QUANTIFYING GROUNDWATER NITROGEN IMPACTS UNDER AN ONGOING BMP INVESTIGATION

Final Report
to the Environmental Farm Plan

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STUDY PURPOSE

This project arose out of a concern that groundwater impacts from agricultural chemicals were being inadequately monitored and quantified. The use of an inadequate conceptual model of the recharge and migration of non-point source solutes above and below the water table has underestimated some of the possible future impacts and has oversimplified the design and monitoring of investigative research. The frequent use of a single-point piezometer perpetuated an overly simplistic conceptual model, and resulted in inaccurate monitoring of chemical mass in groundwater from a given trial.

Typically BMP evaluations have largely focused on nitrogen fate in the soil and surface water. Groundwater has been infrequently evaluated, and, when considered, almost always entailed the use of a monitoring well with a relatively long screen (e.g. a foot long) which is infrequently sampled. There is strong evidence that wide variations in groundwater nitrate concentrations occur both seasonally and over the length of a single field. These variable nitrate concentrations render the use of an infrequently sampled single-point piezometer questionable. The single point may grossly over- or underestimate the nitrate concentration in groundwater, and does not permit calculation of mass flux of nitrogen to groundwater. Thus, the N input to groundwater cannot be calculated, representing a significant flaw in the groundwater component of BMP evaluations.

The stated objectives of this study were two-fold: to design and implement the first-ever intensive groundwater monitoring network in a field scale BMP evaluation, and to compare the results with the commonly used single piezometer method to evaluate relative groundwater impacts under different farm management practices (BMPs).

The groundwater monitoring network installed in the present study consisted of the monitoring wells typically used in BMP evaluation and three monitoring lines of multi-level sampling devices (MLSDs). The monitoring wells and MLSD monitoring lines were installed on the down-gradient edge of three different fields - each under a different crop and/or management. Each monitoring line has 7 or 15 MLSDs, and each MLSD has 15 intake points. The monitoring lines were located perpendicular to groundwater flow, and were sampled at regular time intervals, providing a 'moving picture' of groundwater nitrate distribution. Over the same time period, groundwater samples from the monitoring wells were also analyzed for nitrate.

Also reported here are the preliminary results of a companion study funded by the Waterloo Centre for Groundwater Research (recently CresTECH), entitled *Agricultural non-point source tracer test: Relining the conceptual model and defining a monitoring strategy.* The study sought to conduct a non-point source tracer test to produce an accurate 3-dimensional view of the groundwater distribution of recently recharged 'non-point source' contaminants. Specifically, the use of a tracer with non-detectable 'background' groundwater concentrations (in this instance bromide) permitted the distinction of recently infiltrated groundwater from `background' or older groundwater. Such a distinction is not possible with nitrate, where elevated 'background' groundwater concentrations can exist, obscuring the distribution of 'recently infiltrated' water.

MATERIALS AND METHODS

Site selection

A rigorous site selection procedure was conducted. Our initial intention to utilize sites performing twinned best management practice (BMP) evaluation was abandoned due to significant differences in hydrogeologic settings between farms, preventing the detailed comparison of different farming practices we hoped to conduct. The Middlesex County Soil and Crop Improvement Association's *Strathmere Lodge Demonstration Farm* was selected since it offered the following:

1) uniform sandy well drained soils
2) absence of tile drains
3) absence of manure application or livestock grazing
4) shallow water table
5) flat water table, horizontal flow
6) excellent cooperator involvement
7) ability to compare conservation system to conventional system
8) existing soil data base
9) complete climate records for past years
10) absence of point source of nitrogen
11) absence of pumping well or other significant hydraulic control

Site description

The site is located within the city limits of Strathhroy. It is bordered by County Road 39 and the County road salt dome and yard to the south, Strathmere Lodge to the west, Cuddy farms to the north, and a residential development to the east (Figure 1).

The Soil and Crop Demonstration Farm has been under no-till management since 1989. It is a relatively flat lying site located on the Caradoc Sand Plain (Chapman and Putnam, 1966). The excessively drained soil, mapped as the Fox fine sand, has a low (2- 2.5%) organic matter content and slightly acidic pH (varying between 5 and 8, with composite sample results between 6 and 6.5). Small variations in soil texture manifests in variable water retention, reflected by measurable differences in crop response (Stockman, 1997). There is an overall increase in height of land from the south end to the north end. The average slope is 0.3% (Figure 2).

The site has 5 different fields in various stages of a 6 year no-till rotation consisting of soy-corn-soy-corn-alfalfa-alfalfa (Figure 3). The Soil and Crop Improvement Association leases the land from the county, and pays a local member and innovative farmer, Nick Stockman, to operate the farm. There are no farm buildings on the site, and in recent history, has not supported livestock.

Preliminary hydroqeologic evaluation

Monitoring well installation and soil cores

Sixteen shallow water table wells were installed over the 90 acre farm (Figure 4) in order to assess depth to groundwater, groundwater flow direction, magnitude of vertical gradient (if any), and groundwater flow rate. These wells are typical of standard groundwater monitoring well installations used to compare the groundwater impacts from different management practices on Ontario farms and some research sites. In this study nitrate concentrations from water pumped from these monitoring wells were compared with the data from the multi-level sampling devices.

The water table wells consisted of 2" or ½ " PVC pipe with a 1 foot slotted screen at the bottom (Figure 5). These were installed to varying depths, but primarily finished 0.5 to 2 m below the water table. These wells were pushed into a hole previously augured by a Giddings drill rig with solid stem augers. The well annulus above the water table, which generally would not cave onto the well, was filled with bentonite to surface.

The water table was found at depths of 1.5 to 4 meters beneath the ground surface, depending on location. Soil conditions encountered beneath the water table during drilling were found to be homogenous to depths of 4 to 5 meters beneath the ground surface, and between bore locations. Vertically nested wells were installed in two locations, and no appreciable vertical gradient has ever been measured.

Contouring the water table elevation from the water table wells revealed a relatively flat water table, sloping from north to south with an approximate hydraulic gradient of 0.003. Groundwater flows south east at 150° (S30° E). Slug tests conducted on 8 different monitoring wells (Hvorslev, 1951) yielded an average hydraulic conductivity of 1.5×10^{-4} m/s (std. dev. = 1.2×10^{-4}). The calculated average linear velocity is about of 13 cm/day or 47 m/a (porosity assumed to be 0.3). Recharge for a Southern Ontario sandy phreatic aquifer is estimated between 15 and 25 cm/a (MacFarlane *et al.*, 1983, Shutter *et al.*, 1994, Howard *et al.*, 1996). Hence, groundwater nominally flows between 45 and 85 cm vertically and 47 m horizontally in a given year.

Average annual precipitation in the region is 80 cm/a (Stockman, 1997). The water table fluctuates seasonally with annual maximum soon after spring recharge occurs, and annual minimum after the growing season (Fetter, 1988; Figure 6).

Soil cores from above and below the water table were obtained to 5m for evaluation of soil texture and hydraulic parameters. The sediments consist of medium grained sand to at least 5m depth.

Multi-Level Sampling Device Design and Installation

Ninety four multi-level sampling devices (MLSDs or piezometers) were initially installed at three locations at the Strathroy site. Each MLSD consists of a ½ inch PVC central tube with 15 narrow diameter (1/8") tubes (each extending to a different depth) clustered around each stalk (Cherry et al., 1983; Figure 5) . The end of each tube is covered with a Nytex® screen to prevent aquifer sediments from clogging the tubes. The tubes are attached to the central tube providing a small diameter (less than 2") installation that allows the collection of groundwater samples from 15 discrete depths. A Giddings auger or Vibracore drill rig was used to auger holes to the desired depth, a two inch PVC pipe with a pointed tip was pushed into the ground to the desired depth. The MLSD was inserted into the 2 inch pipe and pressure on the MLSD center stalk pushed out the pointed tip. The PVC pipe was removed and the surrounding sand was allowed to collapse onto the MLSDs. The remaining annulus that did not cave in (i.e. above the water table) was filled with bentonite to surface. High vertical concentration gradients frequently observed in the MLSDs suggest significant vertical short-circuiting between points is not occurring.

The multilevel sampling devices were installed at 3 locations at the site in 1996. At each location 7 or 15 MLSDs were installed in a monitoring line oriented perpendicular to groundwater flow. They were spaced 0.75 m apart with a staggered installation depth such that the top sampling point ranged between 1 m above the fall water table level to 0.5 m below the lowest water table level (Figure 7). This allowed the monitoring of a maximum vertical extent of the aquifer.

The Main monitoring grid, consisting of four individual monitoring lines of 15 MLSDs each, was installed at the downgradient end of a field that was planted to soy beans in 1996 and to alfalfa in 1997 (Figure 4). The Mid-field monitoring line, also containing 15 MLSDs was located at the downgradient end of a field that was planted to no-till corn in 1996 and soy beans in 1997. The Top-field monitoring line, consisting of 7 MLSDs, was located at the upgradient side of the site in order to monitor the groundwater nitrate emanating from the conventional tillage farm located immediately upgradient. This conventional till field was planted to corn in 1996 and to carrots in 1997.

According to our nominal calculations of equivalent vertical and horizontal groundwater flowpath lengths, the entire upper 3m of groundwater monitored by the main and mid field monitoring lines should have been recharged under the field immediately upgradient. The top field monitoring line is an exception. It is separated from the field of interest by a road allowance, and the 3m vertical interval instrumented by the monitoring line receives groundwater that was infiltrated under 2 or 3 fields located upgradient of the road allowance. An extra MLSD (P94) was installed immediately upgradient of the road allowance to improve our understanding of the road allowance's influence on the groundwater nitrate distribution.

Bromide Tracer Application

Bromide was selected as a tracer to avoid any possibility that tracer concentrations might be attributed to `background' sources (there is no detectable bromide concentration in Strathroy groundwater).

The tracer was broadcast as potassium bromide in a dry granular form at a rate of 170 kg/ha over an area 200 m wide by 200 m long, directly upgradient (but not over top) of the main monitoring line in November, 1996 (Figure 8). The tracer was applied over a relatively wide area to avoid the possibility that slight fluctuations in the water table (and hence groundwater flow direction) might cause the tracer to "miss" the sampling network. The bromide spread was started 2 m upgradient of the main monitoring line.

