IMPACTS OF LIVESTOCK MANURE ON WATER QUALITY IN ONTARIO

An Appraisal of Current Knowledge

Prepared for
Ontario Ministry of the Environment
by
The Centre for Soil and Water Conservation
University of Guelph

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EXECUTIVE SUMMARY

There is a growing concern for the impact of livestock manure management on nitrate ($N0_3$) concentrations in groundwater and for phosphorus (P), ammonia (NH_3), organic materials and microbes in surface water. This report provides: (1) a detailed assessment of our current understanding of the impact of livestock manure management on water quality with particular reference to Ontario; (2) a review of current guidelines on manure management in Ontario with recommended changes where justified by current understanding; and (3) a priorized list of research needs specific to Ontario.

As agricultural specialization has increased in Ontario there has been a separation of livestock and crop production. Disposal rather than efficient utilization of livestock manure has become common. This has increase the potential for contamination of surface and groundwater. Pollutants of major concern are nitrate nitrogen ($N0_3$-N) and phosphorus.

Because of its mobility in soil, $N0_3$-N is of greatest concern in groundwater where concentrations greater than the acceptable 10 mg $N0_3$-N L^{-1} are frequently found in association with intensive manure use, sometimes in combination with fertilizer nitrogen. Phosphorus, being strongly adsorbed by soil, does not leach to groundwater readily but is of major concern in overland flow to surface water. The contribution of ammonium, organic carbon and bacteria to surface water, while potentially damaging, is of lesser concern provided the timing of manure application and incorporation are properly managed.

The primary concern with land application of manure is to avoid over application, which may lead to $N0_3$ contamination of the groundwater or excessive phosphorus in overland flow. Because of the highly variable water and nutrient contents of manures, recommended manure application rates must be based primarily on the amount of ammoniacal and total nitrogen applied. The following uncertainties remain:

1. There is no convenient way for farmers~to determine the N content of manure at the time of application. Because of variation in manure storage and handling, manure N content can vary greatly from expected mean values. The return time from laboratory analyses may result in significant changes to the manure characteristics between sampling and application.

2. No reliable N soil test is available at the present time. The best N recommendations available are based on soil and crop type and past crop and fertilizer history. Whether these recommended rates are compatible with groundwater protection is unknown.

3. Manure N availability is dependent on a number of dynamic factors, and is not well established for varying manure, soil, crop and climatic types.

4. Seasonal variability, which alters crop N requirements, may also alter the percentage of manure N available to the crop.

Producers frequently apply manure and fertilizer N at rates exceeding recommended rates to insure against N deficiencies due to poor N requirement estimates, poor manure N content estimates, poor manure N availability estimates, and/or an a typical growing season.

Current guidelines for manure application in the Agricultural Code of Practice are based on the nitrogen required for corn production. The assumption implicit is that most profitable rates of N from a crop production standpoint are also environmentally safe, ensuring that contributions of $N0_3$-N to groundwater are within acceptable limits. This assumption has not been adequately tested in Ontario.

The maximum acceptable rate of manure and fertilizer N application should be based primarily on the contribution of $N0_3$-N to groundwater, not on the requirement for the crop. The following is proposed as a criterion for the maximum acceptable rate of nitrogen application as manure and/or fertilizer.

THE MAXIMUM ACCEPTABLE RATE OF NITROGEN APPLICATION IS THAT WHICH WILL ENSURE THAT THE ANNUAL VOLUME-AVERAGED $N0_3$-N CONCENTRATION IN WATER LEAVING THE ROOT ZONE DOES NOT EXCEED A SET CONCENTRATION.

If this criterion is accepted, the appropriate upper limit for $N0_3$-N must be established and a system must be developed to predict, for representative soil/crop/management systems, the amount of manure and/or fertilizer N that can be applied without exceeding the criterion. An approach for such a system is proposed although present information and understanding do not permit application of the system at this time.

Although major changes in guidelines cannot be recommended at this time, the following less drastic changes are suggested.

1) Greater emphasis should be placed on total (manure plus fertilizer) N and P application.

2) Distinction should be made between fall- and spring-applied manure.

3) Differences based on soil texture could be defined more precisely.

4) Attention should be given in the very near future to completing revisions of the "Guide to Rural Land Use and Farm Practice" and reaffirming the need for the application of the guidelines related to protection of water resources.

A series of research projects is recommended with the following order of priority.

1) Establishment of the relation between environmentally safe and most profitable rates of N application to crop land.

2) Establishment of a system for predicting environmentally safe rates of manure. Subprojects would include:
   - N transformations and losses during storage and handling
   - prediction of denitrification of manure N after incorporation into the soil
   - mineralization/immobilization of manure N after incorporation into the soil
   - effect of hydrological factors on the transport of nitrogen to groundwater
   - validation of predictive models for nitrogen transport

3) Macropore-transport of dissolved and particulate pollutants.

4) Methods for manure application in no-till systems.

5) Potential leaching of P to groundwater from long-term intensive manure applications.

1. INTRODUCTION

Livestock production is a major agricultural activity in Ontario. The livestock population in Ontario in 1987 is indicated in Table 1. This livestock population would produce 3×10^7 tonnes of excrement containing, based on average analyses, 1.3×10^5, 0.5×10^5 and 1.2×10^5 tonnes of N, P and K respectively each year. These values can be compared to 1.3×10^5 , 0.5×10^5 and 1×10^5 tonnes of N, P and K applied as inorganic fertilizer (person commun., T. Sawyer T.F.I.0). At today's costs, the fertilizer replacement value of these nutrients is $158 million.

Table 1: Livestock Population in Ontario on July 1, 1987.

Dairy		Swine	
Cows	465,000	Sows	378,000
Heifers	240,000	Feeder Pigs	2,982,000
Beef		Sheep	
Cows	345,000	Mature	116,000
Heifers & Steers	649,000	Lambs	88,000
Bulls	26,000	Poultry	
Calves	540,000	Hens	8,603,108
		Pullets	4,500,268
Horses	74,961	Broilers	18,904,799
		Turkeys	3,108,852

Source. Agricultural Statistics for Ontario, 1987. Publ. 20, Ontario Ministry of Agric. and Food, Toronto.

Livestock wastes have traditionally been returned to the land on the farm on which they were produced and on which the major portion of the livestock feed was produced. This reasonably closed system minimized the detrimental effects on water quality, although improper management such as winter spreading of manure created occasional problems.

As agricultural specialization increased, there has been a separation of livestock and crop production. To-day, most of the poultry and swine production and much of the beef production are concentrated on a minimal land base. Feed is imported from considerable distances. Although dairy production remains largely a land-based activity, many dairy operations produce asignificant portion of the feed on rented land some distance from the dairy operation. The result of this specialization has been the application of the livestock wastes to smaller and smaller land areas. Although the fertilizer replacement value of manure is quite significant, the cost of transport beyond a short distance outweighs the value. At this point, disposal of the manure rather than utilization becomes the objective.

Once disposal has become the objective, there is little concern for the effect of rate, timing or method of application on the availability of the nutrients or their use by the crop. This leads to the nutrient content of applied manure being ignored in estimating the fertilizer requirement of the crop. This is particularly true of nitrogen which is much more transient in the soil than are phosphorus and potassium. There is a reliable soil test for phosphorus and potassium which will reveal the increase in availability of these two nutrients in the soil as a result of manure application. There is not a reliable test for nitrogen. In addition, nitrogen in the nitrate form may be leached from the soil or lost as nitrogen gas to the atmosphere through denitrification while nitrogen in the ammonium form may be volatilized. Hence the availability of manure nitrogen to the crop is difficult to predict. Many farmers are not prepared to rely on the manure nitrogen so apply inorganic nitrogen in addition.

A major potential impact of excessive application of manure is increased concentration of $N0_3$-N in groundwater. There are numerous isolated measurements of elevated groundwater $N0_3$-N concentrations in Ontario that are attributable to application of nitrogen, either as manure alone or in combination with fertilizer N. In one instance, groundwater flowing beneath a corn field with a past history of over-application of manure and inorganic nitrogen fertilizer had $N0_3$-N concentration consistently above 30 mg L^{-1} and occasionally above 50 mg L^{-1} during a 3year monitoring period. The groundwater beneath an adjacent field upstream (in terms of groundwater flow) never exceeded 20 mg L^{-1} (Miller et al. 1985).

The concern for $N0_3$ in water arises primarily from the threat to infant health. Nitrate may be converted to nitrite ($N0_2$) in the stomach of infants by bacteria. The $N0_2$ oxidizes blood haemoglobin to form methemoglobin which is incapable of reversibly binding oxygen. This leads to methemoglobinemia, commonly known as "infant cyanosis" or "blue baby". Infants under 3 months of age are particularly sensitive for a number of reasons: (1) their low gastric acidity permits growth of the $N0_3$-reducing bacteria; (2) they have a high fluid intake relative to body weight; (3) fetal haemoglobin, the predominant form up to 3 months is more rapidly oxidized by $N0_2$ than is adult haemoglobin; (4) Infants have low levels of enzymes which convert methemoglobin back to its original state (Fraser and Chilvers, 1981).

Based primarily on the threat to infant health, a safe limit of 10 mg $N0_3$-N L^{-1} (OMOE, 1978) has been established for drinking water. Although there have been suggestions that this level is overly conservative from the standpoint of methemoglobinemia (Black 1989), it is recognized that the objective must be to keep $N0_3$ concentration in groundwater as low as is practical.

In addition to their relation to methemoglobinemia, nitrates and nitrites have been implicated in chronic diseases, particularly cancer, although the evidence is less direct than with methemoglobinemia. Nitrite interacts with amine-containing organic chemicals to form nitrosamines which are known to be very active carcinogens in animals although there is no direct evidence linking them to human cancer (Fraser et al. 1980). Many agricultural pesticides have amine structures which make them potential agents for formation of nitrosamines. Black (1989) has presented arguments suggesting that the potential for significant nitrosamine production as a result of $N0_3$ in groundwater is low.

A third potential health effect from $N0_3$ has arisen from a study in Australia in which malformations of the central nervous system and musculoskeletal system were found to be significantly higher in infants whose mothers drank from wells or a lake as compared to rainwater during pregnancy (Dorsch et al. 1984). The risk was related to $N0_3$ concentration of the drinking water. Compared with women who consumed water with $N0_3$ concentrations less than 5 mg L^{-1}, women consuming water containing 5-10 mg $N0_3$ (1-3 mg $N0_3$-N L^{-1}) and more than 15 mg L^{-1} (>3 mg $N0_3$-N L^{-1}) experienced a nearly threefold and a fourfold increase in risk respectively. These levels are below the established safe level of 10 mg NO 3-N L^{-1}. The authors caution that it is premature to interpret their results exclusively in terms of water nitrate exposure but suggest that their data lend weight to the to the possibility of a real association. Black (1989) has concluded that there is little evidence to support such an association.

Nitrates may also be a concern for health of animals. Swine are most susceptible followed in order by cattle, sheep and horses (Willoughby, 1971). Concentrations of 20 mg $N0_3$-N L^{-1} in water are, however, considered to be safe for farm livestock. Deaths from $N0_3$ or NO_2 poisoning have been associated with concentrations in excess of 200 mg $N0_3$-N L^{-1}. Although livestock frequently depend on localized surface water as a drinking source, humans rarely do. This fact, in association with the higher tolerance of livestock places the greatest emphasis on $N0_3$ contamination of groundwater.

The $N0_3$ contamination problem is perhaps most severe in Britain and Western Europe. A program has recently been established in Britain to restrict farm use of nitrogen fertilizers and control other farming practices to reduce pollution in drinking water. The program will establish "nitrate sensitive areas" (NSA's) where $N0_3$ concentrations in water exceed or are at risk of exceeding the safe limit. Actions in the program include: (1) intensive advisory services to induce farmers to follow good practices such as not applying fertilizer or manure in the fall and planting cover crops during winter to minimize leaching; (2) replacement of some existing cultivated land with unfertilized grassland; and (3) provisions to restrict livestock densities and/or manure applications. The European Community has established a draft directive calling for establishment of NSA's which is even more restrictive than that established in Britain.

In the U.S.A., several states have enacted legislation related to groundwater quality. In Nebraska, in 1986, legislative Bill 894 was passed to deal with the irrigation-based nitrate problem. It empowers the Department of Environmental Control to designate special groundwater protection areas and to prepare a management plan to curtail pollution (Morandi, 1987). Iowa's 1987 Groundwater Protection Act states that all soil test results must carry warnings of hazards to groundwater from overapplication of fertilizers (Johnson 1987).

In Minnesota, agriculturally sensitive areas to groundwater contamination have been identified and intensive educational programs developed. Randall (1988) stated "Regulations on the usage of N fertilizers are not just idle talk but are a reality. We will need to develop precise research projects, with clear interpretation of data coupled with effective educational programs to aid in regulation development. That will be the challenge of the next decade".

Although $N0_3$-N in surface water is not a major concern, concentrations of NH, may be. Fish are highly sensitive to free NH_3 in water. Concentrations of as low as 0.02 mg L^{-1} of un-ionized ammonia (NH_3) have been found to be toxic to rainbow trout (Thurston et al. 1984). Chronic toxicity of fathead minnows did not occur until concentrations were about 0.15 mg L^{-1} (Ruston et al. 1986). Acute toxicities occurred with rainbow trout at a concentration of 0.16 mg L^{-1} (Thurston and Russo 1983) and for fathead minnows at a concentration of 0.75 mg L^{-1} (Thurston et al. 1983). The concentration of un-ionized ammonia is dependent on the concentration of total ammonia (NH_4^+ + NH_3) and the pH of the water. The proportion in the un-ionized form (NH_3) increases rapidly as pH increases above 7.0. At a pH of 7.5, 2-3% of the total ammonia is in the un-ionized form. At this pH, a total ammonia concentration of about 1. 0 mg L^{-1} would give rise to an unionized concentration of 0.02 mg L^{-1}, the value found to cause chronic symptoms in rainbow trout. A concentration of over 5 mg L^{-1} would be required to cause acute toxicity of rainbow trout. Although concentrations of 1.0 mg L^{-1} may occur in runoff from fields with manure on the surface, concentrations of 5.0 mg L^{-1} are unlikely to occur except with direct spills of manure into surface water. Spires and Miller (1978) sampled runoff during spring events from 10 fields to which manure had been applied during the winter. The ammonium N concentration in only one sample exceeded 5 mg L^{-1}. The mean of the remaining 9 samples was 1.8 mg L^{-1}. These concentrations would be greatly reduced by dilution when the runoff entered the stream.

