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Institute of Natural Resource Sustainability
William W. Shilts, Executive Director
ILLINOIS STATE GEOLOGICAL SURVEY
E. Donald McKay III, Interim Director
615 East Peabody Drive
Champaign, Illinois 61820-6964
217-333-4747
www.isgs.illinois.edu

Abstract

Accidental and incidental chemical releases of nitrogen-containing fertilizers occur at retail agrichemical facilities. Because contaminated soil may threaten groundwater quality, the facility may require some type of site remediation. The purpose of this study was to develop soil cleanup objectives (SCO) that are protective of groundwater quality in Illinois for unintentional releases of nitrogen-containing fertilizers Illinois. The concepts of the Soil Screening Levels of the U.S. Environmental Protection Agency were used to derive SCOs for nitrogen as nitrate and as ammonium. The Soil Screening Levels are based on the solute transport mechanisms of sorption, volatilization, and groundwater dilution, and the contaminant-specific groundwater cleanup objective used to derive the SCO. Because nitrate is relatively unreactive, only groundwater dilution could be taken into account in the derivation of a SCO. It may be preferable to use agronomic application rates to derive default nitrate SCOs. A default soil cleanup objective for N as ammonium was based on soil water dilution, and the amount of ammonium sorbed that is in equilibrium with the amount in soil water. The extent of sorption was measured using an uncontaminated, surface-soil sample (0 to 15 cm) of 10 different soil types that occur in Illinois and three gravel-fill samples from three different agrichemical facilities. There has been a concern, however, about the possibility that ammonium in soil and groundwater will rapidly convert to nitrate and nitrite. A field study was conducted following the release of anhydrous ammonia at an agrichemical facility. Soil-fill samples were collected for 488 days after the initial release. Extracts of the fill samples were used as a measure of water-soluble nitrate and ammonium. The results suggested that the rate of nitrification was limited, and may have been the result of alkaline conditions in the spill area. An analysis of groundwater data from three other agrichemical facilities suggested that ammonium in groundwater, following a major release, was relatively persistent. These studies did not support the assumption that nitrification in a spill scenario is rapid. In order to provide an environmentally conservative default SCO that takes into account the potential for long-term nitrification, one third of the US EPA's Lifetime Health Advisory Level for ammonium in drinking water (30 mg/L) was used to derive default SCOs for Class I and II groundwater. This approach can also be used to generate site-specific SCOs for nitrogen as nitrate and as ammonium for site cleanups.

Introduction

An agrichemical facility is a commercial agricultural chemical sales and customer application business. These facilities store, sell, mix, load, and apply fertilizers to fields for farmers. Bulk liquid and dry fertilizers are typically mixed, repackaged, or transferred from one container to another. Accidental and incidental chemical releases such as spills, tank leaks, hose breaks, and transport accidents, can result in significant contamination of surface soils. The gravel fill and soil at an agrichemical facility may contain nitrogen as nitrate (NO_3), ammonia (NH_3), and/or ammonium (NH_4). Because contaminated soil may threaten groundwater quality, the facility may require some type of site assessment and remediation. During a site assessment, typically soil cores are collected in various locations where chemical spills may have occurred such as areas where fertilizers are loaded and or mixed. Discrete samples are then collected from each core and analyzed to estimate the lateral and vertical distribution of the contaminants. However, before site remediation can proceed, some type of remediation or soil cleanup objective (SCO) must be determined. A common remediation approach used in Illinois for nitrogen-containing fertilizers is to remove a sufficient volume of contaminated material such that groundwater quality will not be significantly impacted by the leaching of nitrogen compounds from the remaining material. Therefore, when a volume of material contains the contaminant in concentrations greater than the SCO, the material is excavated. The selection of the SCO is critical to the assessment process because the value applied will determine how much material will be removed, which in turn can be a major factor contributing to the costs of site cleanup. In Illinois, once the volume of contaminated material is estimated, it is excavated then either land applied off-site or placed in a landfill. A guidance document for conducting site assessments at agrichemical facilities is available (Roy and Krapac, 2006).