A second bromide tracer was spread at the same rate in a 2 m strip upgradient, a 2 m strip downgradient and directly over the mid-field monitoring line of MLSDs (Figure 8). Soil samples were taken on a weekly basis (until ground ice made sampling impossible) to monitor the tracer as it was displaced from the soil zone by precipitation.

Monitoring protocol

For the first 7 months of the program all three monitoring lines were sampled every week. All samples were analyzed for nitrate and selected samples for bromide. A maximum of 555 samples was taken per week (fewer if some of the points were finished above the water table or inaccessible due to freezing). Sampling was conducted by attaching each point of a MLSD to a 15-point sampling manifold that allowed all 15 points to be sampled simultaneously. A vacuum pump was attached to the manifold to provide suction to simultaneously draw water from all points. The 60 mL polyethylene sample bottles were rinsed once with 30 - 40mL water and then filled leaving approximately 10 mL head space. The rinse water volume was sufficient to remove 3 - 4 times the stagnant water in the deepest tubes. Shallower installations with a lower standing water volume received more flushing prior to sampling. These volumes are equivalent to the amount of groundwater contained in a 4 cm-spherical volume of aquifer material.

The first 4 months of sampling were conducted in the winter of 1997. In order to prevent tubes from freezing during sampling an ice fishing tent with a propane heater was used to keep the sampling area warm. Tubes were tapped after each sampling event to force water back down into the ground so that it would not freeze. Considerable difficulty was encountered with freezing tubes in the midfield monitoring line and hence data is missing.

Approximately 13% of samples taken were submitted for quality assurance-quality control purposes. On each day of sampling one sample of distilled water was pumped through the sampling manifold and submitted to the laboratory as an equipment blank. For each MLSD a sample of distilled water and a blind duplicate were submitted to the laboratory.

During the winter of 1997, the top 5 points in the main and mid-field monitoring lines were analyzed for bromide. Once the bromide was detected in a significant number of points (April, 1997), all samples from each of the MLSD were submitted for both nitrate and bromide analyses.

Water table monitoring wells located next to each of the monitoring lines were also sampled regularly and samples analyzed for nitrate in order to compare standard monitoring practices to the detailed sampling conducted in the MLSDs. These wells were sampled using a peristaltic pump.

Each week water levels were measured at 9 of the original water table wells and in 6 of the MLSD ½ " PVC center-stalks (2 from each monitoring line).

RESULTS and DISCUSSION

Dissolved oxygen (DO) was measured to evaluate whether denitrification was occurring. Chemetrix© field measurement kits were used. Dissolved oxygen concentrations in the groundwater were consistently high (average = 5.7 mg/L, std. dev. = 1.16, and n = 50). Vertical profiles show a slight decrease in D.O. with depth, but aerobic conditions were maintained throughout the vertical aquifer interval studied. Since denitrification is an anaerobic process, nitrate is assumed to be conserved in the section of the aquifer studied. This means that low concentration zones cannot be accounted for by denitrification.

Nitrate Profiles

Vertical contour plots of nitrate concentrations for each sampling event were constructed using SURFER (Golden Software, 1996). Displaying the time series provides a moving picture of the groundwater concentrations in the subsurface, in essence providing a third dimension. These can be used to display a moving picture of the nitrate concentrations over time. They show great spatial and temporal variability. Since it is impractical to include all of the profiles in this report, we have included a selection of profiles to illustrate our main findings. These findings represent our best understanding of the origin of the groundwater nitrate to date, and are continually being refined as we collect and process more data.

Characteristic nitrate profiles of each monitoring line

Characteristic plots of each monitoring line (Figure 9) illustrate the main difference between the three monitoring lines. The groundwater nitrate concentrations in groundwater sampled from all three monitoring lines were high, with average values close to two times the drinking water standard of 10 mg N/L and maximum concentration of 73 mg/L.

The main monitoring line exhibited consistently lower nitrate concentrations in the upper groundwater, underlain by a high N band at depth (Figure 9 g-i) during the winter and spring of 1997. We believe lower nitrate groundwater may have leached from the 1996 soy crop (which relies on natural N fixation and does not receive fertilizer N).

The mid-field monitoring line was planted to corn and fertilized with mineral N in 1996 (Figure 3). High nitrate concentrations (the highest measured to date in the study) in this monitoring line tended to be confined to small, isolated zones in the very shallow groundwater (Figure 9 d-f). These zones are surrounded and underlain by groundwater with lower nitrate concentrations. In the time sequence, these plumes appear to be long and narrow, or 'cigar-shaped'.

In some mid-field profiles, a second, lower layer of N-rich groundwater is clearly evident and separated from the upper layer by cleaner groundwater (with similar concentrations to the upper groundwater in the main monitoring line). The N-rich layers may have leached from 1996 and 1994 corn crops, and the lower-N interlayer may have leached from the 1995 soy crop.

A road allowance located between the top-field monitoring line and the upgradient field complicates the analysis slightly. The monitoring line and the upgradient field boundary are located at an angle to the road allowance (Figure 10). The nitrate distribution in this monitoring line typically shows slightly elevated concentrations in the upper left (north-west)corner, underlain by a band of relatively clean (low nitrate) water, in turn underlain by a second band of N-rich groundwater. If the MLSDs in the top-field monitoring line and the MLSD immediately upgradient of the road allowance (P94) are projected onto a single cross-section along the direction of groundwater flows, the origin of this particular nitrate distribution can be seen.

The shallowest sampling levels of P76 and P78 are apparently sampling N-rich water leached from the field upon which it is located (the shallowest points of P77 are too deep to intersect the plume). Thus, the elevated concentrations in the upper left hand corner (Figure 9 abc) are from N leached from the Strathmere Lodge Farm (it is worthwhile noting nitrate concentrations in the upper left hand corner behave similarly with time to the mid-field monitoring line, which is consistent with this interpretation). The underlying band of relatively 'clean' water is evidently leached from the road allowance (where no fertilizer is applied). This is consistent with a lack of clean water in the shallowest points of P94, located immediately upgradient of the road allowance. The N-rich band of groundwater located immediately below the road allowance water, is then presumably leached from the field immediately upgradient.

Nitrate concentrations in this top-field band are notably more consistent, both spatially and temporally, than those in the mid-field monitoring line. We attribute the shallow N-rich groundwater in both main and mid fields to 1996 applications of fertilizer. Intuitively, the different groundwater nitrate distributions (isolated zones in the mid-field vs. a more consistent `band' in the top-field monitoring line) could be attributed to the impact of tillage on soil properties. No-till management in the mid-field monitoring line likely enhances soil zone macropores, increasing the degree of preferential flow during recharge, and resulting in recharge of N-rich groundwater at discrete water table locations, causing `cigar-shaped' plumes. Tillage in the top-field, on the other hand, disturbs the macropores and mixes the soil, resulting in a more homogeneous and consistent source of nitrate to the groundwater throughout the year.

Changes in average concentration with time

The average groundwater nitrate concentration measured in any given sampling event is an index of the mass of nitrogen flowing past the monitoring line (i.e. higher concentrations indicate higher masses). Significant fluctuations in the average nitrate concentration observed in each monitoring line over the course of the study (Figure 11) indicate significant temporal variations in groundwater nitrate.

The comparison of the average concentrations over time in the three monitoring lines highlights several similarities and differences in the three monitoring lines. The maximum concentrations were noted in all three monitoring lines in spring 1997. This high concentration could be the result of the flushing of nitrate from the soil zone into the groundwater zone with the spring infiltration event. Concentrations in all profiles continually decrease from spring until fall, however concentrations in the main and mid monitoring lines reach annual lows in the fall. The reason for this "cleanup" of nitrate is unknown, however could be the result of heterogeneous recharge of spring meltwater and rains. Concentrations in the top monitoring line remain relatively higher than the mid and main monitoring lines, this could be the result of the different tillage or the application of manure at that location.

Changes in average concentration in upper meter of nitrate profiles with time

In order to examine the most recently recharged groundwater, the average groundwater nitrate concentrations in the upper meter of each monitoring line are plotted against time (Figure 11; in the top-field, the upper meter is taken to be the 'N-rich' band). Each plot seems to reach a maximum average concentration after which time the concentrations continually decrease. The maximums occur at the same time in the main and mid-field monitoring lines, while the top-field maximum occurs about 4 months later. If the mid- and top-field maximums are both attributable to 1996 fertilizer N applications, the later maximum average concentration in the top field monitoring line could be consistent with i) a deeper vadose zone depth (and residence time), and ii) time required for the nitrate to travel under the road allowance before reaching the monitoring line.

The mid-field monitoring line maximum average concentration, in early January, 1997, was mainly a result of the isolated N-rich zones in the upper groundwater (Figure 9) which we attribute to leaching of fertilizer N applied in 1996. These N rich zones move down in the profile, and are overlain by cleaner water after spring recharge (Figure 9), and surprisingly, disappear by September, 1997.

There are also clear differences between the monitoring lines in the average concentrations. The top field monitoring line has the highest average concentration and received the highest rate of fertilizer application. The main monitoring line received no mineral N fertilizer, and has the lowest average concentration. These concentration differences are reflected in the equivalent amount of N leached from each field (Table 1; amount of N leached is calculated as the average groundwater concentration multiplied the estimated annual volume of groundwater recharged over one hectare).

Thus, it appears that the leaching of spring applied fertilizer (or in the case of the main monitoring line, residual fertilizer or natural soil nitrogen) may result in annual groundwater nitrate concentration maximums that subsequently decrease over the course of the year (Figure 9). This interpretation is, of course, preliminary. We will have soon collected a full year's data on the three monitoring lines discussed here, and are also collecting groundwater samples from a new 'West' monitoring line located downgradient of 1997 growing season corn. These addition data will strengthen our understanding of the system considerably.