Improper management of livestock manure may also impact water quality through excessive phosphorus inputs. Although phosphorus is not considered a threat to human or animal health, it is a key determinant of algal growth in surface water. Excess algal growth reduces the oxygen concentration causing the water to be unsuitable for many fish species and other aquatic life. The algal growth also fouls beaches making them unsuitable for recreation.

The impact of livestock waste on phosphorus in water supplies comes about through processes quite distinct from nitrogen. Phosphorus is strongly adsorbed on most soils; hence leaching is of much less concern than for $N0_3$-N. The major concern with phosphorus is from surface runoff from unincorporated manure or erosion of topsoil greatly enriched with phosphorus from manure application. Phosphorus from manure may also reach surface water through tile drainage systems. Although extremely high rates of manure application would be required to cause leaching of significant amounts of P to tile drains through pores of the soil matrix, many soils in Ontario develop cracks on drying which can provide direct pathways for manure transport from the soil surface into buried drains.

Direct input of organic material and microbial bodies to water through mismanagement of manure is also a significant concern. Organic material increases the microbial activity hence depleting the oxygen concentration in water as with excessive algal growth. Direct input of microbial bodies may introduce pathogenic organisms. Organic matter and microbes in manure may reach surface water in manners similar to phosphorus.

The Agricultural Code of Practice for Ontario was designed, in part, to reduce the impact of livestock wastes on water quality to acceptable levels. The Code of Practice established minimum land areas for livestock operations based on the nitrogen in the manure. Manure-N application rates twice that required by the crop are permitted. This appears to have been based on the premise that only half of the nitrogen would be available to the crop in the year of application. While this may be an acceptable limit for a single year, continued application at this rate may not be acceptable. The Code of Practice also specified requirements for manure storage and prohibited winter spreading of manure where there was a potential for runoff. Incorporation of manure shortly after application was also recommended.

In spite of the acknowledged potential for impact of livestock waste on water quality, only limited research has been conducted specifically on this problem in Ontario. Those. projects that have been undertaken have been quite limited in scope with little possibility of extrapolation to more general levels.

With the major increase in concern for NO_3-N in groundwater, and for phosphorus , ammonia, organic materials and microbes in surface water, it is imperative that increased effort be devoted to assessing and reducing the impacts of livestock wastes.

To help direct this effort this report provides: (1) a detailed assessment of our current understanding of the impact of livestock manure management on water quality with particular reference to Ontario; (2) a review of current guidelines on manure management in Ontario with recommended changes where justified by current understanding; and (3) a priorized list of research needs specific to Ontario.

2.0 SCOPE AND FOCUS OF STUDY

Because of the very broad nature of the impact of livestock waste management on water quality, it was important to define the scope and focus of the study at an early stage.

The study has focussed on an assessment of understanding of the issues rather than an assessment of current impacts in Ontario. The actual impact in a region is a function of the livestock-to-land ratio, soil type and topography, and manure management. To identify specific areas where serious impacts are currently likely occurring would require an assessment of these factors on almost a farm to farm basis. This was clearly beyond the scope of the study. Rather, we have attempted to answer the question "Is our understanding of the processes and interrelationships involved in pollution from animal waste management adequate to allow us to predict with confidence the impact that would occur under a given set of circumstances?" And if the answer to that question is "No," what are the information gaps that must be filled to allow such prediction?

It was also necessary to establish the pollutants of concern. A review by the National Research Council of Canada (1983) on manure in the environment recognized a number of potential contaminants including: nitrogen, phosphorus, and potassium; gaseous irritants and asphyxiants; micronutrients and heavy metals; pesticides, organopollutants, drugs and feed additives; bacterial, fungal, viral, and parasitic human and livestock pathogens; and phytopathogens and weeds. Of these, nitrogen, phosphorus, bacteria and organic matter are generally recognised as the major water pollution concerns from manure. Manure gases are important local atmospheric pollutants, but contribute little to the atmospheric deposition of contaminants to water bodies. Transmission of non-bacterial pathogens via manure contamination of water is important; however bacterial contamination may be considered indicative of potential contamination by other pathogens as well. Micronutrients and synthetic organics typically are present in very low concentrations, and become strongly bound to the soil or may be degraded in manure (Krider, 1987). An exception to this may be copper in swine manure. Copper is added to swine feed in relatively high concentrations and hence is found in much higher concentrations in swine manure than in manure from other livestock. There are no known instances of excess copper in water as a result of swine manure management but concern has been expressed, particularly in copper content of feed produced for sheep which are very sensitive to copper.

This report will be concerned with contributions of nitrogen, phosphorus, organic material and microbial bodies to water from livestock waste. We recognize that the contribution of nitrogen and phosphorus from manure cannot be separated from that of inorganic fertilizers. Although many of the concepts discussed will apply to both sources, we have restricted our consideration to livestock waste, as requested by our client.

A number of reviews discuss management conditions which may lead to water pollution from manure (EPA, 1971; Loehr, 1974;Coote and Zwerman, 1975; OMAF, 1976, 1985; SCS, 1988). The following list summarizes these.

1. STORAGE
    1. inadequate capacity.
    2. permeable floor or liner.
2. FEEDLOT
    1. inadequate capacity or runoff control
    2. permeable floor.
3. LAND APPLICATION
    1. over application.
    2. improper timing of application.
    3. inadequate degree of incorporation
4. PASTURE MANAGEMENT
    1. livestock access to water body.
    2. over grazing, erosion from areas where cattle aggregate.

Each of these conditions will be discussed in subsequent sections of this report. ,

The study has also evaluated approaches to assessment of the pollution potential in a given area. Krider (1987) listed three approaches:

1. Presumptive: Collate all available information on animal concentrations, soil, site, geology, management systems and rank areas accordingly.

2. Modelling: Identify physical processes and interrelationships, convert these to mathematical models. This provides a way to deal with a large number of variables in continuous fashion, but such models are not yet well enough developed to be of much predictive use. Research is required to improve the understanding of the processes involved and to incorporate them into a dynamic model.

3. Monitoring: The measurement of pollution levels at select locations and time intervals over a period of time can be used to identify and quantify specific pollution sources and events. In addition, it provides input data for the above approaches, provided related parameters are monitored as well. Monitoring all potential pollution sources is prohibitively costly.

3.0 LITERATURE REVIEW

3.1 POLLUTION POTENTIAL

Several reviews of the potential for water pollution from livestock manure in Ontario have been published in the past.

1. Black (1967) assessed the magnitude of the problem to 1966, considering animal populations and spatial concentration, quantities and properties of manures, and the emergence of the trend toward treating manure as a waste disposal problem, rather than a fertilizer source.

2. A review by Webber et al. (1968) identified nitrate contamination of the groundwater as the greatest environmental concern with manure. Maximum recommended land application rates of manure were equal to twice the recommended nitrogen fertilizer application rate. While this recommendation was little more than an educated guess, it remains essentially unaltered in recent documents (OMAF et al, 1976, and revision proposals, 1985).

3. Townsend et al. (1970) reviewed the pollution potential of the Ontario beeflot industry. Their major concerns were runoff from poorly designed feedlots, and winter disposal of manure.

4. A 1971 joint Canada-US report on agricultural pollution of the Great Lakes basin recognised the potential for biological oxygen demand, suspended solids, nitrates, eutrophication and pathogen contamination from manure sources, but had no evidence of pollution (EPA, 1971).

In the 1970's the Pollution from Land Use Activities Reference Group (PLUARG) of the International Joint Commission (IJC) carried out extensive studies of agricultural pollution of the Great Lakes and Great Lakes tributaries, publishing over 100 reports (PLUARG, 1979). Several of these specifically addressed manure pollution potential (Bangay, 1976; Beak Consultants Ltd, 1977; Patni and Hore, 1978; Coote and Hore, 1978; Robinson and Draper, 1978). Though the primary concern to emerge from the PLUARG work was P loading of the Great Lakes, significant pollution of streams from manure sources also was observed.

A study of Thames River Basin Water Management (OME and OMNR, 1975) included extensive surface and groundwater monitoring . However manure pollution potential estimates were based primarily on livestock population data. Recommendations included restriction of livestock access to streams, and increased environmental surveillance and enforcement to control runoff pollution from feedlots and storage sites.

3.2 MANURE POLLUTION $N0_3$PROCESSES

3.2.1 Detachment and transport

In general, manure and pollutants from manure are carried to water by water. Transport by air can be considered insignificant in most cases. Wind erosion is an uncommon transport mechanism because of the water content and adhesive characteristics of manure. While volatilization of ammonia is a major factor in manure nitrogen dynamics, and ammoniacal-N deposition to water surfaces from the atmosphere may be a significant pollution process, the fraction of atmospheric ammoniacal-N originating from manure is minimal in Ontario. However, the amount of ammonia volatilization is still important for estimating N leaching potentials because it can reduce the amount of N available for leaching. Direct inputs of manure to a water body may occur with spills, improper drainage or cattle access to water bodies.

As with soil erosion by water, the primary processes involved are detachment and transport. Detachment is a result of impact from rain and friction forces from flowing water, or of dissolution reactions. These processes break down aggregates,. and may result in puddling and surface sealing. Therefore aggregate strength is an important factor in resistance to detachment. Transport is a function of flow volume, slope, path tortuosity, surface infiltration rate, and subsurface drainage. Pollutants are removed from transport by filtration, sedimentation, surface reactions, and precipitation processes. Detachment, transport and redeposition processes occur continuously with water flow (Dickinson et al. 1987b).

Manure runoff and soil erosion processes differ in the following ways:

1. Manure has a wide range of water contents, occurring in solid and liquid forms. Transport loads from manure piles and feedlots are strongly dependent on prior water content. In the case of a spill or overflow, liquid manure may be transported by its own water.

2. The liquid portion of manure has very high concentrations of soluble nutrients and organics, so that transport in solution form is of primary importance.

3. Manure storage facilities may be protected from the possibility of detachment by covering, and from transport by containment.

4. When runoff passes beyond the edge of the manure containment area, manure detachment processes no longer occur: manure pollutant load is controlled by transport and deposition processes only. The path distance between the storage facility and a water body are of primary importance in determining which pollutants, if any, reach the water body.

Soil erosion and pollutant transport processes have been simulated in a number of mathematical models (ANSWERS, see Beasley et al. 1980; HSPF, see Johanson et al. 1980; GAMES, see Cook et al. 1985), and the models applied to various agronomic and environmental questions (Frevert and Crowder, 1987; Crowder and Young, 1988). The applicability of models to water pollution study has been reviewed by DeCoursey (1985) and in the Ontario context by Dickinson et al. (1987b).

As a field research tool, models may provide a framework for data collection, parameter evaluation, and assessment of factor variability effects. Field research in turn suggests alterations of the models, and therefore of our understanding of the processes involved.

Current models are of limited use for predictive purposes, however. One limitation is the data input requirements of those models which attempt to account for some of the complexity of pollutant transport. Watershed or field scale models require detailed dimensional, slope, aspect, vegetation, soil, and hydrological data for each sublocation of the study area, in addition to meteorological data, and descriptions of the distribution and physical, biological, and chemical characteristics of the pollutant of concern. Reliance on literature values for some of these parameters can lead to gross error (DeCoursey, 1985).

Another important limitation is the number of parameters and factors which are not properly addressed in current models. The USLE, an empirical equation on which the erosion component of models is often based, frequently gives grossly misleading predictions in the Ontario context, in part because of a lack of recognition of the effects of freeze-thaw processes, tillage translocation, snow melt, seasonal variation in erodability, and in-field variation of erosion rates. Major effort is currently being extended in the U.S. to develop process oriented models to predict erosion (e.g. WEPP, EPIC). Process models require calibration of some site specific coefficients. Extensive models such as WEPP, EPIC and HSPF suffer from too many such parameters, each requiring years of data collection to calibrate the model for a given location (DeCoursey, 1985).

The GAMES model, developed for the Ontario context, predicts total watershed sediment and P loads, and the contribution from each landscape subunit. The model requires extensive site data, appropriate seasonal USLE factor values, and calibration of site-specific sediment delivery ratio factors. These requirements limit its use to fundamental research applications only (Cook et al. 1985; Dickinson et al. 1987b).

Several models include a soil through-flow component (ANSWERS; GLEAMS, see Leonard et al. 1987) or depict through-soil transport processes exclusively (NTRM, see Shaffer and Larson, 1987; LEACHM, see Wagenet and Hutson, 1987). None can predict the effects of macropore flow on pollutant transport, nor are they designed for predicting absolute values, but rather for comparative effect studies.

NTRM is an extremely comprehensive collection of interrelated sub-models, aimed at describing nitrogen and nutrient dynamics under various tillage and residue management schemes. The focus of the model is crop yield, the time-frame modelled is the growing season (Shaffer and Larson, 1987). The extensive treatment of residue degradation, and soil organic matter and nitrogen dynamics make this model potentially attractive for manure modeling work. Considering manure as a residue, half-life decomposition rates, C:N ratios, water contents, and application rates would be required for each manure type modelled, in addition to soil, crop, management, precipitation and temperature data. The input requirements, growing season focus, and lack of landscape modelling limit this model's applicability to manure pollution studies. In addition, model parameters have not been validated for the Ontario context. However comparative investigation of manure type and water content effects on harvest-time soil $N0_3$ concentrations may be one suitable application.

The GLEAMS model, based on the CREAMS runoff and erosion model, also simulates pesticide loading of groundwater (Leonard et al. 1987). Application of this model to manure pollution studies would require solubility, degradation half-life, and sorption coefficient values for each pollutant, as well as detailed site, soil, and climate inputs, and model validation for the Ontario context.

