 hours or less. Chronic toxicity studies produced results in a range that was distinctly separate and lower than the acute range. Chronic studies involved two trophic levels with several sensitive species of the upper trophic level (e.g. salmonids).

One would expect mild toxicity at a concentration of 1mg/L for nitrate to aquatic biota. Based on these results, one could expect that concentrations of nitrate currently found in the lower Great Lakes would have little direct toxicity to aquatic life. However, indirect food chain effects and oxygen reductions resulting from BOD could have a significant impact to the aquatic community.

The CCREM guideline for nitrate is a narrative criterion stating that "concentrations that stimulate prolific weed growth should be avoided" (CCREM 1987). Nicholls and Carney (1986) found that nitrate concentrations in the Bay of Quinte were conducive to significant blue-green algal blooms when concentrations of phosphorous were low (e.g. N:P ratio < 5:1). Bluegreen algae are responsible for atmospheric nitrogen fixation and the production of toxins. Although their proliferation is dependent upon low nitrogen levels, localized outbreaks have been observed in sheltered areas of the lower Great Lakes.

Increasing nutrient concentrations in the lower Great Lakes will result in increased algal populations. Since algae form part of the diet of such organisms as zebra muscle (*Dreissena polymorpha*), then nitrogen reductions should help reduce populations of such undesirable species.

B. Nitrite

Acute toxicity testing with nitrites reported values ranging from a 96 h LC50 of 0.35mg/L for rainbow trout (Russo et al. 1981) to 616.4mg/L for Guadalupe bass (*Micropterus treculi*) (Tomasso and Carmichael 1986) (Table 2).

Chronic toxicity testing for nitrite produced values ranging from 0.015 mg/L (methaemoglobinemia) for steelhead (Wedemeyer and Yasutake 1978) to 0.06mg/L (NOEL - rainbow trout) (Russo et al. 1974) (Table 2). Methaemoglobinemia results in mortality when more than 70% of fish haemoglobin is converted to methaemoglobin (Williams and Eddy 1988). When exposure to nitrite is terminated, methaemoglobin will be converted back to haemoglobin through the activity of methaemoglobin reductase. No aquatic invertebrate toxicity studies were found for nitrite.

The Canadian Council of Resource and Environment Ministers (CCREM) has developed a surface water quality guideline for nitrite (CCREM 1987). The CCREM guideline for nitrite for the protection of aquatic life is 0.06mg/L (based on the rainbow trout NOEL).

It was originally shown that ambient concentrations of nitrate and nitrite in Lake Ontario and Lake Erie are 400 µg/L and 250 µg/L, respectively. If nitrite forms any more than 15% of the current ambient nitrate/nitrite concentration of Lake Ontario or 24% of that of Lake Erie, then these lakes are out of compliance on the basis of the CCREM nitrite guideline. Nitrite concentrations in the Great Lakes can be expected to increase with decreasing oxygen concentrations. Wetzel et al. (1975) found that nitrite formed a small percentage (e.g. 1-3%) of the nitrate/nitrite content of Wintergreen Lake, Michigan throughout the year.

C. Ammonia

A Provincial Water Quality Objective (PWQO) of 0.02mg/L has been established for un-ionized ammonia. The un-ionized form has
been found to be the toxic ammonia component. The ammonia PWQO is temperature and pH dependent.

The toxicity of ammonia is well understood and documented (MOE 1979). There is no clear distinction between the acute and chronic toxicity ranges for ammonia with fish.

**CONCLUSIONS**

To assess the impact of nitrogen to aquatic ecosystems, a nitrogen budget should be developed. Once established, the benefits of various control options could be assessed for the Great Lakes basin. Without the establishment of a nitrogen budget, the significance of reductions from any one source cannot be accurately estimated.

Research must be undertaken to develop a fuller understanding of the aquatic ecological effects of inorganic nitrogen. Currently, there is a lack of information on the acute and chronic effects of nitrates and nitrites to fish and invertebrates in freshwater.

Abatement measures must be undertaken to reduce the constant rate of increase of nitrogen levels in the lower Great Lakes. Significant nonpoint sources must be identified and appropriate action taken to curb pollution from this source.