Given our interpretation of the groundwater nitrate data to date, we hazard some predictions about what we expect to see in average groundwater nitrate concentration in the upper meter in the next 6 months data:

i) Top field monitoring line (carrots planted and fertilized in 1997): should behave similarly to 1996 - average nitrate concentrations should maximize in spring, 1998, then decrease again;

ii) Upper left hand corner of Top-field monitoring line (now under permanent alfalfa growth) - will no longer behave as per Mid-field monitoring line because section of mid-field immediately upgradient of monitoring line will not longer be fertilized. The upper left hand monitoring line MLS points should become "clean" in absence of fertilizer application.

iii) Mid-field monitoring line (planted to soy in 1997, no fertilizer): concentrations should continue to decrease since the Jan, 1997 maximum; concentrations, may rise if significant nitrate is leached from soy bean die off;

iv) Main monitoring line (planted to alfalfa in 1997 and1998, no fertilizer). Since the alfalfa will over-winter, we predict average nitrate concentrations in the main monitoring line should continue to decrease.

Comparison of standard monitoring well data to MLSD data

The concentrations observed in the standard monitoring wells are clearly inconsistent with those observed in the MLSDs. Average concentrations measured in the standard monitoring wells are the reverse of those observed in the upper meter of each monitoring line. The top field monitoring line (where the monitoring well inadvertently intersected groundwater leached from the road allowance) exhibits the lowest nitrate concentrations, while the main monitoring line nitrate concentrations are consistently the highest.

Different MLSD points have been selected to i) included the MLSD point which is closest to the standard monitoring well, and ii) illustrate the variability in groundwater nitrate concentrations in MLSDs vs. the foot-long monitoring screens. It is also relevant to point out here that groundwater samples in BMP evaluations are typically only taken on a quarterly or annual basis and yielding still less data than collected here.

These results clearly indicate that the use of single point monitoring wells does not provide accurate estimates of nitrate concentration in groundwater. The practice is highly questionable, and a more rigorous monitoring strategy is required for the groundwater component of BMP evaluations.

Non-point source tracer test data

The bromide tracer was applied in November, 1996. It was first detected in the groundwater in December, 1996. Early groundwater concentrations were low and sporadic, and gradually increased with time (Figure 13). It was not until April/May that significant numbers of points showed significant concentrations of bromide. Beginning in May the bromide concentrations were plotted in a similar fashion to the nitrate concentrations. Bromide profiles in May, June and September are shown on Figure 14.

The bromide data show conclusively that the mass applied to the field surface arrives in the groundwater much sooner than would be predicted by estimating the `piston flow' movement of a solute through the unsaturated zone (between 1.4 to 3 years). The solute obviously is "short circuited" by some kind of preferential flow mechanism.

Main monitoring line bromide results

The June bromide profiles in the main monitoring line show the heterogeneous nature of the recharge at the site (Figure 14). There is one location where the bromide is concentrated immediately below the water table as would be expected, however the second location shows bromide in the lower west side of the monitoring line, over 2 meters below the water table. The elevation of this bromide is significantly lower than would be expected at a site with an estimated annual recharge between 15 and 25 cm, where the horizontal travel distance is on the order of 2 meters (between the edge of the bromide spread and the main monitoring line). By September bromide has spread throughout the entire thickness being measured on the west side of the monitoring line.

The groundwater distribution of the bromide tracer indicates recharge is highly heterogeneous. The presence of elevated bromide concentrations 2 meters below the water table is unexpected and suggests focused recharge events inducing significant vertical flow. In late February, 1997 the snow and ice that had accumulated since January melted and infiltrated over the course of several days. It is possible that this large volume of melted water was perched on impermeable ground ice until the latter was first breached (by melting through its entire thickness). The highly focused infiltrating water could create significant vertical gradients, forcing the water (and bromide) relatively deeply into the saturated zone.

Mid-field bromide results

The bromide profiles over time in the mid-field monitoring line behave more as expected. The highest concentrations occur at the water table and remain within the same 0.75 m for over 4 months (Figure 14) . The highest bromide concentrations at this location appear to pass by the monitoring line in July, 1997. The consistency of the depth of this profile over time suggests that solute density is not the mechanism causing bromide transport in the main monitoring line to occur deeper than 3m.

The mid-field bromide data indicate the spatial variability of depression-focused recharged. In this case the nature of bromide arrival at the top of the water table occurs more evenly, and follows the standard conceptual model for groundwater recharge. The expected time for the water to flow the two meters over which the tracer was applied is 20 days, however it took approximately 90 days from the time a significant amount of bromide reached the water table to the arrival of the center of mass. The arrival of "piston flow" bromide should not occur until summer of 1998. This difference in predicted vs. real flow times suggest early bromide may be transported through preferred flow paths, while later arrival may have entered into immobile flow zones, causing significant dispersion , and retarding transport of the centre of mass.

CONCLUSIONS

1. Agricultural nitrate concentrations in groundwater in a simple hydrogeologic regime show great temporal and spatial variability.
2. The average nitrate concentration is almost double the drinking water standard. Concentrations as high a 73 mg/L were measured although the extremely high concentrations tended to be confined to relatively thin (less than a meter) bands or zones of N-rich groundwater.
3. Because of the crop rotation at the site, ongoing monitoring should demonstrate the effects of continuous alfalfa production on nitrate concentrations. This practice is predicted to result in relatively low nitrate concentrations possibly similar to those found leached from road allowance.
4. Comparisons of MLSD data to data retrieved from standard monitoring wells shows that standard wells are insufficient to evaluate the relative groundwater effects of different farming practices. Nitrate concentrations from the standard monitoring bore no relation to the average concentrations in the MLSD monitoring lines. Standard monitoring wells also failed to show the variability in nitrate concentrations observed in the MLSDs.
5. Differences in groundwater nitrate concentration and distribution were observed. A) Comparison of no-till to conventional till corn crop suggests that both farm management practices result in the leaching of significant concentrations of nitrate to the groundwater zone. Nitrate concentrations were higher and more spatially variable in the no till case, however they were more consistent and persistent in the conventional till case. These differences may or may not be the result of the different tillage practices alone. B) Significantly less (and continually decreasing) nitrate concentrations were observed under a soy field where no fertilizer was applied.

ONGOING ACTIVITIES

The following activities are ongoing at the Strathmere Demonstration Farm:

1. Groundwater samples continue to be taken from all three monitoring lines (main, mid- and top-field) on a monthly basis for nitrate analyses, and a bimonthly basis for bromide analyses.

2. A 48' monitoring line, the ‘West monitoring line' has been installed. In addition to nitrate monitoring, a third tracer test is being conducted. For this tracer test, the dissolved KBr was applied *concurrently* with liquid UAN fertilizer in May, 1997. Since bromide and nitrate are both mobile and conservative in the section of the aquifer studied, this tracer test will allow us to 'fingerprint' nitrate leached from the 1997 UAN fertilizer application. Based on our current hydrogeologic analysis, we should expect to see ‘cigar-shaped' zones of the fertilizer nitrate in the west monitoring line by December, 1997. In addition, spring

application of the tracer in this instance more realistically mimics fertilizer application.

3. Groundwater samples have been submitted for CFC dating.

4. Groundwater samples have been taken from 5 points to see if ^{18}O in nitrate can be used to discriminate between naturally fixed nitrogen (soy and alfalfa) and fertilizer nitrogen (corn).

5. Groundwater samples have been taken for ^{13}C in DOC to see if one can differentiate between corn carbon (a C_4 plant with $\delta^{13}C$ of 10-14°/$_{00}$) and soy and alfalfa (C_3 plants with $\delta^{13}C$'s of 22-27°/$_{00}$).

6. Funds have been granted from the University of Calgary to conduct geophysics at the site to look for vadose zone structure that may be responsible for the observed preferential flow.
7. Funds have been granted from CresTECH to carry the study through early 1998. Additional funds have been applied for the 1998-1999 period.

FUTURE ACTIVITIES

1. A geostatistical analysis is warranted to i) determine the minimum sample set required to characterize temporal spatial variability under given farm field, ii) determine characteristic vertical and horizontal correlation lengths of (and changes in) nitrate concentrations, etc. It is estimated that fewer MLSD could be used and the data extrapolated to yield similar results. However, this very detailed work was required to adequately assess how many wells would be sufficient.

2. Conduct groundwater modeling of the data to estimate factors such as the variability of recharge and how nitrate concentrations could be affected, attempt to explain why concentrations decreased so significantly in the summer of 1997, evaluate different preferential flow mechanisms causing `cigar-shaped enclave', etc.

3. A soil zone monitoring study is needed to elucidate the times of year when soil nitrate is indeed available for leaching, and relative differences between fields.

ADDITIONAL COMMENTS on STRATHMERE DEMONSTRATION FARM POTENTIAL

The Strathmere Demonstration Farm is now hydrogeologically extremely well characterized. It is now set up to conduct Best Management Practice verification with meaningful groundwater components. While we intend to continue evaluating the effects of the different crops that are included in the current rotation (and to refining our understanding of groundwater nitrate impacts), parts of the farm could be used to conduct groundwater BMP evaluations to evaluate other farming practices. It is our opinion that such BMP evaluations that would be highly successful in the context of elucidating relative groundwater impacts.

PRELIMINARY MONITORING STRATEGY FOR GROUNDWATER COMPONENT OF BMP EVALUATIONS

To accurately evaluate the relative effects of different farming practices on groundwater quality, and even to assess any non-point source impacts on groundwater quality at all, improved monitoring must be implemented. This study has shown high variability of nitrate in groundwater both in time and space, and the shortcomings of standard monitoring practices. The recommended strategy described below is a preliminary effort. Ongoing research (particularly at different sites with different soil conditions, etc.) will clarify the number of monitoring wells and samples that would be required to adequately assess nitrate in groundwater.

It is our belief that the method of site `pairing' in current BMP evaluations (i.e. location of paired sites within a few kilometers of one another) is inadequate for groundwater evaluations due to variability in subsurface sediments, groundwater flow rates, vadose zone depths and residence time, etc.. Meaningful evaluations of relative groundwater impacts from different farming practices is best done by employing the different farming practices on the same field.

At any given site:

1. 3 water table wells should be installed to estimate the groundwater flow direction and to allow ongoing recording of water table fluctuations. Hydraulic conductivity should be estimates to provide average monitoring linear velocity calculation. This information should be used to estimate the depth of groundwater that recharged under the field of interest at the downgradient end of the field.