3.2.2 Overland transport

Rainfall impact breaks soil or manure particles, releasing them for transport, and causing puddling and sealing of the surface, reducing infiltration. When total rainfall exceeds infiltration and surface detention capacities, overland runoff occurs (Dickinson et al. 1987b).

Manure runoff studies generally involve measuring volumes and concentrations leaving the plot area. Such studies don't describe the amounts which reach the stream system. Studies of pollutant concentrations in streamflow cannot identify the source of the pollutants, or attenuation rates on route. The natural stream pollutant load during a storm event may be quite high, and must be taken into account in stream monitoring studies of manure pollution (Robbins et al. 1971).

Feedlot runoff studies were reviewed by Loehr (1974), and Nye (1982). Unpaved feedlots develop a hard impervious floor which reduces leaching losses but increases runoff. Runoff volumes are linearly related to total precipitation; however prior manure water content is a very important factor in determining manure surface strength and susceptibility to detachment. Runoff is greatest when a storm event follows a period of low intensity rainfall, such that the manure is already water soaked. Highest pollutant concentrations occur in the initial runoff from a given runoff event. Transport by overland runoff leaving the feedlot accounted for 0.1 - 6.6% of the total N and 0.8 - 12.5% of the total P output from the feedlots reviewed.

Land application of manure generally improves soil aggregate strength, reducing erosion losses (Coote and Zwerman, 1975), increasing the soil infiltration rate (Mitchell and Gunther, 1976). However manure spread on a relatively impermeable surface is more susceptible to runoff losses. Spreading manure on frozen land results in overland runoff losses during thaw or thaw/rain events of up to 20% of the N and 12% of the P applied (reviewed by Loehr, 1974). Bhatnagar et al. (1985) observed that the P enrichment in runoff was lower from manured than unmanured plots due to preferential adsorption of P by larger aggregates which were less susceptible to erosion (Bhatnagar and Miller 1985). This factor may partially compensate for the increased phosphorus concentration in the surface soil as a result of manure application.

3.2.3 Through-soil transport

Through-soil pollutant transport differs from transport in overland runoff in the following ways:

1. Greatly increased path tortuosity and greatly reduced flow rates permit the soil to function as a filtration matrix.

2. The reactivity of soil particles, and increased time of contact with the soil matrix increase the importance of precipitation and sorption reactions, and biological uptake or degradation of pollutants.

The factors controlling through-soil pollutant transport are pollutant particle size, solubility, and reactivity, and hydraulic conductivity of the soil. Nitrate leaching from storage facilities and from field-applied manure is the primary through-soil manure pollutant concern. Bacteria, organic particles, and phosphate tend to be removed from solutions flowing through soils.

The importance of macropore flow in soil transport processes is currently a major research focus. Three soil physics sessions at the 1988 annual Soil Science Society of America meetings were dedicated to the topic. The assumption of complete displacement of the soil water, on which most transport models are based, has been shown to be theoretically inadequate because of the importance of macropore flow (Scotter, 1978). In field and lab studies, nitrate, chloride and pesticides have been observed to depths far greater than predicted by miscible flow theories (Kanchanasut et al. 1978; Thomas and Phillips 1979; Keeney et al. 1986). Because of this, phosphate, organic matter and bacteria contamination of tile drain water and groundwater may occur at rates not predicted by soil transport principles.

3.3 NITROGEN

3.3.1 Nitrogen content of manure

Tables of concentrations of nutrients in various manures are available in a number of reports (Webber et al. 1968; OMAF, 1976; Kolenbrander et al. 1981; MacLean et al. 1983). Manure composition varies with animal type, feed type, management, and age. Nutrient contents of liquid dairy manures at application time are commonly as much as 30% higher or lower than published expected values (Phillips, 1983). Overcash et al. (1983) found beef manure N contents of 2-4% of total solids, of which 16-40% was in ammoniacal form. Poultry manure was 4-14% N, of which 49-62% was ammoniacal-N, and swine manure was 6.5% N, with 50% in ammoniacal form. A recent review (Chescheir et al. 1986) of published values for ammoniacal N as a percent of total N, in different manure types, suggested ranges of 0.2 - 2.7% for old, composted beef and dairy manures, 3-13% for dry manures, 37 - 43% for liquid dairy manure, 39-72% for liquid swine manure, and 23-68% for poultry manures.

The water content of manure is especially variable over time, and depends on collection and storage methods. Old composted manure may be 80-95% dry matter by weight, while liquid manures may have total solids contents of 4% or less (Cheschier et al. 1986). Three poultry barns with different manure pit ventilation systems had manure total solids contents which varied from 27-82% over a four-month period. During that time 48-75% of the initial N content was lost to volatilization (Bulley and Lee, 1987).

The N content of manure at incorporation time is difficult to predict because of volatilization losses. The average N content of freshly excreted beef manure is 4.8% of total solids, with 58% of the N from the urine. The urea N is quickly converted to ammoniacal N, and NH_3 volatilization losses begin immediately (Chescheir et al. 1986). Losses continue until soil incorporation, and are accelerated with manure drying (Paul and Beauchamp, 1989). The amount of N lost is a function of temperature, water content, aeration and storage ventilation factors, and time. Data on manure N loss between excretion and soil incorporation, reviewed by Burton et al. (1984) showed high variability in total percentage lost. Estimates of NH_3 volatilization losses vary from near 0 to near 100 %. If land-applied manure is not immediately incorporated, 30% of the remaining N may be volatilized in the first 24 hours (Pain and Thompson, 1988). Nitrogen volatilization losses from three swine barns in Ontario ranged from 5 to 27% of the total manure N, varying greatly between management systems. In general, the greater the exposure during collection and storage, the greater the NH_3 volatilization (Burton and Beauchamp, 1986). During a seven-day period after field application of manure and before incorporation, Beauchamp et al. (1982) found volatilization losses of 24 - 33 % of the initial manure ammoniacal-N in early May studies over four years, at the Elora Research Station. Little ammoniacal-N moved into the soil below, while the manure was on the surface. The method of application of manure (ie .surface vs injection) will determine the amount of N lost through volatilization and hence not available for crop use (Beauchamp, 1983).

Within liquid manure storage, nutrient concentrations vary with depth. Ammonium concentrations are higher in the supernatant, and P is. associated with the particulate material. Nutrient contents may differ significantly between loads when a storage facility is emptied for land application unless agitation is thorough (Burton and Beauchamp, unpublished data, Univ. of Guelph).

3.3.2 Reaction of manure-N in soil.

Manure nitrogen inputs to the soil system differ from inorganic fertilizers in several ways which may be important to the nitrate leaching dynamics:

1. A significant component of manure N is in organic form. This N becomes available only over a period of months or years.

2. Considerable ammoniacal-N loss may occur before incorporation of the manure into the soil.

3. Manure has variable water content, and often is applied in liquid form.

4. Timing of application may differ. While inorganic fertilizers are timed to meet the crop needs, manure application may be timed according to a manure disposal schedule.

5. Manure total and organic N contents at time of application vary significantly even within a management operation. As well, application rate variability is much greater with manure than with inorganic fertilizers.

Recent work suggests much of the organic N component of manure is not readily mineralized, but becomes a part of the resistant organic matter. Beauchamp (1987) studied the residual effects of various manures on subsequent corn yields on a silty loam soil in Ontario. Though 50% of the nitrogen in cattle manure was in organic form, the residual effect one year later was less than the residual effect of urea-N applied at a similar rate. Liquid poultry manure, with 83-95% of the nitrogen in ammoniacal form, had a residual effect similar to that of urea. Little effect was seen from any treatment in subsequent years. In an earlier study, the residual effect of 112 kg liquid poultry manure N ha^{-1} y^{-1} was similar to that of 224 kg inorganic N ha^{-1} y^{-1}. Inorganic nitrogen applied in addition to the manure did not increase the residual N effect on yield (Ketcheson and Beauchamp, 1978).

The organic-N content of manure can be conceptually subdivided into a rapidly mineralizable component, mineralized within months, and a residual portion which behaves similar to the soil residual organic matter (Kolenbrander, 1981; Lindemann et al. 1988; Beauchamp and Paul, 1988). Chescheir et al. (1986) add a third class, the urea N, but this is so rapidly hydrolyzed it can be considered a part of the ammoniacal N pool. In incubation studies, Lindemann et al.

(1988) found about 65% of the total N added to soil from sewage sludge was mineralized, regardless of previous sludge applications. Prior sludge amendment history did increase $N0_3$ leaching losses, however.Incubation studies of 9 manures used to amend two soils (Chescheir et al. 1986) showed an average of 49% of the applied N to be available on sandy soil, and 26% on a silty loam. Swine manure had the most available N (51-66%), dairy manure had 39-54% available, and poultry manure 40-44%. Poultry manure had the most rapid mineralization rate.

Some of the ammoniacal N in manure incorporated into soil may be immobilized, depending on the C:N ratio (Beauchamp and Paul, 1988). In laboratory studies, on clay loam and sandy loam soils, liquid pig manure additions resulted in an initial period of increased N immobilization. The rate of immobilization and the length of the period were functions of temperature and water content; however the total nitrogen immobilized was consistently about 40% of the NH_4-N content of the manure. No period of increased mineralization followed (Flowers and Arnold, 1983). Manure N available to plants is approximately equal to the sum of the ammoniacal N (minus net immobilization) and the rapidly mineralizable organic N, a small portion of which may not be available until the second season. Bitzer and Sims (1988) estimated available manure N to equal 80% of the mineral N plus 60% of the organic, in field and lab studies of a sandy loam soil. Beauchamp (1986) showed that the response of corn was mainly to the ammoniacal-N present in three manures.

Over the long term, however, the above simplification may not hold, as the residual organic matter content is increased due to manure additions. Soil organic matter decay rate and consequent nitrogen mineralization rate are functions of soil texture and farm management system, rather than of initial organic matter source (Johnston et al. 1988). After 130 years of manure application at a rate of 238 kg N ha^{-1} y^{-1} at Rothamsted, England, the soil organic matter has increased two to three fold, and annual $N0_3$-N leaching losses are 124 kg ha^{-1} (Powlson et al. 1988).

Denitrification, the conversion of $N0_3$ to gaseous nitrogen compounds, is an important component in nitrogen balances for soils. Spatial variability studies show that denitrification in soil occurs in small, localized, anaerobic "hot spots". These locations are associated with particulate organic carbon (Parkin, 1987; Parkin et al. 1987; Goulding and Webster, 1988). The requirements for denitrification are a carbon source, nitrate, denitrifying bacteria, and anaerobic conditions (Paul and Beauchamp, 1988). Arah and Smith (1988) show that the existence of "hot spots" is predicted by physical principles. Manure incorporation can be expected to increase the concentration of such locations in a soil, thereby increasing the denitrification rate. The presence of both carbon sources and nitrate in manure leachates to shallow groundwater has been shown to result in increased denitrification (Gillham, 1988). It is possible, then, that manure application results in increased denitrification losses, compared with inorganic fertilizers applied at similar available-N rates. In the manure disposal context, this would permit a higher recommended application rate.

Field studies have confirmed that denitrification losses from soil are greater when manure is the fertilizer source. Goulding and Webster (1988) measured 0.6 kg N ha^{-1} day^{-1} lost from inorganic-fertilized fields and 1.3 kg from manure-treated fields. However the amount of N denitrified varies greatly. In a European study, applying cattle slurry manure at a rate of 100 kg NH_4-N ha^{-1}, Thompson and Pain (1988) found denitrification losses accounted for 0-54% of N applied as cattle slurry manure, depending on soil type, application time (spring or fall/winter), and application method (surface applied, or injected). Poorly drained soils had negligible losses, while fall/winter applications to freely drained soils resulted in the greatest denitrification losses. Reduction of NH_3 volatilization losses, by injecting the manure into the soil, resulted in greater denitrification losses.

3.3.3 Nitrogen runoff to streams

PLUARG research included monitoring of 11 Ontario watersheds over a 2 year period. Livestock density correlated positively with stream P and N concentrations, and negatively with sediment load; however these correlations were not statistically significant (Coote et al. 1982). The significant factors in sediment loading were soil clay content and % row cropping (Wall et al. 1982). Nitrogen loading was highly correlated with fertilizer plus manure application rates. Of the total N load, 75% was in $N0_3$ form. Storage facilities were considered not to be major contributors to watershed N loads, but winter spreading of manure was identified as a major concern (Neilson et al. 1982).

In a six-year, field plot study of manure application at three rates, and four spreading schedules, on a sandy clay loam soil, Phillips et al (1981) and Culley and Phillips (1982) found total N, total P and total sediment concentrations in storm runoff did not correlate with manure or fertilizer application rates. More than 97% of the total N and P in the runoff was associated with the sediment. Only winter application of manure had a negative effect on overland runoff water quality. Concentrations of N, P and K in spring runoff were strongly correlated to the amount of manure winter applied. Fecal bacteria counts were significantly greater, as well. Manure application rates supplied 224 - 897 kg N ha^{-1} $y^{-1.}$

Few studies have measured overland runoff from feedlots or storage sites in Ontario. Early work at the Central Experimental Farm in Ottawa showed concentrations in manure pile leachate of 30-1000 mg N L^{-1}, and 1240 mg P L^{-1}. Exposed solid manure storage resulted in 9% greater organic matter, 17% greater N and 12% greater P losses than shed-stored manure, primarily due to runoff losses (Atkinson et al. 1951).

Irwin and Robinson (1975) monitored overland runoff from a well designed paved beef feedlot to a holding pond. Runoff depth correlated linearly with 15-day total rainfall. Runoff occurred if rainfall exceeded 8.4mm; runoff was 60-78% of the excess rainfall.

As part of the PLUARG project, Coote and Hore (1977, 1978) studied overland runoff from two paved feedlots and 2 manure storage sites over a 2 year period. For a given site, runoff quantity per rainfall event varied linearly with total precipitation. Antecedent wet and dry conditions were considered separately. Antecedent water increased the slope of the curve, and reduced the amount of precipitation required for runoff to begin. Minimum precipitation requirements to induce runoff ranged from 1.5 - 7.1 mm; runoff was 60 - 77% of the excess rainfall. Runoff quality was less predictable. Suspended solids concentration was highly variable, and was highly dependent on flow rate. This relationship was particularly strong at feedlots, where the thin manure layer over the feedlot floor was more easily removed. Other water quality parameters correlated fairly well with suspended solids concentration. Nitrogen in runoff from storage accounted for 2% of the total N produced; in runoff from feedlots 6% was accounted for. Runoff contained 0.4 and 1.5% of total P produced, from storage and feedlots respectively. Mean BOD varied from 1370-5000 mg L^{-1} from the 4 sites, and varied with suspended solids and season, reflecting biological activity. None of the runoff met quality standards for stream discharge.