**REFERENCES**

Anderson, B.G. 1944. The toxicity thresholds of various substances found in industrial wastes as determined by the use of *Daphnia magna*. Sewage Works Journal 16(6):1156-1165.


Dowden, B.F. and H.J. Bennett. 1965. Toxicity of selected chemicals to certain animals. Journal of the Water Pollution Control Federation 37(9):1308-1316.


Table 1. Aquatic concentrations of nitrate and resulting aquatic effects.

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Table 2. Aquatic concentrations of nitrite and resulting aquatic effects.

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WORKSHOP PROGRAM
AGRICULTURAL NITRATES AND IMPACT ON WATER QUALITY IN ONTARIO

Sponsored by the
Ontario Water Management Research and Services Committee
Lamplighter Inn
London, Ontario
March 24-25, 1992

Tuesday, March 24, 1992

8:00 - 8:30  
Registration

10:00 - 10:30  
"Sales and Trends of Nitrate Fertilizers in Ontario"  Mr. T. Sawyer, The Fertilizer Institute of Ontario

8:30 - 9:30  
WORKSHOP OPENING - Welcome and Introduction  
- Dr. Lloyd A. Logan, MOE and Mr. Ralph Clayton, OMAF

10:30 - 10:50  
COFFEE BREAK

10:50 - 11:20  
"The Role of Fertilizer in Sustainable Agriculture with Influence on Water Quality"  Mr. H. Neutens, Kent County Fertilizer Limited

9:00 - 9:30  
"Nitrate Movement to Groundwater as Influenced by Soil Management on Coarse Textured Soils"  Dr. D. Burton and E. Beauchamp, Land Resources Science, University of Guelph

11:20 - 11:40  
"Performance of Riesling Grapes and the Leaching of Nitrate on a Clay Loam Soil"  Dr. R.A. Cline, Horticultural Research Institute, OMAF

9:30 - 10:00  
"Nitrate Losses in Tile Water from Conventional and Conservation Tillage Plots"  Dr. D. McKenny, C. Drury and W. Findlay, Department of Chemistry & Biochemistry, University of Windsor and Harrow Research Station, Agriculture Canada

11:40 - 12:00  
"The Land Application of Liquid Swine Manure and its Effect on Tile Drain Water Quality"  Ms. M.E. Foran, Ausable Bayfield Conservation Authority

12:00 - 13:30  
LUNCHEON
TECHNICAL SESSION 2

MODERATOR: Mr. Tim Lotimer, Lotowater Ltd.

13:30 - 14:00

"Geochemical and Physical Processes Relevant to Agricultural Nitrate Contamination of Groundwater in Southern Ontario" Ms. Cathy Ryan and R. Gillham, Waterloo Centre for Groundwater Research, University of Waterloo

Wednesday, March 25, 1992

8:00 - 8:30

Registration

TECHNICAL SESSION 3

MODERATOR: Dr. John O'Sullivan, OMAF

14:00 - 14:30


8:30 - 9:00

"Nitrogen Management - A Budgetary Approach" Dr. M. Goss, Department of Land Resources Science, University of Guelph

14:30 - 15:00

"Nitrate in Agricultural Drainage From a Mixed Farm Operation" Dr. N. Patni, Centre for Food & Animal Research, Central Experimental Farm, Agriculture Canada, Ottawa

9:00 - 9:30

"Impacts of Nitrate on Groundwater Learning from Experiences in the United Kingdom" Dr. K. Howard, Department of Geology, Scarborough Campus, University of Toronto

15:00 - 15:30

COFFEE BREAK

15:30 - 16:00

"Trends of Total Nitrate Concentration in Ontario Streams" Mr. B. Whitehead and L. Logan, Water Resources Branch, MOE

16:00 - 16:30

"Evaluation of an Integrated Soil, Crop and Water Management System on Nitrate Loss" Dr. C.S. Tan, C. Drury, M. Soultani, T. Oloya, J. Gaynor and T. Welacky, Harrow Research Station, Agriculture Canada

10:00 - 10:30

COFFEE BREAK

10:30 - 11:00

"Modelling Nitrate Movement for Management" H.R. Whiteley, W.N. Stammer and J. Sakupwanya, School of Engineering, University of Guelph