2. At least 3 MLSD should be installed at the downgradient end of any monitored field with 3 to 10 points or depth intervals. The depth intervals should be no greater than 50 cm apart and the total interval monitored should represent twice the vertical depth that groundwater flowing beneath the entire field would occupy.

3. All sampling points should initially be sampled biweekly to estimate the distribution of nitrate in the subsurface. This frequency should be reduced to monthly if variation in the data support such a change.

4. Water levels should be measured at the same frequency as the MLSDs are sampled.

5. Monitoring should continue for at least a year to fully assess the impacts from any treatment. If the water table is deep at the site (>3 m) or the soils have low permeability then a delayed arrival at the water table should be estimated and sampling altered accordingly.

6. When possible, a non-point source tracer should be applied to permit an understanding of the amount of time elapsed between ground application and groundwater observation of the tracer (and nitrate).

REFERENCES

Chapman, L.J., and D.F. Putman, 1966. The physiography of Southern Ontario. 2nd Edition. University of Toronto Press.

Cherry, J.A., R.W. Gillham, E.G. Anderson, and P.E. Johnson, 1983. Migration of contaminants in groundwater at a landfill: A case study. 2. Groundwater monitoring devices. J. Hydrology. 63:31-49.

Fetter, C.W., 1988. Applied Hydrogeology. Macmillan Publishing Company, New York. 592p.

Howard, K.W.F., N. Eyles, P.J. Smart, J.I.Boyce, R.E. Gerber, S.L. Salvatori, and M. Dought, 1996. The Oak Ridges Moraine *of* Southern Ontario: A groundwater resource at risk. Geoscience Canada. 22(3):101-120.

Hvorslev, M.J., 1951. Time lag and soil permeability in groundwater observations. US Army Corps Engrs, Waterways Exp. Sta. Bull. 36, Vicksburg, Miss.

MacFarlane, D.S., J.A. Cherry, R.W. Gillham, and E.A. Sudicky, 1983. Migration *of* contaminants in groundwater at a landfill: A case study. 1. Groundwater flow and plume definition. J. Hydrology. 63:1-29.

Shutter, S.B., E.A. Sudicky, and W.D. Robertson, 1994. Chemical fate and transport in a domestic septic system: Application of a variably saturated model for chemical movement. Env. Toxicol. Chem. 13(2):223-231.

Stockman, N., 1997. Personal Communication.

Figure 1. Strathmere Demonstration Farm Location,

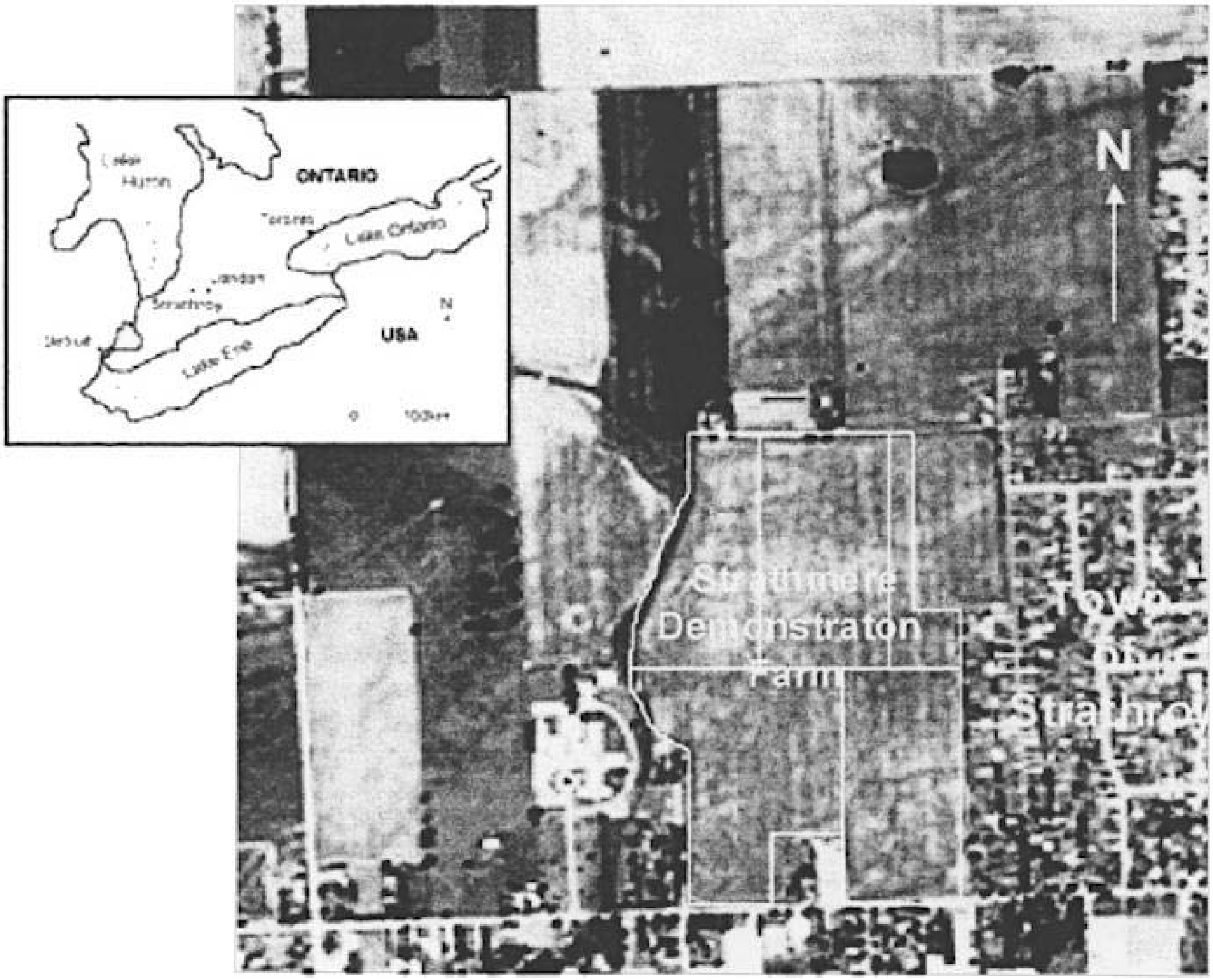


Figure 2. Topographic contours of Strathmere Demonstration Farm.

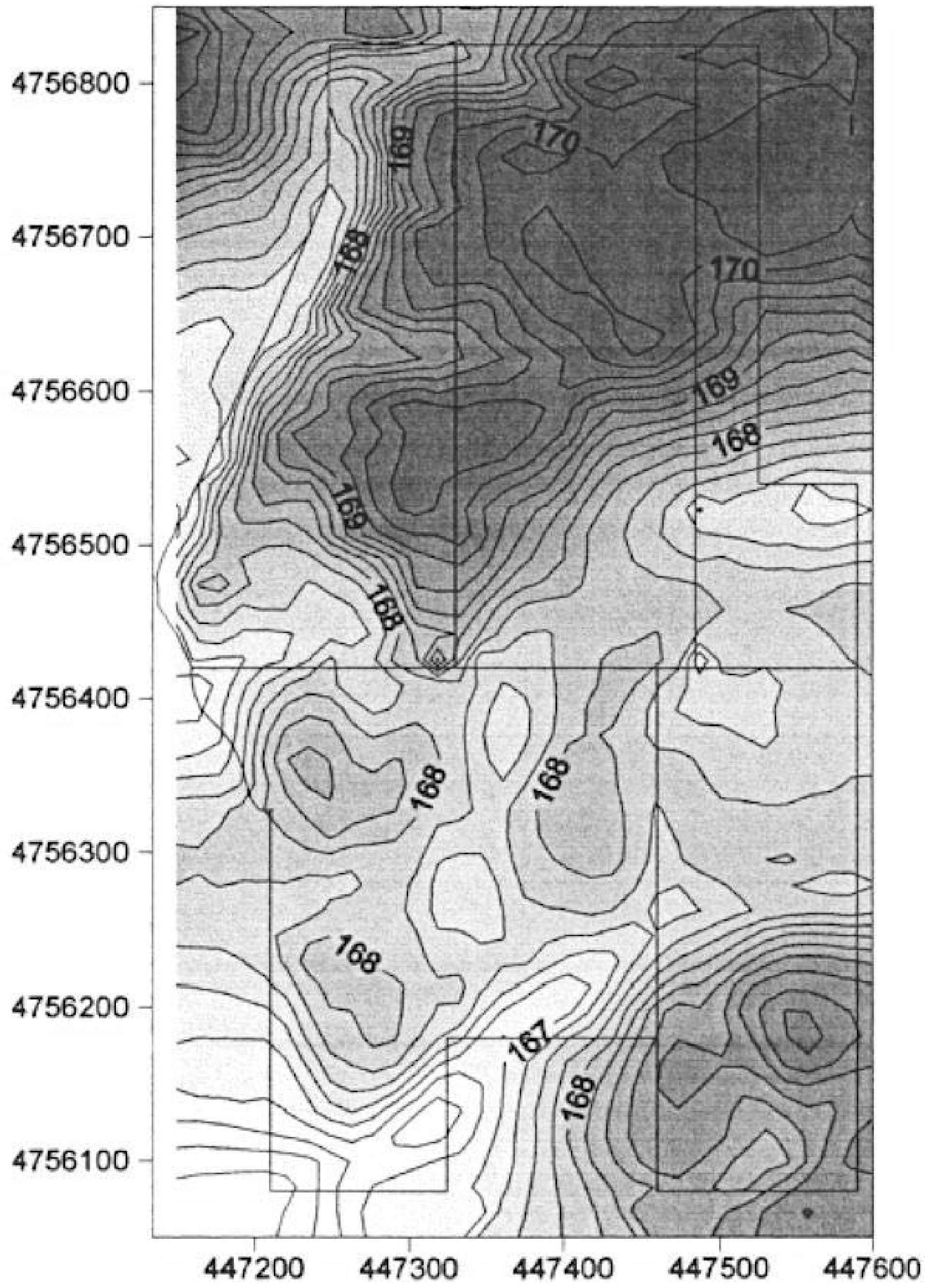
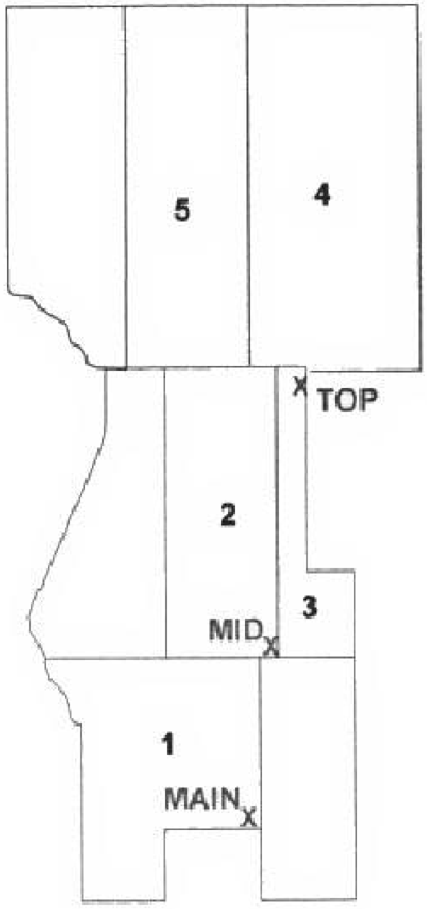


Figure 3. Cropping rotation upgradient of groundwater monitoring installations. Rate of annual fertilizer nitrogen application included in brackets (in kg N/ha).