3.3.4 Nitrate leaching to groundwater

Nitrate contamination of groundwater from land application of manure must be understood in the context of the larger nitrate leaching problem in agriculture. Wherever water moves through a soil with a significant nitrate concentration, nitrate is transported by the water. Soil nitrate concentrations are determined by the soil nitrogen cycle. If 1) conditions are aerobic, 2) plant uptake is not sufficient to use the net mineralized nitrogen, and 3) water is leaching through the soil, then nitrate will be leached. These conditions occur in most intensively-cropped agricultural systems in semi-humid to humid environments, or under irrigation.

Krider (1987) identified several areas in the US where nitrate contamination of groundwater from agriculture has been well documented.

1. In Iowa, in a Karst region, groundwater $N0_3$ content varies with fertilizer rate.

2. In Delaware, fertilizer N application to produce maximum corn yields in a sandy soil region are not compatible with safe groundwater $N0_3$-N levels.

3. In Pennsylvania, a nutrient management program reduced $N0_3$ levels in the groundwater.

4. In a California region, groundwater $N0_3$ concentrations were about 3 mg L^{-1} greater after the groundwater passed under an intensive dairy region. At the livestock operation sites, levels to 70 mg L^{-1} were measured.

Regions with limestone bedrock, with supplemental irrigation of sandy soils, or with intensive corn production have been identified as especially susceptible to $N0_3$ contamination of the groundwater (Keeney et al. 1986). Under some conditions as little as 15.1 kg $N0_3$-N ha^{-1} y^{-1} allowed to pass below the root zone may result in groundwater contamination (Coote and Zwerman, 1975). In Europe, annual nitrogen application rates of 200 kg N ha^{-1} are expected over the long term to result in groundwater nitrate concentrations greater than 10 mg NO_3 -N L^{-1}. An estimated 20-30% of total fall-applied manure N is leached below the root zone (Kolenbrander, 1981). Poultry manure applied at rates of 900-2800 kg N ha^{-1}, in a lysimeter study under corn resulted in N leaching losses of 24-27% of the N applied (EPA, 1971).

The OME groundwater pesticide monitoring program has included measurement of major ion concentrations and bacteria and fecal coliform counts; however data relating sampling sites and livestock industry are not collected (OME, 1987; Beck, 1988). In May 1983, the Ontario Ministry of the Environment (OME) reported $N0_3$ contamination of wells in Hensall, Ontario. The NO_3-N concentration in one well had increased from about 5 mg L^{-1} in 1972 to 12 mg L^{-1} in 1982. A second well became the main source of water in 1982 but concentrations began to increase in that well so a deep well was drilled in 1984. A study was conducted by a consulting firm, Gartner Lee Ltd., in an attempt to determine the source of the nitrates. Although evidence was not conclusive, the report concluded that local land use practices were the reason for elevated nitrates. Potential sources included chemical fertilizers and manures applied to the land, infiltration of contaminated runoff from fertilizer fields, infiltration of leachate from manure storage piles or liquid manure storage (Gartner Lee Ltd. 1987).

Several Ontario studies have monitored groundwater quality beneath and in the vicinity of feedlots or manure storage sites (Gillham, 1968; Sowden and Hore, 1976; Coote and Hore, 1978; Culley and Phillips, 1982b; Miller et al. 1985), or have monitored subsurface tile drainage from livestock farms (Beak Consultants, 1977; Patni, 1978; Patni and Hore, 1978; Culley and Phillips, 1982a). Other studies have addressed nitrogen dynamics and denitrification in soils (Burton, 1982; Beauchamp, 1982, 1983, 1985; Paul and Beauchamp, 1988) and in groundwater (Gillham et al. 1978, 1984; Starr and Gillham, 1988),.

Nitrate contamination of groundwater under a manure storage pile site on a loam soil, in use for more than 50 years, was studied by Gillham and Webber (Gillham, 1968; Gillham and Webber, 1969). They observed a plume of contamination skewed with groundwater flow direction. Nitrate concentrations were greater than 15 mg L^{-1} close to the pile, and greater than 5 mg L^{-1} to a 90-m distance in the direction of flow. No significant P contamination was observed.

Miller et al (1976) found NH4 concentrations under hog manure lagoons at background levels at 20- to 30-cm depth under a 2-year old lagoon on fine soil, but found high concentrations to depth of sampling (1.5 and 4 m) under older lagoons on medium and coarse textured soils. Because of the anaerobic conditions, $N0_3$ levels were low in all samples. Phosphate concentrations were high immediately under the lagoons, but were at background levels in the 20- to 30-cm depth zone, except in coarse soil, where they remained above background levels to 60-cm depth.

Sowden and Hore (1976) found high $N0_3$ concentrations at 122-cm and 275cm depths beside 30-year old manure storage piles, on concrete or gravel pads. Below the water table level, at 275-cm and 425-cm depths, concentrations were low, suggesting denitrification at or near the water table. Soil profile $N0_3$ beneath feedlots may be 30 times greater than that of surrounding soils (Loehr, 1974).

Coote and Hore (1978, 1979) found a distinct contamination plume in groundwater under a six-year old unpaved feedlot on loam soil. Elevated total carbon, $N0_3$, C1 and Na concentrations were observed, with an estimated $N0_3$ contamination zone to 60 m in the direction of groundwater flow. Organic N mineralization and nitrification were the dominant processes in the first 20 m distance from the feedlot, at which point the organic N was exhausted, and $N0_3$ concentrations peaked. Denitrification dominated the nitrogen dynamics thereafter, with $N0_3$ concentrations decreasing more rapidly than $C1^-$ concentrations. The contamination plume may be larger under larger feedlots. (>100 head), in more permeable soils, or in shallow soils over bedrock. Elevated P concentrations were not detected beyond the lower edge of the feedlot.

Miller (unpublished data, Univ. of Guelph) observed low $N0_3$-N concentrations in soil at a location where feedlot runoff ponded temporarily with each runoff event. The alternating aerobic and anaerobic conditions may have led to alternating nitrification and denitrification respectively, of the manure N, thus reducing the $N0_3$ concentrations in the soil.

Another research concern is nutrient leaching beneath earthen manure storage structures. Because of the relatively high cost of impermeable manure storage facilities, and the popularity of earthen storage structures, much research has focused on the suitability of the latter on different soils and for different manure types. Even on sandy soils, earthen storage facilities quickly form a seal with very low hydraulic conductivity (Culley and Phillips, 1982b; Miller et al. 1985; Barrington et al. 1987, 1987b).

Miller et al. (1985) reported almost complete sealing of a storage pond in a sandy soil within three months of addition of liquid beef manure. Rowsell et al. (1985) concluded that the sealing was due primarily to a physical blocking of pores. Although there was direct input of chloride from the pond to the groundwater during the first three months, there was no input of NO_3-N because there was no $N0_3$-N in the liquid manure. In fact the $N0_3$-N concentration in groundwater beneath the pond was lower during this three-month period than that in the surrounding groundwater. This was attributed to denitrification associated with input of soluble organic carbon

(Miller et al. 1985). However, Culley and Phillips (1989a, 1989b) showed ongoing $N0_3$ transport below liquid dairy manure storage pits over a five-year period, on three soils with initial K_S values between 10^{-4} and 10^{-7} ms^{-1}. They showed from physical principles that nitrate movement to groundwater might be expected within a 10- to 20-year lifetime of a storage facility even on soils with an initial K_S value of 10^{-8} ms^{-1}. This would be true, however, only if the stored manure contained $N0_3$-N or the soil beneath the storage facility became aerobic so $N0_3$ was formed.

Nitrate contamination of groundwater has been observed to be closely associated with well-drained, fertilized agricultural land (Brown, 1982; Gillham et al. 1978; Howard and Falck, 1986). In fine textured, poorly drained material, groundwater flows only a few millimetres to centimetres per year. Because modern agriculture is relatively young, and poorly drained soils are commonly tile drained, soil leachate sufficient to result in elevated $N0_3$ concentrations may not have reached the groundwater in these areas (Gillham et al. 1978). An OMAF study to identify regions of southern Ontario with a high potential for groundwater pollution was based largely on soil texture (OMAF, 1987).

In the Ontario environment, leaching is assumed to occur in the fall, after harvest to freeze-up, during extended mid-winter thaws and again with spring thaw. Little occurs during the growing season. Webber and Elrick (1968) reported an average groundwater recharge in 8 Ontario watersheds of 75-200 mm y^{-1}, based on watershed budgets. Such regional average estimates don't address through-soil leaching under specific soils, vegetation, or agricultural management systems. Few Ontario data are available on actual annual through-soil leaching quantities (OMAF, 1987). Lysimeter studies at the University of Guelph showed annual groundwater recharge ranging from 0 to 260 mm y^{-1}, over ten years. Groundwater recharge occurred only in the late fall and early spring, and only in 4 of the 10 years monitored (Webber and Elrick, 1968). Much of Ontario's agricultural land is tile drained. No data were found on recharge amount to below-tile groundwater from tile-drained soils.

Soil nitrate levels have been observed to decrease rapidly after harvest to a background level in late fall (Webber et al. 1968; Burton, 1982). The amount of $N0_3$ remaining in the soil at harvest correlates with the amount of N applied in excess of the recommended fertilizer rate in that season (Beauchamp, 1983). At a site with pumped tile drainage effluent, Miller (1979) found $N0_3$-N concentrations rarely exceeded 10 mg L^{-1} if fertilizers were applied at or below recommended rates, and rarely found concentrations less than 10 mg L^{-1} when fertilizer was applied at above recommended rates. Excess N applied showed up in drainage water the following fall through spring. Two hypotheses are suggested by these observations:

1. The nitrate available for leaching at harvest is the sum of net inputs and net mineralization- immobilization, minus that removed in harvest and that lost by denitrification and volatilization during the growing season.

2. The nitrate leached annually is the amount available at harvest plus the difference between net mineralization-immobilization and denitrification and volatilization losses, during the fall and winter.

The validity of the second hypothesis may be questioned. A European symposium estimated an average of 70% of the after-harvest available N is leached during the fall and winter (Kolenbrander, 1981). In the US corn belt region, high rates of fertilizer application over years has resulted in a significant buildup of $N0_3$ in the profile (Keeney et al. 1986). Recent unpublished Ontario data show significant nitrate concentrations in the soil in early spring (Beauchamp and Kachanoski 1988). Nitrogen cycling throughout the fall and winter period may be highly significant. Total nitrogen inputs and crop off takes are fairly easily calculated; however the net mineralization-immobilization and denitrification-volatilization losses are not easily measured or known. Data from Alberta suggest mineralization, nitrification and denitrification continue to occur at significant rates in soils frozen to 60 cm depth (Heaney and Nyborg, 1988).

Patni et al. (1981) showed five years of manure application, at an average rate of 840 kg N ha^{-1} y^{-1}, on a deep sandy soil under corn, resulted in elevated $N0_3$ concentrations of 30 - 40 mg N L^{-1}, and NH_4 levels of 9 - 29 mg N L^{-1}, to 6-m depth.

Phillips et al. (1981) found spring drain-tile discharge $N0_3$ concentration reflected manure application rates for all application schedules, while $P0_4$ concentrations reflected application rate if the manure was applied in the winter. Phillips et al. (1982) showed drain tile discharge $N0_3$ concentrations reflected nitrogen input rates. Under corn, with 570 kg N ha^{-1} y^{-1} from manure, discharge $N0_3$ concentration was 19 mg L^{-1}. Corn grown with recommended inorganic fertilizer rates resulted in drain water $N0_3$ concentrations over 10 mg L^{-1}. Drainwater-dissolved $P0_4$ represented less than 1% of inorganic fertilizer P, and less than 0.2% of manure-P applied, and did not reflect application rates. $N0_3$-N concentrations were highest in early spring.

Tile drain $N0_3$ concentrations will not necessarily correlate with groundwater contamination. Work by Gillham and co-workers (Gillham et al. 1978, 1984; Starr and Gillham, 1987) has shown that denitrification frequently occurs in shallow groundwater, because of the transport of organic carbon along with $N0_3$ to the water table. If the water table is at greater depth, and organic compounds are oxidized or adsorbed by soil before reaching the water table, denitrification will not occur. Recent studies at the Agriculture Canada Animal Research Centre, Ottawa, have shown tile drain and shallow groundwater $N0_3$ concentrations consistently in excess of 10 mg L^{-1}, while at 6-m depth the groundwater $N0_3$-N concentration is less than 1 mg L^{-1}, under fields receiving manure at fairly high rates since 1971 (Patni, Agriculture Canada, Ottawa personal communication).

On a sandy clay soil under five years continuous corn production, Culley et al. (1981) found total profile (0-120 cm) inorganic N content at harvest reflected annual and cumulative manure application rates. Lower profile (60-120 cm) inorganic N content at harvest depended on application time as well, with higher values from fall applied manure than from spring applied. The effect of an annual rate of 224 kg N ha $^{-1}$ from manure was similar to that of 134 kg inorganic fertilizer N.

Beak Consultants (1977) found that 95% of drain discharge N was $N0_3$ and 3% organic, while in surface runoff 85% was $N0_3$ and 13% organic. This reflects the filtration effects of through-soil transport.

Patni and Hore (1978) found some increase in drainwater $N0_3$ concentrations over three years of manure application at rates of 500 kg N ha^{-1} y^{-1}. Concentrations increased from 1.6 to 5.1 mg L^{-1}. However crop removal and tile drain discharge accounted for only a portion of the total N applied. A buildup of soil organic N due to manure application was suspected. Annual drain tile discharge was one-fifth of total precipitation on a clay soil, and one-tenth on a sandy soil.

