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Bacteria Release and Transport from Livestock Manure Applied to Pastureland

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Abstract. *A comparative field investigation was conducted on release and transport of bacteria from plots treated with cowpies, turkey litter, and liquid dairy manure. Rainfall conditions were simulated and runoff samples were collected to determine concentrations of E. coli, FC, and enterococcus present in runoff. The turkey treatment had the highest percentage of source bacteria released by rainfall, ranging from 1.3% for enterococcus to 14.5% for FC. The cowpie follows with percentages ranging from 0.3 to 0.6%. Runoff samples collected from the transport plots treated with cowpies averaged 137,000 cfu/100 ml for E. coli and over 165,000 cfu/100 ml for FC during two rainfall simulations. Bacteria concentration in runoff from plots treated with liquid dairy manure decreased between the two simulations, while the bacteria concentration from the plots treated with turkey litter increased. The percent of the bacteria that is initially released by rainfall that is transported to the edge of the field in overland flow was highest for the cowpie treatment (95 to 121%), followed by the turkey (41 to 138 %) and liquid dairy treatments (32 to 86%). Results indicated that among the animal waste types investigated, cowpies have the greatest potential to contributed E. coli, FC, and enterococcus to streams and waterways.*

Keywords. Fecal Bacteria, Agricultural waste, Nonpoint pollution, Land Application, Bacteria Release and Transport

Introduction

The transport of fecal bacteria from point and nonpoint sources to surface waters is becoming an increasing concern in the U.S. Elevated concentrations of fecal bacteria in drinking water can be detrimental to human health; potential diseases include Salmonellosis, Anthrax, Tuberculosis, Brucellosis, and Listeriosis (Azevedo and Stout, 1974). Approximately eight percent of U.S. river miles are impaired by pathogens (USEPA, 1998). A major source of fecal bacteria is runoff from agricultural land where manure has been applied or where animals are allowed to graze. Therefore, an understanding of the overland transport mechanisms for fecal bacteria can have a crucial role on the development of best management practices for reduction of pathogens concentration to surface water bodies.

The transport of bacteria in overland flow is affected by rainfall duration and intensity, method of manure application, fecal deposit age, and adsorption of cells to soil particles. Pathogenic organisms are largely retained at or near the soil surface (Faust, 1982), thus increasing the potential for pollution of surface runoff water. Because manure is less dense than soil, incorporating manure into soil increases the soil's interrill erodibility and thus the amount of bacteria detached by overland flow (Khaleel et al., 1979). Runoff from snowmelt or rainfall can carry bacteria from fresh manure into the stream. Doran and Linn (1979) found that runoff from a grazed pasture had fecal coliform (FC) concentrations 5-10 times higher than from an ungrazed pasture, but the FC counts in runoff from both the grazed and ungrazed pastures exceeded the water quality standard of 200 CFU/100 ml more than 90% of the time.

Thelin and Gifford (1983) placed cowpies on a platform and rained on them to determine the release of FC. Fecal deposits 5 days old or less released FC concentrations into the water on the order of millions of organisms per 100 ml. Fecal deposits that had not been rained on for up to 30 days released FC concentrations on the order of 40,000 per 100 ml. Larsen et al. (1994) placed bovine feces at 0.0, 0.61, 1.37, and 2.13 m from a runoff collection point to evaluate the release of FC. At the 0.0 m distance from the fecal deposit, the runoff bacteria concentrations corresponded to a release of 17% of the total FC in the manure, or between 40×10^6 and 115×10^6 organisms/ml. These values were significantly higher than those measured at the 2.13 m distance from the fecal deposit, where less than 5% of the organisms applied to the plots were present in runoff.

Computer simulation modeling is the primary approach used to develop Total Maximum Daily Loads (TMDL), even though insufficient data exist on several model input parameters related to the release and transport of fecal bacteria in runoff. Previous studies often focused on a single manure source and did not provide comparative results from different sources under similar climatic conditions. In addition, the detachment or release of fecal bacteria from land applied sources is not well-documented. Improvements in understanding the overland processes will improve modeling of fecal bacteria transport, and provide a basis for a more realistic evaluation of management practice implementation.

The overall goal of this study was to quantify the release and transport potential of three fecal bacteria indicators, *E. coli*, enterococcus, and fecal coliform (FC), from land applied manure during runoff events. The specific objectives of this study were to identify differences in bacteria transport among various livestock manures by comparing edge of field bacteria levels in runoff from pasturelands treated with liquid dairy manure, poultry litter, and cowpies. In addition, this study evaluated bacteria release rates for different types of manure applied to pasturelands with different history of previous manure applications. The data from this study will serve as a baseline from which the release and transport of fecal bacteria from agricultural watersheds to surface waters can be modeled.

Methodology

Field plots were constructed on existing pastureland in and around Blacksburg, VA. Two sets of plots were established; one set for the study of in-field bacteria release and one set for the study of bacteria transport. Release plots were used to measure available fecal bacteria concentrations in runoff. Four manure treatments (turkey litter, liquid dairy manure, cowpies, and none) and three land type treatments: pasture with a history of poultry litter application (Turkey Farm), liquid dairy manure application (Dairy Farm), and no manure application (Tech Research Farm) were studied. A total of 36 release plots were constructed for three replications of the four manure treatments and three land type treatments.

The transport plots were used to measure the concentrations of fecal bacteria present in overland flow at the edge of the field. The transport plots were only constructed at the Tech research farm due to the labor intensiveness of this component of the research. The release of bacteria from plots applied with liquid dairy, dried poultry litter, and standard cowpies were compared to control plots on which no animal waste was applied. A total of eight transport plots were constructed; two replications of each treatment (turkey litter, liquid dairy manure, cowpies, and control).

Plot Construction

Twelve release plots were constructed at each of the three sites for measurement of fecal bacteria concentrations available to runoff. Each release plot had the dimensions of 1 m by 1 m. Pre-fabricated steel borders were placed into the soil along the plot boundaries to prevent water movement into or out of the plots. Runoff drained through a small flume and was collected down-slope in a bucket. The runoff volume was determined by weighing the bucket.

Eight transport plots were constructed at the Tech research farm. Each transport plot was 3 m wide by 18.3 m long on an approximate 5.5 percent slope. Plywood borders were placed to a depth of 15 cm along the plot boundaries to prevent water movement into or out of the plots. A "V" shaped outlet was placed at the down slope end of each plot to direct runoff into a 0.15 m (6-inch) H-flume equipped with an FW-1 stage recorder for flow measurement. The FW-1 stage recorder recorded runoff depth continuously.

Animal Waste Collection and Application Methods

The state of Virginia requires phosphorous-based application of manure on crop and pasture lands. This method uses the residual phosphorous levels in the soil and the phosphorous levels in the manure to determine the manure application rate. The P_2O_5 application rates recommended for Orchardgrass/Fescue-Clover Pastures on soil productivity groups I and II (DCR 1995) were 90.7 kg/acre (81 lbs/acre) at the Tech farm and 0 kg/ha (0 lbs/acre) at the turkey and dairy farms, respectively.

Because the turkey and dairy farms have a history of receiving land applications of manure, the phosphorous levels were much higher in these fields. The Department of Conservation and Recreation (DCR) Standards and Criteria (1995) recommendation is that no additional phosphorous be applied to the pasture. Currently, the best solution is to apply the manure at a rate slightly lower than the estimated crop uptake, or to restrict manure applications to every other or every third year. Based on this approach, the experimental design was adjusted so that the manure would be applied to the plots at the rate of 56 kg P_2O_5 per hectare (50 lbs P_2O_5 per acre). Farm equipment used to spread manure cannot spread evenly or accurately if the application rates are too low.

Previous animal waste analysis reports were obtained from the DCR and from the farm managers. The previous analyses were used to estimate the phosphorous content in the dairy and turkey manure that would be applied to the plots. Based on the previous year's manure samples, the waste was applied to the plots at a rate of 56 kg P_2O_5 per hectare (50 lbs P_2O_5 per acre). Table 1 compares the results from the previous manure tests to those for the manure samples collected prior to their application to the plots.

Table 1. Concentrations of P_2O_5 in manure and the application rate and volume of the manure applied to the transport and release plots.

Manure type	P_2O_5 estimate based on samples from previous years	P_2O_5 estimate based on current waste samples	P_2O_5 applied to the plots	Application Rate	Transport Plots	Release Plots
Liquid Dairy	0.67 kg/1000 L	0.67 kg/1000 L	56 kg/ha	81,958.5 L/ha	450.1 L/plot	8.2 L/plot
Cowpie	2.0 kg/t	1.7 kg/t	50 kg/ha	29.4 t/ha	161.6 kg/plot (180 cowpies)	3.0 kg/plot (3 cowpies)
Turkey	20.4 kg/t	19.9 kg/t	54.7 kg/ha	2.8 t/ha	15.1 kg/plot	0.28 kg/plot

The dried turkey litter was collected from the Virginia Tech turkey barns. The litter, comprised of pine shavings and manure, was collected after a flock of turkeys were sent to market. The litter was stacked under a covered shed for a time period varying between 3 and 6 weeks before it was applied to the plots. The litter was uniformly broadcast onto the plots using small buckets.

The liquid dairy manure applied to the plots was obtained from the Virginia Tech Dairy manure storage pond. The storage pond contents are agitated twice a year, to suspend the solids that accumulate on the bottom of the pond. The manure was pumped into a tank and stored throughout the duration of the field experiment. The liquid manure was mixed in the tank before being drained into buckets and applied to the field plots.

"Standard" cowpies were constructed from fresh dairy cow deposits. Each cowpie was standardized by weight and shape, and randomly positioned by project personnel at various locations in the "cowpie" treatment plots. The size and shape of the "standard cowpies" was based on research by Thelin and Gifford (1983), who developed standard cowpies to study FC release patterns. The fresh deposits were formed by taking fresh manure and mixing it in a cement mixer for approximately 15 minutes. The manure was then placed in a mold with a diameter of 20.3 cm and a depth of 2.54 cm. Fecal deposits were placed in the mold until a weight of 0.9 kg was reached. The transport plots were divided into 1 m by 3 m sections. Approximately 9 cowpies were placed in each of the sections. A total of 360 cowpies were applied to the two transport plots. The three cowpies were randomly placed in each of the 1 m by 1 m release plots.

Rainfall simulation on Release Plots

A Tlaloc 3000 portable rainfall simulator, based on the design of Miller (1987), with a ½ 50WSQ Tee Jet nozzle was used to apply rain to the release plots. Rainfall simulations were conducted within 24 hours of the manure application. The plot was rained on until runoff occurred for 30 minutes. After 30 minutes, the rainfall simulation ended and the runoff sample was collected.

This rainfall simulator has been developed as the standard simulator used to test the phosphorous index in various states.

Rainfall simulation on Transport Plots

Due to the unreliability of natural precipitation for short-term field research, the Department of Biological Systems Engineering's rainfall simulator (Dillaha et al., 1987) was used to generate storm events to produce runoff from the field plots. Rainfall was applied at a uniform rate (approximately 4.45 cm/hour) to all pasture plots. A series of rainfall simulations was conducted within 24 hours after manure application. The first simulation (S1) lasted approximately 3 hours. The rainfall continued until a steady state runoff resulted. The S1 simulation represented the bacteria transport during dry field conditions. Before the second simulation (S2) began (approximately 22 hours after the end of the first simulation, S1), soils were saturated. This was due to an overnight natural rainfall of approximately 2.9 cm (1.15 in) and the long simulated rainfall event during the first simulation. Therefore, the second rainfall simulation represented the transport characteristics of bacteria under saturated soil conditions.

The uniformity of rainfall applications was measured using a network of volumetric rain gauges in and around each plot. The uniformity coefficient was determined for both rainfall simulations. The uniformity coefficients for the first and second rainfall events were 93% and 95.5%, respectively

Sampling and Analysis

The total runoff volume was collected from each of the release plots and weighed to determine the volume. The runoff was collected in buckets and a single sample was taken from the total runoff volume. A total of 32 runoff samples were collected from the release plots. Grab samples of runoff water were collected from the transport plots every 3 to 9 minutes during both simulated storm events. A total of 68 samples were collected during S1 and 68 samples were collected during S2.

Samples were analyzed, immediately after collection, for FC, *E. coli*, and enterococcus concentrations in runoff. The samples were analyzed using the Spread Plate (Clesceri et al., 1998) and membrane filtration methods (Clesceri et al., 1998 and EPA, 2000).

Statistical Analysis

The release plots were analyzed using a Generalized Randomized Block Design procedure. Tukey's pairwise comparison was used to test significance between the treatments at the $P<0.05$ significance level. Transport plots were analyzed using the repeated measure method (Ott and Longnecker, 2001). The response variable was the concentration of bacteria in the runoff leaving the plot. Tukey's pairwise comparison was used to find significance between the treatments at the $P<0.05$ significance level. The null hypothesis tested for both the release and transport plots was that there was no difference in the concentrations of the bacteria in surface runoff among the treatments.

$$\mu_{turkey} = \mu_{cowpie} = \mu_{liquid\ dairy} = \mu_{control}$$

Results and Discussion

Release Plots

The concentrations of bacteria and TSS in runoff from the release plots are presented in Table 2. The results from the Tech Research Farm are quite different than expected. The Tech Research Farm, which in the past had not received manure applications, had much higher concentrations in the runoff from the control plots compared with the other farms. This could be due to a higher wildlife population in the area and the lack of cattle to discourage wildlife, or to the build up of stable populations in the soil (Faust, 1982). The plots with the liquid dairy and turkey manure applications had lower concentrations of bacteria than the control. The cowpie plots, consistent with the other sites, had the highest concentrations of bacteria in the runoff. The turkey plots resulted in less suspended solids in the runoff than the control, which may partially explain the reason for reduced bacterial loading from these plots.

Table 2. Concentrations of enterococcus, fecal coliform, *E. coli*, and Total Suspended Solids from the release plots.

Tech Research Farm	**Enterococcus (cfu/100 ml)**	**Fecal Coliform (cfu/100 ml)**	***E. coli* (cfu/100 ml)**	**TSS* (mg/L)**
Liquid Dairy	17,000	35,000	23,000	86.0
Cowpie	285,000	159,500	152,500	71.5
Turkey	12,075	29,050	18,550	31.5
Control	23,000	29,350	21,300	37.5
Dairy Farm	**Enterococcus (cfu/100 ml)**	**Fecal Coliform (cfu/100 ml)**	***E. coli* (cfu/100 ml)**	**TSS (mg/L)**
Liquid Dairy	3,067	300,000	55,000	166.0
Cowpie	8,133	300,000	300,000	189.3
Turkey	1,880	92,000	28,000	134.7
Control	1	134	1	1.0
Turkey Farm	**Enterococcus (cfu/100 ml)**	**Fecal Coliform (cfu/100 ml)**	***E. coli* (cfu/100 ml)**	**TSS (mg/L)**
Liquid Dairy	1,867	47,667	30,667	110.7
Cowpie	1,007	65,000	37,000	131.0
Turkey	507	9,000	4,733	146.7
Control	23	167	1	51.7

*Total Suspended Solids

The plots at the Turkey Farm and the Dairy Farm had more consistent results. Figure 1 compares the average concentrations of *E. coli* in the runoff at the three different farms. The plots treated with cowpies had the highest *E. coli* concentrations in the runoff. In general, the plots treated with liquid dairy manure had higher *E. coli* concentrations than the plots treated with turkey litter. Statistical analysis performed on the treatments, which accounted for the different site locations, found statistical differences among all treatments except for the turkey treatment, which was not statistically different from the liquid dairy or control treatments.

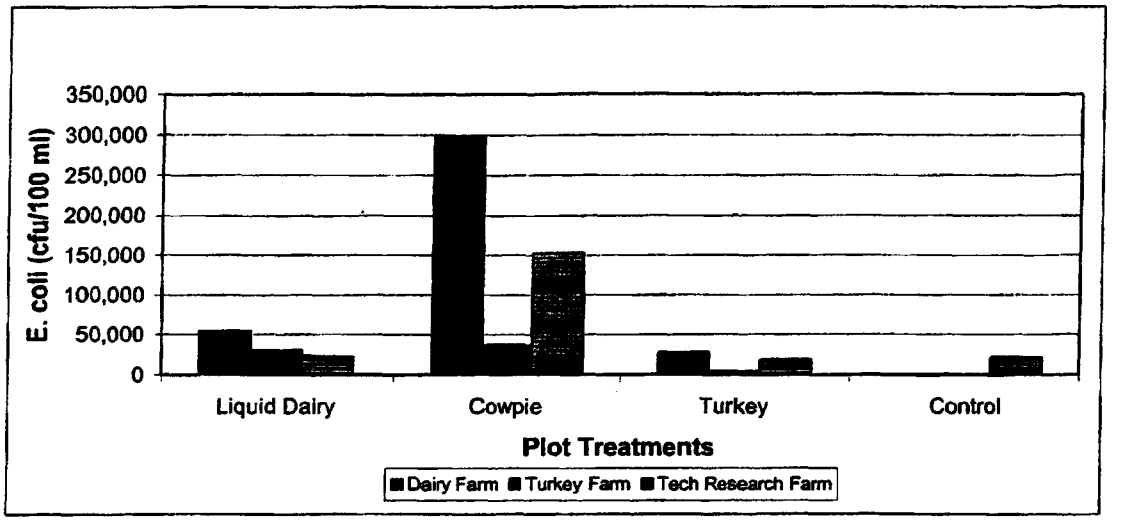

Figure 1. *E. coli* present in runoff from release plots

Figure 2 shows the concentration of FC in the runoff at the three different farms. The plots treated with cowpies, again, had higher FC concentrations in the runoff followed by the liquid dairy and turkey litter. Statistical analysis indicated significant differences among all treatments except for the turkey treatment, which was not statistically different from the control treatment. The FC release concentrations from the plots treated with cowpies ranged from 6.5×10^4 CFU/100 ml to 30×10^4 CFU/100 ml, which corresponds with the values reported in the study by Larsen et al. (1994) who reported the FC release concentrations from bovine feces were between 40×10^4 and 115×10^4 organisms/100 ml.

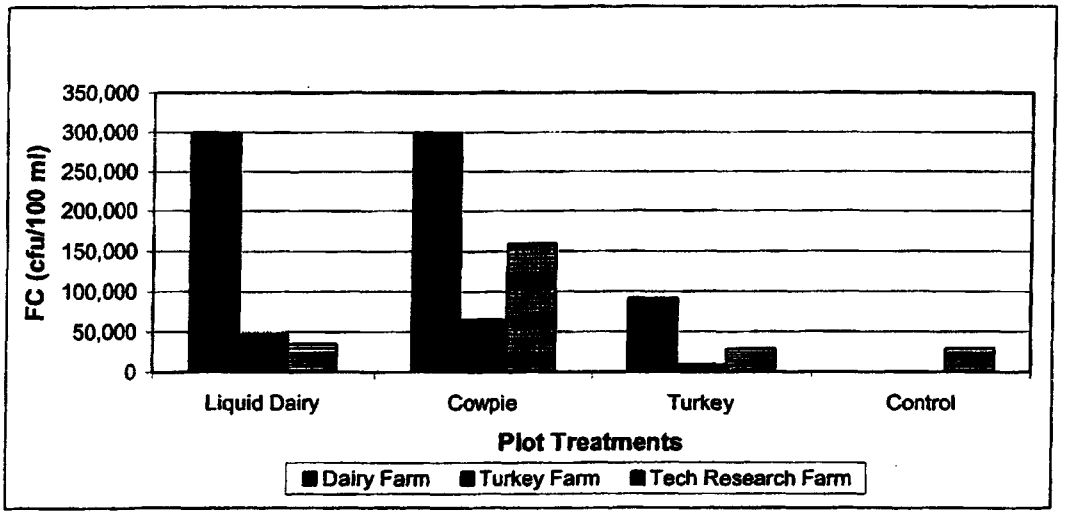

Figure 2. Fecal coliform present in runoff from release plots

In summary, the results from the release plots indicate that during a short but intense rainfall event, the cowpie treatment has the highest bacteria release rate. The liquid dairy treatment had a slightly lower release rate, followed by the turkey litter treatment.

By comparing the bacteria concentrations in the source manure to the average concentrations from the release plots, we were able to determine the percent of the source bacteria that is initially released by rainfall and would potentially be available to be transported to the edge of the field in overland flow. The bacteria concentration in the source manure is initially measured in CFU/gram. This was converted to CFU/100 ml to make the comparison. Table 3 shows the percent of bacteria released from the manure.

Table 3. Percent of bacteria that are released from the manure.

Manure Treatment	% Fecal Coliform released from waste	% *E. coli* released from waste	% Enterococcus released from waste
Liquid Dairy	0.3%	0.1%	0.0%
Cowpie	0.6%	0.5%	0.3%
Turkey	14.5%	5.7%	1.3%

The turkey treatment had the highest percentage of source bacteria released by rainfall, ranging from 1.3% for enterococcus to 14.5% for FC. The cowpie follows with percentages ranging from 0.3 to 0.6%.

Transport Plots

Runoff from the transport plots was measured continuously using FW-1 stage recorders. Figure 3 shows the runoff volume from each of the transport plots. Runoff volume increased during S2, due to the saturated ground conditions before the simulation began. The runoff from the plots varies due to differing soil conditions or compaction levels in the soil prior to the rainfall simulation. Runoff volumes also varied because the time at which runoff began differed among the plots. During S1, the plots treated with liquid dairy had the highest runoff volume, followed by the cowpie, turkey litter, and control treatments. During S2, the plots treated with cowpies had the highest runoff volume, followed by the control, liquid dairy, and turkey litter treatments. The predominant factor affecting runoff volume appears to be the time of between the beginning of the rainfall simulation and when runoff first occurred. The plots with earlier runoff times also had higher runoff volumes.

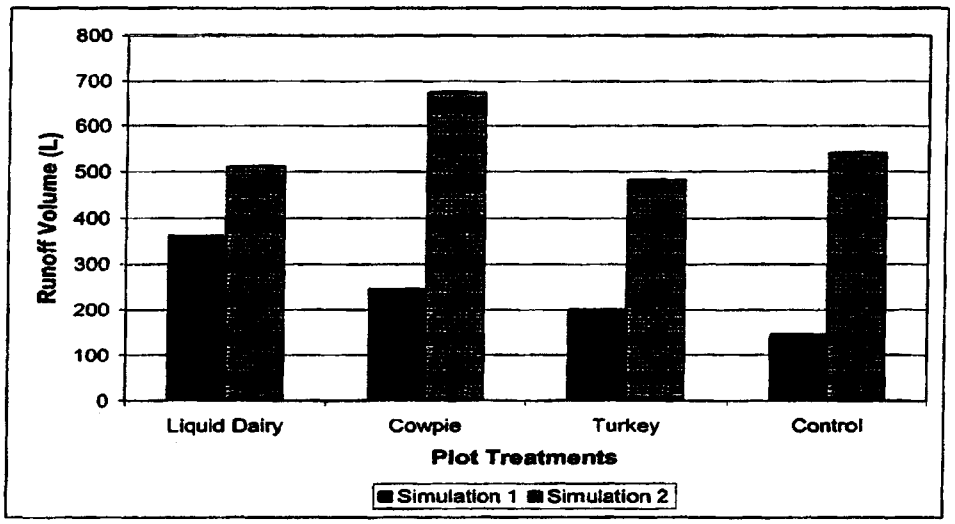

Figure 3. Runoff Volumes from the transport plots.

Statistical analysis was performed on the runoff volumes using the repeated measure method and Tukey's pairwise comparison. No statistical differences in the runoff volumes from the different treatments were found. There were also no significant differences between the runoff volumes during S1 and S2 simulations at the 0.05 error level.

The flow-weighted concentration (FWC) was calculated for the total suspended solids (TSS) in runoff from each of the transport plots (Table 4). The FWC was calculated by multiplying the sample concentration by the volume of runoff that occurred during that time period. These values were then summed and divided by the total volume of runoff from the plot. The addition of the manure to the plots decreased TSS concentrations from the liquid dairy and turkey plots when compared to the control. Gerba et al. (1975) reported that as bacteria and organic substances accumulate on the soil surface, the trapped bacteria become part of the filtration system, and increase the filtration properties of the soil. This may explain the decrease in TSS concentrations from the liquid dairy and turkey litter plots during the first simulation. The cowpie treatment covered just the areas where the fecal deposits were located, but not the entire plot area. The cowpies had higher moisture content than the other waste types, therefore it is possible that after the raindrop impacts disintegrated the cowpies, they were more readily carried off the plots by runoff.

Table 4. Total Suspended Solid concentrations present in runoff from the transport plots.

Treatment	Total Suspended Solids – FWC*		
	Simulation 1 (mg/L)	Simulation 2 (mg/L)	Average (mg/L)
Liquid Dairy	59.9	83.4	71.7
Cowpie	176.7	54.6	115.7
Turkey	37.3	22.5	29.9
Control	85.2	29.1	57.1

*Flow Weighted Concentration

The trends in the TSS concentrations were compared to the trends for the bacteria concentrations in the runoff from the transport plots (Table 5). In general, the plots treated with cowpies and liquid dairy manure had lower bacteria concentrations in the runoff during S2 than S1. The opposite occurred for the turkey litter, except for the enterococcus concentrations. The TSS concentrations, however, decreased during S2 simulations compared with S1 for the cowpies and turkey litter treatments (Table 4), but they increased for the plots treated with liquid dairy manure. These results indicate that higher TSS concentrations in runoff do not necessarily correspond with higher bacteria concentrations. The proportions of bacteria transported in the dissolved form and attached to suspended solids may differ among the different treatments.

Runoff data and sample concentrations from the transport plots were used to calculate the bacteria flow weighted concentrations. Table 5 presents the bacteria FWC for the transport plots for both S1 and S2 simulations.

Enterococcus concentrations in runoff were slightly lower than the *E. coli* and FC concentrations for all treatments. Enterococci are a subgroup of fecal streptococcus. Enterococcus is used as an indicator bacteria because it is often present in recreational water bodies when human illness occurs (USDA, 2000) and is most often used as a fecal indicator in marine waters. Federal standards for primary contact enterococcus is 33 CFU/100 ml. The concentrations reported in this study are much greater since they represent the edge of the field levels as opposed to in-stream concentrations. In-stream concentrations are expected to be lower due to dilution effects and die-off. The cowpie treatment had the highest FWC for both S1 and S2 events. The enterococcus levels in the runoff from the liquid dairy and turkey plots were slightly lower during

S2 compared with S1. Statistical analysis was performed using the repeated measure method and Tukey's pairwise comparison. No statistical differences in the enterococcus concentrations in the runoff from the different treatments were found. There were also no significant differences between the concentrations measured during S1 and S2 simulations at the 0.05 error level.

Table 5. Flow weighted bacteria concentrations in runoff from the transport plots for rainfall simulations S1 and S2.

Treatment	Enterococcus (cfu/100 ml)			FC* (cfu/100 ml)			*E. coli* (cfu/100 ml)		
	S1†	S2‡	Average	S1	S2	Average	S1	S2	Average
Liquid Dairy	9,341	3,179	6,260	74,073	6,817	40,445	31,294	5,526	18,410
Cowpie	187,406	50,465	118,936	234,288	96,045	165,166	200,047	73,235	136,641
Turkey	6,757	6,521	6,639	16,719	18,953	17,836	9,275	16,450	12,863
Control	6	2	4	51	36	43	16	11	13

*Fecal coliforms; †Simulation 1; ‡Simulation 2

Figure 4 presents the *E. coli* results in a graphical form. The *E. coli* FWC decreased for the liquid dairy and cowpie treatments during S2, compared with the S1 values. For the turkey treatment, however, the E-coli concentrations increased during S2. This increase can be partly due to the nature of the poultry waste. The liquid dairy and cowpies wastes are more easily transported, while the turkey litter is dry and may require a more significant runoff event to transport the litter off of the plots. The runoff from the cowpie plots clearly had the highest E-coli FWC for both simulations. Statistical differences were only found between the *E. coli* concentrations in the runoff from the cowpie and the control plots. There were no statistical differences in *E. coli* concentrations between the two rainfall events.

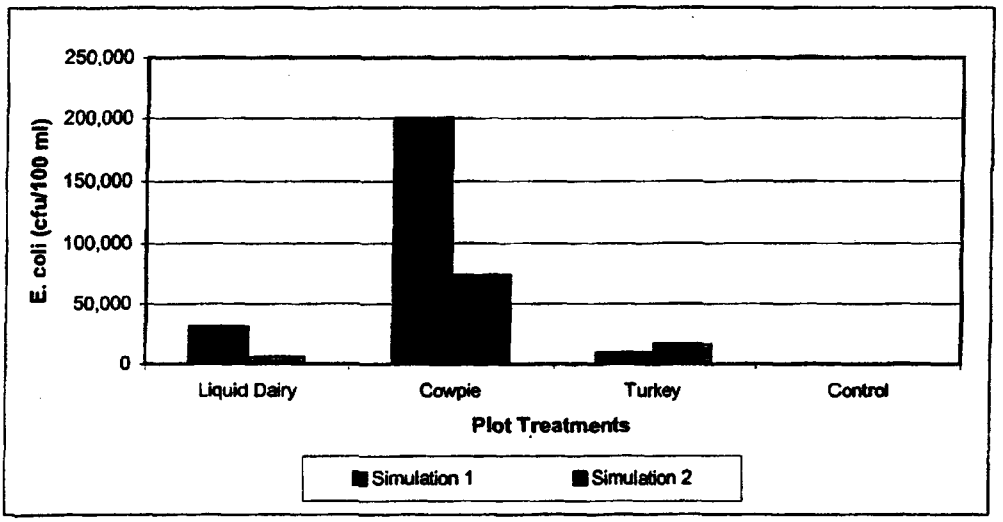

Figure 4. Flow weighted concentrations of *E. coli* in runoff samples from transport plots.

The concentrations of FC in runoff exhibited similar patterns as the E-coli among the different treatments. During S1, the liquid dairy and cowpie treatments had the highest average FC FWC. During S2, the cowpie continued to produce the highest FWC of FC, but the runoff from the plots treated with turkey litter had the second highest FWC, followed by the liquid dairy. The runoff FWC of FC from cowpie treatment were statistically different from all of the other

treatments. There were no statistical differences in FC FWC for each treatment between the two rainfall simulations.

Federal standards for primary contact for FC is 200 CFU/100 ml, much less than the levels present in runoff from the manure treated plots. Baxter-Potter and Gilliland (1988) reported that the typical range of FC present in runoff from pastureland were 1,000 to 57,000 CFU/100 ml. The average value for the two simulations from the pasture treated with cowpies in this study was 1.65×10^5 CFU/100 ml. The cattle stocking density is not provided in the previous studies, therefore it is not possible to compare the results. Furthermore, this study was designed to evaluate bacteria losses from edge of the field in small plots under intensive rainfall conditions. The bacteria concentrations reported in this study are expected to be much higher than those produced under natural rainfall from large pasture fields or watersheds. FC concentrations from grazed pasture in south central Nebraska contained concentrations of 1.21×10^5 CFU/100 ml (Schepers and Doran, 1980), which is similar to the results obtained from this study. Fecal bacteria concentration in runoff from grazed pasture is dependent on both the stocking density and the proximity of the cattle to streams. Cattle loafing in shaded or feeding areas produce high concentrations of cowpies in a smaller area and therefore higher bacteria concentrations in runoff. McCaskey et al. (1971) found FC concentrations to range from 1.4 to 21.7×10^6 CFU/100 ml in runoff from dairy waste applied to pasture plots by a tank wagon. They also reported that runoff from the control area had FC concentrations of 9.9×10^5 CFU/100ml. These values are much greater than the concentrations of 4.0×10^4 CFU/100 ml measured in our study.

To determine a relationship between the bacteria release and transport, the average concentrations from the release plots were compared to the average FWC from the transport plots. By comparing the concentrations from the release plots to the concentrations from the transport plots, we were able to determine the percent of the bacteria initially released by rainfall that is transported to the edge of the field in overland flow. Table 6 shows the percent of the released bacteria that is transported in overland flow.

Table 6. Percent of releas[illegible] bacteria that are present in overland flow.

Manure Treatment	% Released fecal coliform present in overland flow	% Released *E. coli* present in overland flow	% Released enterococcus present in overland flow
Liquid Dairy	31.7%	50.8%	85.6%
Cowpie	94.5%	83.7%	121.3%
Turkey	41.1%	75.2%	137.7%
Control	0.4%	0.2%	0.1%

The cowpie treatment had the highest percentage of released bacteria present in overland flow with percentages ranging from 95 % for FC to 121% for enterococcus. The turkey treatment follows with percentages ranging from 41% to 138%. The differences between the three species are related to the survival characteristics of the bacteria. Enterococcus is able to survive longer in the environment than FC and *E. coli*. The transport concentrations may be higher than the release concentrations because of background bacteria present in the soil.

In recent years significant changes have occurred in the livestock industry. Animal production areas are highly concentrated, resulting in more manure applications to the fields. In addition, the indicator organisms have changed over the years. Many previous studies provided information on total coliforms, fecal streptococcus, and FC concentrations in runoff. The State of Virginia is currently using *E. coli* as the primary indicator organism in fresh water and enterococcus as the primary indicator organism in marine waters (Virginia DEQ, 2002).

Previous studies rarely provide information on *E. coli* or enterococcus. Runoff from the transport plots treated with manure greatly exceeds the Federal Standards for primary contact.

Summary and Conclusions

Field plots were constructed on existing pastureland in southwest Virginia. Two sets of plots were established; one set for the study of in-field bacteria release and another set for the study of bacteria transport. The plots were treated with turkey litter, liquid dairy manure, and standard cowpies. Rainfall was simulated and runoff samples were collected to determine concentrations of *E. coli*, FC, and enterococcus present in runoff.

The runoff collected from the release plot treated with cowpies had higher concentrations of fecal bacteria indicators than those treated with liquid dairy manure or turkey litter. The turkey treatment had the highest percentage of source bacteria released by rainfall, ranging from 1.3% for Enterococcus to 14.5% for FC. The cowpie follows with percentages ranging from 0.3 to 0.6%.

The bacteria flow weighted concentrations in runoff samples from the plots treated with cowpies were over 200,000 CFU/100 ml of *E. coli* and almost 235,000 CFU/100 ml of FC. Runoff from plots treated with liquid dairy treatment had greater fecal bacteria concentrations in runoff during the first rainfall event (S1), which was applied within 24 hours after manure application. These concentrations however were reduced during the second rainfall event (S2), which occurred one day after the initial rainfall. During S1, the concentrations were 31,000 CFU/100 ml for *E. coli* and 74,000 CFU/100 ml for FC, but they decreased to much lower levels during S2 (5,500 CFU/100 ml for *E. coli* and 6,800 *CFU*/100 ml for FC). The turkey treatment resulted in the opposite effect. During S1, the bacteria concentrations remained low (9,300 CFU/100 ml for *E. coli* and 17,000 CFU/100 ml for FC, but increased during S2 (17,000 CFU/100 ml for *E. coli* and 19,000 CFU/100 ml for FC). This is most likely explained by the composition and transport characteristics of the waste. The percent of the bacteria that is initially released by rainfall that is transported to the edge of the field in overland flow was highest for the cowpie treatment (95 to 121%), followed by the turkey (41 to 138 %) and liquid dairy treatments (32 to 86%).

This comparative study clearly indicates that the cowpies have a greater potential to contribute fecal bacteria into streams than the land application of liquid dairy manure or turkey litter; although, runoff from all treatments exceed federal standards for primary contact. These results imply that areas where cattle may congregate, such as in watering or feeding areas, should be moved away from streams, and the buffer zone between grazing cattle and streams should be increased to reduce the loading of fecal bacteria.

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LINKING WATER QUALITY WITH AGRICULTURAL INTENSIFICATION IN A RURAL WATERSHED

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Abstract. Agricultural intensification was linked to streamwater pollution in a case study watershed using GIS and nutrient budgeting techniques. The results showed that surplus nitrogen applications from fertilizers and manure averaged 120 kg ha^{-1} yr^{-1}. In some parts of the watershed surplus applications exceeded 300 kg ha^{-1} yr^{-1}. A consistent increase in pig and chicken numbers (59 and 165% increase between 1986 and 1996) is considered the main reason for the surplus. Water quality was impacted in two ways: nitrate contaminated groundwater contributed to high nitrates in a major tributary during the summer, while in the wet winter season ammonia, phosphate and coliform levels were high throughout the drainage system. Significant negative relationships were found between surplus nitrogen applications and dissolved oxygen while ammonia and nitrate concentrations during the wet season were positively correlated to surplus applications. Soil texture and drainage type were also significantly correlated with the water quality indicators suggesting that it is possible to use the budget/GIS linked techniques for pollution risk assessment from agricultural non-point sources.