The establishment of soil cleanup objectives for nitrogen-containing fertilizers is in a state of flux, and there are few objectives that have been published (Table 1). For example, the Minnesota Department of Agriculture, uses a soil cleanup goal for nitrate as nitrogen of 150 to 200 mg N/kg (MDA, 2003). The soil cleanup goal for total Kjeldahl nitrogen is 5,000 mg/kg for the upper 61 cm, and 1,000 mg/kg for depths greater than 61 cm. Total Kjeldahl nitrogen is a measure of the concentration of reduced forms of nitrogen, principally, ammonium and amino forms of organic

nitrogen. The Illinois Environmental Protection Agency currently uses a SCO of 200 mg/kg of nitrogen for Class I groundwater, and a combination of 200 mg/kg nitrate plus 600 mg/kg ammonium for Class II groundwater (T. C. Hornshaw, personal communication, 2004). The Illinois Department of Agriculture established an administrative rule [8 IAC. Part 259] that outlines the requirements and options for conducting site assessments at retail agrichemical facilities in Illinois. The rule provides information about soil cleanup objectives for the remediation of pesticide-contaminated soil at agrichemical facilities. Commercial fertilizers were recently added to the scope of the rule [415 ILCS 60/19.3]. Prior to this study, however, contained no criteria for the derivation of soil cleanup objectives for nitrogen-containing fertilizers. The purpose of this study was to derive Soil Cleanup Objectives (SCO) for nitrogen as nitrate and as ammonium that are protective of groundwater quality in Illinois.

Approach

The Illinois Department of Agriculture established an administrative rule [8 AC. Part 259] that provides default SCOs for pesticides and a method for developing site-specific SCOs. These SCOs were derived from the application of Soil Screening Levels (SSLs) developed by the U.S. Environmental Protection Agency (U.S. EPA, 1996a,b). The resulting soil-cleanup levels are based on the movement and chemical fate of a given contaminant in soil-water systems, viz.,

$$SCO = C_w[(K_{oc}f_{oc}) + (\theta_w + (\theta_a H))\rho_b^{-1}] \quad [1]$$

where,

- SCO = soil cleanup objective (mg/kg)
- C_w = target groundwater concentration (mg/L)
- K_{oc} = organic carbon partition coefficient (L/kg)
- f_{oc} = organic carbon content of soil (kg/kg)
- θ_w = water-filled soil porosity (L_{water}/L_{soil})
- θ_a = air-filled soil porosity (L_{air}/L_{soil})
- H = Henry's Law constant as K_H (Henry's Law constant as $atm \cdot m^3/mol$)
x [R (gas constant) x T (temperature as degrees Kelvin)]
- ρ_b = dry soil bulk density (kg/L),

$$\text{and } C_w = \{GW_{obj} \times [1 + (KdW/rLW)]\} \quad [2]$$

where,

GW_{obj} = groundwater cleanup objective (mg/L)

K = aquifer hydraulic conductivity (m/yr)

i = hydraulic gradient (m/m)

d = mixing zone thickness (m)

r = groundwater recharge rate (m/yr)

L = length of contaminated soil parallel to groundwater flow (m)

W = width of contaminated soil perpendicular to groundwater flow (m)

To derive SCOs for nitrogen fertilizers, Eq. [1] was recast in a different form. Neither nitrate nor ammonium volatilize from water significantly, and hence H was set to 0.0. Eq. [1] was derived for hydrophobic solutes whereas nitrate and ammonium are ions. However, because $K_{oc}f_{oc} = K_d$, a sorption constant, and $C_w \times K_d =$ the amount sorbed per mass of sorbent (x/m), then it follows that

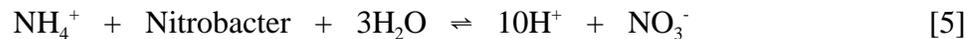
$$SCO = x/m + (C_w \theta_w)/\rho_b \quad [3]$$

In this study, the default values (see rule Roy and Krapac, 2006) for water-filled porosity (0.2 L/L) and for dry bulk density (1.6 kg/L) were used. The amount sorbed was evaluated at $C = C_w$. Hence, SCOs were calculated from