In summary, the amount of manure which can be safely applied to a given soil on an annual basis depends on the safe mineral N fertilizer application rate for the given soil, crop, management, and climatic conditions, and:

1. the ammoniacal and total *N* contents of the manure at time of application,
2. the size of the rapidly mineralizable organic N fraction,
3. the net immobilization of the mineral fraction,
4. the effect of the manure on denitrification,
5. application time, and
6. long term organic matter accumulation effects.

3.4 PHOSPHORUS

Phosphorus is a major contributor to the eutrophication of surface water bodies. The P load from agricultural land to surface waters can be measured in overland runoff, in drain water, and in streamflow. Contributions from tile drains and groundwater, while small, may be significant (Ryden and Syers, 1973).

Beak Consultants (1977) found that 80-100% of the P in drain tile discharge was in soluble form, while in overland flow 60-85% was soluble. The latter value appears high because the majority of the P removed from soil by runoff is normally in particulate form, because of the low solubility of P and the strong P fixation of most soils. However the dissolved P is much more bioavailable and therefore potentially more damaging to surface water. Dissolved P concentration in runoff is greater from fields with high fertilizer application rates or with manure on the surface (Spires and Miller, 1978).

The amount of P delivered to a stream from a mineral soil by runoff is a function of the P content of the surface soil, the total sediment load, the P enrichment ratio, and the sediment enrichment ratio (Spires and Miller, 1978). The P enrichment ratio is included because the finer soil components, which are preferentially transported, also have a higher P content than the bulk soil. The sediment delivery ratio is dependent on soil texture, topography, and distance from the soil source to the stream (Dickinson et al. 1987).

For a given soil, the phosphorus concentration in the surface soil will increase with manure application because the rates of manure application normally add more phosphorus than is removed by the crop. Culley et al. (1981) observed a buildup of bicarbonate-extractable P in the soil profile at the 0- to 15-cm depth corresponding to total cumulative manure application over a five-year period. Accumulation was observed in the 15 to 30-cm depth increment only where the total cumulative application exceeded an apparent threshold value. The bicarbonate-extractable P is a reliable indicator of the plant-available P and is the basis of fertilizer P recommendations in Ontario. Thus the effect of manure on the available soil P is measured directly and is accounted for in making fertilizer P recommendations. When the available soil P reaches a value where additional P is not required for crop production, further applications of manure will increase the P in the surface soil and hence the P in surface runoff without any beneficial effect on crop production.

Phosphorus transport through soil is limited by the low solubility and strong adsorption of P. A site-dependent linear correlation has been observed between P application rate and total P concentration in drain tile effluent (Coote and Zwerman, 1975). The major portion of the effluent P was in particulate form. The relationship was dependent on the permeability and P fixation of the soil, and as such was highly dependent on soil type. Macropore flow may play an important role in P transport to tile drains and groundwater.

Under extreme P loads, saturation of the P fixation complex will occur to some depth in the soil profile, permitting increased transport of soluble P to lower depths (Coote and Zwerman, 1975). Under a 13-year old feedlot on sandy alkaline soil in Manitoba, Campbell and Racz (1975) found evidence of organic and inorganic P movement to 150-cm depth, with P saturation of the upper 30 cm, and organic P mineralization in the 0- to 90-cm zone.

There has been considerable concern in Western Europe, particularly the Netherlands, with the contamination of shallow groundwater with P under intensive manure application. It has been suggested that "disposal of P in manure will be legally constricted in the near future" (van der Zee and Bolt, 1988). Models are being developed and calibrated to predict the time required to increase the P concentration in the soil material overlying the groundwater to the point that P movement would be a serious problem (van der Zee and Riemsdijk, 1988; van der Zee and Bolt, 1988). We believe that it will be many years before this is a problem in Ontario with the possible exception of sandy soils with a shallow groundwater. We cannot, however, ignore this potential problem.

Phosphorus pollution of surface water from manure sources can be largely controlled by:

1. proper manure containment facilities,
2. restriction of livestock access to water courses,
3. dry-weather manure application and immediate incorporation, and
4. application of soil conservation and erosion control measures.

No adequate method has been developed for manure incorporation in notill planting systems. Manure left on the surface greatly increases nutrient loads in runoff (Dickinson et al. 1987).

Several Ontario studies have specifically addressed P forms and loads in runoff. Much of the P in runoff is associated with suspended solids. Miller and Spires (1978) found dissolved reactive P averaged 24% of total P, but was as much as 90% when sediment load was less than 100 mg L^{-1}. Overland runoff from fields with manure on the surface had dissolved reactive P concentrations 8 times greater than those from soils without surface manure. Concentrations generally correlated with the P concentration in the soil. Patni (1982b) found the soluble P portion of the total P load increased in runoff from manured soil compared with non-manured soil. Larger soil aggregates preferentially sorb particulate manure (Bhatnagar 1979; Bhatnagar and Miller, 1985; Bhatnagar et al. 1985). Manure incorporation has been shown to increase soil aggregate stability. Therefore runoff P load from manured soils is more sensitive to rainfall intensity than that from non-manured soils, because heavier rainfall is required to break and transport the stable, relatively P- rich aggregates of the manured soil (Marsh, 1986).

Beak Consultants (1977) studied the effects of agriculture on stream water quality in the Little Ausable River Basin watershed, monitoring 26 sites for two years. The soils were primarily of clay texture. Daily flow rates during spring melt were more than 50 times greater than the average daily flow rate; during storm events, daily flow rates often exceeded 10 times the annual average. Pollutant loads were greatest during these periods. More than 50% of the annual P load was delivered in the months of March through May. Average annual P loads were 0.33 kg ha^{-1} from areas without livestock, and varied from 0.33 to 2.3 kg ha^{-1} from areas with livestock. This inter-farm variability in P load was attributed to farm distance to watercourse, feedlot and manure storage location, drainage, winter spreading of manure, artificial channel reconstruction, and cattle access to streams.

Other Ontario research has supported the observation of the importance of spring runoff and storm events to pollution loads. Monitoring at the Central Research Farm in Ottawa showed very few days each spring accounted for the major part of total annual transport (Patni and Hore, 1978; Phillips et al. 1981; Patni, 1982a,b). Patni and Hore (1978) found snow-melt total P concentration exceeded 0.07 mg L^{-1} in all samples. Tennant et al. (1972) found fecal bacteria numbers in watershed drainage water peaked closely with heavy rainfall events, in presence or absence of manuring activity, at the Greenbelt Research Farm in Nepean Township.

Hore and Ostry (1978a, 1978b) monitored point-source pollution of the Saugeen and Grand River basins. They found water quality reflected agricultural activity, and P loads were correlated to soil texture and manure use. High fecal bacteria loads were associated in part with livestock operations.

Robinson and Draper (1978) modelled P loading of the Great Lakes from manure sources, using an aerial photo survey of manure locations relative to waterways, and runoff load and attenuation distance estimates from literature values. Based on this model it was estimated that 20% of the P loading in the Ontario portion of the Great Lakes watershed came from livestock operations. Due to lack of data, the model was limited by assumptions of a single, linear attenuation rate for the province, straightline runoff flow, and constant runoff loss percentages from feedlots, winter spreading and solid storage.

3.5 Organic Matter

Manure contains a broad range of soluble and suspended organic components, as well as large bacterial populations. Manure contamination of water can greatly increase biological oxygen demand, resulting in fish kills. Organic material also discolors water.

Organic matter is a pollution problem when manure or runoff from manure enters a water course. As such, it occurs in association with P and bacterial pollution. Measures to control P contamination of surface water, listed previously, also will control organic matter contamination.

Manure organics leached to groundwater are not seen to be a health hazard, and may promote denitrification, thereby reducing $N0_3$ contamination. However they may result in discoloration of the water, and are indicative of possible bacterial contamination. Through soil transport of organics from manure is controlled by the transport principles described previously. Fine particulate and soluble organics, which are more readily transported also are more readily decomposed by soil microorganisms. This removal mechanism reduces the organic matter concentration expected to reach groundwater. However macropore flow may result in organic matter transport at rates and to depths not predicted by transport models.

3.6 Bacteria

Water is an important transmitter of several livestock and human diseases, the bacterial agents of which may be found in manure (MacLean, 1983). Bacterial populations in manure are such that runoff from manure to streams generally will result in bacterial contamination of the water. Direct manure inputs, from spills or cattle access to waterways are also important causes of water contamination (Loehr, 1974, Hayman, D.G. 1989).

Ontario studies also have shown fecal coliform numbers in liquid manure decrease with time in storage (Tennant et al. 1972; Patni and Hope, 1978).

Culley and Phillips (1982b) observed higher drain tile discharge fecal bacteria counts for several days following fall or winter application of liquid manure, at a sandy clay loam site. Fecal streptococci numbers were higher from plots with winter applied liquid manure than from plots with other application schedules. Coliform bacteria apparently did not survive the surface freezing. The drain tiles in these research plots were at an average depth of 75 cm.

In a recent review of bacterial runoff from agricultural land, Baxter et al. (1988) concluded:

1. Between-study comparisons are difficult because of differences in watershed size and homogeneity of land use.

2. Fecal coliform and fecal streptococci counts are indicative of water contamination from fecal material, while the ratio of fecal coliforms to fecal streptococci may indicate whether the contamination is from human or animal sources. This ratio is unstable with time in stream flow, however. Nor does it distinguish between domestic and wild animal sources (White, 1979).

3. Fecal bacteria counts from pasture land may tend to be slightly higher than those from corn fields; however the variability is large, and both are frequently over the recommended safe levels. Loehr, (1974), found that fecal bacteria counts in runoff from agricultural land relate more to storm intensity than to livestock grazing.

4. Runoff loads from feedlots are 1000 times higher, on average, than those from agricultural land.

5. In general, bacterial loads are greatest during early peak flow of storm runoff events. However the factors in the relationship between bacteria density and stream discharge include temperature, hydrologic proximity to the manure source, manure age, livestock management, wildlife activity, and channel bank storage of bacteria source material.

6. Loading functions developed to date do not adequately account for the factors noted above.

Field and laboratory studies have shown that bacteria tend to be removed from solutions flowing through soils. Loehr (1974) found 98% of the coliform and enterococci were removed from waste water by 36 cm of soil. However fecal bacteria population increases may be observed in tile drain water minutes to hours after irrigation with liquid manure (reviewed by Patni et al, 1984). Escherichia coli and Clostridium welchii have been transported to depths of 10-15 m in sandy soils (reviewed by Coote and Zwerman, 1975).

3.7 Control Technologies

In many cases, management is observed to be the primary factor determining the extent of water pollution from manure (Robbins et al. 1971; Beak Consultants, 1977; White, 1979; MacLean et al. 1983; Overcash et al. 1983). In general, such observations are based on available technology and methodologies, without explicit consideration of economic factors which a farmer may perceive as prohibitive to acceptance of improved management.

Potential for water contamination from barnyards, animal housing and paved holding facilities can be controlled with proper design and management. Problems remain with:

1. leaching from earthen storage and treatment facilities,
2. overland runoff and leaching from unpaved areas of animal concentration, and
3. overland runoff and leaching from areas with excessive land application rates (Krider, 1987).

Nye (1982) described the design requirements for feedlot runoff control, using settling basins and vegetative infiltration areas for small feedlots, and holding pond and irrigation spreading systems for larger feedlots. With proper design and management, and adequate land for waste disposal, such a system should control water pollution from feedlot runoff.

Wherever pasture animals congregate, manure is concentrated and susceptibility to erosion increased. Proper pasture management involves locating feeding areas, salt licks, shaded areas and watering sites such that erosion and surface water contamination are minimized (SCS, 1988). Overcash et al. (1983) found "poor" pasture and rangeland management resulted in a 6-fold increase in runoff N concentration, a 10-fold increase in biological oxygen demand, and P and fecal bacteria increases to 100 and 1000 times, respectively, compared with well-managed pasture and range land.

A program to permit maximum land disposal of manure without causing contamination of groundwater must be a part of a larger nitrogen management. program, including minimal fallow periods, and nitrogen input rate and timing matched to the crop needs. To avoid manure P contamination of surface water, manure management must be combined with soil conservation and erosion control methods. However a tradeoff may exist between minimizing P pollution of surface water and N pollution of groundwater. Reduction of overland runoff will increase leaching flow through the soil. Conservation tillage methods may actually increase P loads to surface water, particularly where applied manure is not incorporated into the soil.

The possibility for balancing crop production concerns with protection of quality in streamflow and groundwater when manure management is combined with conservation tillage has been examined by Walter et al. (1987). They conclude that conservation tillage and manure management are compatible and that losses of manurial components in surface runoff are greatly reduced by a little incorporation. They note that in many dairy farms in New York state there is more manurial nitrogen available than can be used on the farm to achieve a balanced nutrient status in the soil over a long period. Manurial phosphorus is greatly in excess of what can be used on these farms. Because manure is not generally utilized as a fertilizer source they caution that there is a potential groundwater problem with any tillage system.

The occurrence of algal blooms and extensive episodes of anoxia in large areas of Chesapeake Bay has focussed attention on the role of agricultural activity in creating increased phosphorus and nitrogen loading to the Bay (United States Environmental Protection Agency, 1983). Investigations and remedial measures in Maryland have been focussed on reduced phosphorus loadings through conservation tillage although some attention has been paid to construction of manure storage facilities as part of a best management practices strategy (Staver et al. 1989). These authors report that in corn production areas nitrogen transport to groundwater of 20 to 30 kg/ha produces wide spread nitrate contamination of shallow groundwater. They recommend use of fall-sown cover crops to reduce soil nitrate concentrations during fall and winter recharge episodes. They are critical of an alternate proposal for reduced nitrate movement in groundwater, the use of forested buffer zones along streams, because most groundwater would pass under these strips unaffected in the geological and topographic settings in Maryland.