Keywords: agricultural intensification, agricultural pollution, animal waste, land-water interactions, nitrate, nitrogen surplus, nonpoint source pollution, water pollution, watershed management

1. Background

Agriculture is rapidly emerging as the greatest contributor of non-point source (NPS) pollution to streamwaters in North America (EPA, 1996; Kellogg *et al.*, 1994) and in many other parts of the world (Braden and Lovejoy, 1990; Isermann, 1990) where intensive agriculture occurs. While we have been relatively successful in controlling point sources of pollution, the NPS problem is far more challenging because it is very difficult to isolate the contribution from individually dispersed sources in a scientifically and legally defendable manner. One challenge is how to address cumulative effect over different temporal and spatial scales. This challenge becomes most evident in areas where both animal and crop production have intensified in the same area. Fertilizers are applied to crops in addition to manure because manure does not promote plant growth as effectively as fertilizers during critical periods of the growing season. Also, in areas with intensive animal feed units the disposal of animal waste becomes even more of an issue during periods when crops do not require extra nutrients. Animal waste is primarily applied to soils,

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which generally have a good capacity to absorb, store and slowly release nutrients. However, with continuous application of excess nutrients, the storage capacity of the soil is quickly exceeded and the nutrients find their way to streams and lakes. This leads to ammonia toxicity for fish, elevated nitrates in groundwater, eutrophication affecting the oxygen supply for aquatic biota, and microbial problems that impact water use and consumption (Cooper, 1993). Much has been written about groundwater (Spalding and Exner, 1993) and streamwater (USEPA, 1996) contamination from agriculture by excess nutrients but recent concerns about the human impact on the nitrogen cycle (Vitousek *et al.*, 1997), and eutrophication (Abrams and Jarrel, 1995; Daniel *et al.*, 1998) suggest that the problem is increasing. The use of nutrient budgets (Barry *et al.*, 1993) and GIS (Corwin and Wagenet, 1996) have been advocated as new tools in assessing the problem, and nutrient detention and removal by buffer zone vegetation is a popular mitigation practice (Lichtenberg and Shapiro, 1997; Jordan *et al.*, 1993; Hill, 1996). However in the long term, source control is likely the most cost-effective management option.

The goal of the article is to show how Geographic Information Systems (GIS) can be used in combination with a nutrient mass balance calculation to predict water quality conditions in a watershed context. The specific aims are to:

1. Document changes in agricultural land use and intensity between 1964–1995 using GIS in combination with digital aerial coverage, census data and farm surveys;
2. Illustrate annual surplus applications of nutrients using a mass balance calculation;
3. Document water quality conditions over an annual cycle; and,
4. Show relationships between surplus nitrogen applications, soil and site conditions, and water quality in the stream.

2. Study Area

The Sumas River watershed (Figure 1a) located in the Fraser River Lowland in Washington State and British Columbia, contains some of the best and most productive agricultural land in Canada. The main stem of the Sumas River originates in the coastal mountains in Whatcom County in the United States, and joins the Fraser River east of Abbotsford in British Columbia. The lower portion is dominated by a flat lake bed that was drained in the early 1930's and is known as the Sumas Prairie. A major aquifer (Abbotsford Aquifer) is located to the west and water from that source enters the Sumas River via Marshall creek (Figure 1b). The rainfall and runoff regime in the watershed is one of dry summers and a distinct wet period from November to March when low pressure systems from the Gulf of Alaska dominate the Pacific coastal climate.

Because of the favorable climate, excellent infrastructure, and proximity to

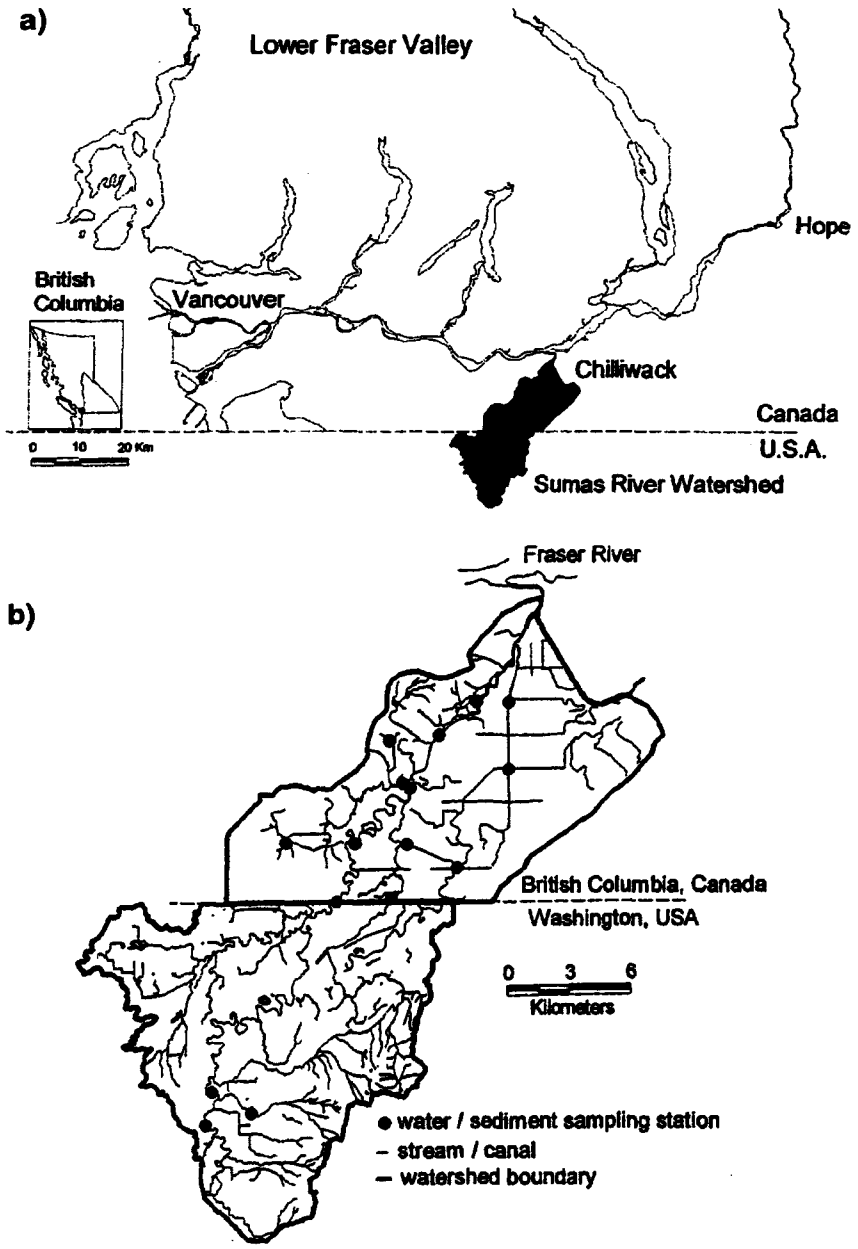

Figure 1. a) Location of Sumas River watershed. b) Stream network and water quality sampling stations.

the urban center of Vancouver the area has undergone significant agricultural intensification. There is considerable concern in this part of North America about sustaining salmonid populations and maintaining good drinking water supplies. Eutrophication of streams and nitrate contamination of groundwater are becoming major environmental issues, leading resource managers and regulatory agencies to continually seek assessment techniques that can help guide effective management practices. Given the differences in data availability and evaluation methods between the U.S. and Canadian portion of the watershed, the analysis of the Canadian portion of the watershed is presented in this study. A comparison between the US and Canadian portion of the watershed is provided elsewhere (Berka, 1996).

3. Methodology

The watershed was delineated using 1:20 000 scale digital terrain data (B.C. province TRIM database). A comprehensive georeferenced GIS database was developed that included: topography, digitized soil survey information, and land use. Land use was compiled using the 1995 digital orthophoto and from an analysis of historic aerial photos from 1954, 1963, 1979, and 1988 (scale 1:10 000). The land use and location of all farms in the Sumas Prairie were identified and digitized for each of the 5 different time periods. In addition, data from a waste management survey (WMS) of 130 farms (IRC, 1994) was used to arrive at animal stocking densities. The agricultural census data for 1986–1996 was used to document agricultural intensification. All information was georeferenced and incorporated into the GIS database. Database queries and GIS overlay analyses were used to examine changes in land use and in the intensity of human agricultural activity within the watershed.

Nutrient mass balance calculations were carried out using the model developed by Brisbin (1995). All sources (i.e., airborne, fertilizers, manure, biological conversion) and sinks (i.e., manure exports, volatilization, uptake by crops and organisms, denitrification, and management losses) were determined for each farm and each contributing area, or subcatchment. The contributing areas were delineated based on topography and location of water sampling stations. The arable land area within each contributing area was determined from the GIS land use analysis. In the mass balance model, manure nitrogen production was calculated by multiplying the nitrogen production rate in kg yr^{-1} for each type of animal by the number of animals on each farm. Based on the farm survey data it was estimated that 30% of broiler manure was exported from the watershed. Nitrogen losses to the air, land and water were determined using manure management conversion factors developed by Brisbin (1995) for each animal type, and different nutrient uptake rates were used for each crop type. Finally, a 9 kg ha^{-1} yr^{-1} rate was used for background atmospheric deposition and a 30% return rate was used for volatilized nitrogen due to management losses. A 10% loss was assumed to account for de-

TABLE I

Changes in land use and farm numbers between 1954 and 1995

Year	No. of farms	Area (ha)		
		Agricultural	Forested	Urban
1954	224	9687	4685	154
1963	233	9426	4645	416
1979	248	9647	4533	569
1988	271	9646	4452	757
1994	283	9751	4463	1542

nitrification of applied manure (Brisbin, 1995). Subtracting total losses from total sources provided surplus or deficit application rates in kg ha^{-1} yr^{-1}.

Water quality was determined at 16 stations shown in Figure 1b over the course of one year on eight different sampling dates. Dissolved oxygen (DO), pH, dissolved orthophosphate, nitrate-N, ammonia-N, and fecal coliform were the key water quality parameters measured. DO was measured with a YSI Model 57 Oxygen Meter, a Hanna digital pH meter was used to measure pH, orthophosphate, nitrate and ammonia were measured with the Lachat flow injection analysis, and fecal coliform was measured with the membrane filtration techniques. Given the high seasonal and diurnal variations the data was pooled for the wet season (November–March) and the dry season (June–August). Wet season and dry season average values were computed for each of the 16 sampling stations. These values were used to determine relationships between land use activities, surplus nutrient applications, site conditions in the watershed, and water quality. Spearman rank correlations were used to determine significant relationships between the variables.

4. Results

4.1. Land use dynamics

The historic land use in the watershed was determined between 1954 and 1994. As shown in Table I, the area under agriculture has remained constant over the entire period. This is mainly due to the introduction of the Agricultural Land Reserve (ALR) in 1973, that prohibits the conversion of highly capable agricultural land into other uses. While the area of agricultural land remained constant the number of farms increased by 26% over the same time period from 224 farms in 1954, to 283 in 1994. This increase in the number of farms on a fixed land base clearly reflects intensification.

TABLE II

Changes in animal numbers in the Sumas Prairie 1986–1996[a]

Year	Reported area farmed (ha)	No. of cattle	No. of pigs	No. of chickens	Expenditures in $ (lime + fertilizer)
1986	7405	18318	26049	323903	1,482,091
1991	7698	18535	38862	481542	1,698,707
1996	7596	18293	41329	859050	2,139,066

[a] Source Agricultural Census, Statistics Canada 1986–1996.

A second indicator of intensification is the increase in animal densities in the watershed. The agricultural census data for 1986, 1991 and 1996 showed that over the 10 yr period the cattle population has remained constant (–0.1%), while the number of pigs has increased by 59% and the number of chickens by 165% (Table II).

As a result of the NPS pollution problem, fertilizer application rates have generally leveled off or declined in many parts of the world (Hallberg and Kenney, 1993). However, this does not appear to be evident in the Sumas River watershed. No direct or adequate information was available on changes in the rate of fertilizer application. However, the fertilizer plus lime expenditures, as reported by the agricultural census and shown in Table II, have increased from $ 1.4 million to $ 2.1 million over the 10 yr period. This suggests that the total use of fertilizers has not declined in the study area.

4.2. Nutrient Budgets

Based on farm surveys it was established that about 30% of the broiler manure is exported out of the watershed while all other manure is applied to nearby arable land within the Sumas Prairie. Using the GIS database and Brisbin's model it was then possible to determine the nutrient inputs, crop uptake and losses for all agricultural areas in the watershed. The surplus or net application rates were determined using three different data sources for the total farmed and the total cropped area within the watershed: farm survey data, agricultural census data, and waste management surplus data published by Brisbin. All calculations resulted in similar overall surplus nitrogen application rates for the watershed ranging from 120–160 kg ha^{-1} yr^{-1} (Figure 2). The same calculation was also conducted for each of the contributing areas delineated in the GIS database. Figure 3 illustrates the spatial distribution of surplus N application rates, which in one contributing area exceeded 300 kg ha^{-1} yr^{-1}.

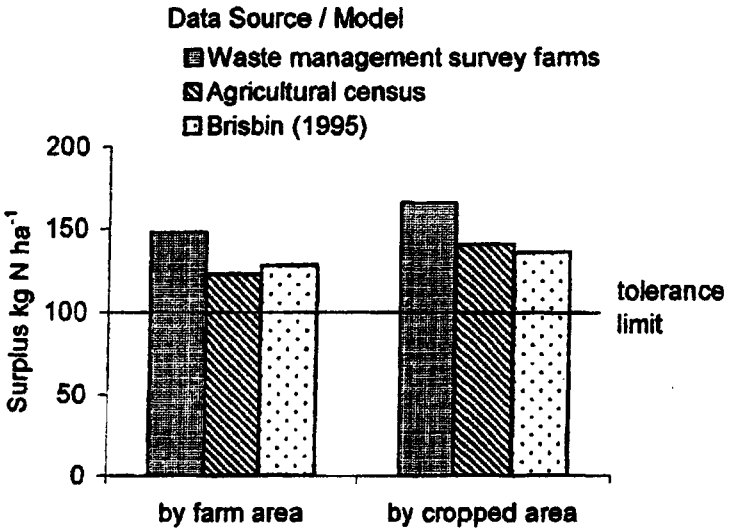

Figure 2. Overall nitrogen surplus applications for the Canadian portion of the Sumas watershed. Comparison between three different data sources (a surplus of 100 kgN ha^{-1} yr^{-1} is considered the upper tolerance limit for water quality impacts).

4.3. STREAMWATER QUALITY

With these excess application rates, the potential for streamwater contamination is high. Nitrate, dissolved orthophosphate concentrations and fecal coliform counts are displayed in the upstream to downstream direction for the wet and dry seasons in Figures 4, 5, and 6, respectively. Nitrate concentrations are of particular interest. Along the mainstem of the Sumas river elevated nitrate levels were measured during the winter months and lower concentrations during summer low flow conditions. In contrast a major tributary (Marshall Creek) showed the opposite trend, with high nitrate values at the upper stations during summer low flow conditions and low values during the wet winter season (Figure 4). While the Sumas headwaters and other major tributaries originate on mountain slopes that fall steeply to the flat agricultural valley, Marshall Creek flows from the major unconfined aquifer in the region (Abbotsford Aquifer). As shown by Liebscher *et al.* (1992), this groundwater resource is heavily contaminated with nitrates from agricultural sources. In summer low flow conditions the main water source for the creek is groundwater, while during the winter dilution occurs with the addition of wet weather runoff.

Orthophosphate levels (Figure 5) exceed water quality criteria guidelines of 0.01 mg L^{-1} generally considered conducive to eutrophication in lakes, and wet season coliform counts (Figure 6) exceed the recreational use Canadian water quality guideline of 200 MPN 100 mL^{-1}. Some of the major tributaries of the

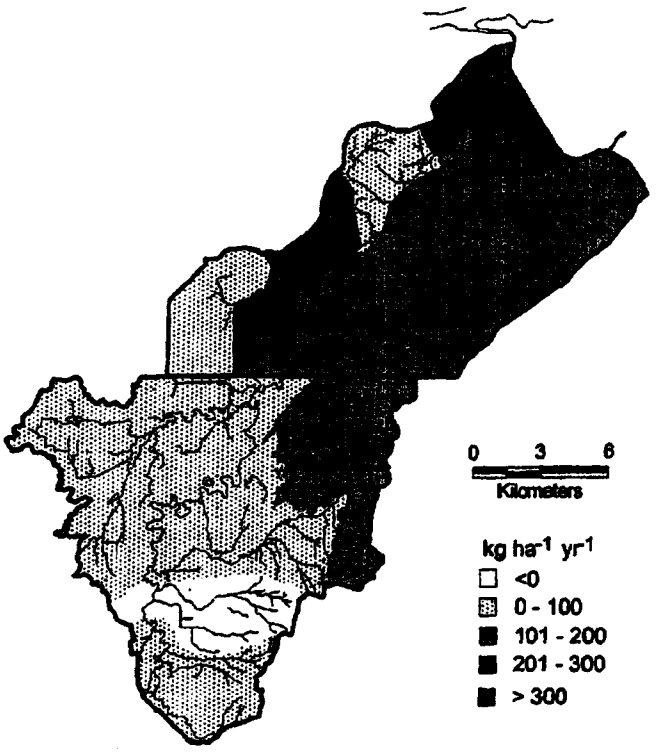

Figure 3. Spatial distribution of surplus nitrogen application in the Sumas River watershed (surplus values are in kg ha^{-1} yr^{-1} above crop needs and after subtractions of management losses).

Sumas River also showed significant dissolved oxygen and ammonia problems, particularly in autumn at the end of the growing season. At this time of year farmers apply large quantities of manure in order to have sufficient manure storage during the winter, when field applications are not possible due to wet soil conditions.

The effect of agricultural intensification on water quality over a longer period of time can be demonstrated by plotting historic wet season concentrations of nitrate-N, available only for the sampling station in the mid-section of the watershed (Figure 7). The plot shows the trend of an increasing spread of data values between 1970 and 1995, and a general trend in the upward direction. Although a clear trend is difficult to establish due to the scarcity of data, large data gaps and the variation of concentration with discharge, plots of chloride, other nutrient levels, dissolved oxygen, and pH showed similar trends in the direction of deteriorating water quality. These trends are indicative of the increasing types and intensities of land use activities occurring in the watershed.

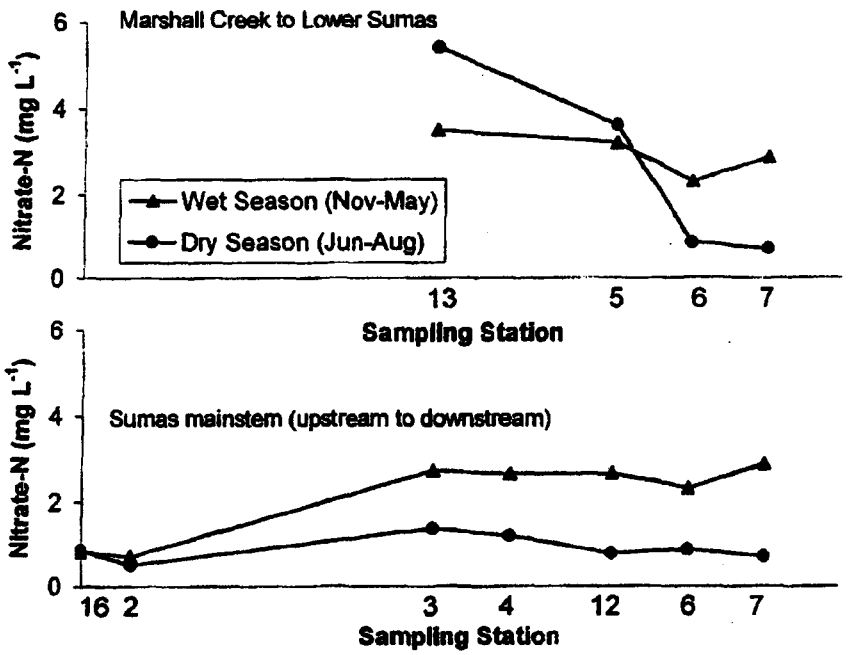

Figure 4. Comparison of nitrate variability in streamwater: Wet versus dry season, main stem stations versus Marshall Creek tributary stations.

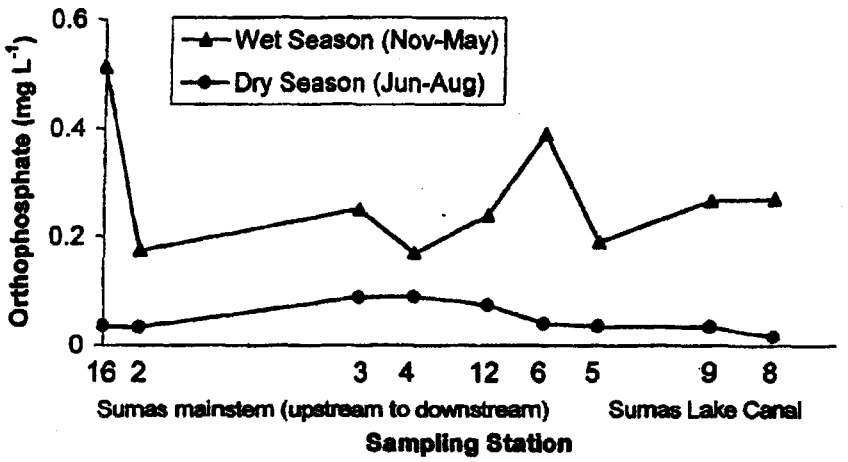

Figure 5. Differences in orthophosphate in streamwater: Wet versus dry season.

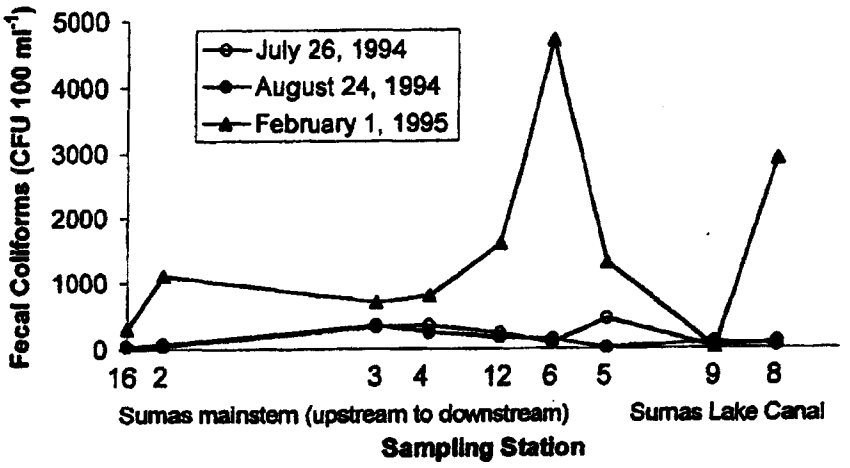

Figure 6. Differences in coliform counts in streamwater: Wet versus dry season.

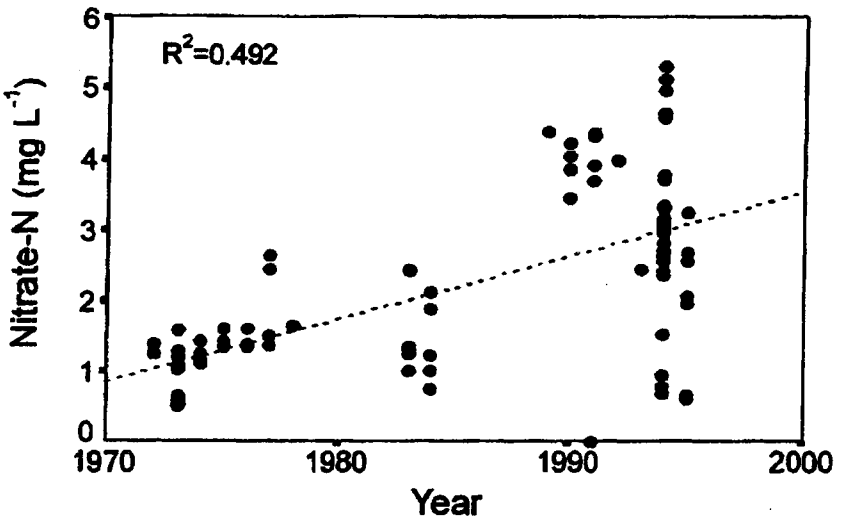

Figure 7. Historic changes in wet season nitrate concentrations in streamwater: 1970–1995.

Relationships between land use and water quality The Spearman rank correlation coefficient was used to identify relationships between the GIS based land use information and water quality. Land use was represented by various land indicator values calculated for each of the 11 contributing areas within Canada. Indicators that showed the best relationships are provided in Table III. Stronger relationships were obtained using wet season water quality indicators (ammonia-N, nitrate-N,

TABLE III

Significant relationships[a] between water quality, land use and site conditions ($p = 0.05$)

Land indicators	Ammonia-N	Nitrate-N		Dissolved oxygen	
	Wet season	Wet season	Dry season	Wet season	Dry season
Surplus N ha^{-1}	0.76			–0.84	–0.63
Pig density	0.67			–0.76	
% Loam texture	0.80			–0.89	–0.71
% Organic		0.75	0.74		
% Well drained				0.71	0.61
% Very poor drainage		0.82	0.83		

[a] $n = 11$, $p < 0.05$, and Spearman rank correlation coefficient shown in all cases.

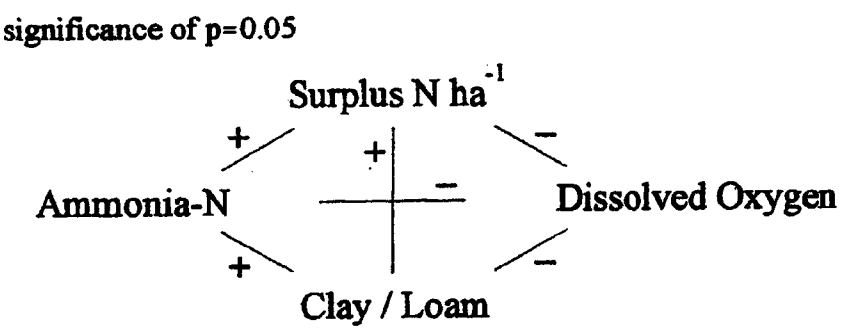

Figure 8. Wet and dry season relationships between surplus nitrogen application, water quality and site conditions.

and dissolved oxygen), surplus N applications, and site conditions (soil texture, soil drainage type, parent material, and animal density). Figure 8 represents the relationships found. Dissolved oxygen levels are negatively correlated, and ammonia levels are positively correlated, to surplus N application rates and the amount of finer textured soils within a contributing area. These results suggest that runoff during the winter is a key water quality problem in the watershed, likely influenced by heavy manure application rates at the end of the dry season and leaching from saturated soils.

The amounts of organic soils and the areas classified as poorly drained soils were also significantly related to nitrate values during the wet and dry seasons. This provides important information as to site vulnerability to N losses. Since N is much more dynamic than P it would be useful to examine if such relationships also occur with phosphorus. Sharpley (1995) suggests that dissolved orthophosphate values exhibited a poor relationship with the land use indicators and this may be explained

by the low solubility of phosphorus and its association with sediments. Sediments are mainly moved from land to the stream system via surface erosion from runoff and to determine its effect a different sampling design is needed. Monitoring of total and soluble phosphorus should be conducted during storm events, particularly during the late fall when soil surfaces are exposed and manure is being applied to the land to make room for winter manure storage. Access for land application is not possible in the winter because of saturated soil conditions. Since only one such event was captured at the beginning of the rainy season, significant relationships between land use and P could not be determined.

5. Conclusions

Agricultural intensification is leading to significant water quality problems in rural watersheds of the Lower Fraser Valley. Based on a case study in the Sumas River it was shown that the increase in animal units, and thus manure production and application, on a fixed land base is primarily responsible for the non-point source pollution problem. Over a 40 yr period the number of farms has increased by 26% and over the past 10 yr the number of pigs has increased by 59% and the number of chickens by 165%. Using a GIS database linked to a mass balance model it was determined that the overall surplus nitrogen application on the agricultural land in the watershed is 120 kg ha^{-1} yr^{-1} and reaches levels of more than 300 kg ha^{-1} yr^{-1} in the most intensively used area of the watershed.

Agricultural intensification is impacting streamwater quality, particularly during the wet winter season resulting in low dissolved oxygen, high ammonia and nitrate levels, and high fecal coliform counts. An evaluation of the historical nitrate concentrations in the watershed demonstrated that wet season levels have increased steadily.

A significant negative relationship was found between surplus N applications and dissolved oxygen and a significant positive relationship was found between surplus N and ammonia during the wet season. Similar relationships were found between these streamwater quality parameters and fine soil texture. Since soil texture is an important factor in the leaching process it is suggested that the GIS/budget technique can be used as, or contribute to, a risk assessment evaluation for streamwater pollution from agricultural non-point sources. Areas with high surplus applications and fine textured soils have the greatest risk of impacting streamwater quality.

It is also suggested that late summer/autumn land application of animal waste is likely one of the key sources of water pollution. Although some improvements to water quality may be achieved through manure management options such as increased storage and/or better timing of application, continued agricultural intensification will require that excess animal waste be processed or applied to nutrient deficient land outside the watershed in order to protect streamwater quality.

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Management effects on nitrogen leaching and guidelines for a nitrogen leaching index in New York.

by H.M. van Es, K.J. Czymmek and Q.M. Ketterings

Management practices may affect the potential for nitrate leaching from agricultural systems. Two studies are discussed that used plot-size lysimeters on loamy sand and clay loam soil in Northern New York. One was conducted from 1991 to 1994 and involved sod plowing and the use of three rates of fertilizer on maize (Zea mays L.). The other study was conducted from 1997 to 2000 and quantified N-leaching losses under maize and orchardgrass (Dactylis glomerata L.) as affected by the timing of manure application. These studies showed that timing and rate of N fertilizer and manure additions, timing of green-manure incorporation, and soil type strongly influenced N-leaching losses. Losses from fall-applied N sources were high, especially on coarse-textured soils. Lower N losses in fine-textured soils were primarily the result of higher denitrification losses, rather than reduced percolation rates. It was concluded that the current N Leaching Index ignores important processes and requires a more dynamic a pproach that includes management factors. In the interim, we established a set of best management practices for N to reduce the potential for N leaching losses.

Keywords: Manure, N fertilizer, nitrate leaching, N Leaching Index, soil type, timing of application

Great improvements in water quality were made from regulation of point sources in the 1970s, and the majority of states now cite agriculture as the primary contributor to water-quality impairments. Besides causing nuisance aquatic vegetation, nutrient contamination of surface waters has broader ecological impacts. Hypoxia, or dissolved oxygen depletion, is a significant problem in many freshwater lakes in the Northeast and has reached alarming proportions in several of the nation's estuaries and the Gulf of Mexico (ERS 1997). The U.S. Geological Survey found 1% of community wells and 9% of rural, domestic wells contaminated with [NO.sub.3]-N levels above the 10 mg [L.sup.-1] Maximum Contaminant Level (MCL, Mueller et al. 1995). The proportion of contaminated wells was as much as 26% in areas with land used for intensive agriculture. In the Northeast, groundwater nitrate contamination typically appears in localized areas and can often be related to intensive agricultural or urban land uses on coarse-textured soils (Poe et al. 1998).

Leaching of N fertilizers. [NO.sub.3]-N leaching under agricultural crops has been studied extensively, primarily through the use of suction lysimeters, monolith lysimeters, and subsurface drainage lines, with increasing amounts of spatial integration, respectively. Maize has been the most widely studied crop to determine the effects of agricultural management practices on nutrient losses (e.g., Baker and Johnson 1981, Kanwar et al. 1988, Kladivko et al. 1991, Randall and Iragavarapu 1995, Randall et al. 1997), generally showing higher nitrate levels in shallow groundwater with increasing fertilizer N levels. [NO.sub.3]-N levels were often well above the 10 mg [L.sup.-1] MCL under the optimum economic fertilizer levels as recommended by the state extension services.

A number of studies quantified [NO.sub.3]-N leaching potential under different crops (e.g., Robbins and Carter 1980; Bergstrom 1987; Owens 1990; Randall et al. 1997). In general, they found the highest nitrate-N levels under maize, intermediate levels under less-fertilized annual crops (e.g., soybeans (Glycine max L.) and wheat (Triticum aestivum L.) and the lowest levels under perennial crops (e.g., alfalfa (Medicago sativa L.) and grasses). In fact, [NO.sub.3]-N levels under the latter crops were generally well below the MCL. Besides changes in [NO.sub.3]-N leaching losses, soil hydrologic patterns also varied among crops. Randall et al. (1997) found drainage from row crop systems to exceed that from perennial crops by 1.1 to 5.3 times, primarily as a result of different timing of crop-water uptake and rooting depths. Bergstrom (1987) similarly found higher drainage under barley (Hordeum vulgare L.) compared with fescue (Festuca Rubra L.) and alfalfa. Therefore, the process of [NO.sub.3]-N leaching under di fferent crops involves complex interactions between soil hydrology, climate, crop-water and nutrient uptake, and management practices. Use of perennial crops is often suggested as an alternative to row crops when [NO.sub.3]-N leaching is of great concern (e.g., Schertz and Miller 1972, Meek et al. 1994, Randall et al. 1997, Yiridoe et al. 1997).

Leaching of manure N. Manure N includes a somewhat unstable component as urea in the liquid portion and a relatively stable organic N component in the feces, whose relative fractions may vary among manure types (Klausner et al. 1994). If

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manure is surface-applied and not incorporated, the urea converts fairly quickly to ammonium ([NH.sub.4]) through hydrolysis and ammonia (NH.sub.3]) as the pH increases and the manure begins to dry. It may then be lost by NH3 volatilization, depending on ambient conditions. If the manure is effectively incorporated, most of the urea is converted to ammonium and nitrate, thereby making it plant-available or subject to leaching or denitrification losses. The organic N component of manure mineralizes and becomes more gradually plant-available, typically represented by a decay series (representing the fraction of organic N that is available in each subsequent growing season, Magdoff 1978, Klausner et al. 1994). For New York conditions, it is assumed that 100% of the [NM.sub.4]-N is available to maize if applied as a late-spring sidedress (although this is perhaps a slight overestimation, Paul and Beauchamp 1995), 65% if applied as spring plowdown (reduced if incorporation is delayed), and 0% otherwise (Klausner 1997).

The decay series for the organic N fraction is 0.35-0.12-0.05-0.02 (unless high in dry matter). However, it is recognized that the rate of N mineralization is strongly affected by variations in soil, climate, manure composition, and management factors (Barbarika et al. 1985, Douglas and Magdoff l99l, Klausner et al. 1994, Jackson and Smith 1997). Estimates for mineralization of the organic manure N fraction are lower for manure applied to poorly drained soil or left on the surface (compared with manure incorporated on well-drained soil). Magdoff (1978) estimated that manure N mineralization rates on a poorly drained Panton clay were about half those on a well-drained Calais loam, although the lower net mineralization rates may actually be the result of higher denitrification losses.

Besides estimates of plant availability, these studies also provide insight into potential environmental losses. Since urea-to-[NH.sub.4] and [NH.sub.4]-to-[NO.sub.3] transformations may occur within a time period of several days (Kirchmann 1991), an incorporated fall manure application when soils are warm and crop uptake is nonexistent is likely to result in considerable nitrate leaching losses during the following winter and spring. Paul and Zebarth (1997) evaluated such leaching losses from fall-applied dairy cattle slurry on two soil types in coastal British Columbia (a poorly drained, coarse-textured soil and a well-drained, medium-textured soil, respectively) and determined them to average 40 kg [ha.sup.-1] (35.7 lb [ac.sup.-1]) above the no-manure treatment. Denitrification accounted for only 17% of the total nitrate losses and, therefore, was less significant than leaching. Smith and Chambers (1993) in England also determined that the application of high-N manures in the fall tends to result in excess ive [NO.sub.3]-N leaching losses and recommended against application during September to December. Manure applications also cease after Aug. 15 on the Dutch experimental farm, De Marke, which is managed to minimize [NO.sub.3]-N leaching to groundwater (Aarts 1996).

Early spring manure application may result in [NO.sub.3]-N release in advance of crop N uptake (Durieux et al. 1995) and may also result in leaching losses. Similarly, timing within seasons may have significant impacts on leaching potential. A late-fall application, when soil temperatures have decreased, may result in different N release patterns from early-fall application.

N-leaching losses from organic sources also strongly vary among cropping systems, although crops other than maize have not been extensively researched in the United States. Nonleguminous, cool-season, perennial hay crops have higher N demands (Klausner 1997), have longer active growth periods, and require different manure application schedules than maize. In addition, manure applied on grass is typically not incorporated, thereby reducing the availability of the urea-N fraction. Kaffka and Kanneganti (1996) measured greater crop response to manure application in a year with abundant rainfall than in a dry year with reported N recoveries as high as 60%.