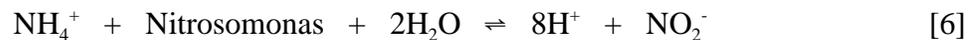
$$SCO \text{ (mg/kg)} = x/m \text{ (mg/kg)} + 0.125 \text{ (L/kg)} C_w \text{ (mg/L)} \quad [4]$$

Potential for Nitrification of Ammonium

The application of these SCOs, however, is complicated by the potential for ammonium to be converted into either nitrate or nitrite. Bacteria called Nitrobacter can convert ammonium into nitrate:



Ammonium can be converted to nitrite via Nitrosomonas bacteria, although the reaction is slow.



Once ammonium is sorbed via ion exchange by colloidal materials, it is no longer available to leach into groundwater. However, if the bound ammonium is desorbed back into solution, it may be subject to conversion to nitrate. Buss et al. (2003) generalized that if the movement of ammonium is not limited by cation exchange such as would be expected in sand and gravel aquifers, that nitrification can be the major process controlling the fate of ammonium in groundwater. The soil chemistry of nitrogen in agricultural fields has been studied extensively, but the potential for aerobic and anaerobic nitrification of ammonium in a chemical spill scenario has not been studied as well. Laboratory studies were conducted in this investigation to determine the ammonium sorption characteristics of selected soils commonly found in Illinois.

Field Study

On November 13, 2003, there was an accidental release of about 8,000 gallons of anhydrous ammonia at an agrichemical facility in east-central Illinois. The identity of the site is confidential. A worker was unloading a semi-truck when a transfer hose ruptured from a bulk tank. A large plume of ammonia gas formed, and residents in the area were evacuated. The local volunteer fire department sprayed water in the spill area before the residents were allowed to return. The solubility of ammonia in water is about 34% at 25°C. Because of the water applied to the spill area, some of the ammonia was likely converted to ammonium hydroxide which can dissociate resulting in a relatively alkaline area:



The dissociation constant for ammonium hydroxide is 1.77×10^{-5} at 25°C. At a pH of 9.25, about 50% of the chemical is as dissolved ammonia (NH_3), and 50% occurs as the ammonium cation (NH_4^+).

The areal distribution of ammonia was easily defined by the odor of the chemical at the surface. The odor threshold for ammonia is about 18 $\mu\text{g}/\text{L}$ in air. Because the spill was documented and the area impacted was relatively small, it appeared that the facility could be used as a study site to provide

information on the biogeochemical fate of nitrogen in the context of establishing soil cleanup objectives.

The goal of the field study was to provide information on the extent and rate of nitrogen conversions in a chemical spill scenario. It was not within the scope of the effort to delineate the 3-dimensional distribution of ammonium in the spill area for the purpose of site remediation. The results of this study may, however, provide much of the information needed to form a remedial plan to remove the contaminated fill and soil.

Methods and Materials

Laboratory Study

The sorption of ammonium was measured using an uncontaminated, surface soil (0 to 15 cm) sample collected from 10 different soil types that occur in Illinois and three uncontaminated gravel-fill samples from three different agrichemical facilities (Table 2). Of the ten soil types, six cover areas greater than 40,000 ha in Illinois. Five are Mollisols (prairie soils), two are Alfisols (forest soils), and two are Entisols (lacking a well-developed B horizon). These ten soils represent about 22.4% of the acreage in Illinois. The three fill samples were described in Roy et al. (1999). The sorption of ammonium by soil and fill samples was measured using the batch equilibrium procedures given in Roy et al. (1992). In brief, the soil samples were air dried and sieved to pass a 2-mm sieve. A series of soil:solution ratios ranging from 1:4 to 1:500 (mass/volume) were made in 125-mL polyethylene bottles, and mixed with 13 solutions containing $\text{NH}_4\text{-N}$ concentrations ranging from 0 to 400 mg $\text{NH}_4\text{-N/L}$. Care was taken to minimize head space within the bottles. The solutions were mixed in a shaking water bath at 25°C for 24 hours. Kinetic data (not shown) indicated that 24 hours was sufficient to attain equilibrium. The liquid phase was then separated using a temperature-controlled centrifuge. Water-soluble ammonium was determined using an ion selective electrode (APHA, 1992). The amount of ammonium sorbed was calculated by difference between the initial solute concentration and the final solute concentration after mixing. In this study, the sorption isotherms were used to calculate the amount of ammonium sorbed that is in equilibrium with the amount of ammonium in solution, C_w .