In Pennsylvania reductions in phosphorus and nitrogen loadings to Chesapeake Bay are an objective of state programs. Lanyon and Beegle (1989) report on the role of on-farm nutrient balance assessments in an integrated approach to nutrient management. Attempts to achieve targeted reductions in phosphorus and nitrogen moving to streams involve a mobile laboratory that allows individual test results for soil fertility status and nutrient content of manure to be given to each farm operator for his/her particular circumstances (Anderson and Graybill, 1987). Farmers are encouraged to install manure storages and to apply manure in the late spring after the damage of runoff and leaching are past and at times of maximum crop demand for nitrogen.

3.8 Summary of Literature Review

Provided the timing of manure application and incorporation are properly managed, primary concern with land application of manure is to avoid over application, which may lead to $N0_3$ contamination of the groundwater or excessive phosphorus in overland flow. Because of the highly variable water, and nutrient contents of manures, recommended manure application rates must be based primarily on the amount of ammoniacal and total nitrogen applied. The following uncertainties remain:

1. There is no convenient way to determine the N content of manure at the time of application. Manure N content can vary greatly from expected mean values, and the return time from laboratory analyses may result in significant changes to the manure characteristics between sampling and application.

2. No reliable N soil test is available. The best N recommendations available are based on soil and crop type and past crop and fertilizer history. Whether these recommended rates are compatible with groundwater protection is unknown.

3. The relationship between manure N availability and inorganic N fertilizer availability is dependent on a number of dynamic factors, and is not well established for varying manure, soil, crop and climatic types.

4. Seasonal variability, which alters crop N requirements, may also alter the percentage of manure N available to the crop.

Producers frequently apply manure at rates exceeding recommended rates to insure against N deficiencies due to poor estimates of N requirement, manure N content, manure N availability, and/or an atypical growing season.

Minimum distances from waterways for land application of manure are described in most legislation and guidelines. However these are seldom based on sufficient quantitative data, nor are the effects of variables such as slope, vegetative barriers, and application rate adequately addressed (Loehr, 1974).

A potential, tradeoff between surface water and groundwater protection has been observed. The nature of this relationship must be understood. While soil is generally recognized as an excellent natural filtration system for many pollutants, this recognition must be weighed against the problem of $N0_3$ contamination of groundwater in intensive agriculture, and the role of manure disposal in $N0_3$ contamination.

4.0 INVENTORY OF RECENT AND ON-GOING ONTARIO STUDIES

On initiation of this study, all Conservation Authorities, Federal and Provincial agricultural research institutions and relevant Universities were requested to submit reports on recent studies and a listing of on-going projects. A listing and brief summary of the responses are presented in Appendix A.

5.0 FRAMEWORK FOR ESTIMATING ACCEPTABLE RATES FOR LAND APPLICATION OF MANURE

The current draft "Guide to Rural Land Use and Farm Practice" contains recommendations for land application of manure. The basic rate recommended is annual application of manure from 2.5 "animal units" per hectare of land. Although it is not directly stated, the rate is based on nitrogen required for continuous corn production. The guide suggests manure "disposal at up to twice this rate (ie 5 animal units per hectare) is allowable without environmental detriment on loam soils or finer. The maximum rate for sandy soils is 3.3 animal units per hectare.

The guide contains general statements on the proportion of N, P and K fed to animals that will be present in fresh excretia. General estimates of losses to the atmosphere of gaseous forms of nitrogen are given as are estimates of availability of nitrogen to crops in the year of application. The higher availability of nitrogen from poultry manure is especially noted. There is no direct effect on recommended application rates of methods of manure collection, storage, distribution and incorporation or of time-of-year of the application.

We agree that the acceptable rate for land application of manure should be based primarily on nitrogen. Although contribution of organic matter and bacteria are recognized as problems, they are associated primarily with overland runoff from fields where manure has not been incorporated or from inadequate storage facilities. Remedial measures for these problems are relatively simple. The current and proposed guidelines do not address the issue of increased P in runoff. This will be discussed later in this section.

The major concern with the current and proposed guidelines in relation to nitrogen is that they are based on the assumption that applications that are recommended in relation to corn production provide adequate protection of groundwater. In our opinion, this assumption is questionable even in a corn production system. It is clearly not valid when associated with crops with a lower nitrogen requirement.

The maximum acceptable rate of manure and fertilizer N application should be based primarily on the contribution of NO_3-N to the groundwater, not the requirement of the crop. We propose the following criterion for the maximum acceptable rate of nitrogen application as manure and/or fertilizer.

THE MAXIMUM ACCEPTABLE RATE OF NITROGEN APPLICATION IS THAT WHICH WILL ENSURE THAT THE ANNUAL VOLUME-AVERAGED $N0_3$-N CONCENTRATION IN WATER LEAVING THE ROOT ZONE DOES NOT EXCEED A SET CONCENTRATION.

The limiting concentration will be at or somewhat above 10 mg L^{-1} because, at least at present, it is the accepted safe concentration for drinking water. There are numerous processes, including dilution and denitrification, that will reduce the concentration of $N0_3$-N as the water moves to and enters the water table. There are, however, no processes which will increase the concentration. Hence this criterion will ensure that groundwater concentrations will not exceed 10 mg L^{-1} due to input from manure application.

Using concentration as the criterion is preferable to stating an allowable absolute quantity of $N0_3$-N entering the groundwater on an area basis because the environmental concern is primarily with concentration in drinking water rather than the total input to the environment. Using concentration does mean, of course, that the absolute amount permitted will vary with the volume of recharge water passing through the soil.

If the stated criterion is accepted, the appropriate limiting concentration must be set and a system must be developed to predict the amount of manure that can be applied without exceeding that concentration. As is apparent from the literature review presented in Sect. 3, the acceptable manure application will vary with type of livestock, manure storage and handling, soil type, crop grown and climate.

Possible approaches to assessment of the pollution potential on a given site as suggested by Krider (1987) were discussed briefly in Sect. 2. One approach, termed the presumptive approach, was to collate all available information on a site and rank sites. A second approach was modelling, in which all the processes and interrelationships would be included in mathematical models. These models provide a way to deal with a large number of variables in continuous fashion. Several models are discussed in Sect. 3.2.1. A severe limitation to application of these models now and in the foreseeable future is that the data input requirements are very great and our knowledge of the processes and interrelationships are incomplete. Reliance on current values for the many parameters in the models can lead to gross errors.

An approach intermediate between the presumptive and modelling approaches appears to be the most promising, at least in the intermediate time frame.

The approach we recommend comprises the development of two independent tables, one of which would allow an estimate of the amount of ammoniacal and total nitrogen reaching the soil from one animal unit, and a second table estimating the amount of ammoniacal and total nitrogen that could be applied to a specific soil/crop system without exceeding the established criterion.

The first table would incorporate the following variables: animal type, time and method of storage, and time between application and incorporation. This table would account for losses, primarily by volatilization of NH_3, between the time of excretion and time of incorporation in soil.

The second table would include the following variables: existing nitrogen in soil, crop to be grown, soil texture and drainage, and time of application. The table would account for transformations, and fate of the nitrogen incorporated as well as incorporate soil hydrologic properties. Knowing the acceptable rate of N application in a particular site from the second table, one could determine the acceptable manure application rate from the first table in terms of animal units per hectare.

We recognize that farmers think in terms of volume or weight of manure per unit area, not animal units. It is difficult to relate animal units to volume or weight of manure because of differences from operation to operation in dilution or use of bedding etc. The first table in the proposed system would be more useful in establishing acceptable livestock/land ratios (ie in policy/regulation development) than in making specific recommendations for manure application by farmers. However the second table could be used directly by farmers in association with an analysis of the manure shortly before application. A possible structure for these tables is presented in Appendix B.

Sufficient understanding of the processes and relationships does not exist to permit the use of the proposed framework in the immediate future. The system, however, would provide a very good framework for research in that it would provide a systematic approach to assessment of the adequacy of current knowledge from the standpoint of environmental protection. It must be recognized that considerations of leaching of $N0_3$-N from manure can not be separated from that of inorganic fertilizer N. If the suggested criteria, ie restricting rates to ensure that $N0_3$-N concentrations reaching groundwater did not exceed a limiting concentration, was adopted for manure it would have to be applied to inorganic fertilizer as well.

We recommend that further effort be devoted to development of the proposed framework. Attempts to establish values for the various cells in the tables will establish more clearly the gaps in understanding.

As indicated earlier there are no guidelines for manure application based on the potential for phosphorus contribution to surface or groundwater. The probability of excessive amounts reaching groundwater by leaching is minimal, at least in the foreseeable future, because of the strong adsorption of P in the soil. The probability of excessive amounts reaching surface water is much greater.

It would be possible to place a limit on manure application based on the P soil test. Guidelines for sewage sludge on agricultural land specify that sludge must not be applied to soil with a soil test value above 60. This value was selected because, at the time, it was the value at which no further P was recommended for any crop. For most crops the value at which no further P is recommended is lower (e.g. 30 for corn). The imposition of such a regulation for sludge is not a serious limitation to farmers. It merely removes the option of spreading sludge. The restriction was placed to reduce the threat to surface water and to encourage more effective use of the phosphorus in sludge.

To impose such a regulation on manure application would be a major limitation to farmers because the land area required for manure application would increase as time progressed if phosphorus imports to the farm in purchased feed exceeded exports. An estimate could be made for a site, based on soil characteristics, of the time period required to reach the designated maximum soil test value. This time period might be unduly short even if manure application rates met the criteria based on nitrogen content. It must also be pointed out that there are no regulations restricting the application of fertilizer P. Any regulation applied to manure P must also be applied to use of fertilizer P.

A limit similar to that for sludge can be criticized on the same basis as the current limit on N i.e. that it is based on the requirement for crop production rather than the requirements for protection of the environment. There is an established maximum acceptable $N0_3$-N concentration (10 mg L^{-1}) in drinking water which can be used as a basis for establishment of environmentally safe rates of manure application. At present there is no such value for P in surface water. It would be possible theoretically to start with a maximum acceptable dissolved reactive P value for surface water and arrive at a maximum acceptable value of total P in runoff from a farm field. The myriad of adsorption, immobilization and transformation reactions that phosphorus undergoes, however, makes such a system much more complex than for nitrogen. It is improbable that such a system could be applied in the foreseeable future.

The most effective control of P contribution to surface water is to control soil erosion. The Soil and Water Environmental Enhancement Program (SWEEP) currently in place in Ontario is designed to address the concerns with phosphorus contributions in runoff from agricultural lands, including concerns those associated with manure use. While there may be a potential for P contribution even if erosion is controlled this is not well defined at this time. We do not think that a limitation on manure application based on phosphorus should be established until the results from SWEEP relating to P transport .to streams have been fully assessed.

6.0 RECOMMENDED CHANGES IN CURRENT GUIDELINES

In general, the guidelines indicated in the draft "Guide to Rural Land Use and Farm Practice" appear adequate with respect to storage and handling of manure. It would be highly desirable to avoid late summer and fall application completely which would require 12 months storage. It is probably not practical to do so for the majority of farmers. Requirements of at least 9 months storage would be beneficial in that it would allow most of the manure to be applied in the late spring and early summer. Guidelines with regard to control of runoff from feedlots etc. appear adequate.

The guide indicates that manure should be incorporated into the soil within 24 hours "whenever possible". This is feasible with most tillage systems but not with a no-till system. There is, at present, no method to incorporate manure while maintaining a no-till surface; thus it is contradictory to require incorporation and, at the same time, recommend notill on livestock farms. There is a need to develop satisfactory systems for manure application in a no-till system.

The guide also states that "spreading on frozen ground is not recommended". We would like to see this statement strengthened to indicate that, not only is it not recommended, but that it is not considered an acceptable practice.

There is little or no discussion in the draft guidelines on the need to consider manure and fertilizer N together. In many cases the value of manure N is down played if not ignored in decisions as to how much fertilizer N to apply. More emphasis should be placed on this aspect in the revised guidelines.

The current guidelines do not distinguish between a single application and repeated annual applications. Since the guidelines establish the land area required per animal unit, we presume the guidelines are for repeated annaual applications. We strongly suspect that application at the maximum rates allowed are contributing significantly to groundwater contamination with $N0_3$-N, particularly when applied to crops other than corn. A more detailed assessment, and possibly more research, is required to resolve this question.

Our major concern with the current "Agricultural Code of Practise" and the proposed "Guide to Rural Land Use and Farm Practice" is the basis on which acceptable rates of land application of manure have been established.

The intermediate term objective in developing guidelines for land application of livestock manure should be the implementation of the framework proposed in Sect. 5. As indicated in Sect. 5, however, our understanding of the processes and interrelationships are currently not adequate to allow this system to be used.

Although we do not have sufficient data to justify a major changes in the approach to guidelines at this time, some less drastic changes could be made.

We recommend that:

1) A section be included related to the effect of manure application on requirements for fertilizer N. This section could include some elements from Table 2 Appendix B in relation to crop, type of manure and time of application.

2) Distinction be made between fall and spring application. Application in the fall will result in a greater proportion of the nitrogen being subject to leaching than would spring application unless special precautions such as cover crops are used to retain the nitrogen. One possibility would be to state that not more than 25% of the allowable manure application should be applied in the fall. This would require a nine-month storage capacity.

3) A distinction be made between coarse- medium- and fine-textured soils with the maximum permissible rate on coarse-textured soils being one-half and on medium-textured soils being three-quarters of that on fine-textured soils.

4) That attention be given in the very near future to completing revisions of the "Guide to Rural Land Use and Farm Practice". We do not have a good understanding of the degree to which the current guidelines are being applied. Our impression is that it is primarily the odour issue that is recognized in applying the code of practice, ie. distance from residential areas, rather that the land requirement. We recommend that the need to apply the guidelines on rates of application be re-affirmed and strengthened.

7.0 RESEARCH NEEDS

The understanding and technology exist to control many of the water quality impacts of animal waste. Limiting access of livestock to streams, provision of adequate manure storage and handling facilities, adherence to guidelines on spreading etc. would greatly reduce the impact. These aspects have been discussed in a recent report prepared by the Upper Thames River Conservation Authority (Hayman, 1989).

There are several areas, however, where understanding and technology are not adequate. These are outlined in this section in order of priority. The general approach that would be most suitable is indicated along with some indication of appropriate funding and research agencies.