Methods and Materials

Fertilizer study. During the period 1989 to 1997, we monitored nitrogen dynamics in plots on two sites that are <1 km (0.6 mi) apart at the Cornell University Willsboro Experimental Farm in northern New York. One site, with 16 plots of 14x14 m (45x45 ft, Figure lA), was located on a Cosad loamy sand (sandv over clayey, mixed, mesic Aquic Udorthent), while the other with 16 plots of 18x18 m (60x60 ft) was built on a Kingsbury clay loam soil (fine, mixed, frigid, Aeric Ochraqualf). Each plot was surrounded by 0.8 mm (0.031 in) impermeable PVC geomembrane to a depth of 1.8 m (6 ft, Figure 1B), which made the plots hydrologically independent and allowed for leaching studies using replicated treatment allocations. Each plot was underlain by a very slowly permeable clay layer and had one drain line (loamy sand site) or three parallel drain lines (clay loam site) at a depth of 0.9 m (3 ft), allowing them to function as plot-size lysimeters. The central drain lines exited into a manhole, facilitating water sampling (Sogbedji et al. 2000)

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From 1988 to 1991, a mixture of alfalfa (Medicago sativa L., cv. Oneida VR) and timothy (Phleum pratense L., cv. Climax) was grown on the clay loam site in a study to evaluate the effects of water management practices (Buscaglia et al. 1994) and the potential for preferential chemical movement (van Es et al. 1991). [NO.sub.3]-N levels in drain outflow (representing shallow groundwater quality) remained between 1 and 2 mg [L.sup.-1] (ppm) throughout this time period, in 1991, the loamy sand site was constructed into a preexisting grass sod.

Sods were plowed in September 1991 for the clay loam soil and in April 1992 for the loamy sand. During 1992 to 1995, maize was grown on the plots under three N fertilizer levels, 22, 100, and 134 kg [ha.sup.-1] (20, 90 and 120 lb [ac.sup.-1]), with 22 kg [ha.sup.-1] (20 lb [ac.sup.-1]) applied as starter and the remainder as side-dress N. The 100 and 134 kg N [ha.sup.-1] (90 and 120 lb [ac.sup.1]) treatments were replicated three times, while the 22 kg [ha.sup.-1] (20 lb [ac.sup.-1]) plot was replicated twice. The remaining eight plots at each site had obstructed drains and are not reported on here. Drain effluent rates were monitored continuously using 22.5[degrees] V-notch weirs combined with Telog Instruments high-sensitivity water-level depth pressure transducers and data loggers installed in the manholes (Soghedji et al. 2000). Water samples were obtained when drain lines flowed at time intervals ranging from 2 hours to 4 days, depending on flow rate.

Results. Intensive monitoring of crop N uptake, soil N dynamics, and groundwater [NO.sub.3]-N concentrations measured under the different N management systems provided insights into N-mass balances and the fate of N under these climate conditions (Sogbedji et al. 2000, 2001a, 2001b): Average [NO.sub.3]-N levels remained near or under the 10 mg [L.sup.-1] (ppm) MCL, even for the 134 kg [ha.sup.-1] (120 lb [ac.sup.-1]) rate, the highest Cornell University-recommended rate for maize on these soils (Figure 2). [NO.sub.3]-N levels were generally similar for the 22 and 100 kg [ha.sup.-1] (20 and 90 lb [ac.sup.-1]) N rates, averaging 3.8 mg [L.sup.-1] (ppm) for 1993 and 1994 under the clay loam, and 6.0 mg [L.sup.-1] (ppm) under the loamy sand. However, [NO.sub.3]-N levels significantly increased for the 134 kg N [ha.sup.-1] (120 lb [ac.sup.-1] rate, averaging 9.3 and 12.1 mg [L.sup.-1] (ppm) for the clay loam and loamy sand, respectively (Sogbedji et al. 2000). This suggests that with increasing fertilizer levels, N is efficiently taken up by the crop until a threshold value is reached (about 100 kg N [ha.sup.-1] or 90 lb [ac.sup.-1] in our case), after which uptake efficiency dramatically decreases. Residual soil nitrate levels and therefore nitrate leaching are then increased: 43% of the additional 34 kg [ha.sup.-1] (30 lb [ac.sup.-1]) N applied between the 100 and 134 kg [ha.sup.-1] (90 and 120 lb [ac.sup.-1]) rates accounted for in groundwater under the loamy sand in the wetter year, 1994 (Sogbedji et al. 2000).

Shallow groundwater [NO.sub.3]-N levels under the loamy sand were the highest during the fall (Soghedji et al. 2000), and were strongly correlated with residual soil [NO.sub.3]-N levels, as also measured for similar soils by Hack-ten Broeke and de Groot (1996). For the clay loam, groundwater [NO.sub.3]-N levels were more evenly distributed during the year. This suggests rapid post-season leaching of residual soil [NO.sub.3]-N in a coarse-textured soil.

Despite prominent differences in soil hydraulic properties, the two sites exhibited similar water-percolation rates during wet periods (early spring periods in all years and early summer of 1994, Sogbedji et al. 2000). During dry periods, differential soil retentivity resulted in varying water-percolation rates. Therefore, differential [NO.sub.3]-N leaching losses for the two soil types during both wet and dry periods cannot be explained solely based on water-percolation, but are also strongly affected by N transformation dynamics (mineralization, nitrification, deniritrifaction, etc.), and Fundamental differences in solute flow processes (notably the extent of multidomain flow) among these soil types (Sogbedji et al. 2001 a, 2001b). Indeed, the N-mass balance of data on soil N levels, crop uptake, fertilizer application, and leaching losses (Sogbedji et al. 2000) suggested that the clay loam site generally had much higher denitrification losses. Also, N-transformation rate coefficients were determined by cal ibrating the LEACHM model (Hurson and Wagenet 1992). This similarly showed higher denitrification rate coefficients for the clay loam soil (Table 1, Sogbedji et al. 2001b). This is also evidenced by the lower [NO.sub.3]-N concentrations for the clay loam in the wetter year, 1994, compared with the drier 1993 (Figure 2).

The highest groundwater [NO.sub.3]-N levels measured during this study occurred on the clay loam site after alfalfa plowdown in September 1991 (Sogbedji et al. 2000). Average flow-weighted [NO.sub.3]-N levels increased from 2 mg [L.sup.-1] (ppm) before plowdown to 10 mg [L.sup.-1] (ppm) within weeks after plowing to 16 mg [L.sup.-1] (ppm) during the next spring. This suggests that considerable [NO.sub.3]-N leaching losses occur from organic sources, especially if the timing of application/plowdown occurs out of sync with crop uptake. The fact that leaching losses associated with this practice resulted in the highest [NO.sub.3]-N leaching losses during the experiment merits attention, and it initially

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appears to contradict conclusions made by others (e.g., Schertz and Miller 1972, Smith et al. 1990, Meek et al. 1994, Randall et al. 1997) that alfalfa suppresses nitrate leaching. However, it appears that the organic N releases associated with crop transitions should be the primary concern. Indeed, researchers h ave found that alfalfa plowing often produces large amounts of [NO.sub.3]-N through N mineralization of residues (Bruulsema and Christie 1987, Fox and Piekeliek 1988). Additionally, the timing of such N release relative to the establishment of a succeeding crop (Campbell et al. 1994, Francis et al. 1994), periods of water percolation, and the occu. ence of N-transformation processes greatly influences the [NO.sub.3]-N leaching potential.

The plot-size lysimeters provided good information on nutrient dynamics in this study. The results clearly suggested that more research emphasis needs to be placed on the management of organic N, which is the main source of N in the Northeastern United States and the focus of the second study.

Manure study. The plot-scale lysimeters were again employed from October 1997 to October 2000 to investigate nutrient leaching as affected by the timing of manure application, soil type (clay loam and loamy sand), and crop type (maize and orchardgrass, Figure 1). Liquid manure was applied starting in fall 1997 using a Nuhn Industries (Sebringville, Ontario) manure applicator. Four application periods were scheduled under the maize crop for each soil type: early fall (target date Oct. 1), late fall (Nov. 1), early spring (April 15), and a split early spring application and late spring sidedress (April 15 and June 15). Manure containing about 1.3 kg of organic N and 1.0 kg of ammonium-N per 1,000 L (10.9 and 8.4 lb per 1000 gal, respectively) was applied at a total annual rate of 93,800 L [ha.sup.-1] (10,000 gal [ac.sup.-1]) and was disk-incorporated within 2 hours of application. The split application treatment on maize received 46,900 L [ha.sup.-1] (5,000 gal [ac.sup.-1]) twice. Each manure treatment allocati on was spatially balanced (van Es and van Es 1993) and replicated twice. Maize plots received an additional 30 kg [ha.sup.-1] (27 lb [ac.sup.-1]) fertilizer at planting and sidedress N based on the results of a Pre-Sidedress Nitrate Test (Magdoff 1991).

One set of three grass plots received 46,900 L [ha.sup.-1] (5,000 gal [ac.sup.-1]) of manure in early spring (April 15), and another set of three received that amount after the third cutting (Oct. 1). Plots not receiving manure at these times were fertilized with 71 kg N [ha.sup.-1] (65 lb [ac.sup.-1] using ammonium nitrate. Both treatments also received such quantities of manure after the first and second cutting, for a total of 140,700 L [ha.sup.-1] (15,000 gal [ac.sup.-1]).

Drain-water samples were collected weekly during low-flow periods when drains were flowing and at least twice a week during periods after each manure application (for appropriate plots) and during high-flow periods. Maize was harvested for silage, while orchardgrass was harvested through three annual cuts.

Results. Preliminary results of this study show that crop type and timing of application strongly influence N leaching potential (Table 2). On both sites, nitrate concentrations under maize were significantly higher than those under orchardgrass, which were at very safe levels relative to the MCL. This may be attributed to higher ammonia volatilization from the lack of incorporation, as well as high N uptake potential and longer active growing period for the cool-season grass.

Flow-weighted mean concentrations in drain outflow under maize showed consistent effects of the timing of application (Table 2). Concentrations for early fall applications on the loamy sand soil were highest at 23.44 rag [L.sup.-1] (ppm) [NO.sub.3]-N indicating unacceptably high N leaching losses. A one-month delay resulted in a 4 mg [L.sup.-1] (ppm) reduction in [NO.sub.3]-N concentrations, presumably caused by lower mineralization and nitrification potential in cooler soil. [NO.sub.3]-N concentrations under spring manure applications were approximately 12 mg [L.sup.-1] (ppm), and significantly lower than fall applications. This demonstrates the benefits of N application closer to the time of crop uptake. No differences were observed between early spring application and the split early-late spring application. A similar pattern was observed for the clay loam soil where [NO.sub.3]-N concentrations for each treatment under maize were 6 to 8 mg [L.sup.-1] (ppm) lower than the respective treatment on the loamy s and, although still in some cases above the MCL. [NO.sub.3]-N concentrations under orchardgrass on the clay loam soil were generally low with flow-weighted mean values under 3 mg [L.sup.-1] (ppm).

Treatment effects on groundwater [NO.sub.3]-N concentrations were generally consistent throughout the study period. However, temporal variations were considerable. Mean [NO.sub.3]-N concentrations during the fall and winter of 1999 ranged from 25.41 to 57.66 mg [L.sup.-1] (ppm) among treatments on the loamy sand site (data not shown), which is

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explained by a very dry 1999 growing season that resulted in high residual soil N levels subject to leaching during the following time period. This supports the notion that more attention needs to be paid to adjusting supplemental N applications based on weather conditions (Sogbedji et al. 2001c)

Results and Discussion

Management guidelines. The results of the fertilizer and the manure studies indicate that the environmental loss potential of N is strongly influenced by soil factors; weather; and amount, timing, and method of application. The associated processes are quite complex, but management guidelines can be established to help reduce N-leaching risk. The current Natural Resources Conservation Service (NRCS) N Leaching Index (based on precipitation and soil hydrologic grouping) separates soils of high and low leaching potential but ignores important processes such as management effects, preferential flow, and denitrification. A more dynamic approach is needed that allows for more accurate determination of N-leaching risk. This can be accomplished through the use of well-calibrated simulation models such as those used in this study (Sogbedji et al. 2001a, 2001b), and evaluating relative leaching potentials for various soil, climate, land-use, and management scenarios.

In anticipation of this, we have developed interim N management guidelines based on the current NRCS N Leaching Index that address the immediate concerns associated with certain N management practices, especially on soils with a high leaching potential. NRCS standard 590 categorizes N Leaching Index results as low (< 2), medium (2 to 10) and high (> 10). Producers are expected to implement best management practices if the Leaching Index score for a field is high (> 10). Producers are expected to consider these practices if the LI score for a field is medium (2 to 10). Based on the studies described above, these best management practices would be to:

* Avoid incorporating sod crops in the fall. Chemical sod killing may be carried out when the soil temperature at a 10 cm (4 in) depth approaches 8[degrees]C (45[degrees]F). Depending on location, this probably will not rake place until early October.

* Minimize fall and/or winter manure application on good grass and/or legume sod fields that are to be rotated the next spring.

* Plant winter hardy cover crops whenever possible, especially when fall manure is applied (e.g., rye, winter wheat, or interseed ryegrass in summer).

* Apply manure in the fall on a growing crop with discretion. Judicious amounts of manure can be applied to or in conjunction with perennial crops or winter hardy cover crops. Applications should generally not exceed the greater of 55 kg [ha.sup.1] (50 lb [ac.sup.-1]) of first-year available N or 50% of the expected N requirement of next year's crop.

* Note that frost incorporation/injection (van Es et al. 1998, van Es and Schindelbeck 2000) is acceptable when soil conditions are suitable, but winter applications should be made in accordance with the New York Phosphorus Index.

* Unless the New York Phosphorus Index identifies the need for P-based fertility management, base manure and fertilizer application rates on Cornell University guidelines for meeting crop N needs.

The last guideline is based on the results of our fertilizer study showing considerably higher N-leaching rates when N fertilizer in excess of the agronomic optimum is applied. Other best management practices associated with the current N Leaching Index are:

* For maize, pre-plant (other than starter fertilizer) and early post-plant broadcast applications of commercial nitrogen without the use of nitrification inhibitors are not recommended.

* Sidedress applications should be made after maize has at least four true leaves.

* If starter N must be broadcast (e.g., for small grains or new seedings of grass), apply fertilizer as close to expected planting date as possible (ideally within 3 days or less).

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* For row and cereal crops, including maize, maintain starter fertilizer N rates below 55 kg [ha.sup.-1] (50 lb [ac.sup.-1]) actual N under normal conditions.

* Manure and fertilizer applications should be adjusted based on information provided in Nitrogen Recommendations for Field Crops in New York, Cornell University Department of Crop and Soil Sciences Extension Series E01-4 (Ketterings et al. 2001).

* Evaluate the need for sidedress N applications based on PSNT or other soil nitrate-nitrogen tests.

* Appropriate ammonia conservation is encouraged. Losses can either be reduced by immediately incorporating manure or eliminated by directly injecting manure as a side-dress application to growing crops.

* Manure N application on legumes is acceptable to satisfy agronomic requirements when legumes represent less than 50% of the stand. When legumes represent more than 50% of the stand, manure may be applied at a rate not exceeding 165 kg [ha.sup.-1] (150 lb [ac.sup.-1]) of available N.

Summary and Conclusions

This study confirmed that timing and rate of N management and soil type are important factors affecting N leaching, and high-risk management practices can be identified, as also suggested by work of Paul and Zebarth (1997), Smith and Chambers (1993), and Randall et al. (1997). The current NRCS Leaching Index does not address the dynamic nature of the N-leaching process. A new N Leaching Index needs to be developed, based on mechanistic simulation modeling results, that addresses interactions of soil type with land-use and management practices. In the interim, guidelines for N management in New York have been developed, based on the current NRCS N Leaching Index, that address the concerns related to practices with high nitrate leaching potential.

[FIGURE 2 OMITTED]

able 1

Rate coefficients for nitrification and denitrification from LEACHM calibrations (Sogbedji et al. 2001b).

Soil type	Rate coefficient Nitrification	Rate coefficient Denitrification
	[d.sup.-1]	
Cosad Loamy Sand	0.391	0.004
Kingsbury Clay Loam	0.240	0.106

able 2

Flow-weighted mean groundwater [NO.sub.3]-N concentrations as affected by timing of manure application and soil type for the period fall 1997 to spring 2000.

Timing of Manure Application

	Early fall	Late fall	Early spring	Split application
		mg [L.sup.-1]		
Loamy sand				
Maize	23.44 a +	19.33 b	11.76 a	12.56 c
Grass	2.64 e		2.11 e	
Clay loam				
Maize	15.44 a	11.79 b	6.15 c	7.16 c
Grass	1.04 d		1.62 d	

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+ Means for crop-timing combinations within each soil type followed by the same letter are not significantly different at _=0.05.

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B

CESSPOOLS OF SHAME

How Factory Farm Lagoons and Sprayfields Threaten Environmental and Public Health

Author
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Natural Resources Defense Council and the Clean Water Network
July 2001

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C

Swine Manure Management Plans in North-Central Iowa: Nutrient Loading and Policy Implications

L.L. Jackson, D.R. Keeney, and E.M. Gilbert

ABSTRACT: Expansion of production in north central Iowa has occurred in dense clusters of confined feeding operations (CAFOs), which poses questions about how manure nutrients are distributed on the landscape and whether manure management regulations are sufficient to protect water quality. Public records were used to document the manure management practices of 10 CAFOs housing 59,700 finishing hogs in a 1,554 ha area of Hamilton County, Iowa. Together, the CAFOs generated an estimated 811,500 kg of nitrogen (N) each year, more than 70% of which volatilized into the atmosphere. CAFOs minimized the area required for applying manure by underestimating manure N content, projecting above average crop yields, and applying manure to soybeans. Some fields were claimed by more than one CAFO, and some field sizes were overestimated. Manure application based on crop demand for phosphorus would require 9,350 ha of cropland, compared to the 990 ha used by CAFOs. Several policy changes could alleviate the nutrient management problems inherent in CAFOs.

Keywords: Ammonia volatilization, CAFO, confined animal feeding operation, hog confinement, livestock concentration, manure management, nitrogen, nonpoint source-pollution, phosphorus, swine production

In recent years the farrowing and finishing of swine has undergone a remarkable transformation from small, on-farm operations to highly concentrated animal feeding operations (CAFOs). While CAFOs have apparent economies of scale, they are not necessarily the most economical of units. They demand a large infusion of capital, usually from outside sources. Ownership may be corporate or hogs may be raised on contract. This approach, patterned after the North Carolina model of the early 1990s, bears little resemblance to the traditional family farm (Schwab 1998). The dramatic change has created social and economic problems for farmers and communities that were adapted to the smaller systems. It also has aggravated problems of odor and nutrient management. Of particular concern is protecting the quality of surface water and groundwater and the integrity of the natural resource base of the countryside (Jackson et al. 1996).

Large scale facilities concentrate animal production in a small area. Most nutrients shipped in via animal rations (corn, soybean meal, feed supplements) must be returned to the land in the form of manure, but often there is not enough cropland within economic hauling distances to put manure back on the land at agronomic rates. Livestock concentration can lead to manure management practices that minimize the nutrient value of manure and increase the risk of nonpoint source-pollution (Lanyon and Beegle 1993; Hoag and Roka 1995).

Excessive nitrogen (N) and phosphorus (P) can lead to eutrophication of receiving water bodies, and high nitrates can make drinking water unfit for human consumption. Ammonia emissions from animal feeding operations are known to cause ecosystem damage by stimulating growth of nitrophilous plant species (Berendse et al. 1993). Although the nutrient problems associated with swine operations may occur regardless of the size of a swine operation, concentration of animals raises special problems, particularly with the quantity of manure and ammonia production.

In recent years, the Iowa swine industry has shared in the expansion of concentrated animal feeding systems. Development of the industry has concentrated in the dominant grain producing areas of central and north central Iowa. Soils in this region are dominantly recent till of the Des Moines lobe, formed under prairie and wetland vegetation. They are highly productive for rain fed grain crops. Corn and soybeans dominate the cropping systems. The soils have slow internal drainage, requiring that they be tile drained to maintain productivity. Because of changes in drainage patterns and the high levels of nitrate developing in the root zone, nitrate is rapidly transferred by tile drainage systems to surface water. There is little opportunity for N processing through denitrification. Most N and P for crop growth are supplied as fertilizers.

Between 1992 and 1997, three north central Iowa counties—Hardin, Hamilton, and Wright—experienced a 2.6 fold increase in hogs, from 610,600 to 1,602,800, and a 36% decline in the number of farms with hogs (National Agricultural Statistics Service 1999). Interestingly, from 1990 to 1997, the human population declined from 2 to 5% in these counties (Goudy et al. 1998). A short term study by Padgitt et al. (1998) indicated that expansion in swine production had not yet affected social and economic status in Wright County.

In the typical concentrated swine system, pigs are farrowed on sites with as many as 4,000 sows. The pigs are weaned at 4.5 kg (10 lbs) and fed to about 20 kg, then moved to finishing facilities. They are sold at 115-120 kg. Finishing facilities typically house 1,100 pigs per building, and buildings are arranged in groups of three to as many as 20. Buildings are ventilated and climate controlled. Swine are fed on concrete floors. Urine and feces are handled as liquids, using water to flush the material into storage lagoons or ponds.

Iowa and other states have felt it necessary to regulate manure storage and management practices. Iowa adopted manure management rules in May of 1995 and amended them in 1996 (Iowa Administrative Code Rule 567-65.1 [455B]; Iowa Administrative Bulletin 1996). The rules stipulate that swine CAFOs with an animal weight capacity of at least 90,800 kg submit a manure management plan (MMP) as a condition for obtaining a construction permit. The animal weight capacity is calculated by multiplying the maximum number of animals confined in an animal feeding operation at any one time (animal capacity) by the average animal weight during the production cycle. The MMP must project the manure N available after losses from ammonia

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volatilization and demonstrate that sufficient cropland is available for manure application at agronomic rates, based on N needs of the crops grown on this land. MMPs are evaluated by the state on a case-by-case basis without regard to regional concentration of livestock.

The recent pattern of swine expansion in north central Iowa into dense clusters of confined feeding operations, housing thousands of animals each, led us to question whether the manure management regulations were sufficient to protect water quality. We selected for study a 1,554 ha (six mi^2 or 3,840 ac) area of Hamilton County, Iowa that had 10 CAFOs, four of them within a 1 km radius.

Our study had three objectives. First, we sought to characterize the concentration of swine in this area and estimate both the total manure nutrients [N, P, potassium (K)] and the total N volatilized into the atmosphere. Second, we wanted to test the hypothesis that in areas of high livestock concentration manure nutrient value would be minimized, increasing the potential for nonpoint source-pollution. To evaluate this, we adopted as a standard the most recent recommendations of Iowa State University agricultural engineers (Lorimor et al. 1997) and compared the calculations of MMPs in the study area with this document and state regulations. Finally, we sought to determine whether enough land existed in the study area to recycle manure N or P, based on the projected needs of a conventionally managed corn/soybean rotation.

Methods

Six adjacent square mile sections (21, 22, 23, 26, 27, 28) in Rose Grove Township (T88N R23W; 42°25'N, 93°32'30"W), Hamilton County, Iowa, were selected for study based on the high density of swine facilities in this area. We obtained the entire record on file for each CAFO, including construction permit application, construction permit, and manure management plan (MMP), from the Iowa Department of Natural Resources (IDNR). Each file included the animal space capacity (number of spaces for hogs in the facility), the method of manure storage and application, projected annual volume and N content of manure after storage losses, and manure N available for field application. The file also included target crop yields, the planned crop and manure application schedule for at least three years, and the location and size of each field.

We confirmed each CAFO in the study area by observation from public roads. Several smaller swine facilities also existed, but these were not included in our study because they were not required to submit an open file MMP. These smaller facilities will add to the nutrient load within the study area, however.

Calculations

We examined each MMP in detail and recalculated the manure N estimate to verify the assumptions used in the calculations. We then used the most recent guidelines for manure nutrient management planning (Lorimor et al. 1997) to make independent estimates. Tabular values and estimation procedures in Lorimor et al. (1997) were based on a number of original scientific studies (Fleming et al. 1993; Killorn and Brenneman 1993; Killorn 1985; Sutton et al. 1993) as well as standard agricultural engineering references (e.g. Midwest Plan Service 1993).

Total manure N available annually for crop production ("NCP") was estimated as

$$NCP = (TMN - V_t) * p$$

where: TMN is the total manure N generated by the facility
V_t is the amount of N volatilization into the atmosphere (in kg N)
p is the proportion of manure N that will mineralize and become available for plant growth in the growing season following application (usually estimated to be about 75%)

The remaining 25% is accounted for later in crop N demand as "carryover N" from previous years' manure applications. State regulations allow CAFOs to use a variety of methods to estimate NCP.

Total manure N (TMN in kg N) generated by the operation is estimated as

$$TMN = ASC * h$$

where: ASC (animal space capacity) is the total number of animals that can be placed in the facility at any one time
h (in kg N per animal space) is a variable representing the kg manure N produced per animal space over 1 yr's time

For finishing hogs raised from 18 to 118 kg (40 to 260 lbs), Lorimor et al (1997) assumed h = 14 kg (30 lbs) N per animal space. This number is based on the amount of N excreted by an animal over its life in the facility, assuming a "turnover" of 2.48 animals occupying a given space each year. Turnover refers to the number of times, on average, that a unit is filled with pigs in a year. Thus, 1,100 capacity building will, on average, house about 2,800 hogs per year.

Nitrogen volatilization (almost exclusively as ammonia) occurs during manure storage and field application. This transfer of N to the atmosphere is the reason for the difference between TMN and NCP. Total N volatilized (V_t, in kg N) is calculated as

$$V_t = (TMN*V_s) + (TMN*(1-V_s) * V_a)$$

where: V_s is the proportion of N volatilized in storage
V_a is the proportion of N volatilized during field application

Lorimor et al. (1997) estimated that for anaerobic lagoons about 80% of total manure N is volatilized into the atmosphere, compared to about 30% for earthen basins. The difference between the two storage methods is due to greater dilution and larger surface area relative to the depth in anaerobic lagoons, leading to increased microbial activity (Zhang et al. 1994). Similarly, sprinkler irrigation of swine manure results in the loss of an estimated 40% of its N content, while injecting the slurry directly into the soil causes only a 5% loss (Lorimor et al. 1997). All of these projections are average values that will vary widely because of many factors, including weather and animal rations. State law does not require estimation of V_t or TMN.

We used the method described by Lorimor et al. (1997) to calculate area needed for manure application under two common cropping systems in central Iowa: continuous corn and a corn/soybean rotation. Total land area (LA in hectares) needed for spreading manure was calculated as

$$LA = NCP/[a * d - (l + f + c)]$$

where: a is a variable [in kg N (grain yield in m^3)$^{-1}$] used to estimate N fertilizer requirements for corn harvested as grain (equal to 15.1 kg N m^{-3} target yield in central and eastern Iowa)

d in m^3 of corn is the target yield for next year's crop. Nitrogen credits include:

l which is the N remaining from the previous year's legume crop.

f is chemical fertilizer applied, such as ammonia, prior to corn planting.

c is carryover N from the previous year's application of manure fertilizer

State regulations allow for a variety of means to estimate "d," but specify values for all other variables.

We inspected each CAFO's land area projections to identify the assumptions and then compared those assumptions to our own projections. We assumed Hamilton County average yield values from 1990 through 1994, excluding low yields in 1993 (Sands and Holden 1996), to estimate total crop N demand. We assumed that soybeans needed no N fertilization because of biological N fixation; therefore, "a" equaled zero.

We also estimated total P and K in swine manure and amounts applied to fields using average values for P and K content of swine manure generated per hog space and stored by different methods. Sequestration of P and K in manure storage facilities was estimated by comparing the P content of solid versus liquid manure. The annual production of P per animal space is 3.44 kg for manure stored as a solid, but only 0.86 kg for anaerobic lagoon effluent and 3.01 kg for earthen storage basin effluent (Lorimor et al. 1997). Differences between solid and liquid storage must be part of sludge solids. Thus, on an annual basis, anaerobic lagoons should accumulate 2.58 kg P per animal space and 6.64 kg K per animal space. Earthen basins would accumulate 0.43 kg P and 4.15 kg K per animal space.

The area needed for spreading manure, based on removal of P or K with corn grain and soybean grain harvest, was estimated using average county yields, a corn/bean rotation, and removal of 10 kg P m^{-3} and 19 kg K m^{-3} for soybeans and 5 kg P m^{-3} and 4 kg K m^{-3} for corn (Lorimor et al. 1997).

Location and size of fields designated for manure application. Using the MMPs, we noted the location and size of each field designated for manure spreading. This made it possible to determine whether any fields were being claimed by more than one operation.

Limitations of study methods. Our study used public records submitted by large scale swine operations and published values for the average operation developed by Iowa State University to estimate nutrient use, accumulation, and transfers on a landscape scale. The advantage of this approach is that data collection is rapid, inexpensive, and does not require the cooperation of CAFOs. A disadvantage is that we cannot assess the variation in actual performance of CAFOs because of differences in management. For example, some CAFOs may reduce P accumulation and N emissions by varying rations (Honeyman 1993). The validity of our study rests on the assumption that the published tabular values reflect average conditions among swine facilities in the study, and this assumption cannot be verified with the methods used.

Iowa law does not require CAFOs to follow their MMPs. Actual manure management practices must be documented and kept available for inspection by the IDNR, but they are not available to the public. Thus, our study reports what each CAFO has said it would do to comply with state law and assumes that each CAFO operates within the law. Because there is a strong economic incentive to dispose of manure inexpensively and because monitoring of CAFO practices is limited, our study probably offers conservative estimates of livestock concentration, ammonia volatilization, and nutrient application rates.

We deliberately chose an area with high concentrations of hogs. Thus, our numbers do not reflect average conditions on the Iowa landscape. By choosing this area, however, we establish a highly concentrated scenario that is useful for reviewing regulations.

Results

Swine concentration, total manure N, and manure management. Construction permits indicate a capacity for at least 59,700 finishing hogs in the 1,554 ha study area (Table 1). Assuming a turnover rate of about 2.48 hogs per animal space per year, we estimate that, over a year's time, 148,000 hogs could be fed in this area. The ensuing waste could contain up to 811,500 kg of N yr^{-1} (Table 1). Manure from the five largest CAFOs, representing 72% of the animal capacity, was stored in anaerobic lagoons and applied to fields using sprinkler irrigation (Table 1). The remaining CAFOs used an earthen storage basin and applied manure by immediately incorporating it into the soil (injection). One CAFO used injection on half of its four fields.

Table 1. Confined Feeding Operation date of permit, animal space capacity (all finishing swine), manure storage method, and manure application method based on building permit and manure management plan.

ID	Date	Animal Capacity	Manure Storage	Manure Application	*Total Manure N (kg)	Storage + appl. losses (kg)
1	1994	15,000	lagoon	irrigation	203,897	179,429
2	1994	15,000	lagoon	irrigation	203,897	179,429
3	1991	4,400	lagoon	irrigation	59,809	52,633
4	1991	4,400	lagoon	irrigation	59,809	52,633
5	1991	4,400	lagoon	irrigation	59,809	52,633
6	1990	3,300	basin	irrig. and inj.	44,857	18,952
7	1991	3,300	basin	injection	44,857	15,027
8	1991	3,300	basin	injection	44,857	15,027
9	1993	3,300	basin	injection	44,857	15,027
10	1995	3,300	basin	injection	44,857	15,027
Total		59,700			811,506	595,817

* "Total Manure N" is the amount of nitrogen produced by hogs in each facility on an annual basis, assuming 14 kg N animal space (Midwest Plan Services 1985). Storage losses are calculated as 80% of total N for anaerobic lagoons and 30% for earthen basins. Application losses are calculated as 40% of N for irrigation and 5% for injection (Lorimor et al. 1997).

Table 2. Comparison of estimates of total manure N available (kg yr^{-1}) using Lorimor et al. (1997) vs. manure management plans projections. Discrepancy in available N between Nos 5 vs. 3 and 4 is due to an apparent clerical error in the MMP.

ID	This Study available (kg)	Manure Management Plan				Comparison
		Basis	kg N L^{-1}	L hog^{-1} d^{-1}	avail. N (kg)	% of est.
1	24,468	published statistics	0.015	1.1	24,468	100%
2	24,468	published statistics	0.015	1.1	21,116	86
3	7,177	lagoon test	0.02183	0.5	5,188	72
4	7,177	lagoon test	0.02183	0.5	5,188	72
5	7,177	lagoon test	0.02183	0.5	5,188	72
6	25,905	published statistics	0.135	0.3	19,670	76
7	29,830	published statistics	0.135	0.3	21,572	72
8	29,830	published statistics	0.135	0.3	21,572	72
9	29,830	published statistics	0.135	0.3	21,572	72
10	29,830	published statistics	0.135	0.3	21,572	72
Total	215,692				167,106	77%

Ammonia Volatilization (V_t) and N available for Crop Production (NCP). We estimated that 70% of the total manure N was volatilized, thus contributing to nonpoint source pollution (Table 1). CAFOs employing anaerobic lagoon storage and spray irrigation transferred an estimated 88% of manure N into the atmosphere; this compared to 34% by CAFOs using earthen basins and soil injection of liquid manure.

Confined feeding operation estimates of NCP were, on average, 77% of our estimates (Table 2); only one CAFO estimate was identical to ours. There were three sources of discrepancies between our calculations and theirs. CAFO 2 changed the capacity of the facility stated in the original construction permit from 15,000 to 13,000 hog spaces; this reduced all other calculations of manure N and land area. Confined feeding operations 3, 4, and 5 projected that they would produce less than half of the manure volume estimated in Lorimor et al. (1997; Table 2). While the second cell of each lagoon had a measured N concentration of 0.0006 kg l^{-1}, or 45% higher than the published value of 0.0004 kg l^{-1}, the total projected manure N was 72% of our estimates. Finally, CAFOs 6-10 calculated NCP using one source for manure volume and another for manure N concentration. They cite Lorimor (1995) for a manure volume of 7.9 l per animal space per day and Sutton et al. (1991) for manure N concentration 0.004 kg N l, less than 70% of the concentration reported by Lorimor et al. (1997). Each of these CAFOs also reduces its total animal space capacity by a "livability factor" of 0.97 before calculating NCP. This term is undefined, but appears to refer to hog mortality while in confinement.

Land Area (LA) required for manure application. We calculated that if all manure generated in the study area were spread on land in a continuous corn rotation, according to the N requirements of corn using county average target yields, 1,138 ha or 73% of all land in the study area would be required each year (Table 3). For a corn/soybean rotation, the land required was estimated as 3,016 ha or 193% of the study area. This assumes that soybeans would require no manure N. Actual land area used by each CAFO for manure application was 28-73% of our estimates (Table 3). In the aggregate, CAFOs estimated that they needed only 991 ha or 63% of the land in the study area.

All CAFOs accounted for manure N carryover, and all took N credit for soybeans planted the previous year, as required by law. The difference between CAFO estimates and our own estimates had several causes. First, CAFOs consistently said they had less N to apply than

Table 3. Estimates from this study of land area required for spreading available manure based on two possible crop rotations, continuous corn (c-c) and corn-soybeans (c-b) using average manure N production per animal space (Lorimor et al. 1997); planned crop sequence, area used for manure spreading, average rate of N application from data reported in manure management plans, and comparison of MMP vs. our estimates of land area needed.

ID	This study		Data from MMPs			Comparison
	Estimated area (ha)					MMP land area as
	c-c	c-b	rotation	ha	kg N ha	% of this study[1]
1	127	351	c-b	153	170	45%
2	127	351[2]	c-b	131	172[2]	38
3	37	103	c-c	28	201	74
4	37	103	c-c	24	228	65
5	37	103	c-c	27	205	73
6	155	371	c-b	106	200	29
7	155	427	c-b	131	176	31
8	155	427	c-b, c-c	118	196	28
9	155	427	c-b	141	164	34
10	155	427	c-b	132	175	32
Total	1140	3090		991		

[1]Comparison of land area needs estimated by the MMP and the estimate made in this study, using the appropriate crop rotation for each CFO.
[2]Based on 15,000 hog spaces as indicated in the CFO's original permit for a waste lagoon, rather than 13,000 hog spaces as indicated in the manure management plan.

our calculations indicated (Table 2). Second, 8 out of 10 CAFOs made more optimistic estimates of target yield. Target yields determine the amount of N that can legally be applied to each field, so this is a critical number. We assumed an average county yield of 12.87 m^3 ha^{-1} (144.7 bu ac^{-1}) corn and 3.91 m^3 ha^{-1} (44 bu ac^{-1}) soybeans. (This is average for the period 1990-1994 with 1993 omitted because of extremely poor yields.) Four CAFOs assumed a yield 10% higher than these county averages during that same period.

Other CAFOs were even more optimistic. CAFOs 3, 4, and 5 assumed a yield of 14.2 m^3 ha^{-1} (160 bu ac^{-1}) corn with no justification. CAFO No. 8 assumed a corn yield of 16.1 m^3 ha^{-1} (180.8 bu ac^{-1}) corn and 4.46 m^3 ha^{-1} (50.1 bu ac^{-1}) soybeans. Five years of past yield data were provided in supporting documents to justify these target yields. But the five year average of recent yields, excluding 1993, was only 13.1 m^3 ha^{-1} (147.1 bu ac^{-1}) for corn and 4.13 m^3 ha^{-1} (46.4 bu ac^{-1}) for soybeans. Thus, target yields were 22% and 8% greater than the recent past corn and soybean yields, respectively.