Soil texture was determined using the pipette method of Indorante et al. (1990) on the < 2-mm fraction of each soil sample. Cation exchange capacity (CEC) was determined using the method of Sumner and Miller (1996) in which the CEC is measured at the reaction pH of the soil using an unbuffered NH_4Cl solution. Surface area was measured by nitrogen sorption using a Monosorb single-point surface analyzer (Quantachrome Instruments, Boynton Beach, Florida). Organic carbon was determined using a CE 440 CHN Elemental Analyzer (Exeter Analytical, Inc, North Chelmsford, Massachusetts). This was a diverse group of sorbents (Table 2). Reaction pH, for example, ranged from 4.8 to 8.5. The cation exchange capacity ranged from 0.04 meq/100 g (Fill 28-4) to 21.8 meq/100 g (Drummer silty clay loam).

Field Study

In the field study, gravel fill-soil cores were collected at the facility during eight sampling efforts corresponding to 19, 98, 137, 208, 251, 347, 396, and 488 days after the anhydrous spill. During each effort, the soil cores were collected by driving 60-cm long Shelby tubes into the surface using a fence-post driver. The tubes were sealed immediately after they were pulled out of the ground and stored in a cold room at 4° C. The tubes were cut open using a plasma torch, and sampled at 5- or 10-cm intervals. Each of the samples removed from the cores was extracted with deionized water to provide a measure of water-soluble ammonium and nitrate. A 20-g sample from the cores was mixed with 130 mL of deionized water and mixed for 24 hours using an NBS Rotating Tumbler. The suspensions were then centrifuged at 5000 rpm for 15 minutes. The liquid phase was characterized in terms of pH, specific conductance (EC), and ammonium using specific ion electrodes. The solution concentrations of chloride, nitrate, and sulfate were determined using a Dionex 2110i ion chromatograph, following U.S. EPA Method 300.0 (Pfaff et al., 1991).

Results and Discussion

Laboratory Study

Nitrogen as Nitrate Default SCO

Nitrate is a monovalent anion, and as such, is not significantly sorbed by the dominantly negatively charged soils. Nitrate is generally considered to be mobile in saturated soil-water systems. Consequently, if we set $x/m = 0$ in Eq. [4], then

$$\text{SCO (mg/kg)} = 0.125 \text{ (L/kg)} C_w \text{ (mg/L)} \quad [5]$$

The groundwater quality standard for nitrogen as nitrate in Class I groundwater is 10 mg N/L [35 IAC. Part 620]. Using the default groundwater dilution factor of 30 (see Roy and Krapac, 2006), and the nitrate standard as the groundwater objective ($G_{w_{obj}}$) results in a target groundwater concentration (C_w) equal to 300 mg/L (10 mg/L x 30 dilution factor), and then the default nitrate SCO = 37.5 mg N/kg via Eq. [5]. For Class II groundwater, $G_{w_{obj}}$ is 100 mg N/L and C_w equals 3,000 mg N/L. Hence, the default nitrate SCO for Class II groundwater would be 375 mg N/kg.

Nitrogen as Ammonium Default SCO

Ammonium was sorbed by the 10 soil and three gravel fill samples (Fig. 1, for example) in the relative order Drummer > Flanagan > Tama > Muscatune > Fayette > Catlin, Elsay > Cisne > Fill 38-1 > Pike County 2 > Fill 7-1, Plainfield, Fill 28-4. The sorption isotherm data for seven of the sorbents were best described by the Freundlich Equation:

$$x/m = K_f C^{1/n} \quad [6]$$

where x/m is mass of sorbate per mass of sorbent

K_f = the Freundlich constant (L/kg)

C = the equilibrium concentration of ammonium (mg/L)