Research Priority No. 1. **Relation Between Environmentally Safe and Most Profitable N Rates.**

Problem

Recommendations for N fertilizer in Ontario are currently based on obtaining the greatest economic return (yield response). There is not, at present, a reliable soil test for nitrogen although preliminary work at Guelph suggests a spring $N0_3$-N may be reliable. Recommendations are now based on the crop to be grown and management practices including manure application. The assumption implicit in much of the literature, and in the Agricultural Code of Practice, is that if N applications do not exceed the recommended amount, leaching of $N0_3$-N will not exceed acceptable levels. This assumption has not been adequately tested. If it is valid, development of a reliable soil test in conjunction with analysis of manure before application would provide adequate protection of groundwater from $N0_3$-N contamination. If the assumption is not valid, a more detailed system of establishing acceptable manure application rates such as that proposed in section 5 would be required. The highest priority in research on livestock waste should be given to testing this assumption in association with the development of a reliable soil test for N.

Desired outcome: The establishment for various soil/crop systems, of the relation between the most profitable rate of N based on a soil test and the amount of $N0_3$-N contributed to groundwater.

Research Approach. A series of field plot experiments representative of major soil characteristics and cropping systems would be required. Variable rates of N, either as inorganic N or a combination of manure and inorganic N would be applied to the same plot each year for several years. The crop response would be used to calibrate the soil test. Contribution of $N0_3$ to groundwater would be monitored and related to the N application. We anticipate that with some soil/crop systems the most profitable rate will also be an environmentally safe rate but with others it will not.

Action Agencies: This research should be supported principally by the Ministries of Agriculture and Food and the Environment. Because of the long-term nature and intensive sampling and analysis involved, a well established agricultural research group such as at Guelph or the Colleges of Agricultural Technology would be the most appropriate agency to conduct the experiments.

Research Priority No. 2 **Establishment of System for Predicting Environmentally Safe Rates of Manure.**

Assuming that under at least some soil/crop systems, the most profitable rate of N will exceed the environmentally safe rate, research should be initiated to develop a more elaborate system such as that described in Section 5 and Appendix B.

Desired Outcome: A reliable, systematic approach to establishment of environmentally safe rates of manure application that can be used in development of regulations regarding land requirements and in development of recommendations for on-farm manure management.

Research Approach: A more detailed development of the approach outlined in Section 5 and Appendix B should be undertaken to establish more clearly the research gaps. There are, however, five areas that definitely require further effort.

a) <u>Prediction of denitrification of manure-N</u>.

Nitrate may be converted to N_2 gas via the denitrification process. When manure N is applied to the soil, the ammonium component is converted to nitrate. In addition carbon substrate is added in the manure which would provide bacteria with the necessary substrate to carry out the denitrification process. This can be contrasted with fertilizer N where carbon is not simultaneously added. Thus if oxygen supply is restricted by water saturation, for example, manure will enhance nitrate transformation to gas and thereby decrease the nitrate available for leaching. There is a need, however, to determine the extent of this process in quantitative terms so that the predicted contribution of manure to nitrate leaching may be improved.

b) <u>Mineralization/immobilization of manure N</u>

The relative quantities of ammoniacal and organic N in manures can be quite variable depending on source and management. The ammoniacal fraction is considered to be as available to crops or eventually for leaching as fertilizer N. Limited information suggests that the availability (mineralization) of the organic N fraction is quite variable depending on both manure and soil characteristics. Moreover, the ammoniacal and organic N fractions may interact in mineral ization/immobilization processes. This, in time, would affect nitrate availability for crops or leaching. Further, a time factor is involved whereby immobilization of N in manures with low ammoniacal content may be dominant shortly after application whereas mineralization gradually becomes dominant. Thus the overall release of mineral N (including nitrate) may be affected by ammoniacal and organic N composition. Research is needed to evaluate this process quantitatively.

c. <u>N transformation and losses in manure systems</u>

Limited research done in Ontario and elsewhere indicates that manure management in livestock facilities or in the field can greatly affect N transformations and losses thereby affecting available N for the crop or for leaching and/or denitrification. The loss of ammonia by volatilization is the most important process in management of manure until it is incorporated into the soil. Further research is needed to define more accurately the extent of losses and associated N transformations that occur with various manure management practices. This information would contribute significantly to the estimates required for the approach suggested in Sect. 5 and Appendix B.

d) <u>Effect of hydrological factors on the transport of nitrogen to groundwater</u>.

The concentration of nitrate-nitrogen in water seeping from the root zone and entering the watertable is determined by processes in the root zone that regulate the amounts and forms of nitrogen there. Some of these processes, especially mineralization, nitrification, and denitrification are controlled in part by the amount of soil water present, a hydrological factor. In addition the annual discharge-weighted concentration of nitratenitrogen in recharge to groundwater can only be determined when the amount of recharge to groundwater with each event is known. For these two reasons good information on the amount of soil water storage and of the amount of recharge to groundwater must be available on a year-round, day-to-day basis for all soils and crops to which manure is applied. There is considerable information in Ontario on soil water storage conditions during the growing season for well-drained soils. There is much less information on conditions during the winter and spring periods and for locations with shallow depth to watertable. There is very little collected information on the patterns of recharge to groundwater for different soils and climate zones on a day-today basis within seasons although estimates of annual total amounts are available. In order to prepare good recommendations for locations and amounts of manure application systematic data collection and analysis focussed on soil water conditions and amounts of groundwater recharge are required.

The research should produce a solid base of data on temporal patterns of soilwater status and rates of groundwater recharge that can be used to select appropriate controls on manure placement. Much of the use of the data will be to validate and calibrate models that predict nitrogen transport to groundwater.

The steps required to obtain these data are first to assemble all available observations on soilwater status and rates of groundwater recharge that have been made in Ontario. The second step is to analyze the collected data and identify the major shortcomings of the data in terms of missing sets of soil/climate conditions and of detailed variation within seasons. The third step would be to commission data collection .to fill in the important gaps. This data collection would need to be done for extended time periods, a minimum of five years or long enough to cover a representative set of wet and dry years.

e) Validation of predictive models for nitroeen transport

In the longer-term, the objective should be to establish process Driented models to predict N transport. There are good mathematical model structures available that can be used to calculate the movement of nitrogen to and through groundwater from cropland that has received manure applications. The main needs in Ontario are to select a small number of appropriate models to test, to calibrate and validate these models for conditions of soil, crop, climate and application procedures that apply in Ontario, and to run the models to produce simulations of nitrogen loading to groundwater from varying patterns of applications of manure. The results from the models can then be used as part of the process of decision-making about appropriate procedures for manure application to land.

The research would consist of an assessment of available modelling procedures, a selection of a small number of the procedures, the testing of these procedures through calibration and validation against measured nitrogen movement, and the use of the models judged best to predict nitrogen transport from selected patterns of manure application. In the selection of models to test the criteria would include: ability to directly.represent the most important processes, adaptability with least effort to Ontario conditions of manure application and climate, ability to produce simulations using only available input data.

Action Agencies: This research program will require the expertise of a range of scientists and research ranging from empirical data collection to research aimed at a better understanding of the processes involved. Expertise in development and testing of computer models will also be required. Although units such as Federal and Provincial research laboratories, consulting firms and Universities could participate it is critical to the success of the approach that the program be centrally coordinated.

Funding would come from several agencies including OMAF, OME, Agriculture Canada and NSERC.

Research Priority No. 3. **Macropore Transport of Dissolved and Particulate Pollutants.**

Problem: There is considerable evidence to suggest that transport rates of dissolved or suspended material through some soils are considerably greater than would be predicted if all of the water-filled pore space was involved in the transport. Movement of material through large continuous macropores, channels or cracks is thought to be responsible for this behaviour. Macropore transport may be most significant for phosphorus, bacteria and organic carbon associated with suspended soil material, but it is also significant for leaching of $N0_3$-N. Research is needed on the occurrence and importance of preferred flow of these materials through soil under different. tillage and cultural practices.

Desired Outcome: An ability to predict, for a given soil/management system, the degree to which macropore transport will occur.

Research Approach: Detailed transport studies using conservative tracers to map out flow paths through soil and their relationship to arrival times of manure constituents need to be carried out under a range of management (tillage, crop etc.) conditions.

Action Agencies: A range of agencies could appropriately fund this project since it has both practical and theoretical aspects.

Research Priority No. 4. **Manure Application in No-till Systems**

Problem: To make the maximum efficient use of manure 'and reduce the risk of direct transport of pollutants to surface waters, manure should be incorporated either during or immediately after application. However, tilling the soil reduces surface residue cover which in turn increases the risk of erosion. In a no-till system even partial incorporation using a chisel plough is not an option. Research is needed on manure application systems for no-till fields which incorporate manure with minimal disturbance.

Desired Outcome: An effective method for incorporating manure on no-till fields that is acceptable to producers.

Research Approach: Systems of direct injection are available but are largely untested. The application systems need to be tested with respect to their potential for reducing the risk of surface transport of both soil and manure constituents.

Action Agencies: Some research is being conducted under SWEEP but further research will undoubtedly be essential. This project could be conducted by innovative farmers with assistance from research scientists

Research Priority No. 5 **Potential Leaching of P to Groundwater from Longterm Intensive Manure Applications**

Problem: Most soils have a large capacity to adsorb P but it is finite. Serious concerns have arisen in Europe that significant leaching of P is occurring. Models are being developed to predict the time frame within which leaching of P may become an environmental problem. This will depend on both the adsorption capacity of the soil and depth to groundwater. Research is needed to assess this potential problem in Ontario.

Desired Outcome: A model to predict, for major soils in Ontario, the relationship between manure application rate over extended time periods and P movement in the soil profile.

Research Approach: Models currently available or under development should be evaluated and, if necessary modified, for use for this purpose. the necessary parameters should be determined for a range of Ontario soils and the model(s) tested under field and/or laboratory situations.

Because of the need to apply manure over an extended time period, this research will require a minimum of five years for completion.

Action Agencies: This problem is related primarily to a direct threat to water quality. Thus it would be most appropriately funded by Ontario Ministry of the Environment.

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APPENDIX A

RECENT AND ON-GOING ONTARIO STUDIES ON IMPACT OF LIVESTOCK MANURE ON WATER QUALITY.

1.1 Recent Studies by Conservation Authorities

1.1.1 Ausable-Bayfield

1. **Shaus, 1982: Water Quality Inventory.**
Watershed area 2500 km2.
-Water quality monitoring, 28 sites, monthly, 14 parameters.
-Found high P levels in summer, high bacteria loads at 3 sites. Parkhill Creek showed especially poor quality.
-Recommendations emphasize education. The fragmentation of pollution control responsibilities between government agencies was identified as a problem.

2. **Ryan, 1982: Assessment of Pollution Potential from Manure Handling and Storage.**
-Pollution potential identification by air photo. Ground truthing by farm interviews.
-The methodology was found useful, dependent on recent photos. Recommendations emphasize education, and remedial measure demonstration and promotion.

3. **Balint, 1984: Manure Management - Water Quality Program.**
-Continued pollution potential inventory by air photos and ground truthing.
-Found 4.3% of farms showed high pollution potential, 20% showed pollution potential. Liquid storage tank overflow was a common problem. Many farmers were unaware of the problem of pollution from manure.
-Recommended a long-term comprehensive program, to study the adequacy of recommended handling and storage methods, cattle access impact, and effectiveness of remedial programs.

4. **Balint, 1985: Water Quality of the Parkhill Creek Watershed.**
Watershed area 135 km^2.
-Water quality monitoring, 8 sites, and select tile drains.
-Fecal coliform counts exceeding guidelines occurred at all sites. 10% of drains showed contamination, 40% had high nutrient loads. Average P concentration and total coliform counts exceeded guidelines.
-Source identification, remedial measures and effectiveness monitoring were recommended.

5. **Palmateer and Huber, 1985: Lake Huron Beach Studies, 1984-85.**
-Monitoring of 10 streams draining to Lake Huron Beaches.
-Bacteria loads were linked to resuspension of stream sediment during rainfall events. Populations in sediment were much greater than populations in overlying water. Fecal bacteria survived 2 months in stream sediment. Contamination due to cattle access was demonstrated.

Salmonella loads were greatest during periods of heavy irrigation with liquid manure. Runoff was a major bacteria transport process.

6. Ryan, 1987: Manure Management Awareness Program.
-Water quality monitoring, 10 stations, weekly, 16 parameters, farm contacts, and extention.
-All streams had excessive organic matter, P, and bacteria loads. Solid manure storage runoff was identified as a pollution source. Cost was the major limitation to remedial action.
-Recommended studies of cost-effective remedial measures.

7. Hocking, 1987, 1988; Griffiths, 1988: Rural Beaches Strategy Program
-Parkhill Creek Watershed: 21.3 km^2.
-Development and implementation of bacterial contamination abatement strategy; and monitoring of effectiveness. Monitoring 5 sites, 14 parameters, biweekly. Tracer bacteria release study.
-Found remedial measures significantly reduced contaminant concentrations. Domestic septic systems may be contributing more P than are manure and milkhouse wastes. Tracer bacteria were detected throughout 16 km stream flow to Lake Huron.

1.1.2 Grand River

8. Hodgins and Koekkoek, 1987; Koekkoek, 1987; Wilson, 1988: Rural Beaches Program
-Several watersheds: Upper Conestogo, Upper Nith, Guelph Lake, Conestogo Lake.
-Water quality monitoring at 25 stations, bi-weekly, plus 10 Guelph Lake sites; information extention and remedial farm contacts; livestock pollution control planning.
-Stream fecal coliform and P loads commonly exceed guidelines. Livestock fecal waste has impacted water quality. Cost is the major constraint to remedial action implementation.

1.1.3 Grey-Sauble

9) Munk and MacDonald, 1986; Munk and Murray, 1988: Sauble River Watershed Beaches Impact Study
Watershed area 700 km^2.
-Water quality monitoring, 12 stations, weekly, plus select feedlot and storage sites throughout basin, 14 parameters.
-Bacteria quality standards exceeded at all stations. Livestock access was identified as a major factor; also barnyard runoff.
-Recommend identification of point sources, development of remedial plans; monitoring of improvements, and investigation of other watersheds in the area.