Only CAFOs 1 and 2 appeared to estimate yields conservatively. They measured the acres of each soil type in each field and used corn suitability ratings to calculate a composite yield potential for each field.

The most important cause of the difference between our estimates and those of the CAFOs was the practice of manuring soybeans. All operations except CAFOs 3, 4, and 5 grew soybeans and applied manure to this crop. They calculated soybean N demand as 48.1 kg N m^{-3} target yield, citing Lorimor (1995). Table 1 of this document includes the N, P, and K removed with harvest of soybean grain and alfalfa, vetch, and red clover hay as a "fertilizer requirement" and, thus, inadvertently supports the case for manuring legumes. Schmitt et al. (1998) reported that soybeans can remove about 180 kg N ha^{-1}, but they cautioned against adding excessive manure N to the crop.

While land slated for corn was manured to reach the full amount needed for the target yield, manure N on soybean land was treated as optional. Actual rates of manure application to soybeans, 101 to 208 kg ha^{-1}, were often lower than estimated soybean N demand, indicating that soybeans were not viewed as "demanding" full N fertilization.

Phosphorus and potassium loading and sequestration. Calculations of P and K are not required by law, but P clearly has the potential to affect water quality and soil nutrient balance. We estimated that CAFOs were applying from 73 to 188 kg ha^{-1} P and 103 to 436 kg ha^{-1} K every year (Table 4). CAFOs 3, 4, and 5 were the only ones to test the nutrient content in their lagoon. In contrast to published estimates of 0.155 kg P/l lagoon waste they found only 0.043 kg P/l and estimated that they were applying only 23 kg P ha^{-1} to fields. This may reflect greater P accumulation in the sludge of two-cell lagoons.

A substantial proportion of the P in hog manure is adsorbed on particulates and settles to the bottom of storage facilities, rather than being spread in liquid manure. The type of storage facility has a large impact on both the concentration of P spread annually on fields and the amount sequestered in the sludge over several years. We calculated that CAFOs 1, 2, 3, 4, and 5, with anaerobic lagoons and 72% of the hogs, are sequestering in temporary storage at least 92% of the P (Table 4). This is multi-year but ultimately temporary storage because eventually the P must be land applied. There are no management plans required for application of sludge from lagoons (Iowa Administrative Bulletin 1996).

If manure P (including sludge) were applied to fields every year at a rate designed to replace the P removed with corn and soybean grain harvest, assuming no losses due to soil erosion and 100% uptake efficiency, CAFOs in this study would require 9,352 ha annually (Table 4). This is 6.0 times the size of the study area (1554 ha) and 9.4 times the area currently used for manure application (991 ha). The area needed for efficient use of manure K would be similar to those for P (Table 4).

Location and size of fields designated for manure application. Thirty percent of the 991 ha designated for swine manure application was outside the study area, but no field was more than five km from the source lagoon. Seven of the fields were adjacent to or dissected by waterways or flowing streams.

In one case, the same fields were designated for manure spreading by two different MMPs, creating the mistaken impression that the land area for spreading manure was more than adequate. CAFOs 1 and 2 were adjacent to one another and had the same owner. Each operation identified the same two fields totaling 101 ha in its MMPs. CAFO 1 listed fields totaling 153 ha available versus 146 ha needed for manure applica-

Table 4. Annual field application of P and K, and amount sequestered in storage facilities, based on published statistics (Lorimor et al. 1997) (actual tests of lagoon contents in parentheses), and land required to spread waste based on crop uptake of P in a corn-soybean rotation.

ID	Applied to Fields (kg ha yr) P	Applied to Fields (kg ha yr) K	Sequestered (kg yr) P	Sequestered (kg yr) K	Area Required (ha) P needs	Area Required (ha) K needs
1	102	254	37,993	95,899	2350	2552
2	119 [1]	254	37,993	95,899	2350	2552
3	166 (23)[2]	437	11,114[3]	28,130	689	749
4	189 (23)[2]	437	11,114[3]	28,130	689	749
5	169 (23)[2]	437	11,114[3]	28,130	689	749
6	119	139	1,929	14,893	517	562
7	79	111	1,929	14,893	517	562
8	87	125	1,929	14,893	517	562
9	73	103	1,929	14,893	517	562
10	78	110	1,929	14,893	517	562
Total			119,063	350,653	9,352	10,161

[1]Based on an animal space capacity of 15,000.
[2]Numbers in parentheses are based on MMP, which uses actual lagoon tests of nutrient content of the effluent.
[3]Number is conservative. Sequestration of P in each of these lagoons based on measured effluent concentration should actually average 34,560 kg yr.

tion, or 7 ha in excess. CAFO 2 listed fields totaling 232 ha available versus 125 ha needed, or 107 ha in excess. In addition to double listing 101 ha, CAFO 2 included a field in its MMP that has a 13 ha hog building and lagoon site. When these areas are subtracted from the "available" land, CAFO 2 is about 7 ha short of its calculated needs. When the land needs of the two CAFOs are combined, those needs precisely match the total land area available.

CAFOs 3 and 4 likewise have the same owner and are adjacent to one another. CAFO 3 claimed two fields totaling 52 ha available for manure disposal versus 28 ha needed. CAFO 4 claimed two fields totaling 41 ha versus 24 ha needed. Together, then, they claimed 93 ha available for manure spreading and 52 ha needed. The map provided for the two MMPs, however, was identical. The map—hand-drawn and not to scale—showed an 81 ha farm, including two building and lagoon sites totaling 10 ha. The precise boundaries and size of each field were unclear. Based on this map, actual land available for manure application appeared to be only 71 ha instead of the reported 93 ha versus a total of 52 ha needed. Thus, the land area may be adequate for the manure application needs of the two CAFOs combined, but is certainly smaller than the total land area claimed in their MMPs.

Discussion

Should an upper limit be placed on livestock concentration? Regardless of the scale of an individual hog farm, the concentration of many animals in a small area can cause nutrient loading that is too high for the ecosystem to recycle. When this happens, greater flux through the system and sequestration of nutrients will take place. Examples of greater nutrient flux include greater rates of ammonia volatilization from storage facilities, suppression of biological N fixation, and increased rates of nitrate-N leaching and runoff from heavily manured soils. Examples of sequestration include the buildup of P and K in storage facilities and the buildup of organic N and total P in soils. Sequestration in storage facilities is only a temporary solution, however, because these pools will eventually be redistributed either deliberately or by accident, and N and P in soils will mineralize during cropping.

While current regulations require that individual CAFOs account for their N usage on cropland, there is no assessment of concentration at the regional, local, or watershed scale that accounts for the combined impacts of all livestock in an area. Are livestock in the study area too concentrated? By what criteria can this be determined? We found N to be inefficiently distributed. According to our calculations, 3,090 ha of land would be required to apply manure N to corn in a corn/soybean rotation. This is twice the size of the study area (1,554 ha) and three times the area used to apply manure (991 ha).

Even when best management practices are followed, target yields are reasonable, estimates of manure N are accurate, and there are no accidental spills or leaks from lagoons, a growing pool of mobile N will be created in this region. The current recommended rate of N fertilization, 15.1 kg times the yield goal in m^3, is not based on the nutrient removal rate of corn grain (8.8 kg N m^{-3}) but on economics—the return on investment of an extra unit of N fertilizer given prevailing fertilizer costs and corn prices. Inevitably, the difference between N applied and N removed will be left at large in the ecosystem. Manure management plans do not account for this. The excess N is spread among pools of varying mobility: decaying plant matter, soil organisms and organic matter, dissolved nitrate, and ammonia. Nitrate losses to shallow groundwater are related to nitrogen loading of the soil in typical row crop production (Hallberg 1989; Kanwar et al. 1983; Baker and Johnson 1981). Unless the excess N provided to corn is consumed by some other crop, nitrate leaching is likely over the long term.

From the standpoint of P cycling, the case for livestock overcrowding is equally clear. According to the manure management plans, P is being applied at about twice the rate recommended for optimum corn production when a soil test indicates low surface soil P (9-15 ppm, Voss et al. 1996). In the absence of severe soil erosion, P will build up in these soils. When P in sludge is accounted for, the CAFOs in our study would require nearly 10 times the land they currently use to apply manure (Table 4).

We can only rely on individual regulation of CAFOs, without regard to regional livestock concentration, if the law requires: a.) accurate nutrient accounting, i.e., annual measurement of nutrients in soil, manure, and crop; b.) target yields that do not exceed a long term average for the particular soil types in use, and c.) a nutrient balance standard, rather than an agronomic standard, for nutrient application rates. Otherwise, we make the implicit and incorrect assumption that unregulated farms in the region are nutrient sinks.

Manure nutrients: A resource or a waste product? A key change in the most recent amendments to the rules regarding manure management change was to replace the phrase "waste disposal" with "manure application" (Iowa Administrative Bulletin 1996). Despite this effort to treat manure as a valuable resource, several lines of evidence suggest that CAFOs in the study area regard manure as a waste. First, according to our calculations, at least 0.6 million kg N is emitted as ammonia into the atmosphere annually (Table 1). CAFOs 1-5, with anaerobic lagoon storage, lost an estimated 88% of total manure N to volatilization, while CAFOs 6-10, using earthen storage basins and liquid injection of manure into fields, lost an estimated 34%.

More evidence for the view that manure is treated as a waste comes from cropping practices. Seven of 10 CAFOs applied manure to soybeans. This is a significant departure from typical agronomic practices for soybeans grown in Iowa. In 1996, only 10% of soybeans in Iowa were fertilized, at an average rate of 35 kg N ha^{-1}. In contrast, 98% of corn area received N fertilizer, at an average rate of 144 kg ha^{-1} (Sands and Holden 1996). Although Lorimor et al. (1997) does not recommend manuring soybeans or provide an example for calculating soybean N requirements, the practice is not forbidden (Iowa Administrative Bulletin 1996). It is unlikely that manure is valued for its organic matter content in north central Iowa because of the high organic matter content of these soils.

Finally, several strategies were employed to minimize the estimate of total N: Combining manure nutrient concentration and volume from separate published tables, inflating yield estimates, overestimating total available cropland, and reducing planned hog numbers below the design capacity of the confinement. Most of these strategies were clearly made possible by language in the Iowa code (Iowa Administrative Bulletin 1996).

Do current regulations adequately protect against nonpoint source-pollution? Even if loopholes in the law are closed to create more conservative manure management plans, we conclude that the law will still fail to protect Iowa's waters from nonpoint source-pollution.

First, the law fails to recognize the N that is squandered in ammonia emissions. Ferm (1998) calculates that in Europe about half of the volatilized NH_3 is deposited within 50 km of the source through dry and wet deposition. Atmospheric N deposition contributes to eutrophication of estuaries. N fertilization of terrestrial communities favors nitrophilous plant species at the expense of species that are tolerant of low N conditions (Berendse et al. 1993; Inouye and Tilman 1995). There are no inexpensive techniques for removing excessive N from terrestrial systems (Marrs 1993). Among the costs of livestock concentration to communities, we must add possibly irreparable damage to the few remaining natural plant communities left in the state.

Second, the costs of transporting liquid manure rise with distance, and where livestock are concentrated, there is a strong incentive to spread the manure close to its source (Fleming et al. 1998). It makes clear economic sense that large scale operations should maximize N volatilization through the use of anaerobic lagoons and sprinkler irrigation in order to afford the manure disposal costs. Until the cost of chemical fertilizers is much higher, CAFOs will logically treat manure as a waste disposal problem.

Finally, manure management regulations fail to contend with P and K application to fields or buildup in earthen storage facilities. Large imbalances in P supply and demand will lead to continued buildup of organic P in soils and eutrophication of freshwater streams and lakes in this region. K is instrumental in ion exchange across cell membranes, but the effects of annual over-application of K are so far unknown in this region. Soil buildup of K will depend upon leaching rates as well as plant uptake.

Policy Recommendations

Based on our findings, we offer the five following policy recommendations:

1. Increase research and outreach and consider incentives for alternate swine housing systems, particularly hooped structures with deep bedding. Deep bedded facilities have intrinsic advantages compared to liquid waste based systems (Honeyman 1991; 1999; Crabtree 1998; Moulton and Moulton 1998). They are less costly, animal friendly, do not use liquid manure handling systems, and are relatively odor free.

2. Increase regulatory scrutiny of current systems and require a public comment period for all new MMPs and permit applications 60 days before the permit is granted.

3. Increase research on the methods of nutrient management, including ammonia volatilization reduction, and monitor more closely both new and existing systems.

4. Establish statewide zoning regulations, on a township basis, for density of animal units, and consider regulating all new construction, regardless of size.

5. The largest gains to be made in nutrient use efficiency at a landscape scale have to do with crop rotation. A crop rotation that includes small grains and unmanured forage legumes will help to balance N, P, and K demand over time (Beegle et al. 1998). These crops will also protect soils from erosion, improve soil tilth, contribute to overall pest management, and reduce the need for pesticides (National Research Council 1989).

Rather than rely ever more heavily on regulations and monitoring, we should research and demonstrate the economic viability of integrated livestock-crop production systems whose intrinsic features encourage efficient nutrient use and protect our groundwater and surface water.

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Pollutant Movement to Shallow Ground Water Tables from Anaerobic Swine Waste Lagoons[1]

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ABSTRACT

The effect of three anaerobic swine waste lagoons on ground water quality was investigated in the Atlantic Coastal Plain region. The lagoons studied were located on high-water-table soils with different textures. Ground water was sampled from wells, with unperforated casings, located at depths to 6 m and distances to 30 m from the lagoons. These samples were taken monthly from September 1974 through January 1975, and bimonthly thereafter through November 1975. Ground water also was sampled in November 1975 from shallow wells with perforated casings located at distances to 36.6 m from two of the three lagoons. Constituents determined to investigate ground water contamination were density of fecal coliforms and concentrations of Cl, Ca, Mg, NH_4-N, NO_3-N, PO_4-P, and Zn.

Chloride, NH_4-N, and NO_3-N concentrations in ground water samples indicated that seepage entered ground water from each of the three lagoons. Rupture of lagoon seals leading to seepage was attributed to drying of exposed subsoil or embankment soil during recession of lagoon liquid levels and to gas release from microbial activity in soil beneath the seal. Overall, a low level of ground water contamination occurred around a lagoon that was in operation for over 8 years in Myatt very fine sandy loam with a clay subsoil, and a lagoon that was in operation 1 mo prior to this investigation in Dragston fine sandy loam with a sandy clay loam subsoil. Ground water contamination in excess of recommended drinking water standards for Cl and NO_3-N occurred around the third lagoon. This lagoon was in operation for >8 years in a disturbed area consisting of predominantly sandy surface and subsurface soil. Ground water contamination in the embankment area of this lagoon was attributed to seepage and beyond the embankment area to ground water contamination from lagoon overflow.

Additional Index Words: ground water monitoring, lagoon seepage, soluble nutrients, waste disposal.

Swine confinement production units are prevalent in the Coastal Plain region. This intensive swine production creates a disposal problem due to concentration of large quantities of wastes in a relatively small area. Anaerobic lagoons frequently are used for swine waste disposal because of a relatively low construction cost and a minimal maintenance requirement. Effluent from the anaerobic waste lagoons is unsuitable for discharge into waterways. Seepage of the lagoon liquid above a critical level into ground water also would be unacceptable.

Investigations were conducted to study the amount of water movement from dairy waste ponds. Davis et al. (1973) reported that the hydraulic conductivity of sandy loam beneath a pond decreased from 122 to 0.5 cm/day after 4 mo of operation and that the amount of seepage was negligible when evaporation was taken into account. Baier et al. (1974) found that the hydraulic conductivity in soils beneath nine ponds decreased to zero within 60 days of operation and that an inverse relationship existed between length of time for sealing and clay content in pond bottoms. Chang et al. (1974) determined the hydraulic conductivity in columns of various soils that had been located beneath a pond. Hydraulic conductivities decreased to 0.39 cm/hour in quartz sand and to zero in sandy, loamy, and silty clay soils within 29 days of lagoon operation. The initial decrease in hydraulic conductivity was attributed to physical entrapment of waste particles in soil pores, and the final obstruction of water movement from the ponds was ascribed to microbial growth in the pores (Chang et al., 1974; Davis et al., 1973).

Investigations indicated that seepage occurred from some anaerobic animal waste lagoons. Miller et al. (1976) found higher than background levels of soil NH_4-N to depths > 140 cm beneath two anaerobic swine waste lagoons. These two lagoons were in operation for 8 years or more in a medium- and a coarse-textured soil. Sewell et al. (1975) found increased Cl concentrations in ground water at a 6-m depth and 5- to 15-m distances from a dairy waste lagoon. Two months after loading, the Cl concentrations approached background levels. This lagoon was located in silt and sandy loams to a depth of 1 m and quartz sand at 1 to 4 m. Nordstedt et al. (1971) reported above background levels of NO_3-N in ground water at a 2- to 3-m depth and a 15-m distance from a dairy waste lagoon that had been in operation for 8 mo in a clayey soil. Water level measurements by Pate et al. (1974) indicated inseepage of ground water into a beef cattle waste lagoon after a heavy rainfall and outseepage with subsidence of the water table. The lagoon under study was located in an organic soil with a high water table. Nordstedt and Baldwin (1973) concluded that lagoons in high-water-table soils should contain sufficient levels of water to prevent inseepage and, thereby, to allow seal formation.

This investigation was initiated because of the concern that anaerobic swine waste lagoons might contaminate shallow ground waters used for human consumption in the Coastal Plain region. The purpose of the research was to determine the amount and distance of pollutant movement from three anaerobic swine waste lagoons. Two of the three lagoons under study were in operation for over 8 years and each of the lagoons was located in soils with high water tables. The water table levels in lower areas around each lagoon rose to the soil surface during periods of heavy rainfall.

MATERIALS AND METHODS

Ground water quality was determined from samples taken at various depths and distances from the three lagoons. Lagoons selected for the investigation were located in the Coastal Plain region of Virginia at the Virginia Swine Evaluation Center (VSEC), at the Tidewater Research and Continuing Education Center (TRACEC), and at a private

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farm. Selection of the lagoons was based on lagoon age and design, on drainage characteristics and soil texture, and on the suitability of the area for well location.

Lagoon Description

The VSEC lagoon was located in Myatt very fine sandy loam (Typic Ochraquults) with clay subsoil. The lagoon was constructed predominantly below ground level. It was in use for over 8 years. The TRACEC lagoon was located in Dragston fine sandy loam (Aeric Ochraquults) with sandy clay loam subsoil. The lagoon was constructed with embankments approximately 2 m above ground level. Operation of this lagoon began in August 1974. The lagoon located on a private farm was constructed in a disturbed area predominantly in sandy textured surface and subsurface soil. Embankments of the lagoon were 1 to 2 m above ground level. It was in operation for over 8 years.

Well Location

Wells with unperforated polyvinylchloride (PVC) pipe as casings (unperforated wells) were placed at various depths and distances from the three lagoons for ground water monitoring. Two series of wells were placed at depths of 3, 4.6, and 6.0 m at distances of 3, 15, and 30 m from the VSEC and private farm lagoons. The NW well series was located in the region of ground water flow from the VSEC lagoon, whereas the SE well series was located upstream from ground water flow. Well series I was located in a depressional area downstream from ground water flow at the private farm lagoon, whereas well series II was located in a higher, better drained area perpendicular to ground water flow. Three series of wells were placed in the region of ground water flow to depths of 3, 4.6, and 6.0 m at distances of 3 and 15 m from the TRACEC lagoon. Piezometric maps were constructed when ground water tables were relatively high to determine the direction of ground water flow at the lagoon sites.

Shallow wells with perforated PVC pipe as casings (perforated wells) were placed at various distances from the TRACEC and private farm lagoon sites to obtain ground water that possibly was located above impervious subsoil layers. Two series of perforated wells were located at depths of 3.7 m at the TRACEC site and 2.4 m at the private farm site and at 15 distances ranging from 0.0 to 36.6 m from the lagoons. These wells were placed parallel to and approximately 2 m from unperforated well series at the two lagoon sites. Soil around the perforated wells at the 0.0-m well distance was fully exposed when the lagoons were approximately two-thirds full.

Well Construction and Ground Water Sampling

Polyvinylchloride pipe of 5-cm diam was used as unperforated well casings. These wells were water-jetted about 15 cm deeper than the well casing. Pea gravel was inserted into the well to form a 10-cm layer in the bottom of the casing. This procedure was followed to prevent undue sediment uptake during sampling. The wells were fitted with a PVC plastic cap. A sampling tube consisting of 1.8-cm-diam PVC pipe was inserted through the cap to a depth of 10 cm above the pea gravel. Ground water was sampled monthly from these wells from September 1974 through January 1975, and bimonthly thereafter through November 1975. A shallow-well-jet pump fitted with a 46-kg check valve and trap-assembly was used to obtain ground water samples. Wells were purged 1 day before sampling to remove standing water.

Polyvinylchloride pipe of 5-cm diam was used as perforated well casings. The PVC pipe was perforated with 3-mm holes to allow free movement of ground water. These holes were spaced at 2.5 cm in four rows down the pipe. The 5-cm perforated PVC pipe was water-jetted into soil such that the perforations ran from the bottom of the casing up to 0.3 m below the soil surface. These wells were fitted with a removable PVC plastic cap. One ground water sample was obtained from these wells in November 1975.

Ground Water Analyses

Constituents determined in ground water samples from the unperforated wells were density of fecal coliforms and concentrations of Cl, Cu, Mn, NH_4-N, NO_3-N, PO_4-P, and Zn; and from the perforated wells were concentrations of Cl, NH_4-N, and NO_3-N. Fecal coliform densities were determined by the millipore membrane filter technique (APHA, 1971). These analyses were initiated within 2 hours after ground water sampling. Samples for chemical analyses were frozen within 2 hours after collection and, prior to initiation of the analyses, the samples were thawed and passed through 0.45-μm filters. Concentrations of Cl were determined with a specific ion electrode, and of NH_4-N with a gas sensing electrode. These electrodes were coupled to an expanded-scale pH meter. Concentrations of NO_3-N were determined by a phenoldisulphonic acid method, and of PO_4-P by an ascorbic acid procedure (APHA, 1971). The concentration of Cu, Mn, and Zn were determined by atomic absorption spectrophotometry.

Statistical Analyses

Chloride, NH_4-N, and NO_3-N concentrations in ground water from unperforated wells, located at various depths and distances from each lagoon, were evaluated by analyses of variance and by Duncan's Multiple Range Tests (Duncan, 1955). The 10 sampling times served as replicates in the statistical analyses. These analyses were performed on an IBM 360 computer by utilizing the SAS program (Service et al., 1972).

RESULTS AND DISCUSSION

Chloride, Cu, Mn, NH_4-N, NO_3-N, PO_4-P, Zn, and fecal coliforms are either degradation products or constituents of hog fecal material. Concentrations of these constituents were determined in ground water sampled from the vicinity of three anaerobic swine waste lagoons to evaluate whether ground water was contaminated by seepage from the lagoons. Ground water sampled on 10 dates from unperforated wells at the various depths and distances from the three lagoons contained relatively low concentrations of <0.2 ppm Cu, <2.9 ppm Zn, and <0.5 ppm Mn. A gradient did not exist between concentration of the three elements in ground water with either sampling depth or distance from the three lagoons.

Soluble PO_4-P concentrations were <0.06 ppm in all ground water samples from unperforated wells, except those taken from one well located at a 3-m depth and a 3-m distance from the VSEC lagoon. The PO_4-P concentrations in ground water from this well were >2.0 ppm during six of the first seven samplings and decreased to <0.06 ppm during the last three samplings. These data suggested movement of lagoon seepage to ground water and, thereafter, seal formation and soil fixation of PO_4-P in relatively insoluble forms.

Fecal coliform densities were frequently >2,000/100 ml in ground water for the first two samplings from unperforated wells. These fecal coliform bacteria densities probably reflected contamination during well installation. Only four positive fecal coliform densities occurred in ground water from the last eight samplings, and, of these, only one exceeded 2,000/100 ml. This density is the recommended limit for usage of raw surface water for treatment for drinking water (USEPA, 1973). Subsequent discussion indicates that concentrations of Cl, NH_4-N, and NO_3-N gave a better measurement of the amount of ground water contamination from the three lagoons than concentrations of Cu, Mn, PO_4-P, or Zn, or density of fecal coliforms.

Virginia Swine Evaluation Center Lagoon

Mean NH_4-N concentrations in ground water from the NW and SE unperforated well series were not significantly different and, consequently, these data were

Table 1—Ammonium-N concentrations in ground water sampled from unperforated wells at various depths and distances from the VSEC lagoon.*

Distance from lagoon	Well depth, m		
	3	4.6	6
m	ppm NH_4-N		
3	10.8 a	0.1 a	4.1 a
15	0.5 b	0.2 a	0.1 b
30	0.1 b	0.1 a	0.1 b

* Means within columns followed by different letters are significantly different at the 0.05 probability level.

combined for statistical analyses (Table 1). In contrast, mean concentrations of the mobile Cl anion were higher in the NW well series, which was located in the direction of ground water flow, than in the SE well series, which was located upstream from ground water flow (Table 2). The Cl concentrations showed that the major portion of the ground water contamination was in the region of ground water flow on the downstream side of the lagoon, and that the amount of ground water contamination in this area decreased with distance from the lagoon. Decreases in Cl and NH_4-N concentrations with distance from the lagoon indicated that lagoon seepage entered the ground water. The highest concentrations in ground water from any sampling at the 3-, 4.6-, and 6-m depths and at the farthest distance from the lagoon of 30 m were 79 ppm Cl, 0.6 ppm NH_4-N, and 0.4 ppm NO_3-N. The anion concentrations were below the recommended tolerable limits for drinking water of 250 ppm Cl and 10 ppm NO_3-N (USEPA, 1973). These relatively low pollutant concentrations in ground water suggested that seepage from that lagoon in clay subsoil posed little threat to the ground water supply.

Tidewater Research and Continuing Education Center Lagoon

Three series of unperforated wells were located at various depths and distances from the lagoon in the direction of ground water flow. Well series location did not significantly affect Cl, NH_4-N, and NO_3-N concentrations in ground water obtained on the 10 sampling dates and, hence, the data were combined for statistical analyses (Table 3). Mean Cl and NO_3-N concentrations were higher in ground water from the 3-m depth and distance than from the 3-m depth and 15-m distance from the lagoon. The lack of a similar relationship for mean NH_4-N concentrations may have reflected nitrification. Wetting and drying of embankment soil above the wells and the presence of NH_4-N in seepage in the embankment (Table 4) would be conducive to nitrification. Evidence supporting nitrification was provided by the data showing a higher concentration of NO_3-N in ground water from perforated wells at a distance of 0.8 m from the lagoon (Table 4) than in lagoon liquid. The NO_3-N concentrations did not exceed 9.3 ppm in liquid samples from this lagoon in March, June, July, and September of 1975.[3]

Decreases in Cl, NH_4-N, and NO_3-N concentrations in ground water with distance from the lagoon indicated some seepage from the lagoon (Tables 3 and 4). Seepage may have been due to an incomplete biological seal or to disruption of the seal with drying as a decrease in lagoon liquid level exposed embankment soil. The latter conclusion was supported by experimental data showing that the hydraulic conductivities of completely sealed lagoon bottom soils reverted to their original level on drying (Chang et al., 1974).

Mean Cl concentrations were higher in ground water sampled 15 m from the lagoon at depths of 3 and 4.6 m than in ground water from check wells (Table 3). In contrast, mean NH_4-N and NO_3-N concentrations were not higher in ground water sampled at 15 m distance from the lagoon than in ground water sampled from check wells. The greater distance of ground water contamina-

Table 2—Chloride concentrations in ground water sampled from two series of unperforated wells located at the VSEC lagoon site.*

Distance from lagoon	Well depth, m					
	SE series			NW series		
	3	4.6	6	3	4.6	6
m	ppm Cl					
3	60 a	7 a	6 a	109 a	148 a	40 a
15	8 b	6 a	13 a	97 b	75 a	47 a
30	8 b	9 a	7 a	52 b	45 b	18 b

* Means within columns followed by different letters are significantly different at the 0.05 probability level. The overall mean Cl concentration was significantly higher at the 0.05 probability level in ground water from the NW series than in ground water from the SE series.

Table 3—Chloride, NH_4-N, and NO_3-N concentrations in ground water sampled from three series of unperforated wells located at the TRACEC lagoon site.*

Distance from lagoon	Well depths, m								
	Cl			NH_4-N			NO_3-N		
	3	4.6	6	3	4.6	6	3	4.6	6
m	ppm								
3	103a	49a	35a	1.9a	0.1a	0.2a	2.7a	0.6a	0.7b
15	33b	33a	10b	0.7a	0.3a	0.2a	0.9b	0.2a	0.2b
Check	6c	6b	11b	0.2a	0.2a	0.1a	0.3b	0.6a	2.5a

* Means within a column followed by different letters are significantly different at the 0.05 probability level.

Table 4—Chloride, NH_4-N, and NO_3-N concentrations in ground water from two series of perforated wells located at various distances from the TRACEC lagoon.†

Distance from lagoon	Cl	NH_4-N	NO_3-N
m	ppm		
0.0	364	18.8	13.3
0.8	168	3.5	30.7
1.5	160	0.7	3.9
2.3	155	1.1	0.9
3.0	153	0.6	1.2
3.8	79	0.3	0.8
4.6	180	0.5	1.2
6.1	117	0.5	1.0
7.6	103	0.6	1.2
10.7	90	0.7	1.4
13.7	63	3.4	0.6
16.8	67	0.6	0.8
22.9	53	0.2	<0.2
29.0	33	0.2	<0.2
36.6	12	0.3	4.1

† Each value represents the average of duplicate determinations for samples from two series of wells.

tion indicated by Cl, as compared with NH_4-N and NO_3-N (Tables 3 and 4), probably reflected the much higher concentration of Cl in the seepage; denitrification and nitrification relationships; and NH_4-N exchange, fixation, and release. Overall, a very low level of ground water contamination occurred from seepage at this site as evidenced by data showing that tolerable limits for Cl and NO_3-N in drinking water (USEPA, 1973) were not exceeded in ground water beyond 1.5 m from the lagoon.

Private Farm Lagoon

Two unperforated well series were located at various depths and distances from the lagoon. Series I was located in a depressional area in the direction of ground water flow and series II in a higher area perpendicular to ground water flow. Higher mean Cl and NH_4-N concentrations were present in ground water from series I at a 3-m depth and 3-m distance from the lagoon than at the same depth and 15- and 30-m distances from the lagoon (Table 5). A similar relationship occurred for mean NO_3-N concentrations at similar locations in series II. These data indicated seepage from the lagoon to ground water. Higher Cl, NH_4-N, and NO_3-N concentrations in ground water from perforated wells located near the lagoon, as compared with those more distant from the lagoon, provided further evidence of seepage from the lagoon (Table 6). Lagoon seepage may have been caused by an incomplete biological seal or by disruption of the seal with drying as decreases in lagoon liquid level exposed embankment soil.

Chloride concentrations in ground water from perforated wells in series I were higher at the 13.7- and 16.8-m distances from the lagoon than at the 0.8- to 10.7-m distances (Table 6). Also, higher mean NO_3-N concentrations occurred in ground water from some unperforated wells at distances of 15 or 30 m than at 3 m from the lagoon (Table 5). For example, the mean NO_3-N concentrations were 0.2, 11.2, and 17.2 ppm in samples from the 4.6-m depth and 3-, 15-, and 30-m distances from the lagoon, respectively. The latter two values exceeded the recommended tolerable level of 10 ppm NO_3-N for drinking water (USEPA, 1973). Lagoon overflow probably caused the contamination of ground water sampled at distances of 13.7- to 30-m from the lagoon.

Table 6—Chloride, NH_4-N, and NO_3-N concentrations in ground water sampled from two series of perforated wells located at the private farm lagoon site.†

Distance from lagoon	Series I			Series II		
	Cl	NH_4-N	NO_3-N	Cl	NH_4-N	NO_3-N
m	ppm					
0.0	256	51.3	49.3	123	45.3	66.9
0.8	93	12.3	68.1	118	65.1	100.8
1.5	62	63.8	66.7	54	7.5	90.5
2.3	23	15.9	41.6	37	3.7	68.6
3.0	8	8.9	35.5	15	1.5	23.9
3.8	49	76.3	4.2	27	0.9	9.3
4.6	4	10.6	0.4	23	0.6	15.9
6.1	29	20.7	<0.2	13	0.3	8.6
7.6	51	18.0	<0.2	10	0.2	5.4
10.7	49	13.5	<0.2	18	0.3	7.7
13.7	103	20.8	<0.2	35	0.2	5.1
16.8	204	19.5	<0.2	15	0.2	1.9
22.9	49	18.0	<0.2	8	0.2	3.8
29.0	60	0.5	4.0	5	0.1	1.8
36.6	6	0.4	0.5	7	0.4	0.5

† Each value represents the average of duplicate determinations.

Table 5—Chloride, NH_4-N, and NO_3-N concentrations in ground water from two series of unperforated wells located at the private farm lagoon site.*

Distance from lagoon	Well depth, m					
	Series I			Series II		
	3	4.6	6	3	4.6	6
m	ppm					
	Cl					
3	272a	119a	166a	12a	15a	25a
15	162b	114a	183a	10a	28a	14a
30	153b	72a	92a	7a	7a	9a
	NH_4-N					
3	357.9a	12.5a	40.9a	0.4a	0.1a	0.1a
15	25.4b	63.4a	97.6a	0.1a	0.2a	0.2a
30	16.0b	64.0a	0.3b	0.2a	0.1a	0.2a
	NO_3-N					
3	3.2a	0.2c	0.2b	9.5a	10.7a	1.0b
15	0.2a	11.2b	6.3a	2.9b	4.8b	5.0a
30	1.6a	17.2a	0.2b	3.2b	2.8c	6.0a

* Mean Cl, NH_4-N, and NO_3-N concentrations in columns followed by different letters are significantly different at the 0.05 probability level. The overall mean Cl and NH_4-N concentrations were significantly higher at the 0.05 probability level in ground water from series I than in ground water from series II.

INTERPRETATION

The sealing mechanism of anaerobic swine waste lagoons in high-water-table soils would be similar to that in well-drained soils. A smaller hydraulic head in lagoons in high-water-table soils may slow the initial physical entrapment of waste particles in pores and, thereby, lengthen the time for initial sealing. The seal, once formed, would be subjected to varying hydrostatic pressures due to fluctuations in water table depth, and this may cause seal failure (Nordstedt and Baldwin, 1973). Disruptions in the seal could be caused by drying of embankment soil (Chang et al., 1974), and by gas release from microbial activity in soil beneath the seal (Nordstedt and Baldwin, 1973).

Concentrations of Cl, NH_4-N, and NO_3-N indicated a greater amount of seepage to ground water from the private farm lagoon than from the TRACEC and VSEC lagoons. This may have reflected the coarser texture of the soil surrounding the private farm lagoon and the higher concentrations of nutrients in liquid from this lagoon. The latter was exemplified by the average Cl concentrations of 575, 253, and 157 ppm in liquid sampled in March, June, July, and September 1975 from the private farm, TRACEC, and VSEC lagoons, respectively.[3]

Little ground water contamination occurred from seepage near the TRACEC and VSEC lagoons, which probably reflected their construction in relatively fine-

[3] R. G. Wilhelm. 1976. Swine waste lagoon effluent: its nutrient-supplying potential and efficiency in corn production. M.S. Thesis. Virginia Polytechnic Inst. and State Univ., Blacksburg.

ual wells located at a distance of 30 m from the private farm lagoon frequently exceeded recommended tolerable limits for Cl and NO_3-N in drinking water. The latter relationship was attributed to contamination of ground water from lagoon overflow. This points to a problem encountered in selection of lagoons for assessing the amount of ground water contamination from lagoon seepage. Many lagoons observed were likely to overflow in the direction of ground water flow unless liquid removal preceeded heavy rainfalls. The overflow may pose a greater threat to water systems than seepage. Tunnelling from lagoons by burrowing animals also could cause pollution of water systems.

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Prediction of Algal-Available Phosphorus in Runoff Suspensions[1]

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ABSTRACT

Methods for estimating the potential availability of sediment-P forms are desirable for better assessment of the impact of runoff waters on surface water quality. The P sorbed by resin-affixed hydroxy-aluminum equilibrating with simulated sediment suspensions was evaluated for use as a routine means of estimating relative availability of P. A highly significant relationship ($r = 0.98^{**}$, $n = 5$) was found between availability estimated by resin extraction and that determined by bioassay. For the soils tested, resin-extractable P averaged 98% of that found to be available to algae by bioassay.

Additional Index Words: **eutrophication, phosphate, hydroxy-Al resin.**

Surface runoff from agricultural land is considered to be a major source of phosphorus (P) to surface waters (Task Group 2710 P, 1967). Although soluble inorganic P is readily available to algae and other aquatic plants, there is less certainty regarding the availability of sediment-bound P forms. Because the major portion of P carried in runoff is often attached to sediment, routine methods for estimating the potential availability of such forms are desirable for better assessment of the relative impact of runoff from agricultural land on the quality of receiving waters.