$1/n$ = the Freundlich exponent

The sorption of ammonium for the Fayette, Plainfield, Pike County 2 soils and all the gravel fill samples conformed best to a Langmuir-type equation, viz.,

$$x/m = K_L MC / (1 + K_L C) \quad [7]$$

where x/m is mass of sorbate per mass of sorbent

K_L = the Langmuir constant (L/mg)

C = the equilibrium concentration of ammonium (mg N/L)

M = the Langmuir capacity (mg N/kg)

For each soil sample, the isotherm equation that best fit the data (Table 3) was used to calculate the amount sorbed (x/m) in equilibrium with the target groundwater concentration, (C_w) via Eq. [4]. We adopted the concepts used in developing the US EPA Lifetime Health Advisory Level (LHAL) for ammonium in drinking water of 30 mg N/L to determine an appropriate ammonium groundwater cleanup objective (GW_{obj}). Health Advisories are estimates of the concentration of a chemical in drinking water that is not expected to cause any adverse noncancer effects after a lifetime of exposure. In order to provide a level of safety, and to provide a target groundwater concentration (C_w) that would be representative of the concentrations used to generate the sorption constants for the 13 soil materials, one third (10 mg N/L) of the LHAL was chosen as the GW_{obj} . Using the default groundwater dilution factor of 30 resulted in a target groundwater concentration (C_w) of 300 mg N/L. The resulting soil cleanup objectives for Class I groundwater for the 13 soil materials via Eq. [4] ranged from 94 to 3,001 mg N/kg (Table 4). For Class II groundwater, the C_w (1,500 mgN/L) was determined by increasing the GW_{obj} by a factor of 5 (as per the statute) and using the default groundwater dilution factor. The SCOs for Class II groundwater ranged from 252 to 7,344 mg N/kg (Table 4). Because these soils are representative of the range of soil properties in Illinois, the SCOs are also representative of the likely range of site specific SCOs that would likely be determined at most agricultural facilities.

Relationship Between Soil Cleanup Objectives for Ammonium and Soil Properties

In this investigation we tried to predict soil cleanup objectives based on soil properties such as cation exchange capacity, surface area, organic carbon, clay content, and pH. Potential relationships between the ammonium soil cleanup objectives and soil properties were characterized by the application of statistical models: linear [8], quadratic [9], and cubic [10]:

$$\text{SCO} = aS + b \quad [8]$$

$$\text{SCO} = aS^2 + bS + c \quad [9]$$

$$\text{SCO} = aS^3 + bS^2 + cS + d \quad [10]$$

where S represents the value of a soil property, and a, b, c, and d are empirical constants.

In addition, a multiple regression was performed using all of the soil properties to predict SCO. A regression coefficient was calculated for each model and the significance of that property to predict SCO was determined.

The SCO calculated for each of the 13 soils was positively correlated with cation exchange capacity (CEC), clay content, surface area, and organic carbon (Table 5). For many soils, a direct relationship exists between CEC and organic matter (measured as organic carbon). Stevenson (1982) summarized that organic matter alone can account for 25 to 90% of the CEC of surface soil samples. A correlation between CEC and SCO was expected because the sorption of ammonium by soil is generally dominated by a cation exchange mechanism that is reflected by the CEC of the material. As shown in Fig. 2, CEC alone could account for about 91% of the variance in the Class I groundwater SCO using a cubic model:

$$\text{SCO} = 0.36(\text{CEC})^3 - 8.93(\text{CEC})^2 + 143(\text{CEC}) + 103 \quad [11]$$

The cubic model for all soil properties provided the best estimate of an individual soil property to predict SCO (Table 5). For example the linear model for CEC yielded a correlation coefficient (r^2) of 0.86 while the cubic model's coefficient was 0.91 for Class I groundwater. Clay content and CEC were the best (largest coefficients) soil properties in predicting SCOs. When all of the soil properties were considered simultaneously, the application of multiple regressions resulted in slightly larger coefficients (0.94 and 0.96) than those determined using only CEC or clay content at $P = 0.01$.