1.1.4 Lake Simcoe Region

10. Antoszek et. al., 1986, 1987: Rural Beaches Impact Strategy Study
Watershed area 420 km^2.
-Farm pollution potential assessment; farmer survey; monitoring of 16 stations, weekly, 15 parameters.

-Geometric mean bacteria counts do not increase in proportion to livestock numbers: management and sediment delivery factors are important. Fecal coliform counts exceeded limits at 10 stations, mean P loads at 2 stations. Fecal bacteria and P contamination were strongly correlated with number of farms with pollution potential. Most farmers did not calculate manure nutrient inputs in their fertilizer application program.
-Recommended remedial actions tailored to individual farms. Recommended revision of manure handling regulations.

1.1.5 Maitland Valley

11. Puddister, 1985: Belgrave Creek Non-point Pollution Control Project.

Watershed area 205 km^2.
Study to control bank erosion due to unrestricted cattle access, 1978-81.
Water quality monitoring, monthly.
Detailed inventory of creek bank; remedial installations at 5 sites.
Project considered successful due to interagency and landowner co-operation. Found evidence of reduction of suspended solid loads, and improved fish habitats.

12. Foran and-Fuller, 1988: Maitland Manure Management Program

Watershed area 502 km^2.
-Water quality monitoring, 24 stations, weekly, 13 parameters; inventory of farms with pollution potential; extension; implementation of remedial structures.
-Consistently poor water quality: fecal bacteria and total P. 13% of samples met bacteria standards, 7% met P standard. Three principle sources: Solid storage without containment, unrestricted cattle access to streams, and milkhouse waste.
More than 80% of farmers felt current management was adequate.
-Recommended development of provincial guidelines for by-laws and regulations. The need for similar studies of other watersheds was noted.

1.1.6 Metro Toronto and Region

13. Hindley et al, 1985: Upper Humber River Water Quality Study.

Detailed assessment of 5 areas of the Upper Humber River. Severe fecal bacteria pollution throughout the livestock watershed, in stream flow and in downstream sediment. Fecal counts correlated with pasturing season.

14. Hubbard et al, 1987, 1988: Metropolitan Toronto and Region Rural Beaches Impact Study.

Study of Bruce Creek, Centreville Creek and East Humber River watersheds.
Farmer survey and rural resident questionnaire; public education; demonstration farm.
Water quality monitoring included routine sampling at numerous locations, as well as special sampling.
Dry weather fecal coliform densities were generally below recommended limits, while wet weather levels usually exceeded limits.

Manure storage facility runoff control was found to be generally inadequate. Manure spreading is primarily for disposal purposes.
The importance of suburban bacterial contamination must be investigated.

15. **Rural Beaches Project Staff, 1988: A Water Quality Bibliography** Compilation of over 800 related references, with indexing by author and keywords.

1.1.7 Niagara Peninsula

16. **Laidley, 1988: Binbrook Reservoir Rural Beaches Study**
Watershed area 4140 ha^2.
-Water quality monitoring at 16 stations, bi-weekly, 14 parameters. Farm contacts, information extension.
-Total P exceeded guidelines at all stations. Fecal coliform counts exceeded guidelines at all stations upstream from the Reservoir. .
Farmers felt OSCEPAP II assistance was inadequate incentive to counter economic constraints to remedial measure implementation.

1.1.8 North Bay - Mattawa

17. **1987 Wasi River Watershed Management Study.**
Monitoring of total P loads in the Wasi River Basin.
Manure pollution locations identified, implementation Committee set up to see that remedial actions are taken.

1.1.9 St. Clair Region

18. **Quinlan, 1988: Upper East Sydenham River Watershed Study.**
Water quality monitoring at 8 stations, weekly, 15 parameters. Walking survey of the drain; landowner interviews.
-Elevated fecal coliform, fecal streptococci, P, and N levels were common. Some tile drains were identified as pollution sources. Also contamination from winter spreading suspected.
-Recommended development of pollution abatement strategies for identified sites. Also recommended additional monitoring.

1.1.10 South Nation River

19. **Armstrong et al, 1987: Payne River Watershed Study**
Water quality monitoring, 14 sites, monthly. Education, farm contact. -Fecal coliform counts, total P and dissolved oxygen demand were consistently above recommended levels. Direct stream access by livestock was common. -Further study is required to identify pollution sources.

1.1.11 Upper Thames River

20. **Demal, 1983; Graham and Knight, 1982a,b,c; Dickinson, 1982; OME, OMNR, OMAF, 1984: Stratford-Avon River Environmental Management Project**
Multi-agency comprehensive urban and rural water pollution study, 1980-1982, initiated by the City of Stratford. Extensive water quality monitoring at 16 sites, over 2 years.

-Bacteria, P, and sediment loads exceeded standards throughout the river. Livestock accounted for 12% of total P input. Chronic bacteria contamination was from livestock management, while periodic acute contamination was from spills or sewage by-pass. Cattle access sites showed elevated P, BOD, and geometric mean fecal coliform densities at 3 of 4 sites.
Elimination of cattle access to streams, and control of feedlot and storage site runoff would reduce P loads by 2 and 4%, respectively, and would reduce the chronic bacteria levels in rural parts of the watershed.

21. UTRCA, 1984; Glasman and Hawkins, 1985; Thornley and Bos, 1985: Pittock Watershed Sub-Basin Study.
Watershed area 240 km^2.
Water quality monitoring, 15 sites, bi-weekly, 13 parameters, plus 38 additional sites; 1983-84.
Study of manure spreading impact on tile drain effluent. Total P and fecal coliform counts consistently exceeded guidelines. Major impacts from erosion; manure, livestock and milkhouse waste mismanagement; and tile drains.
Identified potential for surface transport of manure runoff from 2/3 of farms. 40% of farmers winter spread manure.
Illegal tile drainage connections were found at 25% of farms.

22. Hayman and Merkley, 1987: Manure and Waste Management Program
Watershed area 1591 km^2.
Water quality monitoring, 7 sites, weekly and event, 2 years.
Inventory of farms with manure pollution potential.
16% of farms were found to have manure pollution potential.
Identified need for pollution potential evaluation, remedial plan development and cost analysis on by-farm basis.

23. Hayman and Merkley, 1987b; Hayman and Briggs, 1988: Rural Beaches Strategy Program.
Small scale sub-watershed remedial effects assessment, bacteria contamination source identification, bacteria survival studies.

Fecal coliform counts exceeded standards in three of five subwatersheds. Tracer bacteria survived in stream sediment throughout the summer season.

24. Hayman, 1989. A Clean Up Rural Beaches Plan (CURB). for Fanshawe, Puttoch and Wildwood Reservoir in the Upper Thames River Conservation Authority Watershed.

Report prepared for The Ontario Ministry of the Environment to identify the relative impact of pollution on reservoirs beach closures.

-Limiting livestock access to streams found to be a cost-effective remedial measure for control of bacterial contaminations.
-Manure spills /discharges can be the most significant pollution source on a farm.

25. **Hayman, D.G. 1987. Bibliography of selected reading on livestock related non-point pollution sources. Report Upper Thames River Conservation Authority.**

1.2 Ongoing or Proposed Studies

1.2.1 Government and University

Centralia College
Parkhill Creek Monitoring Project. Richard Brunke, OME and OMAF funding.

University of Guelph: E. Beauchamp, H. Whiteley
Manure Nitrogen Fate at Arkell, Elora and Ponsonby Res. Stns. Expected completion: Dec. 1990. OMAF funding.

University of Guelph: Beauchamp, Kachanoski, Elrick, Brown.
Management practices and nitrate contamination of groundwater. Agriculture Canada, Tobacco Diversification Plan. Completion:1992.

1.2.2 Conservation Authorities

Ausable-Bayfield Conservation Authority Proposed Study:
A practical study to determine the effects of Agricultural waste management on receiving water quality with respect to bacteria, P, N, and sediment loads.

Grand River Conservation Authority
Rural Beaches: final report in progress. 1989 funding applied for.

Grey Sauble Conservation Authority
Sauble River Beaches Impact Study.
Third year report and Curb Plan in progress.

Lake Simcoe Region Conservation Authority
Lake Simcoe Rural Beaches Impact Study (Pefferlaw Brook Watershed).

Maitland Valley Conservation Authority
Rural Beaches Project: Third year summary in progress.

Metro Toronto Region Conservation Authority

Load Translation Model.
Using PLOP loading model, and in-situ survival rates. Completion: fall 1989.

M.T.R.C.A. and Lake Simcoe Region C. A.
Bacteria Survival and Bio-tracer Project.
Study of survival characteristics of fecal indicator bacteria in streams, at six sites.

Moira River Conservation Authority
Fencing and Cattle Access Control Program. Through the Erosion and Sediment Control Program.

Niagara Penninsula Conservation Authority
Binbrook Reservoir Rural Beaches Project. Currently in third year of Study.

Otonabee River Conservation Authority
Indian River Study. Through the Rural Beaches Program.

St. Clair Region Conservation Authority
Bear Creek Watershed Water Quality Study. Spring 1989.
Proposed Lake Huron and St. Clair River Shoreline Studies.

Upper Thames River Conservation Authority
Rural Beaches Program: livestock and manure management promotion.

APPENDIX B

PLANNING PROCEDURE FOR DETERMINING AREA OF FIELDS REQUIRED FOR MANURE APPLICATION

Explanation of Approach

This procedure uses the concept of an "Animal Unit", the quantity of nitrogen excreted by a dairy cow and calf in one year ie 75 kg of N/yr. Other types of animals are given animal unit ratings based on the relative amount of nitrogen each animal produces per year compared to this standard.

The minimum land area on which the manure from a set number of animal units can be placed is found using a calculated maximum-allowable number of animal units per hectare of land used for manure placement. The equation that applies is

Minimum land area = (Number of Animal Units)/(Maximum Number of Animal Units/ha)

The maximum number of animal units whose manure can be placed on a hectare of land is calculated as the ratio NA/NM. NA is the maximum allowable application of mineral nitrogen in manure kg ha^{-1} yr^{-1} for a given combination of soil, crop, and watertable condition and NM is the annual amount of mineral nitrogen produced by one animal unit measured as mineral nitrogen in the soil after placement and incorporation in kg/yr.

The value of NA that applies to a particular field depends not only on the crop being grown but also on all the other factors that control the concentration of nitrate nitrogen in water leaching from below the root zone of the field and reaching the watertable below. Besides type of crop these factors include existing nitrogen content of the soil at the time of manure application, time-of-year of application, soil texture, amount of leaching water, depth to watertable and amount of fertilizer to be applied.

The mineral nitrogen content provided to the soil per year by an application of manure from one animal unit varies with the type of animal producing the manure, the collection and storage system, the period-length of storage and the methods of application and incorporation. The annual mineral nitrogen contribution per animal unit NM can be established through use of a table that sets out the proportion of the original nitrogen content of the manure as excreted that will be present at the time of application and the proportion of that nitrogen that will result in mineralized nitrogen in the soil after application. An example of the form that such a table would take is shown in Table 1 attached.

The allowable application amount of mineral nitrogen could be obtained from .a second table that sets out this amount as a sum of components of a nitrogen balance with allowable leaching amount taken from groundwater quality requirements. An example of the form such a table might take is shown in Table 2 attached.

Table 1: Factors to calculate the proportion of total nitrogen content of manure from one animal unit that is present as mineral nitrogen in the soil following application with allowance for effects of storage and incorporation procedures.

Coefficient Cl - Proportion of N remaining after storage

Storage System	Storage Time	Cattle	Poultry	Swine	Sheep	Horses
Solid	1 d					
	1 wk					
	1 mo					
	1 yr					
Liquid	1 d					
	1 wk					
	1 mo					
	1 yr					

Coefficient C2 - Proportion of N available as Mineral N

Spread as	Time to Incorporation	Cattle	Poultry	Swine	Sheep	Horses
Solid	1 hr					
	l d					
	1 wk					
	never					
Liquid	1 hr					
	1d					
	1 wk					
	never					

NOTE[1]: The quantity NM (amount of mineral nitrogen added to the soil from one application of the manure from one animal unit) is found as follows:

NM (kg/yr) = 75 x Cl x C2

NOTE[2]: One animal unit for various types of livestock is defined in the "Agricultural Code of Practice" or in the draft "Guide to Rural Land Use and Farm Practice".

Table 2: Factors used to calculate allowable amount of mineral Nitrogen (NA) for one annual application of manure.

F1 Net Nitrogen removed in crop (kg/ha)
(Removal less amount fixed~may be negative)

Crop	Yield Low	Average	High
Grain Corn			
Silage Corn			
Wheat			
Barley			
Soybeans			
Alfalfa hay			

F2 Nitrogen removed by denitrification(kg/ha)

Soil Type	Internal Drainage (dependent on topography) Below Average	Average	Above Average
Sand			
Sandy-Loam			
Silt-loam			
Clay-loam			
Clay			

F3 Net immobilization of nitrogen (kg/ha) (values may be negative)

Soil Type	C/N Ratio of Manure Below Average	Average	Above Average
Sand			
Sandy-loam			
Silt-loam			
Clay-loam			
Clay			

F4 Existing mineral N status (kg/ha)

Below Average	Average	Above Average
(+)	(0)	(-)

Table 2 (continued)

F5 Allowable leaching (k g/ha)

Soil Type	Net water supply (Precipitation-Evapotranspiration)			
	200 mm	300 mm	400 mm	500 mm
Sand				
Sandy-loam				
Silt-loam				
Clay-loam lay				

F6 Denitrification in Groundwater (kg/ha)

		Depth to Watertable	
Soil Type	>2m	2mto5m	>5m
Sand			
Sandy-loam			
Silt-loam			
Clay-loam			
Clay			

CALCULATION PROCEDURES

The maximum allowable application of mineral nitrogen in manure (NA) in kg/ha

NA = (Fl + F2 + 173) + F4 + F5 + F6 - (Fertilizer N to be applied)

- For April or May application use (Fl + F2 + 173) as tabulated

- For June July application use 0.5 (F1 + F2 + F3)

- For August September application use 0.2 (F1 + F2 + F3)

- For October November application use (Fl + F2 + F3) = 0