The amounts of sediment-bound P equilibrating with several extracting solutions have been studied as predictors of algal-available P (Chiou and Boyd, 1974; Cowen and Lee, 1976; Porcella et al., 1970). Test solutions were evaluated by comparing P extracted with increases in algal cell numbers (assuming a constant level of cell P). Because several days were allowed for cell growth, the relationships between test results and P availability may be invalid for estimating short-term availability. Short-term availability of P is important in surface waters because the time that eroded soil particles are present in the photic zone prior to settling may be relatively short.

Recently, Sagher[3] proposed a sodium hydroxide (NaOH) extraction procedure for estimating runoff P potentially available to algae during periods of up to 48 hours. At solution/sediment ratios of 1,000:1 or greater, a significant relationship existed between dissolved plus NaOH-extractable P and algal-available P determined by bioassay. However, he suggested that the

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[2] Research Assistants and Professor of Soil Science, respectively.

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Impact of Animal Waste Lagoons on Ground-Water Quality*

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ABSTRACT

Ground-water quality was monitored for three years at two sites around clay-lined animal waste lagoons on the Delmarva Peninsula. A swine waste lagoon located in an Evesboro loamy sand soil (excessively well-drained) was having a severe impact on ground-water quality. Ammonium nitrogen concentrations above 1000 mg/liter N have been measured in shallow monitoring wells around the lagoon. Chloride and total dissolved solids concentrations were also high. At the second site which has three lagoons and a settling pond in poorly drained soils, some seepage was occurring. Ammonium nitrogen, nitrate nitrogen, chloride and total dissolved solids were above background concentrations in some of the monitoring wells. There was a strong correlation between nitrate nitrogen and chloride concentrations in the monitoring wells. The results indicated that clay-lined animal waste lagoons located in sandy loam or loamy sand soils with high water tables may lead to degradation of ground-water quality.

INTRODUCTION

Many anaerobic lagoons on the Delmarva Peninsula are clay-lined and installed in either sandy loam or loamy sand soils with a high water-table.

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The Soil Conservation Service (Delaware and Maryland, USA) has been interested in determining if seepage is occurring from these clay-lined animal waste lagoons.

Ritter *et al.* (1984) found that an unlined anaerobic lagoon for swine wastes had some impact on ground-water quality. During the first year of operation, nitrate nitrogen, ammonium nitrogen and organic nitrogen concentrations increased in some of the monitoring wells but decreased to lower levels after the first year. None of the monitoring wells had nitrate nitrogen concentrations above 10 mg/liter N.

In a study of unlined lagoons in the Coastal Plain soils in Virginia, Ciravolo *et al.* (1979) found that two anaerobic swine lagoons caused measurable, but minimum ground-water contamination. A third lagoon contaminated ground water with chlorides and nitrate nitrogen in excess of drinking water standards. Sewell (1978) found nitrate nitrogen and chloride concentrations in ground water taken from wells 15 m from an unlined anaerobic dairy lagoon increased rapidly during the first six months of lagoon operation, and later decreased to levels similar to those before the lagoon was loaded. Median nitrate nitrogen concentrations of all the test wells were below 10 mg/liter N. The lagoon was located in an area with silt loam and sandy loam soils to a depth of 1 m and a quartz sand horizon at 1–4 m. Nordstedt *et al.* (1971) found that nitrate nitrogen concentrations were above background levels in the ground water in wells at a depth of 3·0 m and a 15 m distance from a dairy lagoon in a clay soil that had been in operation for 8 months. At a distance of 15 m, the average nitrate nitrogen concentration in the wells was 14·3 mg/liter N.

Since there have been reports in the literature of unlined lagoons impacting ground-water quality, a research project was initiated in 1982 to determine if clay-lined animal waste lagoons installed in sandy soils with high water-tables were causing ground-water contamination. This paper summarizes the results obtained from the study.

METHODS

Experimental Facilities

In 1982, two sites where anaerobic lagoons with clay liners have been in operation were selected for monitoring ground-water quality. The first site had an anaerobic lagoon for a 500 head hog finishing unit with a flushing system (Fig. 1). A berm was constructed around the lagoon, so approximately 2·3 m of the lagoon depth was constructed above the original surface of the ground. The finishing house is flushed twice a day with fresh water at a rate of 7000 liters/day. The lagoon bottom and sides are lined with 15 cm of

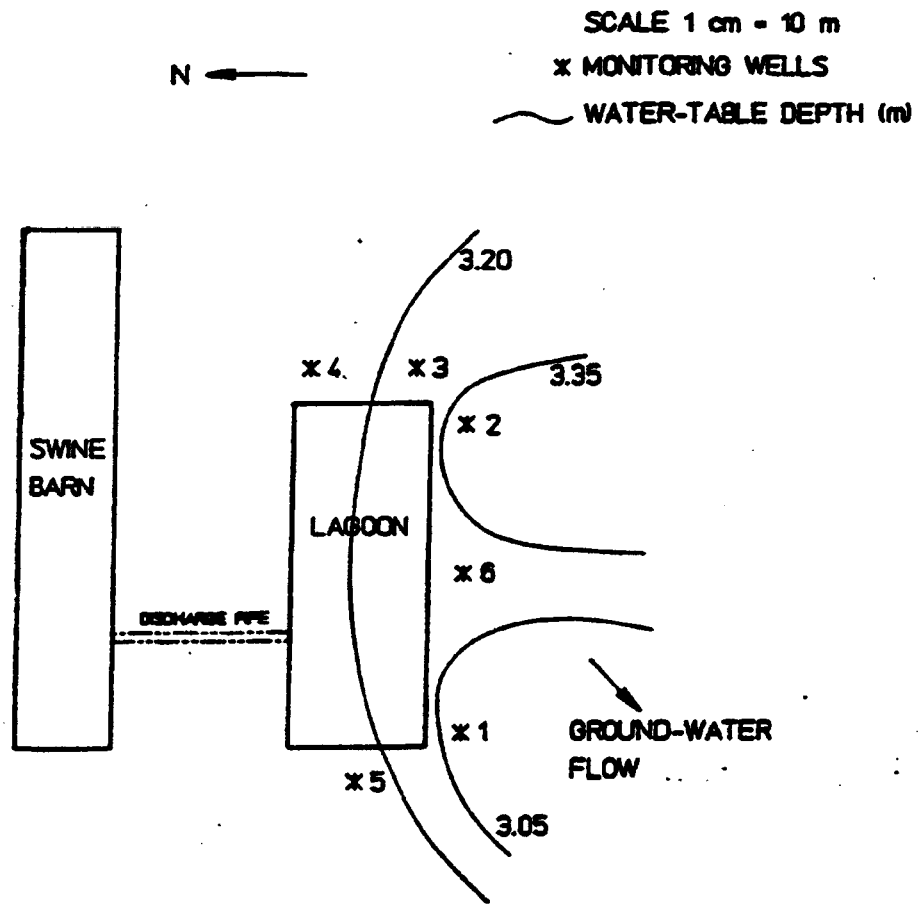

Fig. 1. Ground-water table depths at Site 1 on 4 March 1983.

compacted clay. The lagoon had been in operation for approximately five years when the monitoring wells were installed. The lagoon is completely emptied twice a year and the effluent is applied to cropland by sprinkler irrigation.

The second site was located on a farm that raises beef and hogs and has a slaugher house. There were a total of three anaerobic lagoons and a settling pond on the site (Fig. 2). One of the lagoons received waste from the slaughter house and one received waste from a hog feed operation. Both of these anaerobic lagoons had been in operation for approximately six years when the ground-water quality monitoring was started. In the autumn of 1982, the farmer constructed a new 200 head beef feedlot with a flushing system for cleaning the alleys. The manure is flushed twice a day with 9000 liters/day of fresh water into the settling pond and the effluent overflows into

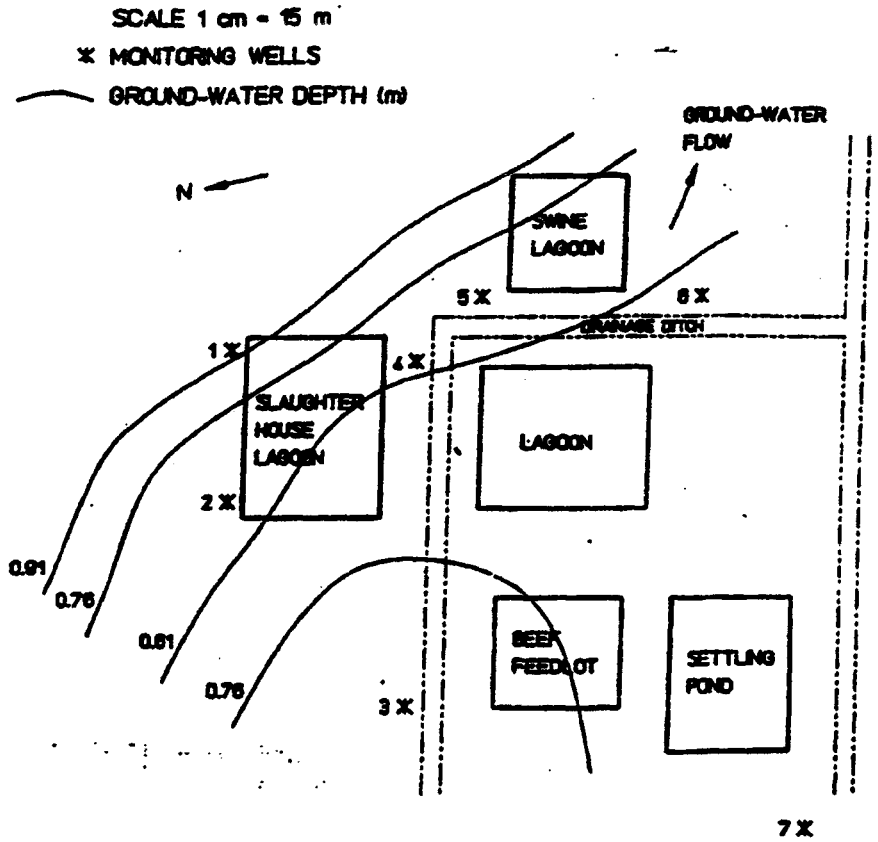

Fig. 2. Ground-water table depths at Site 2 on 4 March 1983.

an anaerobic lagoon. The swine lagoon handles the manure from 200 finishing hogs. The BOD loading rate for the slaughter house lagoon is in the range of 16–32 g/m^3 per day. All lagoons are lined with 15 cm of compacted clay. Solids are pumped from the settling basin and applied to land with a liquid manure spreader. Effluent from the three lagoons is applied to cropland by irrigation three or four times a year. None of the lagoons is pumped below the design operating level of 1·0 m. Depth and volume of the lagoons are presented in Table 1.

Monitoring wells

Six monitoring wells were installed at Site 1 and seven monitoring wells were installed at Site 2. The locations of the monitoring wells are shown in Figs 1 and 2. Polyvinyl chloride pipe of 32 mm diameter was used as well casing. All of the wells had 1·5 m of PVC screen with 0·25 mm slot width. All wells were

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for Lagoon Sites		
Silt (%)	Clay	
	4·0	
	8·0	
6·4	12·0	
14·4	4·0	
13·6	3·6	
5·2		
1·2		
9·0		
8·5		
11·0		
6·0		
22·6		
17·6		
19·6		
10·4		
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68·4		
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siliceous, mesic, typic ochraquults) and Pocomoke sand loam (coarse-loamy, siliceous, thermic typic umbraquults) soils. Both the Pocomoke and Fallsington soils are classified as poorly drained. Soil particle analysis of three cores taken at each site are presented in Table 2. At Site 1 below a depth of 0·9 m, the content is over 90% sand. The soil was relatively uniform over the entire site. Monitoring well logs indicated that from the 1·5–5·5 m depth, fine to medium sand was present. Below a depth of 5·5 m, the sand was coarse and contained some gravel.

At Site 2, the sand increased at the 0·9 m depth. The data presented in Table 2 are similar to the soil in the vicinity of the monitoring wells. Monitoring well logs indicated that from a depth of 1·7 to 6·1 m, there was medium to coarse sand with some thin clay lenses in several locations. Below a depth of 6·1 m, fine to medium sand with traces of silt was present.

Ground-water monitoring

The monitoring wells at both sites were sampled every other month from October 1982 to December 1985. Samples were taken with a battery-operated peristaltic pump. The well casing was pumped dry before a sample was taken. After the samples were collected they were stored at 0°C until they were analyzed. Most samples were analyzed within 48 h of their collection. Water table depths were measured each time wells were sampled. All samples were analyzed for nitrate-nitrite nitrogen, ammonium nitrogen, chlorides and total dissolved solids (TDS). Nitrate-nitrite nitrogen was analyzed by the Devarda's alloy reduction method (APHA, 1980). Ammonium nitrogen was analyzed by steam distillation and nesslerization and chlorides were analyzed by a specific ion probe. Total dissolved solids were analyzed by filtering and evaporation at 102°C.

RESULTS AND DISCUSSION

Geology and hydrology

The lagoons are located in the Atlantic Coastal Plain and the sediments are of the Pleistocene age. The principal aquifer system is the water-table aquifer. From pumping tests, Johnston (1973) estimated the average transmissivity of the aquifer is 994 m^2/day. The average annual recharge is 36 cm (Johnston, 1973).

At Site 1, the water-table depth varied from 3·0 to 5·3 m. The ground-water table depth contours for 4 March 1983 are shown in Fig. 1. The direction of ground-water flow is from Wells 4 and 3 toward Wells 1 and 5.

nitoring period, the water-table was below the elevation of
ıe lagoon.

water-table depth varied from 0·3 to 1·8 m. This means
ıe year the bottom of the lagoons were below the water-table.
water mounding has occurred around the lagoons. The
ɔn of ground-water movement is from Well 3 to Well 6. The
table depth contours for 4 March 1983 are shown in Fig. 2.

/ monitoring

ɛntrations of ammonium nitrogen, nitrate-nitrite nitrogen,
ıd TDS are presented in Tables 3–6.

age concentrations of ammonium nitrogen in the monitoring
ɛ 1 ranged from 7·84 to 960 mg/liter N. The highest ammonium
oncentrations occurred in Wells 1, 2, 5 and 6. Wells 1, 2 and 5 are
ells (4·6 m) and Well 6 is a deep well (12·2 m). These four wells are
ı the direction of ground-water flow away from the lagoon. All
ıllow wells had mean ammonium nitrogen concentrations above
iter N. The variations in ammonium concentration with time for
, 5 and 6 are shown in Fig. 3. The variation in ammonium
ırations in Well 2 was similar to the variation in Well 1. Background
ocated 1500 m from the lagoon had ammonium nitrogen con-
tions below 0·50 mg/liter N. Ammonium nitrogen concentrations in

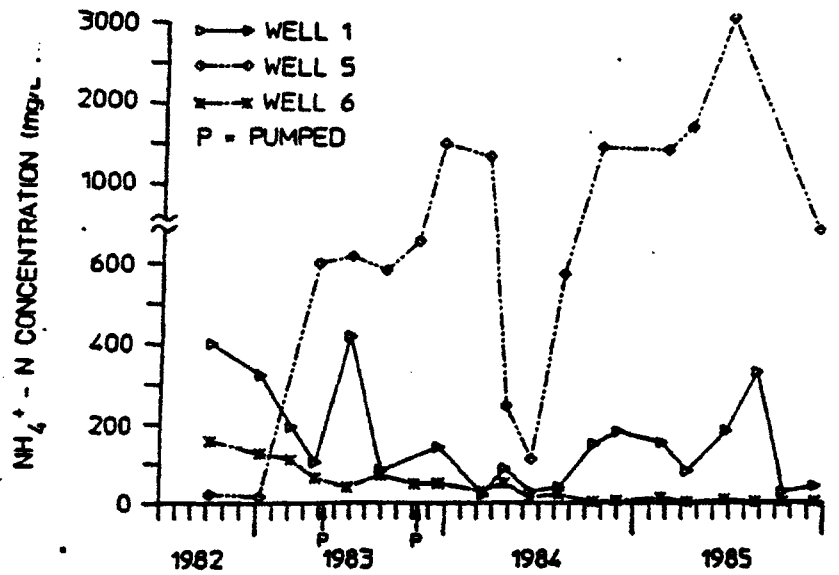

Fig. 3. Ammonium nitrogen concentrations in monitoring wells at Site I.

TABLE 3
Average Ammonium Nitrogen Concentrations of Monitoring Wells from October 1982 to December 1985

Well no.	*Well depth (m)*	*No. of samples*	*Concentration (mg/liter N)*		
			Mean	*Standard deviation*	*Range*
Site 1					
1	4·6	20	155	129	21·3–418
2	4·6	20	504	354	33·5–1 060
3	4·6	20	7·84	10·4	0·12–44·6
4	4·6	17	8·20	9·10	0·44–28·5
5	4·6	18	968	748	16·9–3 050
6	12·2	20	40·7	45·3	1·74–157
Site 2					
1	15·3	20	0·12	0·14	<0·05–0·49
2	4·6	18	0·86	0·46	0·28–1·93
3	4·6	20	0·71	1·35	<0·05–4·81
4	4·6	20	12·1	12·5	0·38–44·0
5	11·3	20	1·44	1·78	0·08–6·78
6	4·6	10	0·26	0·39	<0·05–1·16
7	4·6	20	5·03	9·29	<0·05–39·6

TABLE 4
Average Nitrate Nitrogen Concentrations of Monitoring Wells from October 1982 to December 1985

Well no.	*Well depth (m)*	*No. of samples*	*Concentration (mg/liter N)*[a]		
			Mean	*Standard deviation*	*Range*
Site 1					
1	4·6	20	51·8	182	0·30–814
2	4·6	20	7·79	9·43	0·20–29·1
3	4·6	20	33·9	27·6	1·92–78·1
4	4·6	17	40·5	29·9	0·50–109
5	4·6	18	3·74	2·93	0·72–12·8
6	12·2	20	19·9	14·0	4·28–54·3
Site 2					
1	15·3	20	0·29	0·27	<0·05–1·17
2	4·6	18	9·09	5·45	<0·05–27·7
3	4·6	20	5·88	6·78	0·11–29·1
4	4·6	20	2·78	5·18	<0·05–23·5
5	11·3	20	0·65	0·84	<0·05–3·18
6	4·6	10	1·84	0·86	0·12–3·15
7	4·6	20	0·36	0·39	<0·05–1·59

[a] Includes nitrite nitrogen.

TABLE 5
Average Chloride Concentrations of Monitoring Wells from October 1982 to December 1985

Well no.	*Well depth (m)*	*No. of samples*	*Concentration (mg/liter)*		
			Mean	*Standard deviation*	*Range*
Site 1					
1	4·6	20	97	66	18–281
2	4·6	20	270	156	108–540
3	4·6	20	12	6·0	3·0–25
4	4·6	17	19	14	2·8–46
5	4·6	18	239	106	66–447
6	12·2	20	35	15	18–66
Site 2					
1	15·3	20	11	4·4	6·4–21
2	4·6	18	137	66	9·4–272
3	4·6	20	50	22	11–88
4	4·6	20	54	37	2·9–144
5	11·3	20	26	19	14–88
6	4·6	10	10	2·0	7·9–14
7	4·6	20	30	17	8·5–57

TABLE 6
Average Total Dissolved Solids Concentrations of Monitoring Wells from October 1982 to December 1985

Well no.	*Well depth (m)*	*No. of samples*	*Concentration (mg/litre)*		
			Mean	*Standard deviation*	*Range*
Site 1					
1	4·6	20	860	390	231–1 810
2	4·6	20	2 560	1 170	697–4 400
3	4·6	20	514	314	181–1 570
4	4·6	17	564	283	199–1 040
5	4·6	18	1 280	874	253–3 220
6	12·2	20	370	162	247–451
Site 2					
1	15·3	20	150	85	73–307
2	4·6	18	768	558	383–2 440
3	4·6	20	239	110	86–567
4	4·6	20	453	340	201–1 436
5	11·3	20	189	84	51–301
6	4·6	10	198	217	39–571
7	4·6	20	655	616	120–1 930

Well 5 have been high since April 1982, but were above 1000 mg/liter N from October 1984 until June 1985, as indicated in Fig. 3. The soil has a cation exchange capacity of approximately 3·0 meq/100 g, so with the low exchange capacity, ammonium nitrogen can move through the soil profile to the water-table. The farmer emptied the lagoon twice a year. This would allow the clay liner to dry out and develop cracks, so excessive seepage would occur. Also, it was difficult to determine how well the lagoon was lined. There was no inspection during construction to indicate if the clay liner was installed according to the Soil Conservation Service specifications. Of the four wells with higher ammonium nitrogen concentrations, Well 6 had the lowest concentrations. Since Well 6 is deeper than the other three wells, higher ammonium nitrogen concentrations would be expected in the shallow wells because of less dilution. Ammonium nitrogen concentrations in Well 6 ranged from 1·74 to 157 mg/liter N with an average concentration of 40·7 mg/liter N. The highest concentrations occurred in Well 6 near the beginning of the monitoring period. There were a number of times before the monitoring started and during the monitoring that the lagoon overflowed in the vicinity of Well 6. This may also have contributed to the higher ammonium nitrogen concentrations in Well 6.

The farmer stopped feeding hogs in the spring of 1984, so no further waste was discharged to the lagoon. The lagoon was nearly full when he stopped discharging and it was not emptied, so there was a large hydraulic head that could have caused seepage through the bottom of the lagoon. After the farmer stopped discharging to the lagoon, ammonium nitrogen levels decreased in Well 6 but increased in Wells 1 and 5 until June or August 1985 and then decreased.

Even though Wells 3 and 4 are above the lagoon in the direction of ground-water flow, they are located close enough to the lagoon to have been impacted by it. The average ammonium nitrogen concentrations were 7·84 and 8·20 mg/liter N, respectively in Wells 3 and 4. These ammonium nitrogen concentrations are much higher than background levels found in ground water in the surrounding area.

The results show there is the potential for ammonium nitrogen to move through the soil profile to the ground-water from an anaerobic lagoon in a coarse textured soil. Anaerobic conditions may exist below the bottom of the lagoon so nitrification will not occur. Miller *et al.* (1976) also found that ammonium nitrogen moved below the bottom of two anaerobic hog manure lagoons in medium and coarse textured soils.

The variation of nitrate nitrogen with time for Wells 1, 5 and 6 at Site 1 is presented in Fig. 4. Average nitrate nitrogen concentrations ranged from 3·74 mg/liter N in Well 5 to 51·8 mg/liter N in Well 1. Only Wells 2 and 5 had average nitrate concentrations below 10 mg/liter N.

The high ammonium nitrogen concentrations in Wells 2 and 5 and the low nitrate nitrogen concentrations indicate very little nitrification is taking place in the vicinity of these wells. Based on the high nitrate nitrogen concentrations in the other wells and the lower ammonium nitrogen concentrations, some nitrification is occurring in the vicinity of the lagoon in some areas. Background nitrate nitrogen concentrations in monitoring wells 1500 m from the lagoon in an Evesboro loamy sand soil ranged from 0·55 to 19·6 mg/liter N. Nitrate nitrogen concentrations did not follow any seasonal trends in the monitoring wells around the lagoon. After the farmer stopped discharging waste into the lagoon, the nitrate nitrogen concentrations followed the same general trend as the ammonium nitrogen concentrations.

Chloride concentrations were highest in Wells 1, 2 and 5. Ammonium nitrogen concentrations were also highest in Wells 1, 2 and 5. The correlation coefficient between ammonium nitrogen and chloride concentrations for the six wells was 0·883 which indicates the lagoon is the source of ground-water quality degradation.

Total dissolved solids concentrations were high in all of the monitoring wells at Site 1 compared to background levels found in ground water 1500 m from the lagoon. Wells 2 and 5 which have the highest ammonium nitrogen concentrations also have the highest TDS concentrations. The correlation coefficient between ammonium nitrogen and TDS concentrations was 0·608.

Although there was considerable variation in ammonium nitrogen, nitrate nitrogen, chloride and TDS concentrations in the monitoring wells,

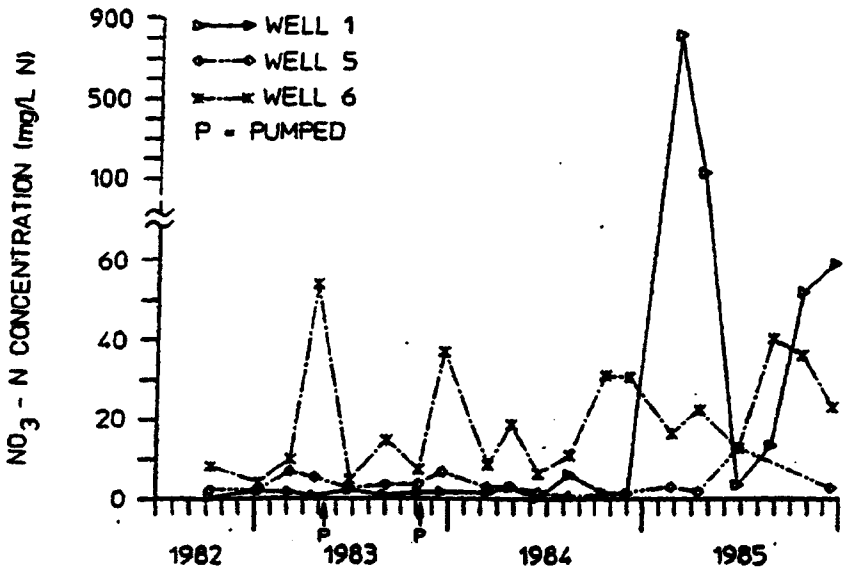

Fig. 4. Nitrate nitrogen concentrations in monitoring wells at Site 1.

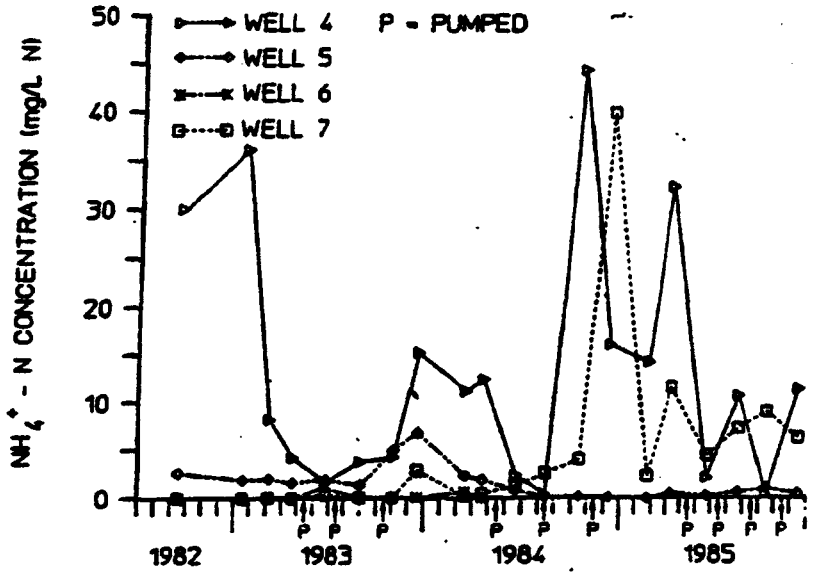

Fig. 5. Ammonium nitrogen concentrations in monitoring wells at Site 2.

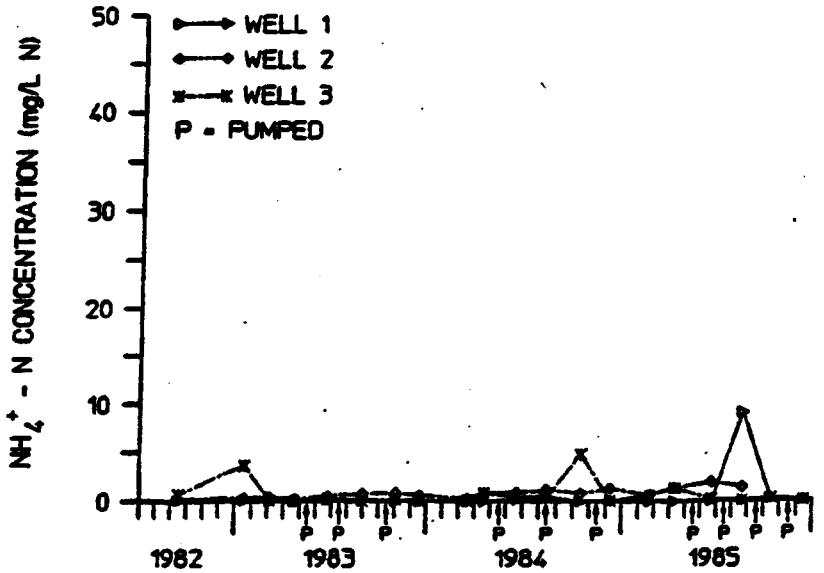

Fig. 6. Ammonium nitrogen concentrations in monitoring wells at Site 2.

all of the wells had concentrations above background levels. Since the lagoon was emptied twice a year, there was a variation in seepage from the lagoon throughout the year that caused the variation in pollutant concentrations in the monitoring wells. Twelve to fifteen months after waste was no longer discharged to the lagoon, ammonium nitrogen, nitrate nitrogen, chloride and TDS concentrations started to decrease in the monitoring wells. If the farmer had maintained the liquid depth at the design level in the lagoon throughout the year, the lagoon should have had less impact on ground-water quality. Research has shown that over a period of time lagoons and earthen manure storage basins will self-seal due to biological action (Miller *et al.*, 1985).

Variation in the ammonium nitrogen concentrations in the monitoring wells at Site 2 is shown in Figs 5 and 6. Well 6 was destroyed in May 1984 so no more samples were collected from it. Only two of the monitoring wells around the lagoon and the background monitoring well had ammonium concentrations above 1·0 mg/liter N. Wells 4 and 5, where ground-water mounding is occurring, had the highest ammonium nitrogen concentrations. Ammonium nitrogen levels in Well 5 decreased since mid-1984 as indicated in Fig. 6. Ammonium nitrogen levels in Well 7 (background) only started to increase in 1984 after a septic tank was installed near the well. Before mid-1984, ammonium nitrogen concentrations in Well 7 ranged from <0·05 to 2·89 mg/liter N.

Nitrate nitrogen concentration data for site 2 are presented in Figs 7 and 8 and Table 4.

None of the monitoring wells had average nitrate nitrogen concentrations above the EPA drinking water standard of 10 mg/liter N. There was a great fluctuation in nitrate nitrogen concentrations in Wells 2, 3 and 4. Nitrate nitrogen concentrations increased in Well 3 in 1985 but decreased in Well 2.

Chloride concentrations were higher in Wells 2, 3 and 4 at Site 2 than the other monitoring wells. Nitrate nitrogen concentrations were also higher in these wells, which indicates some seepage was occurring from the slaugher house lagoon. The highest chloride concentrations did not occur at the same time as the highest nitrate concentrations shown. Chloride concentrations decreased in Well 2 in 1985 but increased in Well 3 from the autumn of 1984. The correlation coefficient between nitrate nitrogen and chloride concentrations was 0·901.

Total dissolved solids concentrations at Site 2 were highest in Wells 2 and 4. Total inorganic nitrogen concentrations (ammonium nitrogen + nitrate nitrogen) were also highest in these wells. The same trend was observed at Site 1, although the ammonium nitrogen, nitrate nitrogen, and TDS concentrations were much higher at Site 1. Total dissolved solids

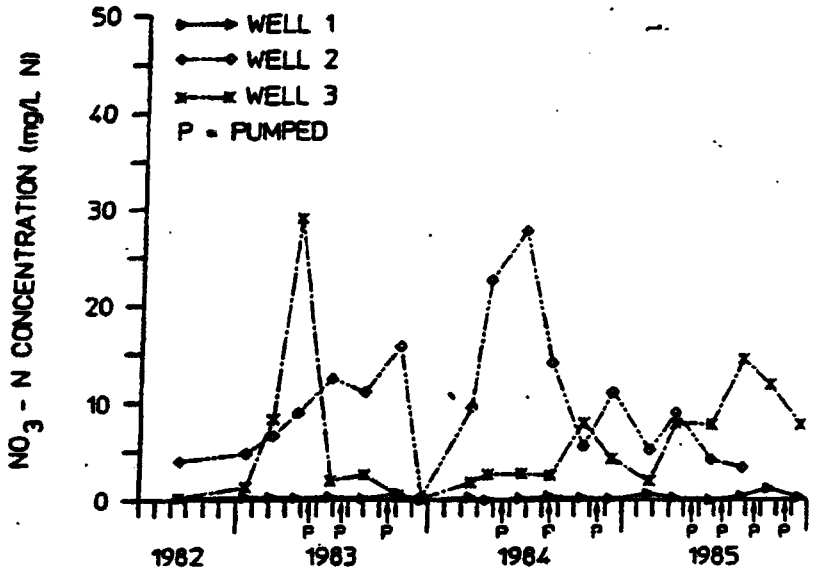

Fig. 7. Nitrate nitrogen concentrations in monitoring wells at Site 2.

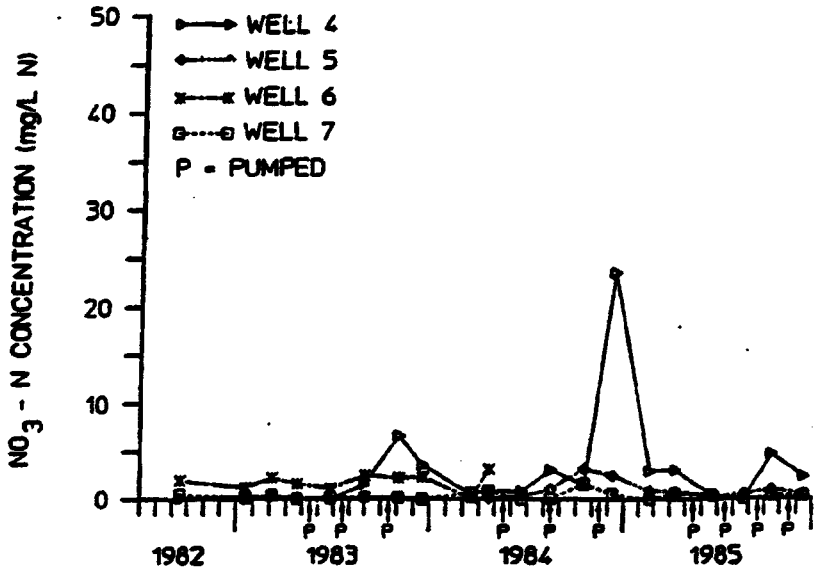

Fig. 8. Nitrate nitrogen concentrations in monitoring wells at Site 2.

concentrations started to increase in Well 7 in 1984 at the same time as ammonium nitrogen concentrations started to increase. The correlation coefficient between TDS and total inorganic nitrogen was 0·610 at Site 2.

The lagoons at Site 2 did not have as severe an impact on ground-water quality as the lagoon at Site 1. A liquid depth of at least 1·0 m was maintained in the lagoons at Site 2. The lagoon at Site 1 is located in an excessively well-drained soil while the lagoons at Site 2 have soils that are poorly drained.

Ground-water quality was impacted at both lagoon sites. There was no seasonal trend in ammonium nitrogen, nitrate nitrogen, chloride or TDS concentrations in the ground water. At Site 2 that had poorly drained soils, the water-table was above the bottom of the lagoons during part of the year. There was a strong correlation between chloride and nitrate concentrations at both sites. Chlorides could be used as a tracer in monitoring seepage from animal waste lagoons on the Delmarva Peninsula.

The research shows that clay-lined lagoons installed in coarse textured soils with high water-tables may cause ground-water contamination if the lagoons are not operated properly. Liquid levels should not be pumped below design levels to protect the clay from drying out and cracking. Clay liners should also be inspected during construction to make sure they are installed according to specifications. On the Delmarva Peninsula in coarse textured soils, ammonium nitrogen concentrations may be higher than nitrate nitrogen concentrations in the ground-water around some animal waste lagoons. Monitoring only nitrate nitrogen concentrations may not give a true picture of the extent of the seepage occurring. In Delaware, farmers are only required to monitor nitrate nitrogen concentration in the ground water as part of their lagoon permit requirements. The research also raised the question of how much ground-water contamination manure storage basins are causing on the Delmarva Peninsula. Manure storage basins are generally completely emptied twice a year. The lagoon at Site 1, where severe ground-water contamination was occurring, was operated more like a manure storage basin than a lagoon, although it was designed as a lagoon and not a storage basin.