18:00 - 20:00

BANQUET
11:00 - 11:30
"Managing Nitrogen in the Environment" **Mr. J. Schleihauf**, Plant Industry Branch, OMAF

11:30 - 12:00
"Environmental Significance of Nitrogen to Aquatic Life" **Mr. N. Bazinet**, Water Resources Branch, MOE

12:00 - 13:00
LUNCHEON

1:00 - 3:00
WORKSHOP SESSIONS

**SESSION 1**  *Education & Incentives*
**MODERATOR:** Peter Mason/Kevin Laidley

**SESSION 2:**  *Users & Suppliers*
**MODERATOR:** Frank Letourneau / Keith Wires

**SESSION 3:**  *Management Directions*
**MODERATOR:** Dr. Ramesh Rudra/Jim Myslik

3:00 - 3:45
Workshop Summary

3:45
Closure
WORKSHOP SUMMARY
AGRICULTURAL NITRATES AND IMPACT ON WATER QUALITY IN ONTARIO

Sponsored by the
Ontario Water Management Research and Services Committee

Lamplighter Inn
London, Ontario
March 24-25, 1992

Attendance:

From the 75 pre-registrants 68 people were in attendance over the two days of proceedings. They represented various agencies; including universities, Agricultural Canada, provincial ministries, consulting firms, industries, farm organizations, agricultural colleges, research stations, agricultural organizations, institutes and farmers.

Purpose:

The purpose was to explore current concerns on nitrate in agriculture, its influence on water quality conditions; and ongoing measures and studies to reduce environmental impacts.

Technical Presentations:

Seventeen papers were committed to the workshop; from which 16 speakers arrived for the presentation. The abstracts of papers were supplied through an Abstract Proceedings.

The papers I presented addressed three theme areas; the first theme related to processes and fate and transport of nitrates, here, Drs. Burton and Beauchamp looked at detailed processes of nitrates, the influence of soil management on nitrate was examined further by Drs. McKenney, Drury and Findlay. They relate nitrate losses in tile drainage from conservation and tillage plots.

Some reasons for trend increase in nitrates in Ontario was outlined by Mr. Sawyer as being due to increase in crop acreage used by farmers; implying that as a depended cause, which could be justified by the nitrogen sales/acreage and trends of nitrates in Ontario. Mr. Neutens, on the other hand, looked at this problem from a different perspective; that is, viewing the broader benefit derived. This was viewed from a sustainable aspects looking at the integration of principles of economic crop production with concerns of environmental protection.

A more direct field investigation on nitrate processes was given by Dr. Cline who tried to value nitrate leaching from clay soils under grapes production, under fertilizer application; while Ms. Foran demonstrated typical manure leaching through tile drains, emphasizing the greater concerns for leaching of other parameters rather than evidence of nitrates.

The other theme dealt with the problem of nitrate and how the processes are interfaced with surface to groundwater systems. Under this theme Ms. Ryan and Dr. Gilliam, described the physical processes of nitrate relevant to denitrification/carbon influences in a southern Ontario case study. In terms of encroachment of urban communities into rural areas they showed how agricultural practices may affect the quality of groundwater, as they demonstrated by a nitrate management strategy in the Paris, Ontario area.
Dr. Patni described and tried to explain the nitrate losses delay time observed in agricultural drainage from mixed farms. Some causative factors were inherent in the nitrate process, yet remain unexplained, which may be responsible for the delay of movement through the system.

On a wider scale the end point of nitrate movements through the system by way of tributary transport were assessed by Mr. Whitehead and Dr. Logan. Status of total nitrates in A more structured controlled experimentation quantified the losses through the system.

The final theme dealt with management and policy issues. Here Dr. Gross described a nitrogen management budgetary approach viewing the practical measures for budgeting nitrogen input and output in the soil-water systems. Looking from a European experience, Dr. Howard presented the results of an investigation on impacts of nitrates in the United Kingdom. Of interest was the chronology and time delay of nitrate associated materials in overburden and the carryover impacts they may be generating over time.

From a management decision making perspective, Dr. Whiteley looked at the modelling of nitrate movement and how this could best be used in management, viewed from a subjective judgement. Mr. Schleihauf viewed the management of nitrate from a plant industry perspective by promoting the development and use of a Nitrogen-Soil Test. Here the benefit to the farmers have been emphasized.