Field	1998	1997	1996	1995	1994	1993
1 NT*	Alfalfa	Alfalfa (0)	Soybean (0)	Corn (154)	Soy (0)	Corn (77)
2 NT	Corn	Soy (0)	Corn (160W**/120E)	Soy (E)/Alf(W) (0)	Alf(W)/Soy (0)	Soybean (0)
3 NT	Alfalfa	Soy (0)	Corn (160)	Soy (0)	Corn (100)	Alfalfa (0)
4 CT	Corn	Carrot (100)	Corn (100)	Corn (100)	Wheat (90)	Unknown
5CT	Corn	Soy (0)	Corn (150)	Soy (0)	Corn (150)	Unknown

* CT = conventional tillage, NT = no-till management
** E = East side, W = West side of field

Figure 4. Water table contours and location of water table monitoring wells (open circles) at Strathmere Demonstration Farm.

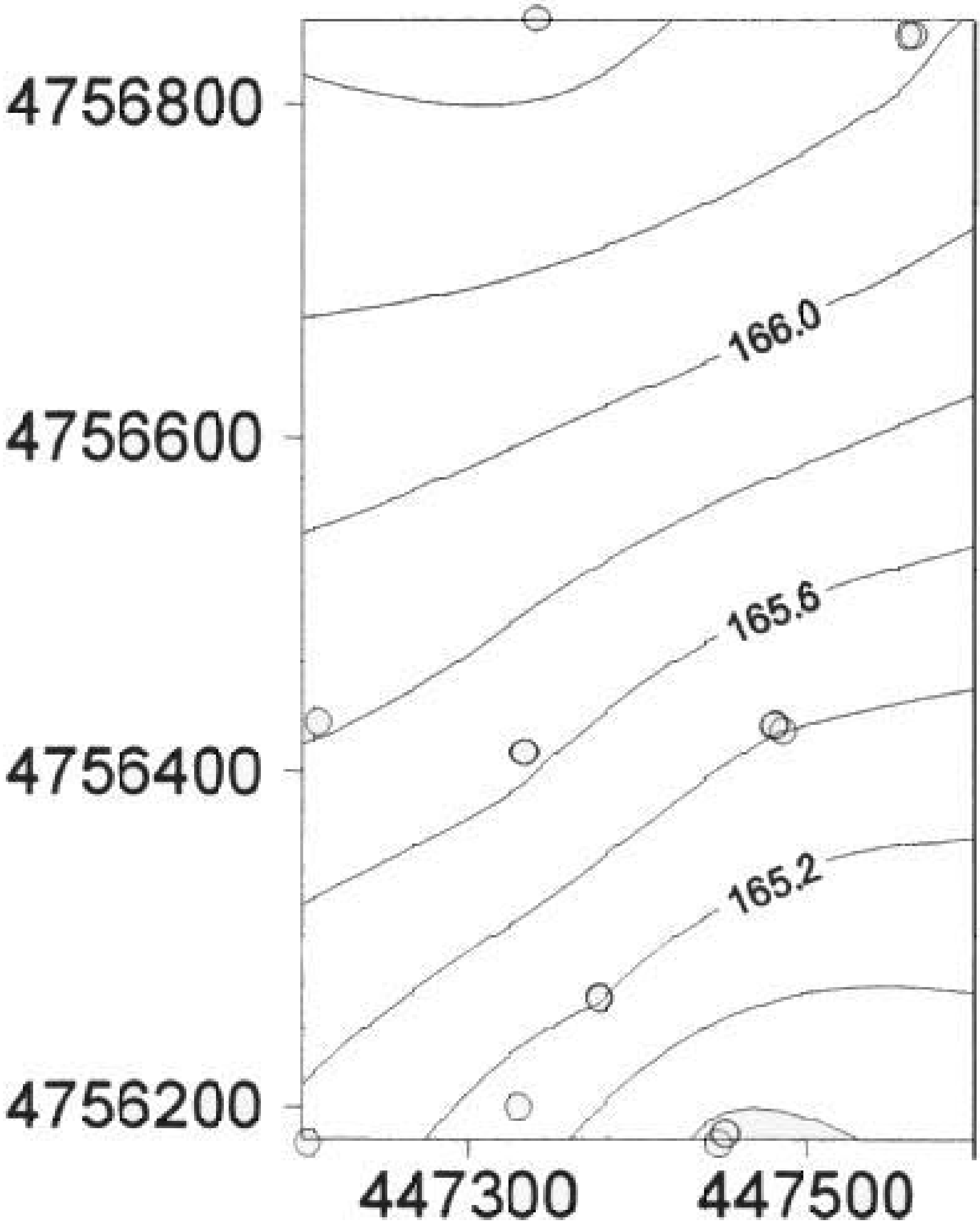


Figure 5. Schematic diagram of monitoring well and multi-level sampling device (MLSD).

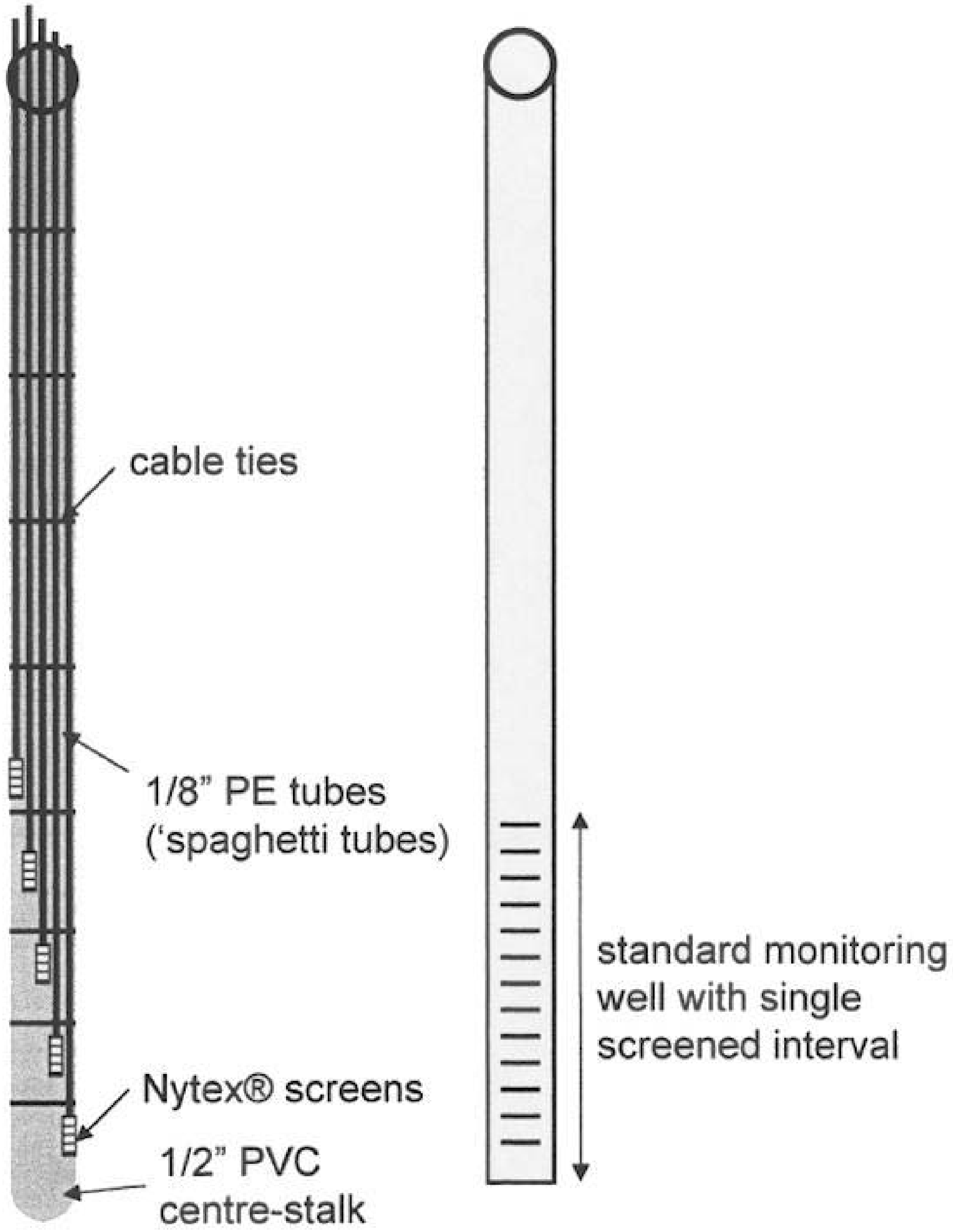


Figure 6. Fluctuations in average water table with time.