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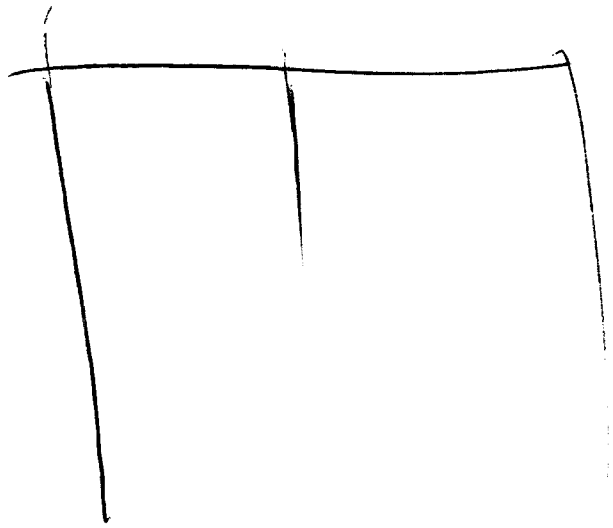

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Contamination in Ontario farmstead domestic wells and its association with agriculture: 2. Results from multilevel monitoring well installations

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Abstract

Multilevel monitoring wells (MLWs) were installed at 144 farms which were part of a province-wide survey of farm drinking water wells conducted in 1991–1992 in Ontario, Canada. The multilevel sites were selected in areas characterized by coarse-textured soils, on farms typifying local agricultural enterprises. The MLWs were installed in cultivated fields adjacent to the location of the drinking water wells on each farm (within 200 m). On 16 of these farms, MLWs were also installed in a woodlot adjacent to the field site. Water samples were collected on two occasions (winter and summer) and analyses were conducted for nitrate (NO_3^-), typical bacteria and a selected suite of common pesticides. At 23% of the sites, concentrations in 50% or more of the monitored intervals exceeded the provincial drinking water standard (MAC) for NO_3^--N during both sampling periods. Significantly higher frequencies of total coliform contamination were encountered in the winter (66%) than in the summer (36%). Very few pesticide detections were recorded. The average concentration of NO_3^- with depth in multilevel wells decreased from approximately 10 mg N l^{-1} near the water table to 3 mg N l^{-1} at a depth of about 6.5 m. Bacteria concentrations remained more uniform with depth but decreased significantly in the summer. For most analytes, contaminant frequency was similar for both the drinking water wells and multilevel wells. The occurrence of elevated levels of contamination in the water wells appeared to be associated more with activities on the cultivated fields than with on-farm point

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sources of contamination. Groundwater quality determined using drinking water wells was consistent with conclusions drawn from multilevel monitoring wells, indicating the utility of water well survey data for assessing groundwater quality within the rural environment. No correlations were observed between the type of cropping practice and the frequency of groundwater contamination. Farms where manure spreading occurred, however, had significantly higher occurrences of contamination from NO_3^- and bacteria than farms where manure was not applied along with commercial N-fertilizer. The uncultivated conditions beneath the woodlot areas appeared to provide an environment conducive to attenuation of nitrate but not bacteria.

Keywords: Wells; Drinking water; Nitrate; Coliform

1. Introduction

Throughout the Province of Ontario, groundwater provides approximately 30% of the demand, but in some communities between 80% and 90% of the supply is drawn from local aquifers. Routine sampling of the municipal wells has resulted in the development of local data bases that can be used to indicate transient trends in water quality. This provides the information necessary for early detection of deteriorating groundwater quality in the municipal system. In contrast to urban users, the rural population depends almost entirely on groundwater, but rarely has access to historical water quality data. The individual users are generally responsible for their own quality monitoring. As a result, the condition of much of the rural groundwater resource is not well documented.

Local- and regional-scale surveys of well water quality have provided some insight into the general condition of rural groundwater resources. Studies such as those completed by Exner and Spalding (1985), Mackay and Smith (1990) and Goss et al. (1998), have shown significant levels of contamination in rural wells. Several reasons for a large proportion of water wells being contaminated have been proposed. The application of commercial fertilizers, animal manures, and a wide variety of insecticides and herbicides to field crops, is often cited as the source of potential nonpoint or diffuse contamination over extensive land areas. At the farm scale, point-sources of contamination such as septic fields, manure storage areas, and feedlots, together with poor well-maintenance programs, are often implicated as contributing to problems of well-water quality. Domestic farmstead wells are often sited close to the barnyard, which enhances the possibility that water quality may be influenced more by point sources than by cropping practices on the cultivated land. Alternatively, wells may also be located under grassed areas away from barnyards and cultivated fields. This raised questions as to the usefulness of surveys of domestic well-water quality in identifying the impact of agricultural practices. Multilevel monitoring wells (MLWs) have been used to investigate contaminants that may come from point sources in discrete plumes, and those that come from diffuse sources, both of which tend to stratify within an aquifer (Sudicky, 1986). These monitoring wells could also be used to assess the quality of groundwater below cultivated fields with typical agricultural management for that farm, thereby helping to relate specific agricultural activities to the occurrence of contaminants in the local aquifer. It was also anticipated that information from these wells could be used to

assist in determining whether contamination of the domestic well has resulted from point and nonpoint sources on the farm.

In Ontario, where nearly all arable rural land is in cultivation, potential impacts of agricultural activities on groundwater quality has become a major concern. The magnitude of the problem and primary factors controlling the occurrence of groundwater contamination in agricultural settings are site- or regionally specific and are still poorly understood. It is difficult to assess whether the groundwater contamination is related to a localized source or a more regional, nonpoint source without extensive field monitoring. In order to evaluate the influence of applying proposed best management practices at the farm scale, the influence of current practices on the shallow groundwater resource needs to be more completely understood.

In the companion paper, (Goss et al., 1998), approximately 1300 domestic wells were sampled and analyzed for the occurrence of nitrate (NO_3^-), bacteria, selected pesticides and petroleum derivatives throughout Ontario. In conjunction with this well survey, at 144 of the farms where the drinking water well was sampled, multilevel monitoring wells were installed in cultivated fields adjacent to the location of the water well, generally within 200 m. Samples from the MLWs were analyzed for the same set of target contaminants.

This paper describes the multilevel well investigation including the site selection process, installation techniques, sampling protocols and general results. One of the main objectives of the work was to investigate the nature of groundwater contamination beneath cultivated fields near the farmstead water wells at locations considered to be outside the influence of contaminant point sources associated with the barnyard. In this way, impacts on groundwater quality from the cropping activity could be assessed. The data are initially examined relative to general levels of contamination and persistence with depth. Another objective was to determine the relationship between the occurrence of groundwater contamination in the multilevel wells and the associated drinking-water wells relative to agricultural land-use practices. The results from both data sets are considered together in order to evaluate the utility of drinking-water-well surveys in assessing regional groundwater quality in the rural environment. In addition, the significance of distributed or nonpoint sources of contamination with respect to the deterioration of groundwater quality in rural wells is investigated within the scope of the data set.

2. Materials and methods

2.1. Site selection

The majority of farm wells throughout the Province of Ontario are completed in unconsolidated glacial overburden sediments. These sediments generally consist of complex sequences of materials ranging from clay tills to coarse gravels. Well types range from shallow-dug wells to drilled and cased wells, some of which are completed in the bedrock underlying the glacial sediments. On a given farm, the well type depends on the local hydrogeologic conditions. Within the scope of the province-wide well

survey reported in the companion paper (Goss et al., 1998), water wells were randomly selected within all common hydrogeologic settings across Ontario. The selection process used for the multilevel sites was somewhat less random than that used in the farm-well survey, although every effort was made to choose locations that were representative of conditions typical of those across the province.

A total of 144 farms, typical of local agricultural enterprises, was selected for the multilevel monitoring well-installation program. Although many of these fields are tile drained, none receive additional irrigation water. At each farm, a MLW was installed in a field adjacent to the house or yard, so as to be close to the farm well but away from the influence of barnyard activities. Approximately 40% of the sites were located in sandy soils and about 20% in fine-textured clays and silts. The remaining 40% of the monitoring wells were placed at sites where the soil texture was highly variable with depth. On 16 randomly selected farms, a second MLW was also installed in a woodlot adjacent to the field site for a combined total of 160 MLWs. The objective of these woodlot sites was to compare the nature of groundwater contamination under cultivated and noncultivated land.

2.2. Multilevel well installation

The multilevel monitoring wells were installed during the winter of 1991–1992 with an auger drill rig. Three different multilevel designs were used depending on the nature of the subsurface geology at a given site (Fig. 1). At sites where the subsurface material was cohesive and not expected to collapse after drilling (i.e., clay and clay-rich loams), solid stem auger flights were used to drill the borehole. Two installation techniques were used in these types of materials. The first involved the placement of individual 2.0-cm-diameter PVC tubes in a single borehole with a sand pack placed at the base of the tube where a screen was positioned, and a bentonite seal placed between the top of the sand pack and ground surface (Fig. 1a). At sites where the subsurface material was extremely stiff, auger flights could cause severe smearing of the borehole wall restricting inflow of groundwater. To minimize the effect of smearing, individual boreholes were drilled to each desired level and the base of the borehole was sub-cored with a thin-walled sampler prior to the installation of the screen.

At sites where the sediment contained less clay material and were less susceptible to smearing, several 2.0-cm-diameter PVC monitoring wells were installed in the same borehole. Individual sand packs were installed around each screen and the vertically staggered sand packs were separated by bentonite seals (Fig. 1b). Generally three to four monitoring points were installed at each site in these heavier soils, to an average maximum depth of 5 m below the water table. All boreholes were completed about 30 cm below ground surface to avoid interference with future cultivation activity.

In noncohesive materials such as sand and gravel, the boreholes were drilled using a rig with a hollow-stemmed auger, and a bundle-type well design was adopted (Fig. 1c). These monitoring wells consisted of a 2.0-cm-diameter PVC centre stock with a 15-cm well screen fixed to the base and a series of 1.5-cm-diameter polyethylene tubes fastened to the centre stock at different levels. Each of the tubes was connected to a small well screen. The device was placed down the inside of the hollow stem augers which were

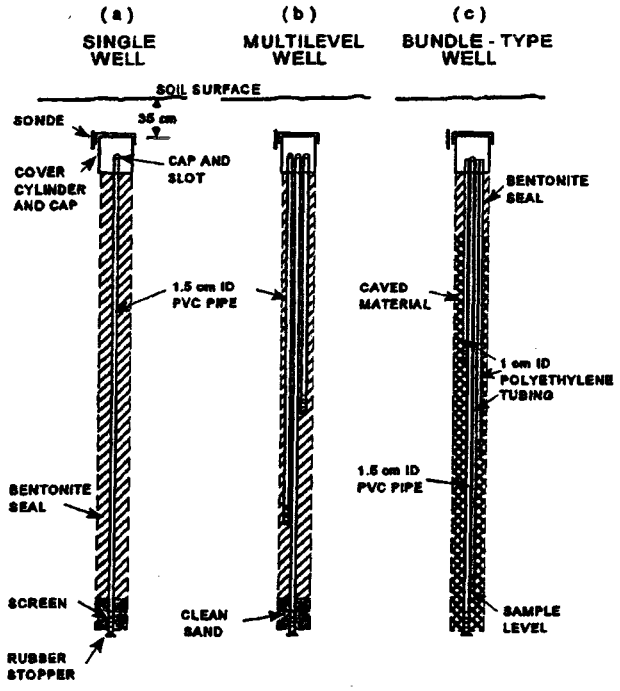

Fig. 1. Construction details for multilevel monitoring wells.

subsequently removed permitting the loose material to collapse around the monitoring well effectively retaining the natural hydraulic conductivity of the in situ sediment around the installation. In these types of materials, the installations ranged between 8 m and 12 m in depth with 5 to 6 individual monitoring levels placed between the water table and the bottom of the borehole. After installation, the multilevel wells were developed to remove fine sediments and to ensure adequate hydraulic response.

2.3. Sampling and chemical analysis protocols

The multilevel wells were first sampled in winter between November 1991 and January 1992. The second sampling was carried out in summer between June and July 1992. Sampling of the multilevel monitoring wells involved a three-step sequence. Prior to sample collection, the water level in each sampling tube was measured to determine the hydraulic head. A length of Teflon tubing was then inserted into each sampling tube and approximately 2 l of water was withdrawn using a peristaltic pump to purge both the well and the sampling system. After the well recovered from purging, the sample was collected using the Teflon sampling tube and peristaltic pump.

The samples were analysed for nitrate, total and faecal coliform bacteria and a specific set of pesticides. Sample handling and analysis procedures were the same as those described by Goss et al. (1998). In brief, samples for analysis of nitrate were

collected in 20-ml polyethylene vials, and samples for bacteriological analysis were collected in 250-ml plastic, sterile, screw-capped bottles containing sodium thiosulphate. Samples for analysis of pesticide content were collected in 1-l amber glass bottles. All samples were stored in coolers for transport to the various analytical facilities.

Nitrate concentration in the samples was measured spectrophotometrically after reduction to nitrite in a copper–cadmium column using a Technicon Random Access Automated Chemistry System TRAACS-800 (Alpha-Laval., Stockholm, Sweden). The bacteriological samples were analysed within 24 h at local Ministry of Health laboratories, where standard membrane filtration techniques were followed for detection of coliforms, faecal coliforms and faecal streptococci (Clesceri et al., 1989).

For the pesticide analyses, atrazine and a metabolite deethylatrazine, alachlor, metolachlor, metribuzin and cyanazine were selected for analysis because of their common use in Ontario. Each sample was subsampled and analysed for the occurrence of atrazine by an enzyme-linked imunoassay system (Agri-Diagnostics, Moorestown, NJ, USA), and for alachlor, metolachlor, deethylatrazine, metribuzin and cyanazine by solid phase extraction followed by gas chromatography. Positive detections were confirmed on the remaining sample by chloroform liquid–liquid extraction followed by gas–liquid chromatography. Detection limits ($\mu g\ l^{-1}$) were: alachlor, 0.2; atrazine, 0.05; cyanazine, 0.2; metribuzin, 0.05; metolachlor, 0.2; and deethylatrazine, 0.1.

Slightly different subsets of the multilevel-well network were sampled for nitrate and the pesticides during the two sampling campaigns as a result of local site conditions. Two samples were collected from the same multilevel well at approximately 90% of the sites. Bacterial analysis was conducted on a subset of approximately 50 of the multilevel wells during both the winter and summer sampling rounds. Of these wells, 70% were located on farms where a manure-management system was present and 30% where there was no manure system.

To evaluate the general level of contamination in samples collected from the multilevel monitoring wells, observed concentrations were compared with the Ontario

Table 1
Drinking water objectives, for the pesticides investigated in this study, nitrate, and coliform bacteria

Contaminant	Ontario (MAC or IMAC)
Pesticides ($\mu g\ l^{-1}$)	
Alachlor	5
Metolachlor	50
Atrazine	5[a]
Metribuzin	80
Cyanazine	10
Nitrate (MAC value) (mg N l^{-1})	10
Bacteria, colony-forming units (100 ml)$^{-1}$[b]	
Total coliform	5
Faecal coliform	0

[a]MAC includes deethylatrazine.
[b]In 95% of samples from a municipal supply.

Drinking Water Objectives as applied to private well supplies (MOE, 1992) (Table 1). Individual monitoring points of the MLWs were deemed contaminated if one or more of the target contaminants were present in excess of the maximum acceptable concentration (MAC) or the interim maximum acceptable concentration (IMAC) in the case of most pesticides. For several groups of bacteria, no current objectives were available, so values suggested by the MOH (M. Brodsky, personal communication, July 1992) were adopted.

2.4. Survey interpretation and statistical analysis

The percentage of contaminated wells in a given category was determined arithmetically, and then a 95% confidence interval (C.I.) was calculated for each percentage contamination value by assuming a binomial distribution. A difference between two was considered to be significant if the values $\pm$ their 95% C.I. did not overlap.

Interpretation of contamination levels in multilevel monitoring wells requires special consideration because of the variation between wells in the number of depths sampled, and the actual depths of ports below the water table. At every multilevel well, several samples were collected, each from a specific depth in the subsurface. These data provided an indication of the distribution of a given contaminant with depth. At over 80% of the farms where the multilevel wells were installed, the drinking-water wells were fairly shallow dug or bored wells. Water collected from the latter wells was a mixture of groundwater entering along the entire length of the well screen and, as such, gave depth-averaged values. To make a more realistic comparison between water wells and multilevel wells, a multilevel site was considered contaminated if 50% or more of the sampled levels exceeded the MAC guidelines. This approach also avoided the inherent bias involved with calculating arithmetic average concentrations of contaminants at sites with different numbers of sampling levels or where a single, anomalously high concentration was recorded.

At each farm selected for multilevel monitoring, a detailed questionnaire was completed with the assistance of the farmer during each sampling episode. The questionnaire sought information on cropping practice, livestock operations, and locations of septic fields, feedlots, manure storage facilities and milkhouse waste-disposal systems. Specific land-use practices were classified according to the cropping system categories defined by the Ontario Ministry of Agriculture and Food (OMAF, 1988).

Questions related to well construction, well-maintenance schedules and historical water quality were also included in the questionnaire. This information, along with the data from the water sample analyses, were compiled in a single data base designed to be GIS compatible.

3. Results and discussion

3.1. General level of contamination

The data are presented and discussed both in terms of the maximum concentrations observed in individual multilevel wells, and as a classification of contaminated sites

based on 50% or more of the ports of a given well yielding water having concentrations above the maximum acceptable for a contaminant. In both the winter and summer about 45% of the multilevel sites exceeded maximum NO_3^--N concentrations of 10 mg l^{-1} (Table 2a). It is also notable that a third of the wells had concentrations from 5 to 10 mg NO_3^--N l^{-1}. The maximum concentration measured in the winter was 78 mg NO_3^--N l^{-1}, and in the summer was 87 mg NO_3^--N l^{-1}. The actual NO_3^- concentrations observed at each of the sites in both winter and summer were very consistent. For example, 81% of the multilevels contaminated with NO_3^- in winter also contained in excess of 10 mg NO_3^--N l^{-1} in the summer. No clear seasonal trend was apparent in the NO_3^- data.

Approximately 80% and 56% of MLWs tested for coliform colonies in the winter and summer respectively contained more than five colonies per 100 ml (Table 2a). The loss in bacterial contamination from winter to summer, in the MLWs, was significant at $p < 0.05$. When the values obtained in the summer were linearly regressed on those obtained in the winter, the correlation was not significant ($p > 0.05$). This indicates that

Table 2

(a) Multilevel results using maximum values, where winter and summer data available

Contaminant	No. of wells tested	Exceeds objectives	
		Winter (%)	Summer (%)
Total coliform	50	80.0±11.3[a]	56.0±14.0
Faecal coliform	47	25.5±12.7	12.8±9.7
E. coli > 0	0	–	–
Faecal coliform or *E. coli*	51	23.5±11.9	11.8±9.0
Total coliform or faecal coliform	46	78.3±12.2	52.2±14.7
Bacteria[b]	50	80.0±11.3	56.0±14.0
NO_3^--N > 10.0	137	45.3±8.5	45.3±8.5
Bacteria or NO_3^--N	50	88.0±9.2	78.0±11.7
Bacteria and NO_3^--N	50	44.0±14.0	26.0±12.4
5.0 < = NO_3^--N < = 10.0	137	38.0±8.3	29.2±7.8

(b) Multilevel well contamination (≥ 50% of intervals > MAC), where both winter and summer data are available.

Total coliform	50	66.0±13.4[a]	36.0±13.6
Faecal coliform	47	12.8±9.7	4.3±5.9
E. coli > 0	0	–	–
Faecal coliform or *E. coli*	51	11.8±9.0	3.9±5.4
Total coliform or faecal coliform	46	67.4±13.8	32.6±13.8
Bacteria[b]	50	66.0±13.4	36.0±13.6
NO_3^--N > 10.0	137	22.6±7.1	24.1±7.3
Bacteria or NO_3^--N	50	72.0±12.7	54.0±14.1
Bacteria and NO_3^--N	50	18.0±10.9	10.0±8.5
5.0 ≤ NO_3^--N ≤ 10.0	137	13.1±5.8	9.5±5.0

[a] ±95% confidence interval.
[b] Bacteria include total coliforms, and faecal coliforms or *E. coli*.

not only were the bacteria concentrations significantly lower in the summer but also a different set of wells was contaminated. Goss et al. (1998) reported a similar finding in the results for drinking water wells. Similar trends were observed in results for faecal bacteria where 26% of the multilevel sites investigated had some colonies present in winter compared to only 13% in the summer (Table 2a).

Using the criterion of half or more of the monitored levels exceeding the MAC for drinking water, approximately 23% of the sites were classified as contaminated with NO_3^- in both the winter and summer (Table 2b). The percentages for total coliform bacteria in winter and summer, 66% and 36% respectively, were about 20% less than the values obtained using the maximum concentration data (Table 2a). Contamination with faecal coliform bacteria also showed a significant decrease under this criterion (13% in winter and 4.3% in summer samples). Based on both approaches to data analysis, contamination with bacteria was observed to be significantly more variable, both spatially and temporally over time than was NO_3^-. This illustrates the complexity of the processes influencing the fate and transport behaviour of bacteria in the shallow groundwater environment, topics that are currently not completely understood.

Detections of pesticides in the multilevel monitoring wells were infrequent. Groundwater at five sites had detectable pesticide residues in the winter, while pesticides were detected at three additional sites in the summer. Most of these detections were either atrazine or deethylatrazine. Pesticide concentrations did not exceed the IMAC values in the winter. However, in the summer, cyanazine (60 $\mu g\ l^{-1}$) was found in one well, which was significantly higher than the IMAC limit of 10 $\mu g\ l^{-1}$. The maximum concentrations of atrazine and deethylatrazine were 4.7 $\mu g\ l^{-1}$ and 11 $\mu g\ l^{-1}$ respectively, with three wells exceeding the MAC value of 5 $\mu g\ l^{-1}$. No significant correlations were observed between the occurrence of detectable pesticide residues and local site conditions.

3.2. Persistence of nitrate and bacteria with depth

The highest concentrations of NO_3^- in multilevel wells occurred within a depth of 3 m below the water table (Fig. 2). A quadratic polynomial model fit ($R^2 = 0.05$, $p < 0.001$) to the cumulative data indicated that average concentration decreased from approximately 10.0 mg N l^{-1} near the water table to 3.0 mg N l^{-1} at a depth of 6.5 m. This profile was not affected by season. To assess the distribution of nitrate contamination with depth, data from the MLWs were ordered by depth below the water table and then divided into five approximately equal size groups. Frequency of sample concentrations above the MAC limits for each group are plotted against average depth in Fig. 3. Based on the results from both sample rounds, the proportion of samples that exceeded the MAC guidelines showed a distribution with depth similar to that noted for NO_3^- concentrations. Over 30% of the monitoring points located near the water table exceeded the drinking water limit. This decreased to about 10% of the samples taken from the 6.5-m average depth below the water table. The form of these concentration profiles remained fairly consistent at each site, indicating stability on a seasonal basis. The variation in depth is similar to that reported for the proportion of farmstead domestic wells contaminated with nitrate (Goss et al., 1998). This decreasing trend with depth

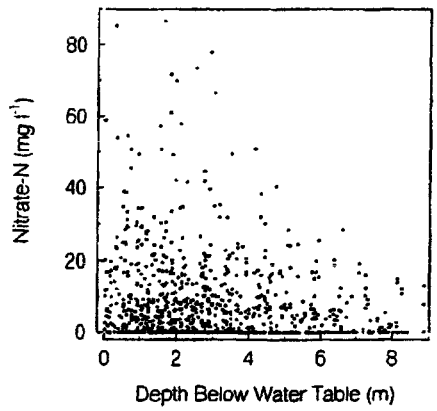

Fig. 2. Scatter plot of nitrate-N concentrations at various depths below the water table, from sampling points of multilevel wells in the winter and summer.

suggests that an attenuation mechanism may be actively influencing the depth persistence of nitrate beneath the cultivated fields. Denitrification has been observed to be enhanced with depth below the water table in several studies including Gillham (1991). More detailed geochemical investigation would be required, however, to evaluate the mechanism or mechanisms responsible for the decrease in nitrate concentration with depth.

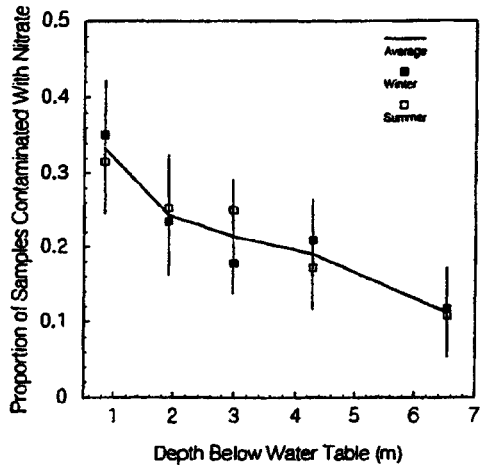

Fig. 3. Variation with depth in the proportion of samples from multilevel wells contaminated by nitrate. The plotted points are the proportion of contaminated wells in the 0.20 depth percentiles for the winter (closed square) and summer sampling (open square). The solid line connects the average of the winter and summer values. Error bars are 95% confidence intervals on the average values.

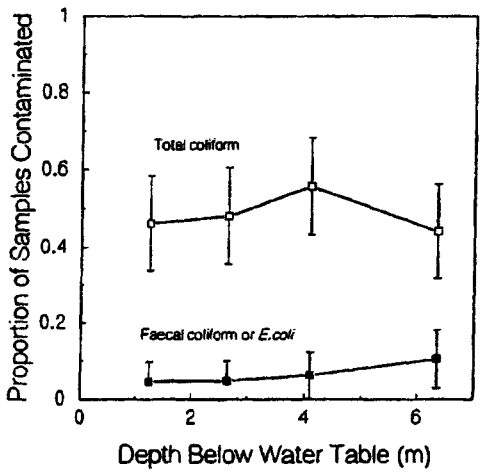

Fig. 4. Variation with depth in the proportion of multilevel wells contaminated by coliform or *E. coli* bacteria. The plotted points are the proportion of contaminated wells in the 0.25 depth percentiles. Error bars are 95% confidence intervals for the plotted values.

The profiles of contamination with total coliform or faecal coliform bacteria are significantly different than the nitrate profiles. The average proportions of sample points that exceeded the MAC limits for total coliform and combined values for faecal coliform and *E. coli*, were relatively uniform with depth (Fig. 4). This contrasted strongly with the progressive decrease with depth observed for NO_3^-. The proportion of samples contaminated with bacteria was greater in the winter than in the summer (Table 2), but this difference did not occur consistently for the various depth intervals. Again this likely reflects the complexity of the processes influencing the fate of bacteria in the subsurface which is beyond the scope of this study.

Although the bacterial data set was relatively small and the majority of the sites were selected in areas with sandy soils, sites on finer-textured soils also showed no significant difference in frequency of contamination or persistence with depth. In fact, bacteria concentrations tended to be slightly greater on average within clayey-soils than within coarser-textured soils.

3.3. Contaminant occurrence in multilevel wells and associated water wells

Both the multilevel wells and their associated drinking water wells were sampled at approximately 40 sites for bacterial analysis and 121 sites for nitrate analysis during both the winter and summer. The frequency of NO_3^- contamination encountered in both the MLWs and water wells was similar for both sampling sets, averaging 24% and 32% respectively (Table 3). When average NO_3^- concentrations, based on all samples from a given multilevel, were compared to the concentration in the wells on the same farms, it was observed that at approximately 50% of the sites concentrations were similar and at

Table 3
Percentage of wells contaminated (≥ 50% of intervals > MAC) at sites with a water well and multilevel well and where samples from both groups of wells were analyzed in winter and summer

Contaminant	No. of wells	Multilevel wells		Water wells	
		Winter (%)	Summer (%)	Winter (%)	Summer (%)
Total coliform	43	65.1 ± 14.5[a]	39.5 ± 14.9	23.3 ± 12.9	32.6 ± 14.3
Faecal coliform	40	12.5 ± 10.5	2.5 ± 4.9	12.5 ± 10.5	17.5 ± 12.0
E. coli > 0	0	–	–	–	–
Faecal coliform or *E. coli*	44	11.4 ± 9.6	4.5 ± 6.3	11.4 ± 9.6	15.9 ± 11.0
Total coliform or faecal coliform	40	67.5 ± 14.8	37.5 ± 15.3	25.0 ± 13.7	35.0 ± 15.1
Bacteria[b]	43	65.1 ± 14.5	39.5 ± 14.9	23.3 ± 12.9	34.9 ± 14.5
NO_3^--N > 10.0	121	22.3 ± 7.6	25.6 ± 7.9	29.8 ± 8.3	33.9 ± 8.6
Bacteria or NO_3^--N	43	72.1 ± 13.7	55.8 ± 15.1	48.8 ± 15.2	55.8 ± 15.1
Bacteria and NO_3^--N	43	16.3 ± 11.3	11.6 ± 9.8	9.3 ± 8.9	16.3 ± 11.3
5.0 < = NO_3^--N < = 10.0	121	14.9 ± 6.5	9.9 ± 5.4	16.5 ± 6.8	16.5 ± 6.8

[a] ±95% confidence interval.
[b] Bacteria include total coliforms, and faecal coliforms or *E. coli*.

26% of the sites, the water well concentrations were higher. Consequently, conclusions related to the regional occurrence of NO_3^- contamination based on samples from drinking-water wells appear to be reasonably consistent with those based on the more sophisticated multilevel wells. This provides support to the representative nature of regional water well surveys as a method for gauging large-scale groundwater quality.

Comparisons between the bacteria data sets from both the multilevel wells and the water wells are less consistent. Substantially more seasonal variability was observed in the multilevel wells than in the water wells (Table 3). During the winter survey, contamination frequency by faecal coliform bacteria was similar in both sets of wells, however in the summer, the number of contaminated multilevel sites dropped significantly. For total coliform on the other hand, a substantially higher frequency of multilevel wells were contaminated in the winter as compared to the water wells, but again this percentage decreased substantially in the summer. Although the water well data were not consistent with the seasonal effects observed in the multilevel wells, the general frequency of contamination occurrence is in reasonable agreement and, as observed for NO_3^-, the water well data likely provides a fairly representative indication of bacteria contamination on a regional basis.

Results from farms where the concentration of NO_3^- in the water well exceeded 3 mg N l^{-1}, considered to be the upper limit of NO_3^- concentrations in groundwater not influenced by anthropogenic activity (Madison and Brunett, 1985), were evaluated further. A simple linear regression of the concentration of NO_3^- in the farm well on the maximum concentration in each multilevel well accounted for 21% of the variance in the data and the slope and intercept values were both significantly different from zero ($p < 0.001$). Contamination by NO_3^- in the farm well was therefore reasonably well-correlated with contamination beneath the adjacent cultivated field. Upon detailed examination of the on-farm conditions, no clear associations could be made between

contamination in the water wells and potential point sources of contamination. As such, it would appear that the activity on the cultivated field can be directly associated with the occurrence of elevated NO_3^- levels in the drinking-water wells. Schepers et al. (1991) reported that irrigation wells in Nebraska reflected the N-fertilizer practices on the fields.

3.4. Land use practices, manure application and nitrogen management

Results from the multilevel wells indicated that no land-use class (OMAF, 1988) had significantly more nitrate contamination than any other. There was a wide range of agricultural activity encountered at the multilevel sites so the population size for any given land-use practice was relatively small. The average level of nitrate contamination was greatest in some of the minor land-use systems such as orchards or tobacco. The majority of the multilevel wells, however, were located in fields with row crop systems, corn systems or tobacco systems. Examination of these specific sites also failed to provide a clear relationship between the type of crop under production and the level of contamination observed in the subsurface. Under all three of these cropping systems, between 40% and 50% of the multilevel wells had maximum NO_3^- concentrations exceeding the MAC limit during both the winter and summer samplings. In addition, there was no obvious correlation between a particular land-use and the occurrence of bacterial contamination in the groundwater, either from the analysis of the water-well results or the multilevel monitoring wells beneath the fields.

Manure application practice was inferred from the presence of a significant livestock operation and manure storage facility being present on the farm. The data sets for this comparison were significantly larger than was the case for the different cropping practices. For bacteria, all the species tested were combined for this assessment. Despite the difference in the sub-sets of wells contaminated by bacteria in winter and summer, there was a clear trend of higher frequency of contamination with both NO_3^- and bacteria under fields where manure was applied (Table 4). When all the data are considered together, contaminated conditions were encountered twice as frequently in farm fields where manure was routinely applied. Excess nitrogen tended to be applied to

Table 4
Effect of manure application practice on contamination of multilevel wells ($\geq$ 50% of intervals > MAC)

Livestock	Winter		Summer		Both	
	No. of wells	> MAC (%)	No. of wells	> MAC (%)	No. of wells	> MAC (%)
Nitrate contamination						
no	99	18.2	98	20.4	86	16.3
yes	45	31.1	41	31.7	36	30.6
Bacterial contamination						
no	20	40	20	35	11	27.3
yes	37	81.1	34	38.2	16	68.8

many of the fields, on which manure was spread, as mineral N-fertilizer was also applied.

A further analysis of the NO_3^- data from multilevel sites considered the nitrogen management on the farm site. The multilevel sites were classified according to the risk of shallow groundwater contamination, and the relationship between this classification and the NO_3^- contamination in the well. Farms where the amount of nitrogen (manure and mineral fertilizer) applied was less than would be recommended for maximum crop production were classified as 'low-risk' sites. Farms where the nitrogen application followed accepted standards were classified as 'moderate-risk' sites, and farms where the application was in excess of the accepted standards were classified as 'high-risk' sites. These standards follow the risk-assessment strategy set forth by the Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA, 1996) and are based on the amounts and type of commercial fertilizer and manure spread on the farm, consideration of the type of crop and its nitrogen requirements for growth, and whether a winter cover crop had been planted to take up excess nitrogen in the winter months.

Sites classified as medium or high risk with respect to on-farm nitrogen management accounted for 93% of sites with maximum nitrate-N concentrations above 10 mg N l^{-1} in both the winter and summer surveys. This suggests that a relatively simple risk assessment of this type can be applied on an individual farm basis to provide an indication of the potential for elevated levels of nitrate in the shallow groundwater system. It should be emphasized, however, that soil type was not included in the assessment, and consequently it could produce misleading results in some cases, particularly where finer-textured soils occur. These results provide additional evidence illustrating the significance of nonpoint source contamination in affecting groundwater quality in agricultural environments.

3.5. Nitrate and bacteria concentration under cultivated and uncultivated conditions

At the 16 sites where a second multilevel was installed in a woodlot adjacent to the field multilevel well, several qualitative observations can be made (Table 5). During the winter and summer samplings, three and two field MLWs, respectively, had maximum NO_3^- concentrations in excess of 10 mg N l^{-1}. In the woodlots, only one site had NO_3^- concentrations in excess of 10 mg N l^{-1}, and this occurred in the summer. Significant leaching of NO_3^- did not appear to be occurring beneath natural woodlots.

In contrast to the results for NO_3^-, bacterial contamination clearly existed and persisted in the groundwater under both cultivated and uncultivated conditions (Table 5). The incidence of total coliform counts above the MAC limits was approximately the same for the field and woodlot sites, comprising 80% of the sites in the winter and about 50% of the sites in the summer. This trend was also seen for faecal coliform bacteria (Table 5).

These results indicated that the conditions beneath the woodlot areas may provide an environment conducive to attenuation of nitrate. In contrast, bacterial contamination moving beneath the woodlot areas as a result of regional groundwater flow conditions does not seem to be influenced by the variable conditions encountered under the uncultivated areas.

Table 5
Incidence of contamination (≥ 50% of intervals > MAC) at sites with both field and woodlot multilevel wells, where winter or summer data are available

Contaminant	Winter			Summer		
	No. of wells	Woodlot wells (%)	Field wells (%)	No. of wells	Woodlot wells (%)	Field wells (%)
Total coliform	5	80.0 ± 17.9[a]	80.0 ± 17.9	5	60.0 ± 21.9	40.0 ± 21.9
Faecal coliform	5	20.0 ± 17.9	20.0 ± 17.9	5	20.0 ± 17.9	0.0 ± 0.0
E. coli > 0	0			4	0.0 ± 0.0	0.0 ± 0.0
Faecal coliform or *E. coli*	5	20.0 ± 17.9	20.0 ± 17.9	5	20.0 ± 17.9	0.0 ± 0.0
Total coliform or faecal coliform	5	80.0 ± 17.9	80.0 ± 17.9	5	60.0 ± 21.9	40.0 ± 21.9
Bacteria[b]	5	80.0 ± 17.9	80.0 ± 17.9	5	60.0 ± 21.9	40.0 ± 21.9
NO_3^--N > 10.0	16	0.0 ± 0.0	18.8 ± 9.8	15	6.7 ± 6.4	13.3 ± 8.8
Bacteria or NO_3^--N	5	80.0 ± 17.9	80.0 ± 17.9	5	60.0 ± 21.9	80.0 ± 17.9
Bacteria and NO_3^--N	5	0.0 ± 0.0	40.0 ± 21.9	5	0.0 ± 0.0	0.0 ± 0.0
5.0 < = NO_3^--N < = 10.0	16	6.3 ± 6.1	31.3 ± 11.6	15	6.7 ± 6.4	20.0 ± 10.3

[a] ± Standard error.
[b] Bacteria include total coliforms, and faecal coliforms or *E. coli*.

4. Conclusions

Data from the multilevel monitoring wells provided insight into the occurrence of groundwater contamination when considered both independently and in conjunction with the associated water well data. The selection of the multilevel sites was biased significantly toward farms where permeable soils prevailed and as such, a higher frequency of contamination was anticipated than would be expected on average throughout the province.