Ontario streams and trend over time were presented. Detailed prototype of nitrate losses/movement under integrated soil and crop management in a controlled condition was shown by Dr. Tan et al. Here, a well structured controlled experimentation quantified losses through the system. On the policy question, Mr. Bazinet examined the significance of nitrogen to aquatic life. In this case, he examined the controversial ideas regarding the development of water quality objectives and guidelines for nitrate. Emphasis was placed on different effect levels that may have to be addressed when considering nitrate occurrence in drinking water in comparison to presence in the surface water surroundings.

**Workshop Discussion Sessions:**

1. Education and Incentives  
2. Users and suppliers and;  
3. Management Directions

The following are brief conclusions derived from the discussions:

**Education and Incentives**

The conclusions were categorized by:

a) A recognition that there is a need for more research, with emphasis towards:
   - Nitrogen-tests for other crops  
   - Manure application and
   - better management practices, e.g. rates, methods, examining various management techniques by site conditions

b) Promotion through public media:
   - Existing extension mechanisms
   - Public discussion meeting
   - Emphasis on the use of Nitrogen-Soil Test

c) Better utilization of research findings in advancement in Agricultural Extension Programs as they become available; for example:
- Nitrogen application rates
- Manure management
- Feasibility of other research innovations in managing nitrates

d) Financial/Regulatory Incentives:

- There was some argument that this may not be required.

**Users and Suppliers**

There were a number of general conclusions:

- need to identify travel time and origin of materials
- need for better groundwater surveillance in Ontario
- need for more qualified hydrologist to tackle emerging problems
- promoting the advancement of a Centre of Excellence, e.g. Centres in Waterloo and Guelph
- develop and identify groundwater quality versus land management systems need for more research on nitrogen cycle
- promote conservation plans for agricultural sectors
- promote soil and chemical water analyses
- need for technology information transfer
- need to involve Conservation Authorities in management solutions
- need to improve application methods, timing, placement, soil responses

**Management Directions**

A number of questions have been raised on which various discussions were generated.

The questions posed were:

1. when evaluating the effectiveness of nitrate management systems, the drinking water criteria of <10mg/L of total nitrate is used; (a) can this be attained?, also, (b) is it a realistic criteria for ground and surface waters?

- the basic response to (a) is yes, but to achieve this criteria may be some cost associated; to (b) are that it should be used as a guideline or an objective to target for; also, some health implications might have been documented for levels at or above 10mg/L; criteria development should be tied in with education and implementation of BMPS.

2. various monitoring programs are in place or completed. Is monitoring pointing a "finger" justly or is it too early to tell?

some responses are:

- surface monitoring is at a point where areas of concerns can be identified on a broad scale; however, more research and detailed study will be required at those areas to explain the fate and pathways of the nitrate. We need supplementing groundwater and atmospheric monitoring to expand our interpretation on the concerns areas. other sources should also be investigated e.g. urban runoff, septic, industrial, etc.

- shallow groundwater monitoring, especially under known agricultural practices over the long-term is limited. Information on deep groundwater monitoring may be inadequate.
- Surface water monitoring carried out by MOEE is now undergoing constraints which may influence future monitoring activities.

- A new thrust on groundwater management strategy is in focus in MOEE.

3. Nitrogen management systems are being studied and assessed at various spatial scales; i.e. at watershed, aquifer, farm field, etc. (a) Which spatial levels are progressing adequately? (b) Which levels need more attention?

- The responses are (a) Research results tend to justify that field scales appeared to be progressing, but adequacy may be hard to conclude at this stage; some efforts seemed to be ongoing on a farm scale; however, efforts on an aquifer or watershed scale appeared to be missing; (b) Aquifer and watershed scale appeared to need more attention; nonetheless, the farm and field scales definitely need enhancement.

4. If nitrogen sources on a farm are categorized as a "chemical" or "organic" (a) Which is more critical? (b) Should research of the two sources be balanced or not?

- The responses are (a) Organic appeared to be more critical; if one uses only chemical nitrogen, then what would be the need for organic. If this tends to be the case, then the organic nitrogen would be treated as a waste rather than a resource; which it appears to be at this point.
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