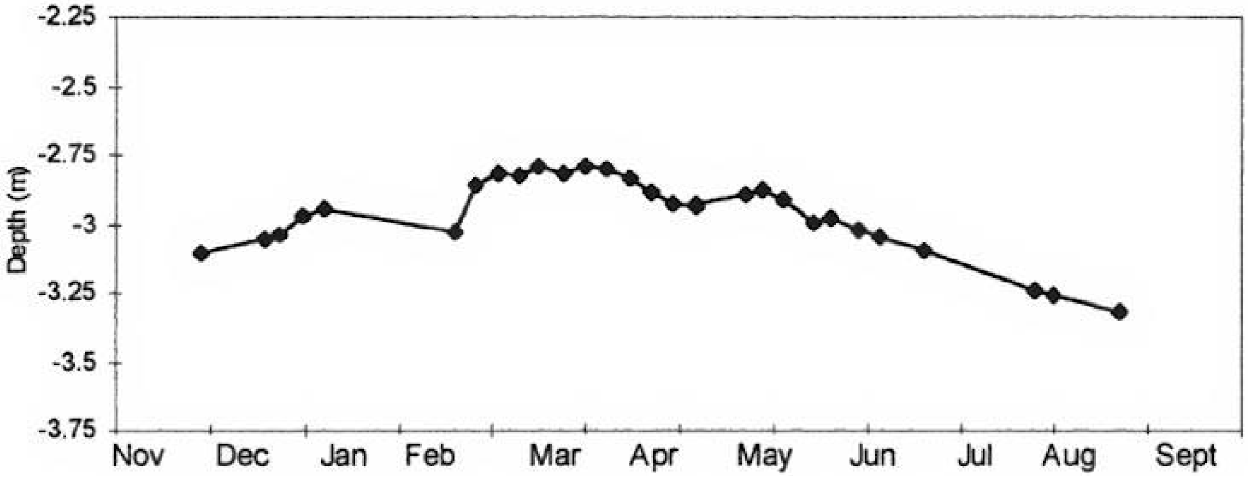


Figure 7. Schematic diagram of MLSD monitoring line.

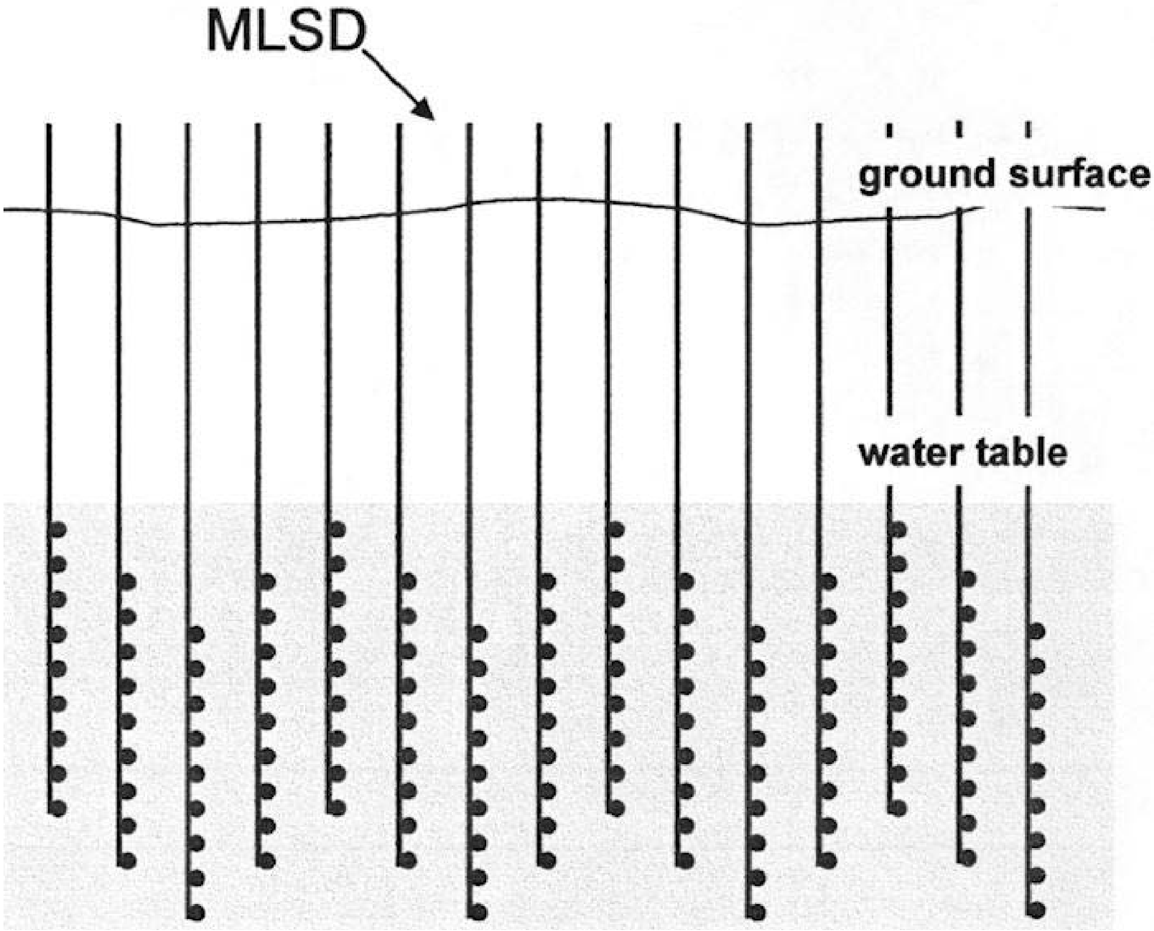


Figure 8. Location of non-point source tracer spreads.

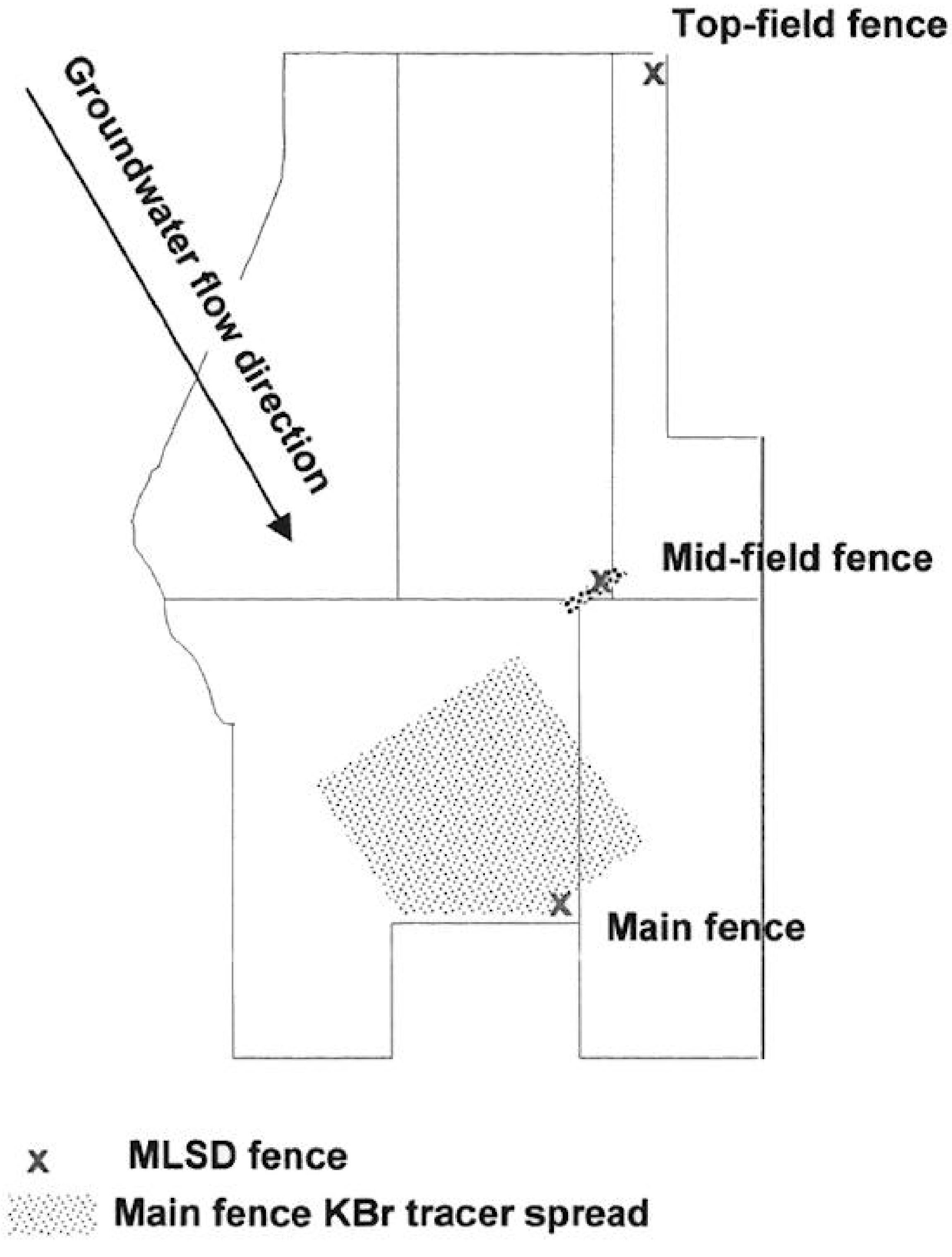


Figure 9. Characteristic profiles of groundwater nitrate (in mg NIL) in main (a, b. c), mid (d,e,f), and topfield (g, h, i) monitoring lines from December, May, and September, 1997.