On average, NO_3^- concentrations tended to decrease exponentially with depth, whereas bacteria were much more uniformly distributed through the entire monitored profile. The NO_3^- concentration profiles may have represented equilibrium conditions, suggesting significant attenuation with depth, which would minimize the impact on deeper groundwater flow systems. These observations remained fairly consistent throughout the year. Bacteria, on the other hand appeared to survive in similar concentrations at all depths below the water table although significantly more temporal variability was observed in the bacteria data with winter concentrations being generally higher than those measured in the summer.

The occurrence of contamination in the multilevel wells correlated fairly well with that observed in the farm wells on the same sites. This provides support for the use of regional water-well survey data as a reasonable indicator of groundwater quality in the rural environment. The drinking-water wells appeared to be more influenced by the distributed source of contamination associated with the adjacent cropped fields than by point sources. The frequency of groundwater contamination in the multilevel wells was not influenced by cropping practice. The scale of this survey was insufficient to identify the influence of cropping practices on groundwater quality. On farms where manure was routinely applied, the multilevel monitoring wells showed a greater occurrence of nitrate and bacteria contamination than on farms without livestock.

With respect to NO_3^-, groundwater quality was better under woodlots than under adjacent farm fields. However, both land uses had similar bacteria contamination. These results suggest that there was some assimilation of NO_3^- from groundwater by the woodlot plants, and this may be an important feature in protection of the regional water quality (Phillips et al., 1993).

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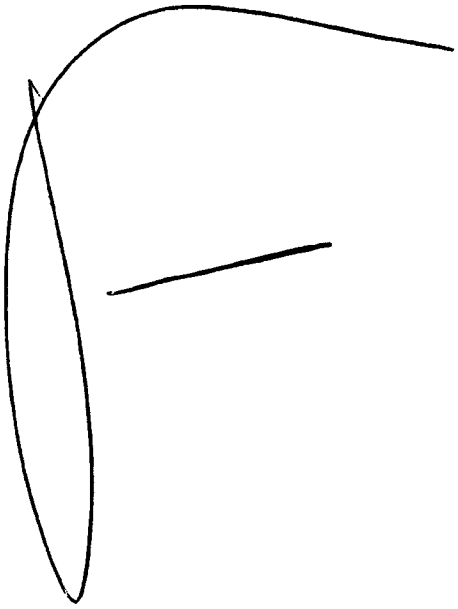

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Effect of Manure Application and Rainfall Timing on the Leaching of Labeled Bacteria through Soil Columns

by

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Summary: Pathogens leaching through the soil profile depends on many factors, including how the manure was applied to the soil and the frequency and timing of rainfall following manure application. The effect of various manure application techniques (broadcasting, incorporation, and pre-broadcast tillage) on bacterial leaching following different rainfall events through intact soil cores was examined by using manure inoculated with E.coli strain RS2, that was genetically tagged with Green Fluorescent Protein (GFP). Steady rainfall consisting of 5 cm rainfall was applied at 4, 8, and 16 days after manure application. The level of RS2 (GFP) in the leachate decreased following each rainfall event and manure application technique did not affect leaching. Inoculum recovery averaged 77% for the first rainfall event for all manure application techniques. Maximum numbers of RS2 (GFP), irrespective of manure application method, occurred in the top 4 inches of the soil profile followed by a rapid decrease in their numbers with increasing depth. Columns subjected to manure incorporation had the highest recovery of RS2 (GFP) from the soil samples. Irrespective of the length of time the manure sits on the surface of soil or the manure application technique, the first rainfall event is crucial in pathogens leaching through the soil. However, manure application practices affect survival of pathogens in soil presumably due to differences in bacterial die-off due to exposure to surface environmental conditions.

Keywords: leaching, manure application, pathogens, rainfall event, recovery, soil columns

Introduction

Land applied manure may constitute a source of pathogens to the groundwater, surface water and soil. The application of manure to agricultural lands may pose an environmental risk even when the application procedures are within the current guidelines. Microbial contamination of water and soil due to land application of liquid manure and other liquid wastes is difficult to treat, because once applied, manure becomes a potential non-point source of pollution, less susceptible to correction than a point source. Problems have been demonstrated in Ontario by Fleming *et al.* (1990) and Palmateer *et al.* (1989) where application of liquid manure to agricultural fields resulted in rapid movement of a tracer bacterium, nalidixic acid-resistant *Escherichia Coli (E.Coli)*, through the soil and under drain systems leading to contamination of surface receiving waters. Many factors have been observed to affect the survival and movement of microorganisms in soil. These are mainly a result of complex interaction between the soil, water, microorganisms and the surrounding environment (Crane *et al.*, 1983; Gerba and Bitton, 1984). The main factors influencing bacterial survival in the soil environment are moisture content, temperature, pH, sunlight, organic matter and competition of other micro flora (Gerba *et al.*, 1975; Crane *et al.*, 1984). In a study by Bell (1976), Fecal coliform populations in soil were completely destroyed by 10 hours of bright sunlight. Furthermore, no decreases in their population were observed in the absence of bright sunlight. Van Donsel *et al.* (1967) found similar results, showing 90% reductions in fecal coliform densities in soil in 3.3 days and fecal streptococcus densities in 2.7 days during summer. Variations in these die-off rates and a better understanding of the soil-bacteria-water relationship may help provide some explanation to the variability seen in the field experiments when dealing with bacteria. Van Donsel *et al.* (1967) offered yet another explanation for this by discussing the possibility of the populations of bacteria in applied manure to increase shortly after application. This observation was made for non-fecal bacteria after a short rainstorm and could probably be attributed to the nutrient levels in the soil matrix and the added water which became available from the rain. When rainfall percolates into the soil, the enteric bacteria present in the manure become suspended and move with water to surface and subsurface drainage. The first runoff-producing rainfall event produces the maximum bacterial loss and the highest concentrations in runoff water. There is a direct relation between the peak rate of rainfall, the ensuing runoff and the bacterial concentrations in that runoff (Crane *et al.*, 1983). Patni *et al.* (1985) stressed the importance of rainfall amount and the intensity of rainfall in the movement of bacteria in runoff. Crane *et al.* (1983) showed that the residence time of bacteria on the soil surface influenced bacteria concentrations in runoff. According to them, the type of waste (either liquid or solid) and waste volume is crucial in bacterial movement. In a study conducted by Abu Ashour *et al.* (1998) on a silt loam soil with and without macropores, bacterial concentrations were found to be highest near the top of the columns at a low moisture content of 10%. At moisture content above 40% before irrigation, bacterial leaching was greatly facilitated. Smith *et al.* (1985) found that 96% of the bacteria applied at the top of the 28-cm columns could be recovered with increasing rates of water application. Van Elsas *et al.* (1991) studied the influence of soil properties on the movement of a genetically-marked *Pseudomonas fluorescens* bacterial strain through 50-cm long soil columns of loamy sand. Bacterial cells were applied at the top of soil columns with moisture content of 5.3% and 13% respectively. Despite, a significant difference in bacterial concentrations between the soil moisture treatments, there was little difference in bacterial concentrations collected from the leachate of the two different

treatments. The degree to which agricultural drainage water becomes contaminated depends on varied factors (including cropping system, rainfall events, soil conditions, and manure management techniques) depending on the field situation. Tillage is also important in determining water quality, since tillage practices enhance the soil's ability to retain water, thus preventing runoff and slowing drainage to the subsurface. McCaskey *et al.* (1971) investigated the quality of runoff from dairy application sites where manure was applied frequently at annual application rates averaging between 20 and 300 metric tons of dry matter per year in liquid, semi-liquid, or solid form. Analyses of the runoff from these areas showed that maximum annual removal of applied total coliforms, fecal coliforms and fecal streptococci was .06%, .07% and .08% respectively. Bacterial losses were the highest for the solids spreading technique and lowest for the liquid manure application method (0.0005 to 0.0012%). These results indicate that manure can indeed be spread with little potential for bacterial pollution under optimal conditions. No clear advantage was shown between manure application methods and rates with respect to bacterial densities of total coliforms, fecal coliforms and fecal streptococci in runoff. Subsurface injection or plowing under wastes should virtually eliminate bacterial losses in runoff as long as the method employed effectively places manure below the soil surface. Subsurface injection could, however lead to increased bacterial losses via groundwater movement due to less contact of bacteria with the more reactive soil surface zone. Application should also be avoided for atleast 24 hours prior to a runoff event (Crane *et al.*, 1980). In many studies, fecal coliform indicator organisms have been detected in receiving and drainage water even when no manure had been applied to land. Cook (1998) conducting a manure management study found that runoff from the control plots (no manure applied) contained higher concentrations of fecal coliform, fecal streptococci, and *E. coli* not significantly different from the manure treated plots. In another study of six watersheds including five of those that had received manure and one that had not, the average bacterial numbers for all the six watersheds exceeded the bathing water quality limits (Robbins *et al.*, 1972).

Soil column studies done in laboratory conditions have been instrumental in providing valuable information on transportability and survivability characteristics of microorganisms through soils. Trevors *et al.* (1990) argued that repacked soil columns provide reproducibility and homogeneity, making it possible to study different factors affecting bacterial movement in soils. At the same time, they conceded that undisturbed soil cores provide better predictive values for field situations than repacked ones. Bitton *et al.* (1979) compared the findings from four field studies that monitored viruses and their removal by soil with results of laboratory soil column experiments designed to simulate the field conditions. In each instance, there was a close correlation between field and laboratory results. Tracer bacteria have been used in such column studies to monitor the movement of fecal bacteria from various sources, since they provide additional characteristics for selectivity and identification. A commonly used method for labeling bacteria is to induce antibiotic resistance in the organism of interest. Presence of the antibiotic restricts the growth of nonresistant organisms while at the same time allowing the tracer to flourish. Smith *et al.* (1985) used streptomycin resistant *E. coli* to trace the movement of bacteria through intact soil columns. At an applied concentration of 10^7 cells/ml, the leachate concentration was constant. Soil structure and water flow velocity were major factors influencing movement of the tracer bacteria. Dean and Foran (1992) introduced a nalidixic acid resistant *E. coli* and rifampicin resistant *S. faecalis* in liquid manure to trace the movement of manure bacteria through soil to subsurface drainage. They had good recoverability because bacterial strains resistant to nalidixic acid and rifampicin are not common in the natural environment or

manure. Joy *et al.* (1998) applied liquid manure spiked with a nalidixic acid resistant strain of *E. coli* and spread on a subsurface drained cornfield. The volume of manure application and concentration of the tracer bacteria in manure did not correlate with subsurface effluent concentration. Concentrations were greatest in effluent when rainfall occurred shortly after application. With little rainfall, very little tracer moved to tile drainage. Since, coliform bacteria occur naturally in the environment there is sufficient reason to believe that tracer bacteria could serve as an effective tool to study the fate and transport of enteric bacteria in soil water profile

Objective: The overall objective of this study is to identify the effect of manure application technique and rainfall timing on bacterial leaching through the soil-water profile in order to minimize bacterial pollution and preserve water quality. Specific objectives are:

1. To assess the survivability and transportability of an inoculated tracer (genetically-marked) bacteria in the effluent resulting from leaching swine manure and water through intact soil cores.
2. Compare and quantify bacterial concentrations (specifically: genetically-marked and other enterics) in leachate and soil from different manure application methods and the timing of rainfall application.

Experimental location

The study was conducted in the Manure Management Laboratory of the Agriculture and Biosystems Engineering department located at the National Swine Research and Information Center on the Iowa Sate University Campus during the fall of 2000. Prior to that, the inoculum's preparation was done at the Soil Microbiology Laboratory of the Agronomy department at Iowa Sate University during summer 2000. In the laboratory, water was applied/ponded to simulate rainfall conditions to evaluate the survivability and transportability of tracer bacteria by examining the effects of manure application technique and rainfall timing.

Materials and Methods

Inoculum preparation

The effect of manure application procedures, and the timing and frequency of rainfall on bacterial leaching was investigated using intact soil cores and a genetically-tagged tracer/indicator bacterium that was originally isolated from manure. All cores were inoculated with a kanamycin and rifampicin resistant derivative of *E.coli* strain RS-2GFP. Enteric bacteria was isolated from manure samples using standard bacteriological procedures, which led to the tentative identification of one isolate as either a *Salmonella* species or an *E.coli* by gas chromatographic analysis of whole-cell fatty acids using the Microbial Identification System Gas Chromatographic and Computer system (MIDI, Newark, DE). Based on the presence of glucuronidase enzyme activity, using the MUG media, it was very likely that this isolate (RS-1) that was chosen for further study and characterization was an *E.coli*. A series of additional biochemical tests indicated that the isolate was not a *Salmonella* species, but the exact species designation was not determined from these tests. Putative strain identification is being done by comparing the nucleic acid sequence of the 16S rDNA gene with those present in the Ribosomal Database Project at Michigan State University. Kanamycin and rifampicin resistance was induced by plating dense suspensions of the parent culture on Luria Bertani (Difco Laboratories) agar supplemented with 100-μg/ml kanamycin and 10-μg/ml rifampicin respectively. Resistance was sufficiently stable, as determined by the comparison of growth on media with and without

antibiotic. A plasmid (pPROBE *PnptII::gfp*) that harbors a constitutively expressing green fluorescent protein (*gfp*) gene whose expression is driven by the neomycin phosphotransferase II promoter was moved into a rifampicin-resistant derivative of RS-1 to derive strain RS-2GFP. It was determined that acquisition of the plasmid and constitutive expression of green fluorescent protein was not detrimental to bacterial growth or survival in soil and manure, which was an essential element for its selectivity as a tracer. The plasmid also confers resistance to the antibiotic kanamycin and all enumeration of RS-2GFP was achieved by plating onto complex media amended with kanamycin. RS-2GFP derived colonies were identified by determining whether the colonies were visibly green or green during exposure to ultraviolet light. Green fluorescent protein excitation is optimal when exposed to ultraviolet light (365 nm) or at 488 nm; emission optimum is 510 nm and hence its green fluorescence. This approach allows for unequivocal identification of the tracer and differentiation from indigenous organisms that also grow on the semi-selective medium.

Soil cores

Thirty soil columns were collected from the Kelly Farm located near the Iowa State University Agronomy and Agricultural Engineering Research Center near Ames, IA. The soil used was Clarion loam under annual corn and soybean rotation. Soil columns were extracted in late summer 2000, before the soybean harvest, using a Giddings probe and 20-cm bit adapter. The 30-cm long columns were extracted in 38-cm long sections of sterilized galvanized tubing that had been sharpened on the down facing edge. The soil columns were then transported to the Manure Management Laboratory at National Swine Research and Information Center on Iowa State University campus. Autoclaved screens were installed on the bottom of each column in order to prevent soil loss. The columns were then arranged in a random block design in a leachate collection apparatus comprised of 25-cm autoclaved funnels and a guide table that held the columns upright. Two such guide tables were used accommodating 18 and 12 columns respectively. The columns were saturated to field capacity by placing them in buckets of water allowing saturation to take place from bottom to top for 48 hours and then allowed to drain for 3 days. Fresh manure was procured from the Swine Nutrition Management and Research Center of the Iowa State University, near Ames, IA. 152 ml of manure was applied to simulate 5000 gal/acre to each of the columns (approximately 0.5 cm depth). Prior to the experiments on the soil cores, RS-2GFP was grown by streaking stock culture cells on Luria-Bertani agar supplemented with kanamycin and rifampicin. A loopful of cells grown on the agar was transferred to a 500 ml Erlenmeyer flask containing sterile water. The culture was incubated at room temperature for 24 hours with gyratory shaking at 200 rpm until the culture became highly turbid. The concentration of this suspension was determined to be $5.6*10^9$ colony forming units/ml (cfu/ml) by the spread-plating technique. 152 ml of manure spiked with 15 ml of the tracer RS-2GFP was applied to the soil columns to simulate three different manure application methods: broadcasting, tillage and incorporation. The concentration of the inoculum determined after mixing with manure was $4.8*10^8$ cfu/ml, confirming that manure was not detrimental to its growth and green fluorescence expression.

Experimental design

A total of 30 columns; 27 experimental and 3 controls were used. The study was divided into three levels, each having three treatments and three replications respectively (9 columns for each

level of rainfall frequency) resulting in 27 columns in a randomized block design. The following three treatments were used for application of manure to soil columns:

- Broadcasting (spreading of manure on top of soil without disturbing it)
- Tillage (tilling the soil and then broadcasting manure)
- Incorporation (broadcasting manure followed by tillage)

Manure was applied to all the columns only once i.e. at the beginning of the experiment. Four days after manure application, the first of the four rainfall events took place on the columns. Water was irrigated to a ponding depth of 5.2 cm (volume= 1650 ml), which is a typical rainfall amount for the month of April in Iowa. The effect of time before the first rainfall and rainfall frequency was examined by applying rainfall at 4, 8, and 16 days after manure application. These time intervals were chosen to represent actual situations that can be encountered in the field (Average no of days in the month of April that receive more than one inch of precipitation =10.7). Table 1 provides a summary of the timing and frequency of rainfall events to the different columns. These rainfall events were simulated in such a manner so that nine of the columns (three from each manure application technique treatment) got rained on once (16 days after manure application); nine were rained on twice (8 and 16 days after manure application) and nine were rained on three times (4, 8, and 16 days). When the first rainfall event took place four days after manure application, 3 columns from each of the manure application methods got rained on along with the controls. On the next rainfall event (after 8 days), the ones that had previously received rain during the first rainfall event (including the controls) got rained on again. Additionally, 3 new columns from each of the three manure application treatments got rained on the first time while manure had been sitting on their surface for 8 days. The procedure was repeated on the 16th day to simulate the third rainfall event.

Table 1: Experimental design of the study

Method of Manure application (TRT#1)	Frequency of rainfall (TRT#2)			Total sets
	Once (On16th day)	Twice (8th & 16th day)	Thrice (4th, 8th &16th days)	
Broadcasting (bd)	3	3	3	9
Incorporation (in)	3	3	3	9
Tillage (ti)	3	3	3	9
Total	9	9	9	27

Data Collection Procedure:

Leachate was collected at the bottom of each soil column in sterile plastic sample bottles and aliquots were directly dilution plated (without any serial dilution) on Luria-Bertani agar supplemented with kanamycin and rifampicin media to enumerate RS-2GFP. These plates were

incubated at 37°C for 24 hours. The bacterium was counted manually and the values obtained for replicate plates were averaged. RS-2GFP colony counts were confirmed by placing the plates under UV light (green flouroscent). At the end of the experiment, the soil in the columns was analyzed to determine both the viability of the RS-2GFP and it's distribution throughout the column. Soil was sampled at four different depths viz; the top inch, 0-4, 4-8, and 8-12 inches respectively. Since it was not possible to assess the moisture content before or after each water application without disturbing the soil columns and introducing macropores in them, moisture content at different depths for all the columns was determined only at the end of the experiment. The leachate samples were also analyzed within 24 hours and stored at 4°C until they were analyzed. Analysis for fecal coliforms (FC), total coliforms (TC) and fecal streptococcus (FS) was done according to the membrane filtration technique described in Standard Methods for the Examination of Water and Wastewater, 18th edition by plating on m-FC agar for fecal coliforms and total coliforms, and m-coli blue for fecal streptococcus. Densities for the tracer bacteria and the enterics were recorded in terms of colony forming units/ml and were later converted to the log values of the total numbers of colony forming units recovered from the leachate or soil samples.

Results and Discussions

Leachate analysis

Leachate samples were collected on the day of each rainfall event. The average volume of leachate collected on the first rainfall event was less than that recovered on the next event in the case of incorporation and tillage techniques respectively. However, the opposite trend was observed in case of the broadcasting technique where a higher volume leached on the first event followed by a lower volume on the second event and finally a slightly increased volume on the last rainfall event. Fig 1 depicts the average volume of leachate collected on the different rainfall events. The leached samples were analyzed for recovery of the RS-2GFP cells. Percent recovery of the applied inoculum was maximum on the first rainfall event followed by a decreasing trend over the next two events (Fig 2). Average percent recoveries ranged from 72.7% for the broadcasting technique to 69.3% for the incorporation technique on the first rainfall event. On the subsequent events, the recoveries were in the range of 45.7% – 37.8%. The maximum recovery on the first event occurred in case of the broadcasting technique, probably due to swift and direct leaching of the pathogens since they had lesser chance of traveling or mixing through the soil profile. RS-2GFP population numbers decreased by more than a thousand fold (applied inoculum concentration @ 10^8 cfu/ml to 10^4 cfu/ml) after manure had been sitting for four days before the first rainfall event occurred. This die-off was the maximum in case of the columns that were subjected to the tillage technique (10^4 cfu/ml on first rainfall event to 10^1 cfu/ml on the third rainfall event). Maximum cfu/ml of the leachate were recovered from the columns that had been subjected to the incorporation technique ($8.6*10^4$ Cfu/ml) on the first rainfall event but differences in the volumes of leachate recovered from the columns somehow compensated for this, so that treatment was not an important variable influencing leaching of the RS-2GFP. Log concentrations of the tracer in leachate for different manure application techniques following different rainfall events are given in figure 3. Generally, maximum concentrations of the tracer were observed following the first rainfall event, irrespective of the manure application method, proving that the first rainfall event is the most crucial in vertical leaching of pathogens. Over time there was a decrease in the total number of RS-2GFP detected in the leachate, with most of

the decrease occurring before the third rainfall event. An ANOVA test was performed to assess the influence of the different variables influencing leaching through the soil columns, the results of which are presented in table 2.

Table 2: ANOVA Table: Effect of variables influencing RS-2GFP numbers in leachate

Effect	Degrees of Freedom	P > F
Manure application	2	.5093
Rain	2	.661
Treatment*rain	4	.0172
Day	2	.0021
Day*treatment	4	.6935

Table 2 reveals that the day of rainfall application was a major factor influencing the bacterial concentrations since rainfall event 1 (rainfall application on 4th day after manure application) was significantly different from event 2 (rainfall application on 8th day) and event 3 (rainfall application on 16th day). The second and third event resulted in lower bacterial densities for all treatments as compared to event 1 (fig 3). The interaction of rain and treatment was also an important influencing variable, since the number of times (frequency of rainfall) a column received rain was an important factor governing the numbers of the tracer present in the soil-water profile. The effect of the first flush of rainfall was consistent over all the treatments on the different events. Columns subjected to incorporation and tillage technique behaved in exactly the same manner with respect to their survival patterns following the first flush of rainfall on the 8th and the 16th day of rainfall, while the broadcasting technique showed an increase in the numbers of populations that were recovered on the 16th day of rainfall as compared to the 8th day (fig 4). Leachate samples were also analyzed to detect enteric organisms such as fecal coliforms (FC), total coliforms (TC) and fecal streptococcus (FS). Population densities of these enterics detected over the different rainfall events for the different manure application techniques is presented in figures 5 through 7. An interesting observation is that leachates from the controls (with no manure application) had higher numbers of the enterics than the manure applied columns. One probable explanation for this can be that the enterics came from the soil columns or the soil, while application of manure had a detrimental effect on transport/survival of certain species or it decreases the survival of indigenous enterics that are leached following a particular rainfall event.

Soil analysis

Upon completion of the simulated rainfall events, the soil cores were sampled to assess the survival of RS-2GFP in the soil at various depths. Maximum colony forming units per gram (cfu/gm) of oven-dried soil were present in columns that were subjected to broadcasting technique. Manure application technique was an important factor (P value of .0007 at 5% level of significance) influencing tracer survivability in the soil since the incorporation method with a log mean of 8.48 was significantly different from both the broadcasting and tillage techniques (means 6.79 and 6.62 respectively). The depth at which the tracer survived was also of importance with a P value < .0001 at 5% level of significance. Most of the viable RS-2GFP in

the soil was found in the top 1 inch and the top four inches of the soil column and in general the population numbers decreased with increasing distance from the soil surface. The mean concentrations of the tracer found in the top inch of the soil profile and in the top 4 inches were not significantly different (means 9.014 and 8.904 respectively), while the lower half of the soil column had significantly lower and different survival rates of the tracer (6.65 and 4.7 respectively). The data also suggests that manure application procedure significantly affects RS-2GFP survival in soil; the greatest survival followed manure incorporation while the poorest survival occurred following manure broadcasting (fig 8). The number of times a column was rained on and the kind of manure application technique it received had a mixed interaction with a P value of 0.076. This mixed effect is evident in fig 9, where the concentrations of the tracer recovered from the soil increased as the frequency of rainfall applications increased. This indicates that increased moisture in the soil enhances survivability. A correlation analysis was performed to assess the effect of initial moisture content on the numbers of RS-2GFP recovered on each event. A correlation coefficient of 0.44 indicates that there was a somewhat limited causal-effect relation between the two variables. The overall composite recovery of the tracer cells from both the leachate and the soil was the highest in case of the incorporation technique (fig 10). Though the overall recovery was not greater than 100%, it means that conducive conditions were present in the soil-water profile within the columns such that the tracer was undergoing the normal cycle of expansion and die-off. This means that incorporation leads to greater survivability of the tracer by facilitating better mixing of the soil and allowing easier transport of the tracer. Broadcasting on the other hand leads to dieoff of the organisms that remain on the surface, since they are exposed to harsh environmental conditions.

Conclusions

Intact soil columns were used to model the movement of a genetically-tagged bacterium to subsurface drainage following broadcasting, incorporation and tillage techniques and application of rainfall timing. In all cases, the first flush of rainfall produced the highest bacterial concentrations regardless of manure application techniques. Although, manure application technique was not a significant factor influencing bacterial concentrations in the leachate, clear differences were identified between treatments during the first, second and third irrigation events following manure application. Analyses of the soil samples revealed that maximum numbers of the RS-2GFP were present in the top four inches of the soil profile and their survivability decreased in the lower half of the columns. Manure application method contributed significantly to survival of RS-2GFP within the soil profile so that maximum numbers of the RS-2GFP were observed in soil columns that had been subjected to the incorporation technique, while the poorest survival occurred in case of the broadcasting technique. Additionally, an interaction between the manure application technique and frequency of rainfall was observed, suggesting that increased frequency of rainfall application led to greater recovery of cells from the soil. Overall recovery of the RS-2GFP from both the leachate and soil was maximum for the incorporation technique and lowest for broadcasting.

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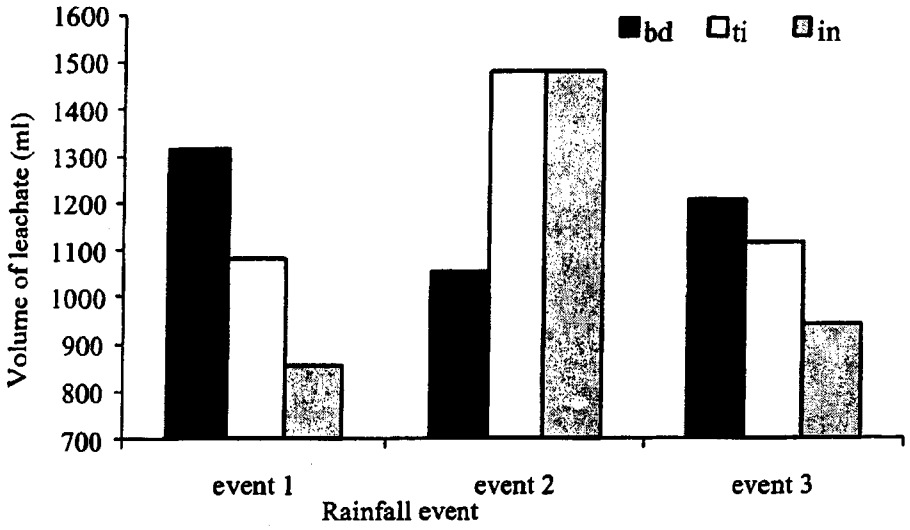

Fig 1.Volume of leachate recovered on different rainfall events

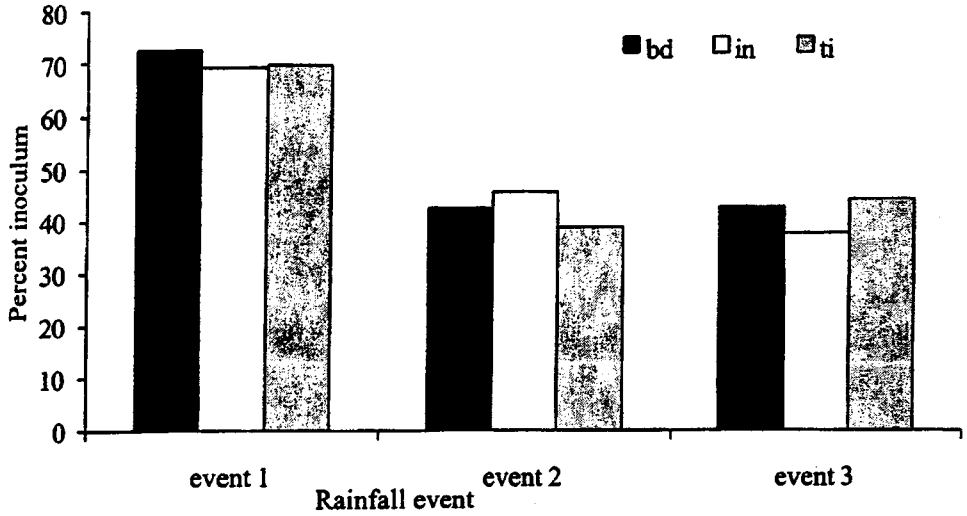

Fig 2. Percent recovery of tracer in leachate

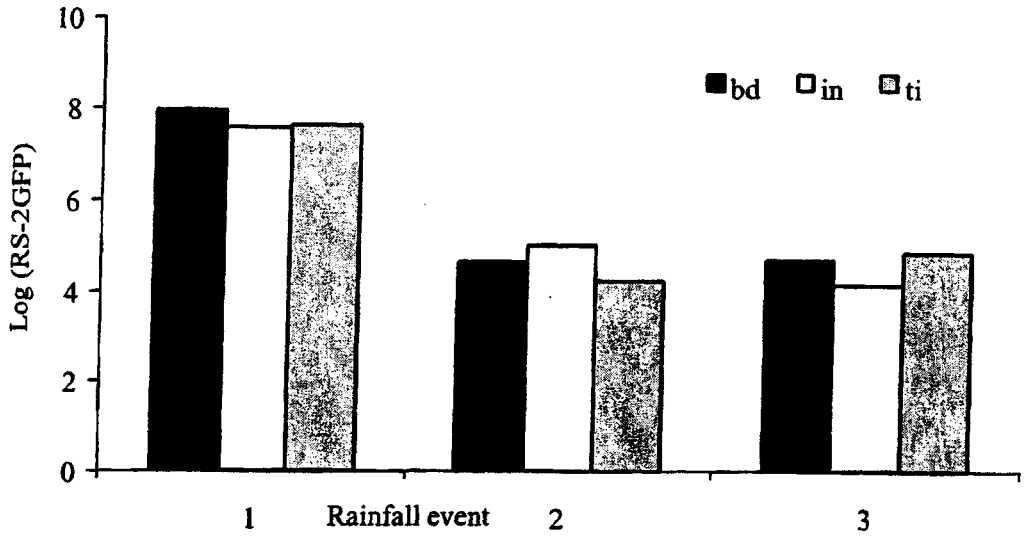

Fig 3. Concentrations of tracer in leachate at different rainfall events

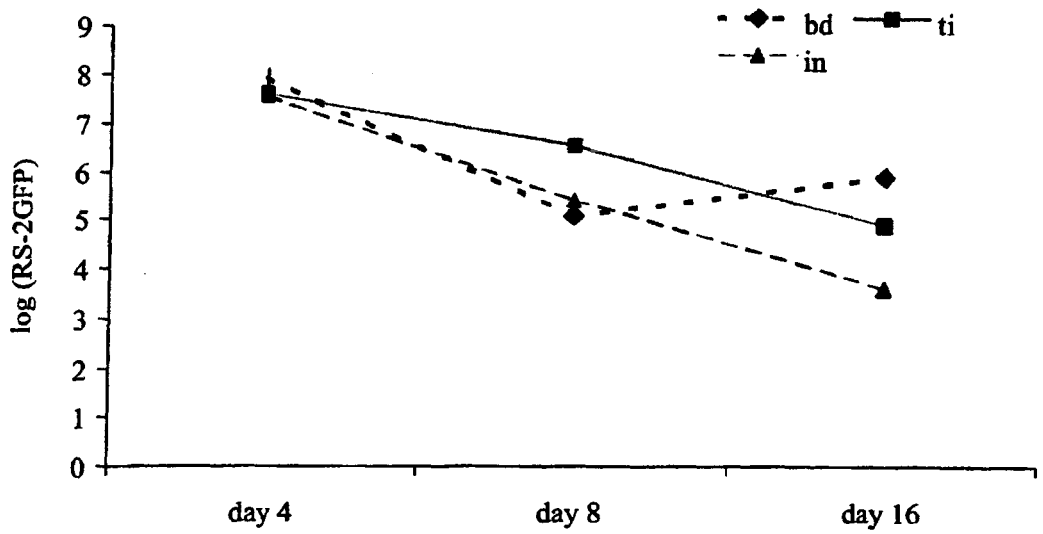

Fig 4. Effect of first rainfall event only on concentrations of tracer

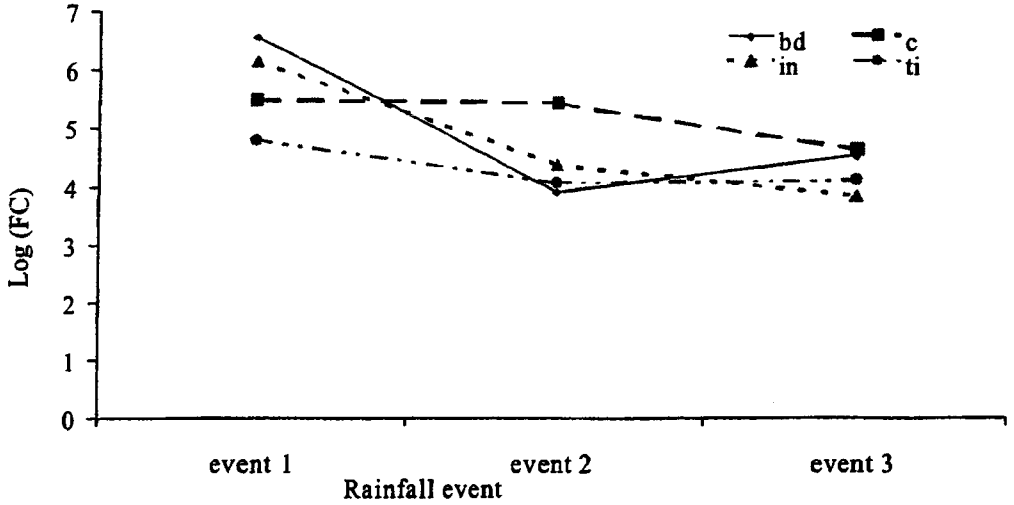

Fig 5. Concentrations of FC in leachate at different events

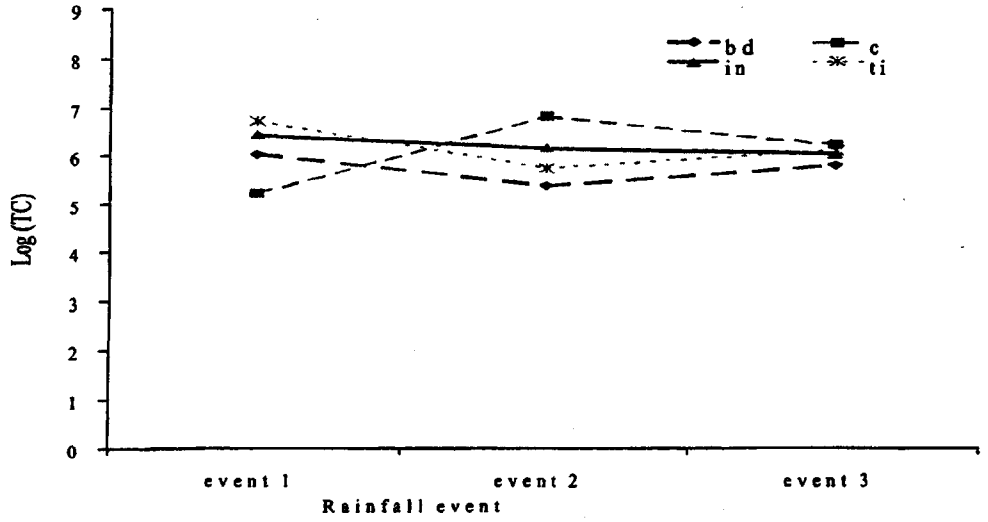

Fig 6. Concentrations of TC in leachate at different events

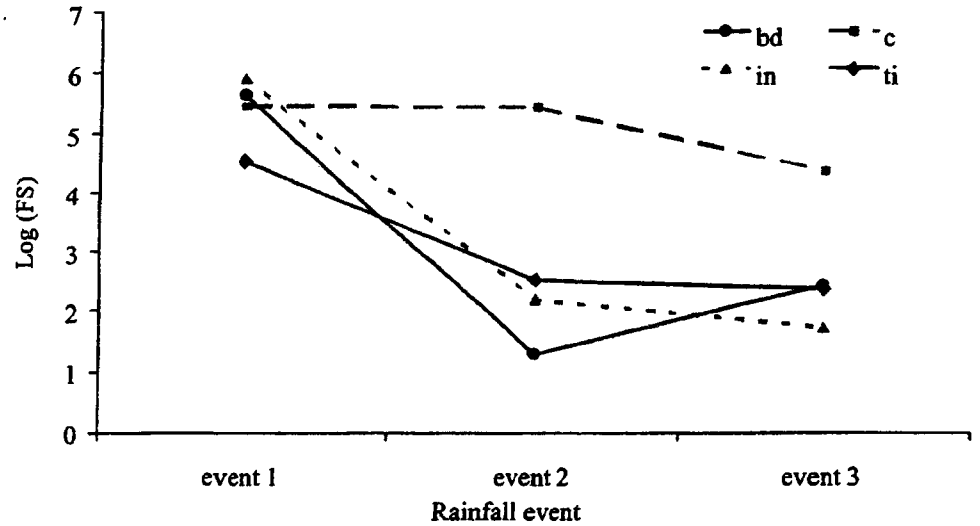

Fig 7. Concentrations of FS in leachate at different events

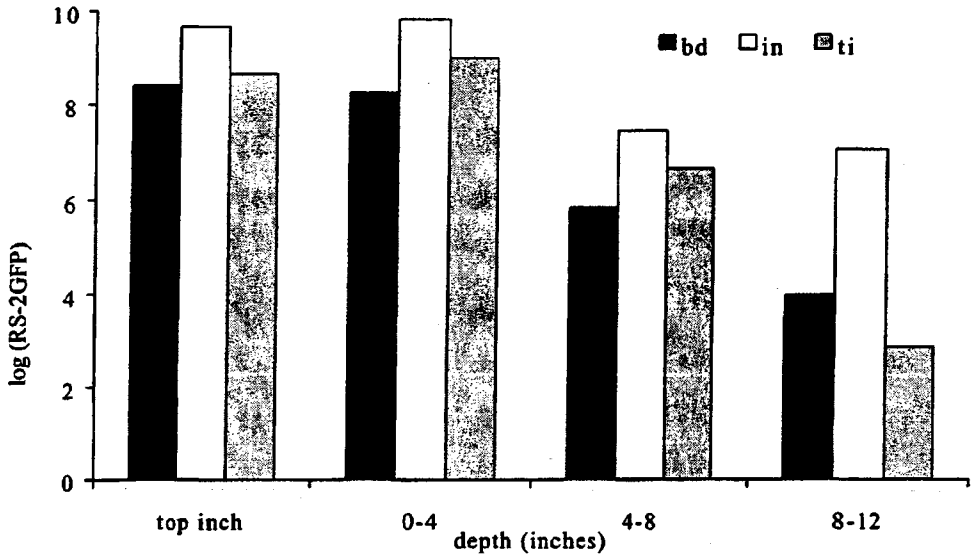

Fig 8. Concentrations of tracer at different depths in the soil profile

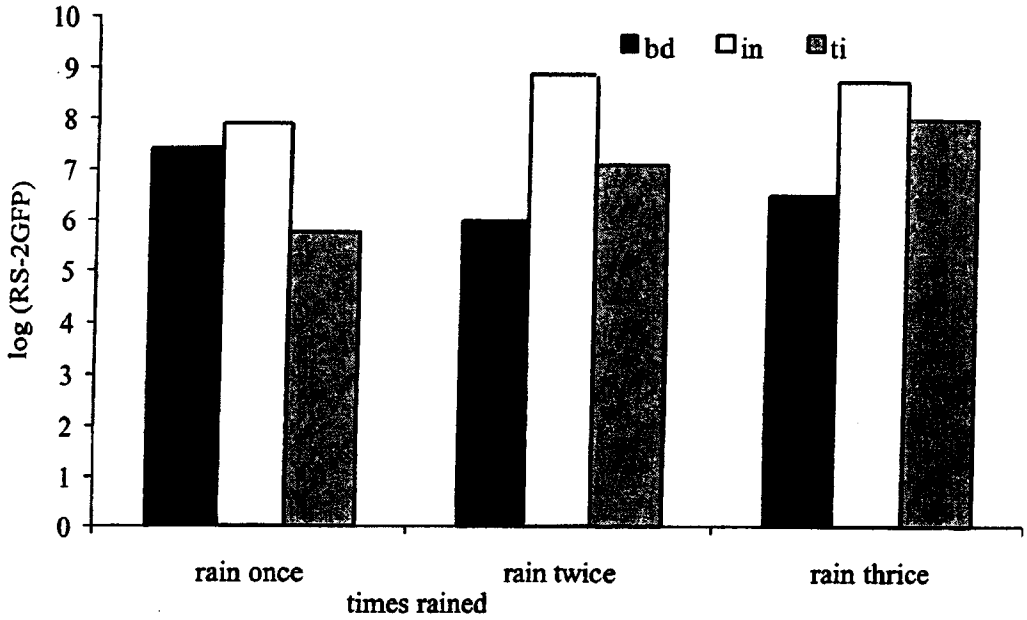

Fig 9. Effect of rainfall application frequency on concentrations of the tracer in soil