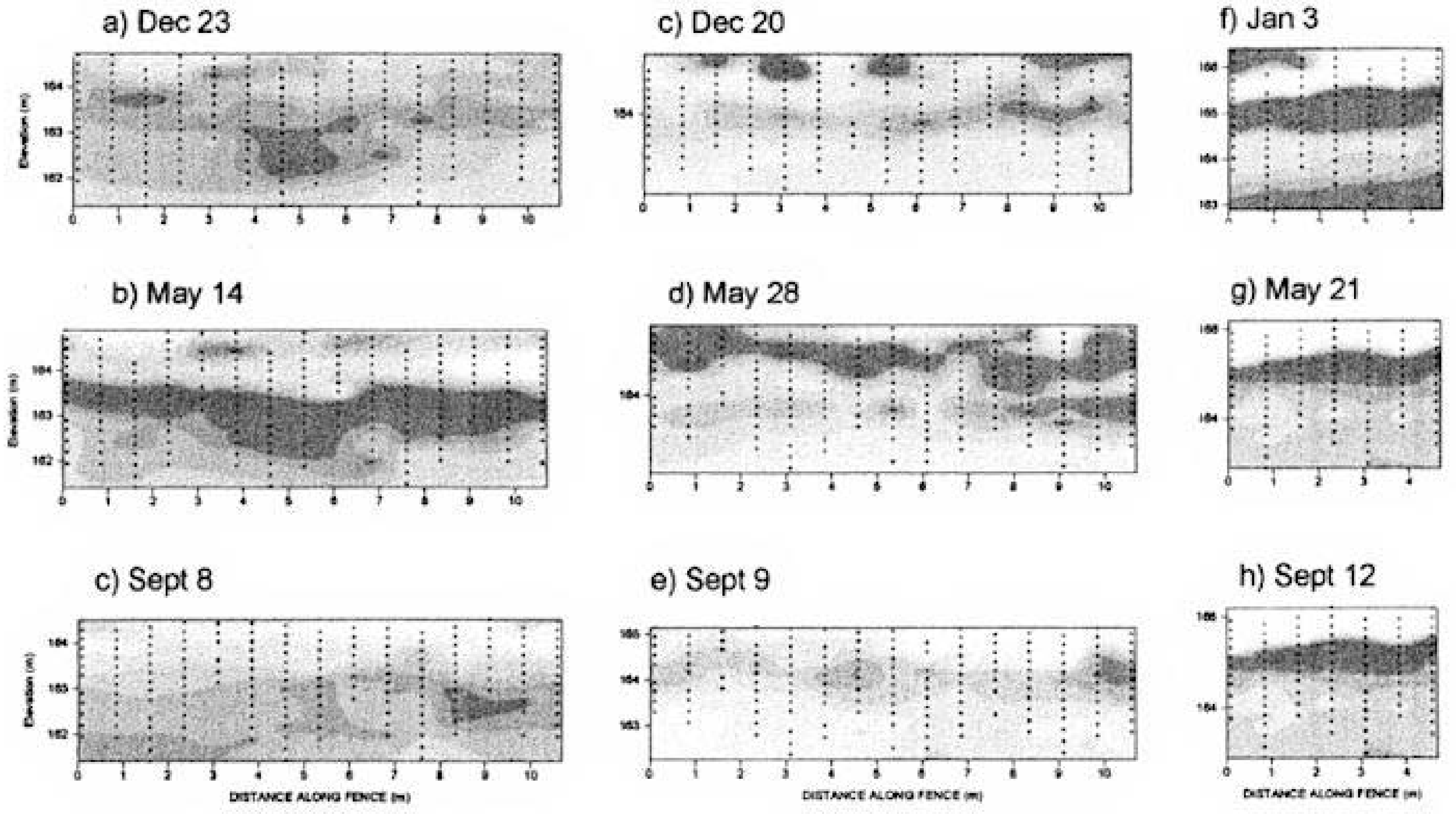


Figure 10. Location of top-field monitoring line with respect to road allowance and upgradient fields

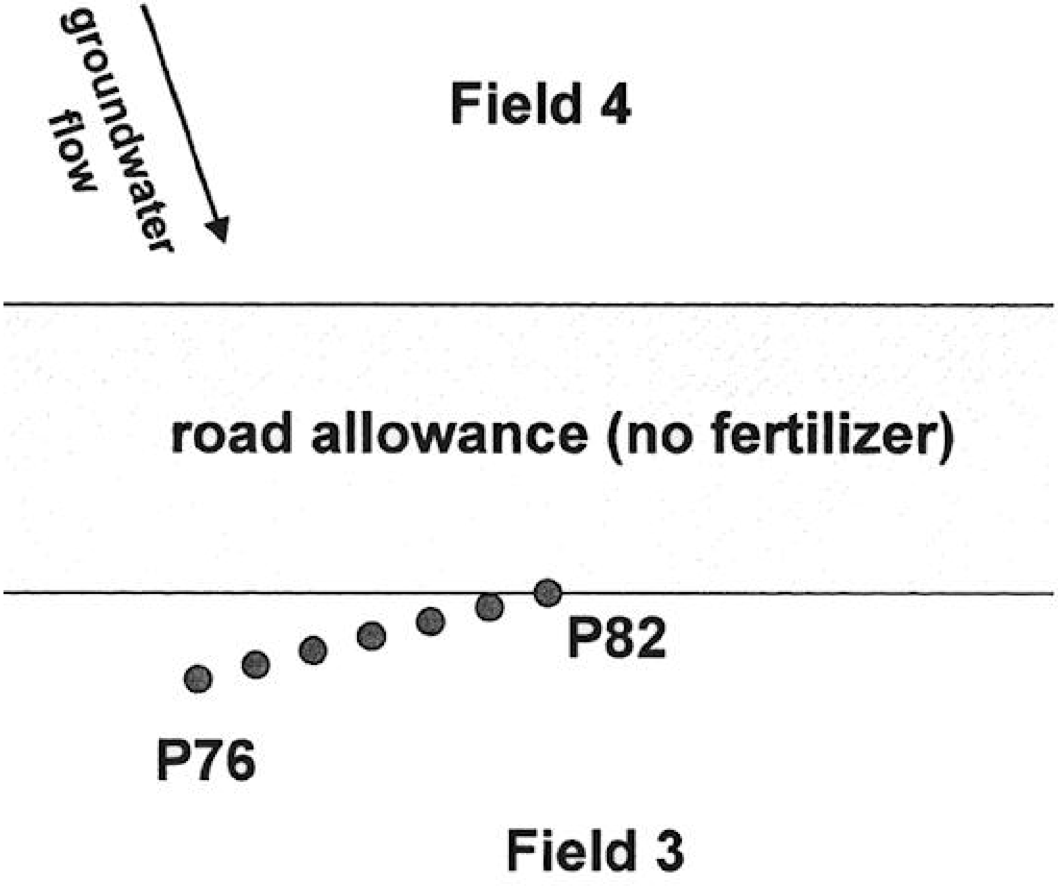


Figure 11. Average nitrate concentrations with time a) in each monitoring line, and b) in the upper metre of the main- and mid-field monitoring lines, and in the N-rich band in the top-field monitoring line

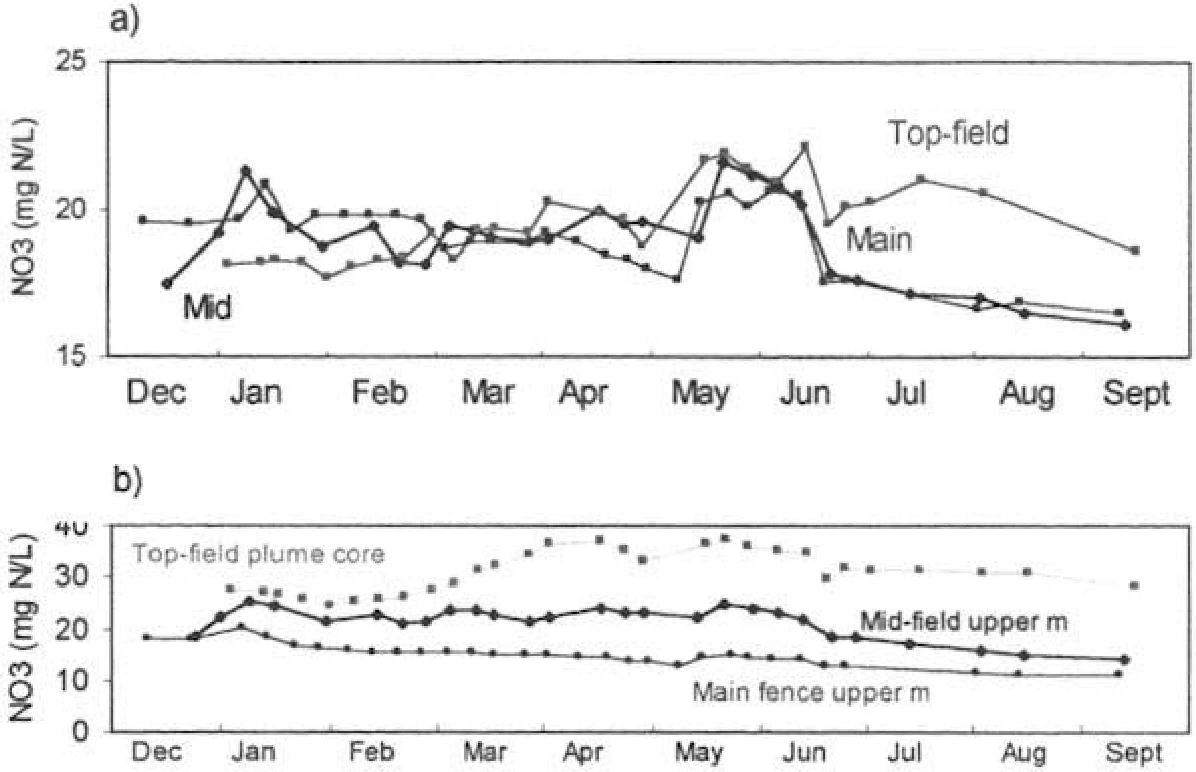


Figure 12. Nitrate concentrations in groundwater sampled from standard monitoring wells and MLSDs. Sampling point elevations shown on RHS in MASL. Blue symbols show water table fluctuation over the course of the study.

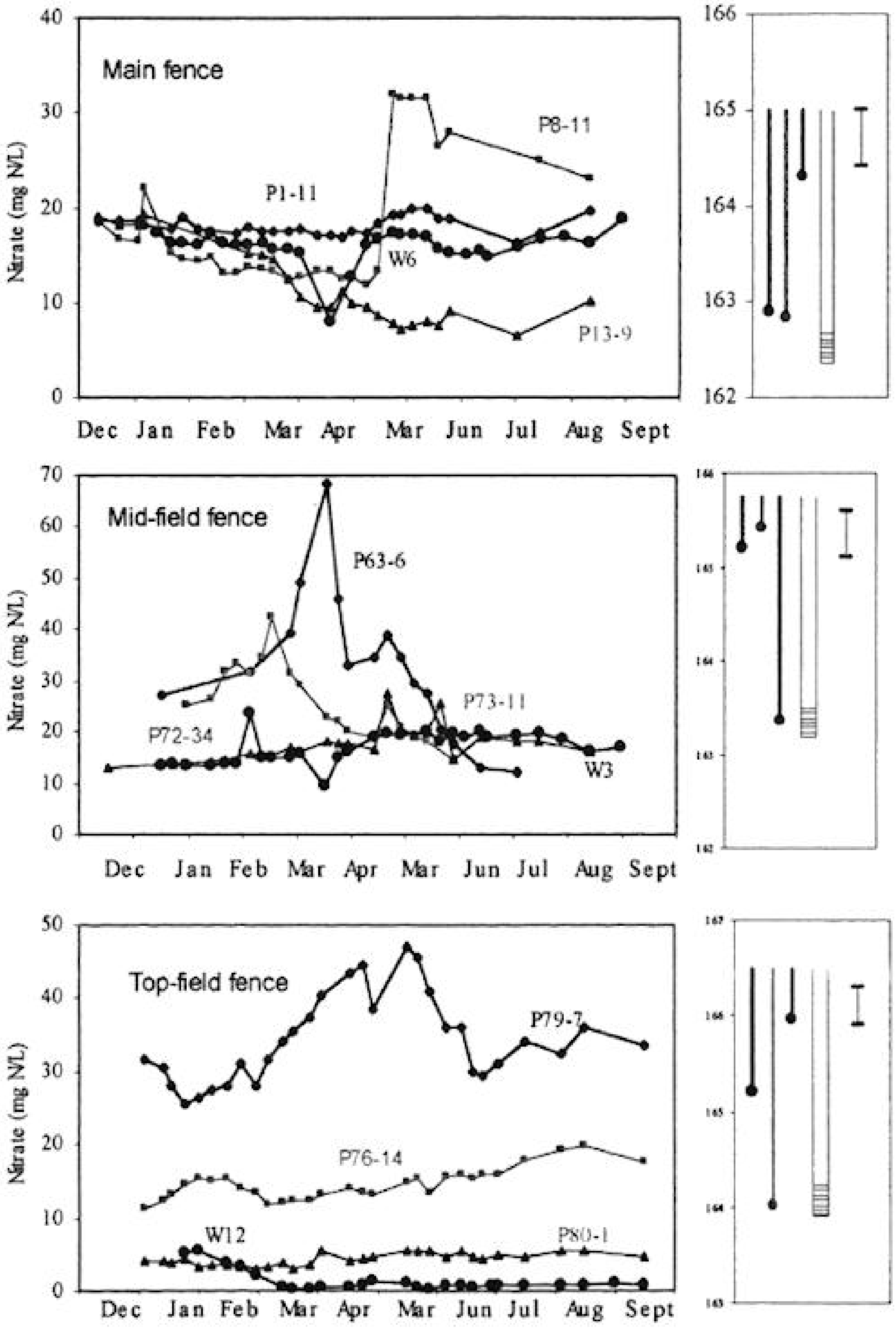


Figure 13. Bromide concentrations in groundwater vs. time (to 160 days after tracer application).

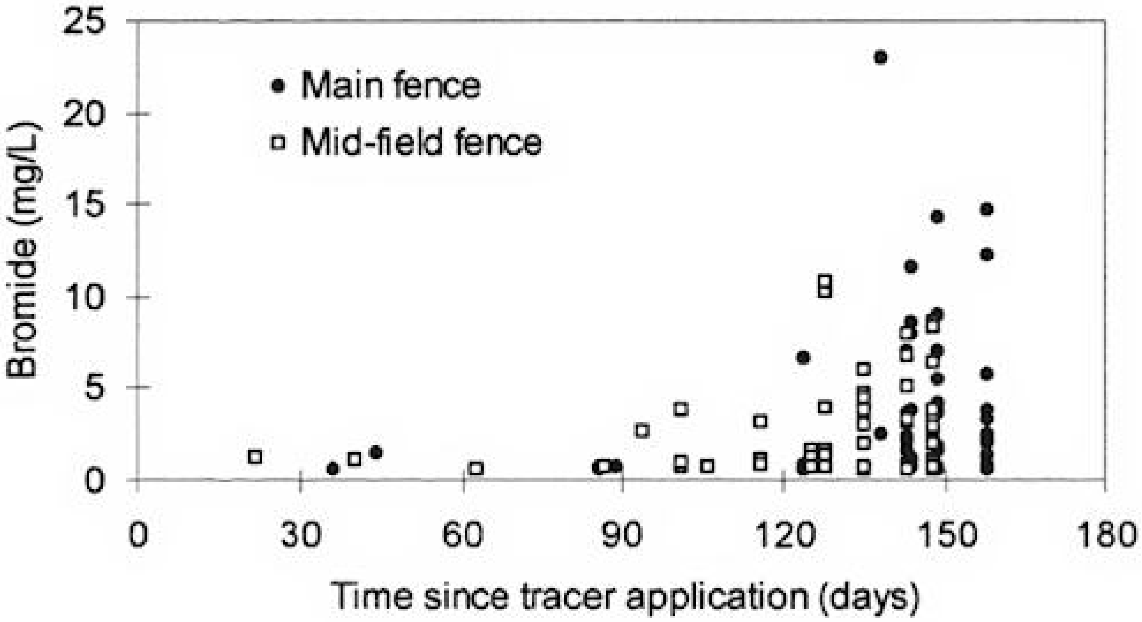


Figure 14. Bromide profiles from main and mid-field fences for May, June, and September, 1997.

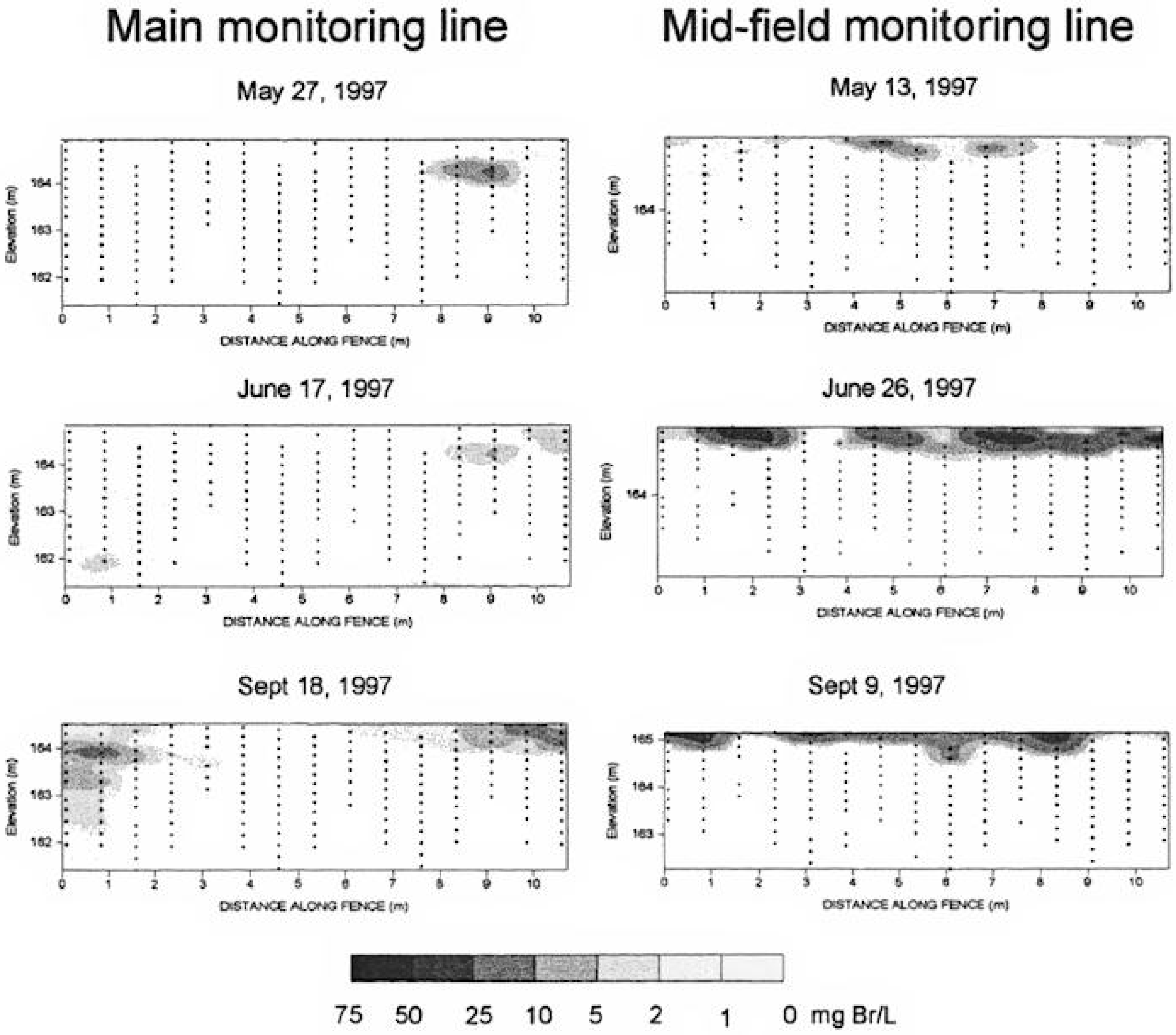


Table 1. Average groundwater nitrate concentration measured in upper metre of main, mid-field, and `plume core' of top-field monitoring line (Dec, 1996 to Sept, 1997), and calculated equivalent leaching rate of nitrogen fertilizer for recharge ranging from 15 to 30 cm/a.

	Average NO_3 (mg N/L) (std. dev., n)	Equivalent N leached kg N/ha	
		15 cm/a recharge	30 cm/a recharge
Main	**14.9** (3.9, 2031)	14.9	29.8
Mid-field	**21.4** (9.0, 1820)	21.5	43.0
Top-field	**31.0** (6.8, 652)	31.0	62.0

Note: Equivalent N leached calculated by multiplying average nitrate concentration by the equivalent volume of water recharged over one hectare in one year.