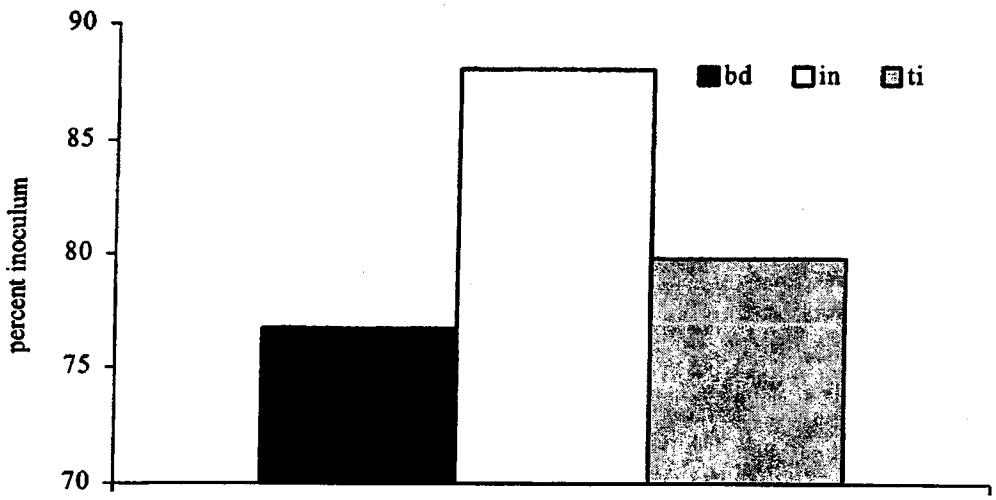

Fig 10. Total composite recovery of RS-2GFP cells (soil+leachate)

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LINKING WATER QUALITY WITH AGRICULTURAL INTENSIFICATION IN A RURAL WATERSHED

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Abstract. Agricultural intensification was linked to streamwater pollution in a case study watershed using GIS and nutrient budgeting techniques. The results showed that surplus nitrogen applications from fertilizers and manure averaged 120 kg ha^{-1} yr^{-1}. In some parts of the watershed surplus applications exceeded 300 kg ha^{-1} yr^{-1}. A consistent increase in pig and chicken numbers (59 and 165% increase between 1986 and 1996) is considered the main reason for the surplus. Water quality was impacted in two ways: nitrate contaminated groundwater contributed to high nitrates in a major tributary during the summer, while in the wet winter season ammonia, phosphate and coliform levels were high throughout the drainage system. Significant negative relationships were found between surplus nitrogen applications and dissolved oxygen while ammonia and nitrate concentrations during the wet season were positively correlated to surplus applications. Soil texture and drainage type were also significantly correlated with the water quality indicators suggesting that it is possible to use the budget/GIS linked techniques for pollution risk assessment from agricultural non-point sources.

Keywords: agricultural intensification, agricultural pollution, animal waste, land-water interactions, nitrate, nitrogen surplus, nonpoint source pollution, water pollution, watershed management

1. Background

Agriculture is rapidly emerging as the greatest contributor of non-point source (NPS) pollution to streamwaters in North America (EPA, 1996; Kellogg *et al.*, 1994) and in many other parts of the world (Braden and Lovejoy, 1990; Isermann, 1990) where intensive agriculture occurs. While we have been relatively successful in controlling point sources of pollution, the NPS problem is far more challenging because it is very difficult to isolate the contribution from individually dispersed sources in a scientifically and legally defendable manner. One challenge is how to address cumulative effect over different temporal and spatial scales. This challenge becomes most evident in areas where both animal and crop production have intensified in the same area. Fertilizers are applied to crops in addition to manure because manure does not promote plant growth as effectively as fertilizers during critical periods of the growing season. Also, in areas with intensive animal feed units the disposal of animal waste becomes even more of an issue during periods when crops do not require extra nutrients. Animal waste is primarily applied to soils,

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which generally have a good capacity to absorb, store and slowly release nutrients. However, with continuous application of excess nutrients, the storage capacity of the soil is quickly exceeded and the nutrients find their way to streams and lakes. This leads to ammonia toxicity for fish, elevated nitrates in groundwater, eutrophication affecting the oxygen supply for aquatic biota, and microbial problems that impact water use and consumption (Cooper, 1993). Much has been written about groundwater (Spalding and Exner, 1993) and streamwater (USEPA, 1996) contamination from agriculture by excess nutrients but recent concerns about the human impact on the nitrogen cycle (Vitousek *et al.*, 1997), and eutrophication (Abrams and Jarrel, 1995; Daniel *et al.*, 1998) suggest that the problem is increasing. The use of nutrient budgets (Barry *et al.*, 1993) and GIS (Corwin and Wagenet, 1996) have been advocated as new tools in assessing the problem, and nutrient detention and removal by buffer zone vegetation is a popular mitigation practice (Lichtenberg and Shapiro, 1997; Jordan *et al.*, 1993; Hill, 1996). However in the long term, source control is likely the most cost-effective management option.

The goal of the article is to show how Geographic Information Systems (GIS) can be used in combination with a nutrient mass balance calculation to predict water quality conditions in a watershed context. The specific aims are to:

1. Document changes in agricultural land use and intensity between 1964–1995 using GIS in combination with digital aerial coverage, census data and farm surveys;
2. Illustrate annual surplus applications of nutrients using a mass balance calculation;
3. Document water quality conditions over an annual cycle; and,
4. Show relationships between surplus nitrogen applications, soil and site conditions, and water quality in the stream.

2. Study Area

The Sumas River watershed (Figure 1a) located in the Fraser River Lowland in Washington State and British Columbia, contains some of the best and most productive agricultural land in Canada. The main stem of the Sumas River originates in the coastal mountains in Whatcom County in the United States, and joins the Fraser River east of Abbotsford in British Columbia. The lower portion is dominated by a flat lake bed that was drained in the early 1930's and is known as the Sumas Prairie. A major aquifer (Abbotsford Aquifer) is located to the west and water from that source enters the Sumas River via Marshall creek (Figure 1b). The rainfall and runoff regime in the watershed is one of dry summers and a distinct wet period from November to March when low pressure systems from the Gulf of Alaska dominate the Pacific coastal climate.

Because of the favorable climate, excellent infrastructure, and proximity to

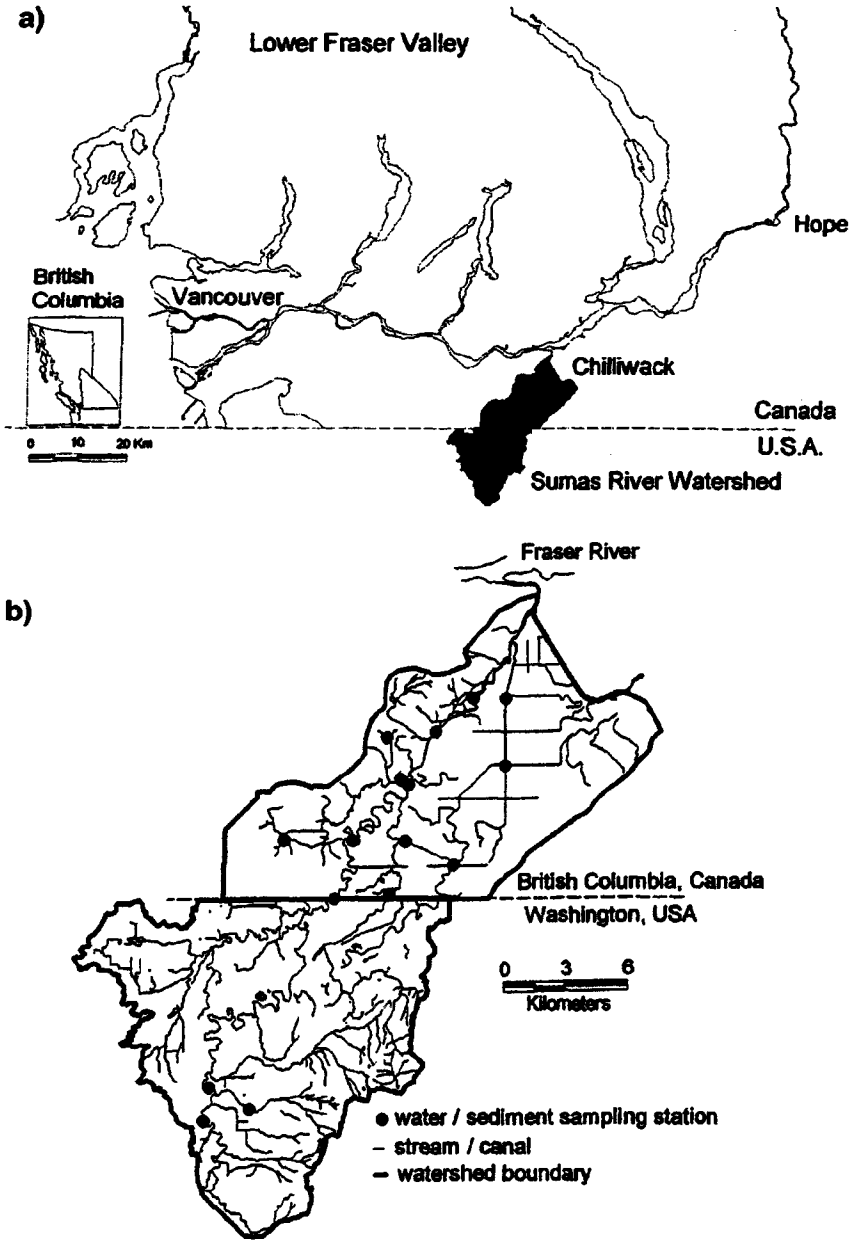

Figure 1. a) Location of Sumas River watershed. b) Stream network and water quality sampling stations.

the urban center of Vancouver the area has undergone significant agricultural intensification. There is considerable concern in this part of North America about sustaining salmonid populations and maintaining good drinking water supplies. Eutrophication of streams and nitrate contamination of groundwater are becoming major environmental issues, leading resource managers and regulatory agencies to continually seek assessment techniques that can help guide effective management practices. Given the differences in data availability and evaluation methods between the U.S. and Canadian portion of the watershed, the analysis of the Canadian portion of the watershed is presented in this study. A comparison between the US and Canadian portion of the watershed is provided elsewhere (Berka, 1996).

3. Methodology

The watershed was delineated using 1:20 000 scale digital terrain data (B.C. province TRIM database). A comprehensive georeferenced GIS database was developed that included: topography, digitized soil survey information, and land use. Land use was compiled using the 1995 digital orthophoto and from an analysis of historic aerial photos from 1954, 1963, 1979, and 1988 (scale 1:10 000). The land use and location of all farms in the Sumas Prairie were identified and digitized for each of the 5 different time periods. In addition, data from a waste management survey (WMS) of 130 farms (IRC, 1994) was used to arrive at animal stocking densities. The agricultural census data for 1986–1996 was used to document agricultural intensification. All information was georeferenced and incorporated into the GIS database. Database queries and GIS overlay analyses were used to examine changes in land use and in the intensity of human agricultural activity within the watershed.

Nutrient mass balance calculations were carried out using the model developed by Brisbin (1995). All sources (i.e., airborne, fertilizers, manure, biological conversion) and sinks (i.e., manure exports, volatilization, uptake by crops and organisms, denitrification, and management losses) were determined for each farm and each contributing area, or subcatchment. The contributing areas were delineated based on topography and location of water sampling stations. The arable land area within each contributing area was determined from the GIS land use analysis. In the mass balance model, manure nitrogen production was calculated by multiplying the nitrogen production rate in kg yr^{-1} for each type of animal by the number of animals on each farm. Based on the farm survey data it was estimated that 30% of broiler manure was exported from the watershed. Nitrogen losses to the air, land and water were determined using manure management conversion factors developed by Brisbin (1995) for each animal type, and different nutrient uptake rates were used for each crop type. Finally, a 9 kg ha^{-1} yr^{-1} rate was used for background atmospheric deposition and a 30% return rate was used for volatilized nitrogen due to management losses. A 10% loss was assumed to account for de-

TABLE I

Changes in land use and farm numbers between 1954 and 1995

Year	No. of farms	Area (ha)		
		Agricultural	Forested	Urban
1954	224	9687	4685	154
1963	233	9426	4645	416
1979	248	9647	4533	569
1988	271	9646	4452	757
1994	283	9751	4463	1542

nitrification of applied manure (Brisbin, 1995). Subtracting total losses from total sources provided surplus or deficit application rates in kg ha^{-1} yr^{-1}.

Water quality was determined at 16 stations shown in Figure 1b over the course of one year on eight different sampling dates. Dissolved oxygen (DO), pH, dissolved orthophosphate, nitrate-N, ammonia-N, and fecal coliform were the key water quality parameters measured. DO was measured with a YSI Model 57 Oxygen Meter, a Hanna digital pH meter was used to measure pH, orthophosphate, nitrate and ammonia were measured with the Lachat flow injection analysis, and fecal coliform was measured with the membrane filtration techniques. Given the high seasonal and diurnal variations the data was pooled for the wet season (November–March) and the dry season (June–August). Wet season and dry season average values were computed for each of the 16 sampling stations. These values were used to determine relationships between land use activities, surplus nutrient applications, site conditions in the watershed, and water quality. Spearman rank correlations were used to determine significant relationships between the variables.

4. Results

4.1. Land use dynamics

The historic land use in the watershed was determined between 1954 and 1994. As shown in Table I, the area under agriculture has remained constant over the entire period. This is mainly due to the introduction of the Agricultural Land Reserve (ALR) in 1973, that prohibits the conversion of highly capable agricultural land into other uses. While the area of agricultural land remained constant the number of farms increased by 26% over the same time period from 224 farms in 1954, to 283 in 1994. This increase in the number of farms on a fixed land base clearly reflects intensification.

TABLE II

Changes in animal numbers in the Sumas Prairie 1986–1996[a]

Year	Reported area farmed (ha)	No. of cattle	No. of pigs	No. of chickens	Expenditures in $ (lime + fertilizer)
1986	7405	18318	26049	323903	1,482,091
1991	7698	18535	38862	481542	1,698,707
1996	7596	18293	41329	859050	2,139,066

[a] Source Agricultural Census, Statistics Canada 1986–1996.

A second indicator of intensification is the increase in animal densities in the watershed. The agricultural census data for 1986, 1991 and 1996 showed that over the 10 yr period the cattle population has remained constant (–0.1%), while the number of pigs has increased by 59% and the number of chickens by 165% (Table II).

As a result of the NPS pollution problem, fertilizer application rates have generally leveled off or declined in many parts of the world (Hallberg and Kenney, 1993). However, this does not appear to be evident in the Sumas River watershed. No direct or adequate information was available on changes in the rate of fertilizer application. However, the fertilizer plus lime expenditures, as reported by the agricultural census and shown in Table II, have increased from $ 1.4 million to $ 2.1 million over the 10 yr period. This suggests that the total use of fertilizers has not declined in the study area.

4.2. Nutrient Budgets

Based on farm surveys it was established that about 30% of the broiler manure is exported out of the watershed while all other manure is applied to nearby arable land within the Sumas Prairie. Using the GIS database and Brisbin's model it was then possible to determine the nutrient inputs, crop uptake and losses for all agricultural areas in the watershed. The surplus or net application rates were determined using three different data sources for the total farmed and the total cropped area within the watershed: farm survey data, agricultural census data, and waste management surplus data published by Brisbin. All calculations resulted in similar overall surplus nitrogen application rates for the watershed ranging from 120–160 kg ha^{-1} yr^{-1} (Figure 2). The same calculation was also conducted for each of the contributing areas delineated in the GIS database. Figure 3 illustrates the spatial distribution of surplus N application rates, which in one contributing area exceeded 300 kg ha^{-1} yr^{-1}.

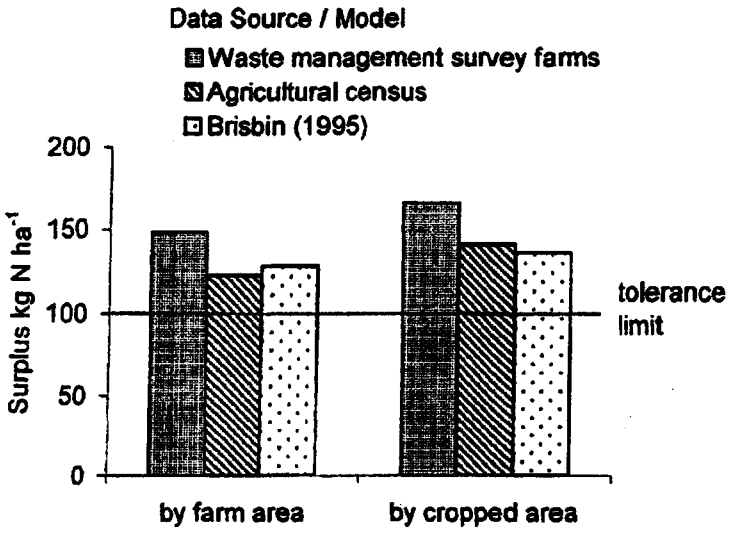

Figure 2. Overall nitrogen surplus applications for the Canadian portion of the Sumas watershed. Comparison between three different data sources (a surplus of 100 kgN ha^{-1} yr^{-1} is considered the upper tolerance limit for water quality impacts).

4.3. Streamwater Quality

With these excess application rates, the potential for streamwater contamination is high. Nitrate, dissolved orthophosphate concentrations and fecal coliform counts are displayed in the upstream to downstream direction for the wet and dry seasons in Figures 4, 5, and 6, respectively. Nitrate concentrations are of particular interest. Along the mainstem of the Sumas river elevated nitrate levels were measured during the winter months and lower concentrations during summer low flow conditions. In contrast a major tributary (Marshall Creek) showed the opposite trend, with high nitrate values at the upper stations during summer low flow conditions and low values during the wet winter season (Figure 4). While the Sumas headwaters and other major tributaries originate on mountain slopes that fall steeply to the flat agricultural valley, Marshall Creek flows from the major unconfined aquifer in the region (Abbotsford Aquifer). As shown by Liebscher *et al.* (1992), this groundwater resource is heavily contaminated with nitrates from agricultural sources. In summer low flow conditions the main water source for the creek is groundwater, while during the winter dilution occurs with the addition of wet weather runoff.

Orthophosphate levels (Figure 5) exceed water quality criteria guidelines of 0.01 mg L^{-1} generally considered conducive to eutrophication in lakes, and wet season coliform counts (Figure 6) exceed the recreational use Canadian water quality guideline of 200 MPN 100 mL^{-1}. Some of the major tributaries of the

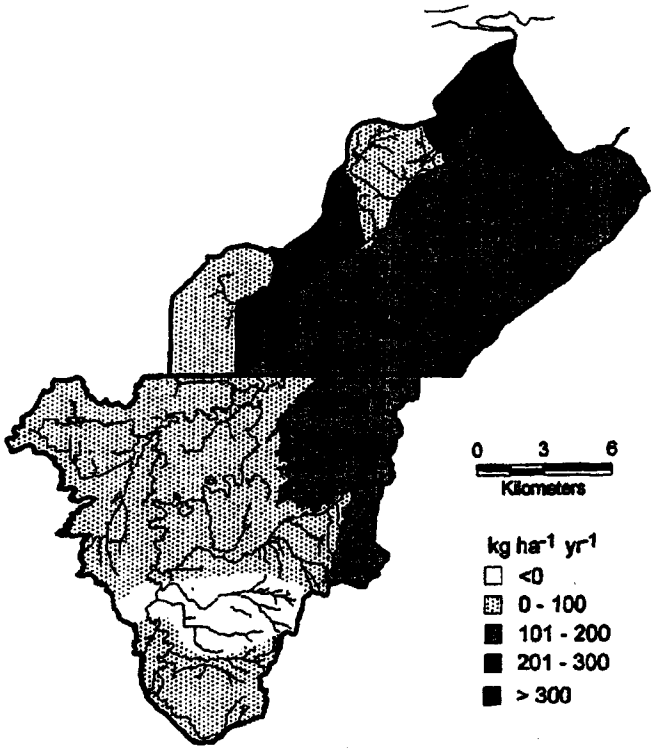

Figure 3. Spatial distribution of surplus nitrogen application in the Sumas River watershed (surplus values are in kg ha^{-1} yr^{-1} above crop needs and after subtractions of management losses).

Sumas River also showed significant dissolved oxygen and ammonia problems, particularly in autumn at the end of the growing season. At this time of year farmers apply large quantities of manure in order to have sufficient manure storage during the winter, when field applications are not possible due to wet soil conditions.

The effect of agricultural intensification on water quality over a longer period of time can be demonstrated by plotting historic wet season concentrations of nitrate-N, available only for the sampling station in the mid-section of the watershed (Figure 7). The plot shows the trend of an increasing spread of data values between 1970 and 1995, and a general trend in the upward direction. Although a clear trend is difficult to establish due to the scarcity of data, large data gaps and the variation of concentration with discharge, plots of chloride, other nutrient levels, dissolved oxygen, and pH showed similar trends in the direction of deteriorating water quality. These trends are indicative of the increasing types and intensities of land use activities occurring in the watershed.

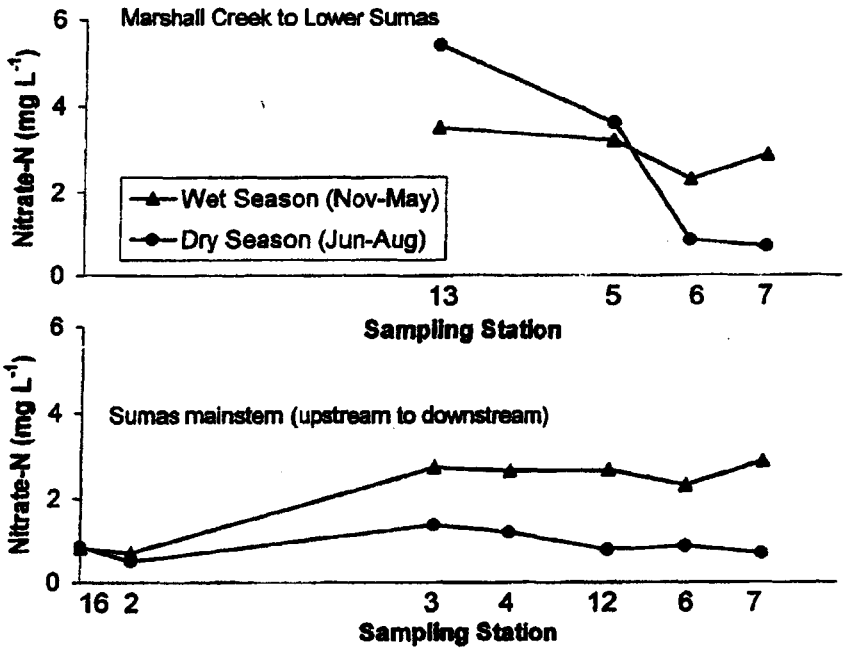

Figure 4. Comparison of nitrate variability in streamwater: Wet versus dry season, main stem stations versus Marshall Creek tributary stations.

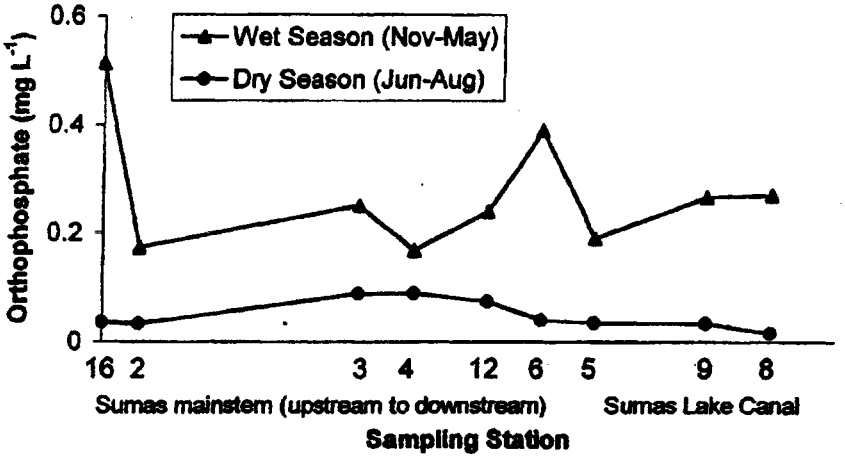

Figure 5. Differences in orthophosphate in streamwater: Wet versus dry season.

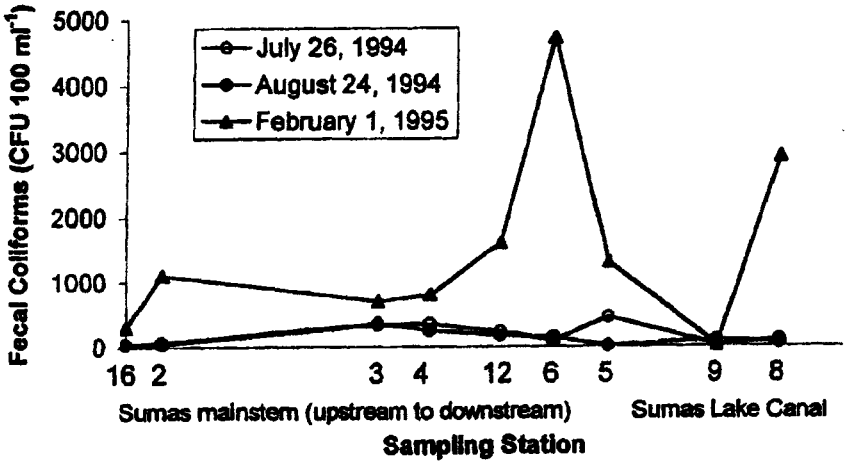

Figure 6. Differences in coliform counts in streamwater: Wet versus dry season.

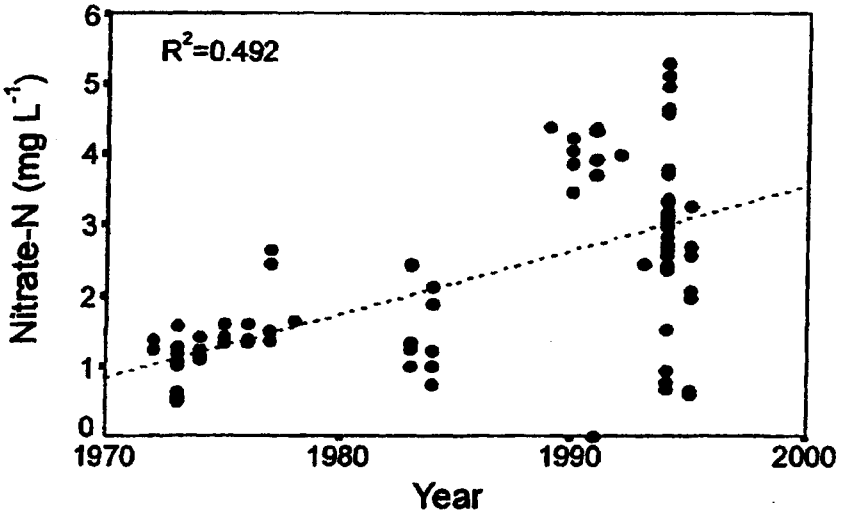

Figure 7. Historic changes in wet season nitrate concentrations in streamwater: 1970–1995.

Relationships between land use and water quality The Spearman rank correlation coefficient was used to identify relationships between the GIS based land use information and water quality. Land use was represented by various land indicator values calculated for each of the 11 contributing areas within Canada. Indicators that showed the best relationships are provided in Table III. Stronger relationships were obtained using wet season water quality indicators (ammonia-N, nitrate-N,

TABLE III

Significant relationships[a] between water quality, land use and site conditions ($p = 0.05$)

Land indicators	Ammonia-N	Nitrate-N		Dissolved oxygen	
	Wet season	Wet season	Dry season	Wet season	Dry season
Surplus N ha^{-1}	0.76			–0.84	–0.63
Pig density	0.67			–0.76	
% Loam texture	0.80			–0.89	–0.71
% Organic		0.75	0.74		
% Well drained				0.71	0.61
% Very poor drainage		0.82	0.83		

[a] n = 11, $p < 0.05$, and Spearman rank correlation coefficient shown in all cases.

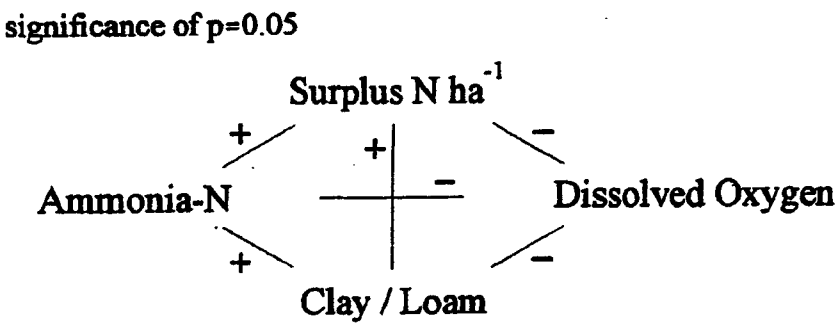

Figure 8. Wet and dry season relationships between surplus nitrogen application, water quality and site conditions.

and dissolved oxygen), surplus N applications, and site conditions (soil texture, soil drainage type, parent material, and animal density). Figure 8 represents the relationships found. Dissolved oxygen levels are negatively correlated, and ammonia levels are positively correlated, to surplus N application rates and the amount of finer textured soils within a contributing area. These results suggest that runoff during the winter is a key water quality problem in the watershed, likely influenced by heavy manure application rates at the end of the dry season and leaching from saturated soils.

The amounts of organic soils and the areas classified as poorly drained soils were also significantly related to nitrate values during the wet and dry seasons. This provides important information as to site vulnerability to N losses. Since N is much more dynamic than P it would be useful to examine if such relationships also occur with phosphorus. Sharpley (1995) suggests that dissolved orthophosphate values exhibited a poor relationship with the land use indicators and this may be explained

by the low solubility of phosphorus and its association with sediments. Sediments are mainly moved from land to the stream system via surface erosion from runoff and to determine its effect a different sampling design is needed. Monitoring of total and soluble phosphorus should be conducted during storm events, particularly during the late fall when soil surfaces are exposed and manure is being applied to the land to make room for winter manure storage. Access for land application is not possible in the winter because of saturated soil conditions. Since only one such event was captured at the beginning of the rainy season, significant relationships between land use and P could not be determined.

5. Conclusions

Agricultural intensification is leading to significant water quality problems in rural watersheds of the Lower Fraser Valley. Based on a case study in the Sumas River it was shown that the increase in animal units, and thus manure production and application, on a fixed land base is primarily responsible for the non-point source pollution problem. Over a 40 yr period the number of farms has increased by 26% and over the past 10 yr the number of pigs has increased by 59% and the number of chickens by 165%. Using a GIS database linked to a mass balance model it was determined that the overall surplus nitrogen application on the agricultural land in the watershed is 120 kg ha^{-1} yr^{-1} and reaches levels of more than 300 kg ha^{-1} yr^{-1} in the most intensively used area of the watershed.

Agricultural intensification is impacting streamwater quality, particularly during the wet winter season resulting in low dissolved oxygen, high ammonia and nitrate levels, and high fecal coliform counts. An evaluation of the historical nitrate concentrations in the watershed demonstrated that wet season levels have increased steadily.

A significant negative relationship was found between surplus N applications and dissolved oxygen and a significant positive relationship was found between surplus N and ammonia during the wet season. Similar relationships were found between these streamwater quality parameters and fine soil texture. Since soil texture is an important factor in the leaching process it is suggested that the GIS/budget technique can be used as, or contribute to, a risk assessment evaluation for streamwater pollution from agricultural non-point sources. Areas with high surplus applications and fine textured soils have the greatest risk of impacting streamwater quality.

It is also suggested that late summer/autumn land application of animal waste is likely one of the key sources of water pollution. Although some improvements to water quality may be achieved through manure management options such as increased storage and/or better timing of application, continued agricultural intensification will require that excess animal waste be processed or applied to nutrient deficient land outside the watershed in order to protect streamwater quality.

Acknowledgement

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