Using the models for Class I groundwater in Table 6, SCOs were calculated for the 10 soils using either clay, CEC, or all soil properties (multiple) (Fig. 3). Generally, using either the clay or CEC models resulted in the smallest SCOs. For 40% of the soils, the clay model predicted smaller SCOs than either the CEC or multiple models. Similarly, for 40% of the soils, the CEC model predicted smaller SCOs than either the clay or multiple models. The average difference in SCOs between all three models was 259 mg/kg. Catlin exhibited the greatest difference in predicted SCOs with the CEC model estimating a SCO 517 mg/kg greater than the clay model.

These results suggest that either the CEC or the clay content of a soil, in lieu of making formal sorption measurements, could be used to predict site-specific soil ammonium cleanup objectives. These soil properties can be measured using standard protocols, and they are commonly preformed by accredited soil testing laboratories (as defined in [8 IAC. Part 259]) allowing these data to be accurately and rapidly collected. The use of multiple soil properties to determine a site-specific ammonium SCO can slightly increase the reliability (greater regression coefficient) to predict sorption, but the costs associated with determining each soil property may outweigh the benefits. Thus, there is the potential to determine a site-specific ammonium SCO based on the local soil CEC and/or clay content and the cubic equations provided in Table 5 rather than using default SCOs. In general, if the CEC or clay content of a site soil is greater than 10 cmol/kg or 21.4%, respectively, and if it is underlain by Class I groundwater, it could be advantageous to the site owner to determine a site-specific SCO rather than to use the default SCO. If a site is underlain by Class II groundwater, a CEC and/or clay content greater than 8.7 cmol/kg and 21.8%, respectively would likely yield a site-specific ammonium SCO that is greater than the default. Using Table A-3 in the appendix and a soil

survey map, an initial estimate of the site-specific CEC can be made with the subsequent analytical determination of soil samples for the CEC to better define the range in CEC at the site.

Soil Sample Collection Within the Spill Area

As shown in Fig. 4, soil cores were collected in four general areas; the spill area where the hose ruptured, and in three areas (nests 1, 2, 3) away from the spill area. The gravel layer at the sample locations varied in thickness from 5 to 11 cm with a mean of 7.3 ± 1.5 cm. The material below the gravel layer was a soil-fill mixture. Some of the mixed layers also contained pieces of concrete, clay tile, metal bolts, slag, and gravel. The texture of selected fill samples ranged from silty clay loam to loamy sand; the surface samples were typically sandier (Table 6). The organic carbon content ranged from 0.49 to 3.09% with a mean of $1.59 \pm 0.99\%$. Roy et al. (1994) reported that the mean organic carbon content of 27 fill samples collected at nine different agrichemical facilities was 0.92%. The surface area of the samples ranged from 2.7 to 16.9 m²/g. Roy et al. (1994) also reported that the mean surface area of the 27 fill samples was 5.57 m²/g.

As shown in Fig. 5, the distribution of water-soluble ammonium in the spill area was extremely variable. For example, 19 days after the release, ammonium was detected in concentrations ranging from 757 to 2,168 mg N/kg. Background cores collected away from the spill (G3 and G15) yielded <0.3 mg N/kg. Similarly, the concentrations of water-soluble nitrate in the upper 25 cm of the profiles did not seem to be related to the time of sampling since the initial spill (Fig. 6). The concentration of nitrate in the extracts, however, appeared to increase with time in samples collected at depths greater than about 30 cm. For example, the amount of nitrate at 50 cm 19 days after the spill was about 0.84 mg N/kg. After 488 days, samples collected in the spill area yielded about 517 mg N/kg at the same depth. An alternative approach of presenting the data is given in Fig. 7. Samples collected at the 32-cm depth yield increasingly greater amounts of nitrate but less ammonium during the 488-day study. The increase in the amount of water-soluble nitrate could have been the product of nitrification that converted the ammonium to nitrate. When the ratio of ammonium to nitrate was plotted as a function of depth for each sampling date, the ratio of the two

forms of nitrogen increased in the lower (deeper than 30 cm) part of the profile (Fig. 8) since the initial release. The increase in the NH_4/NO_3 ratio could have resulted from nitrification.

The interpretation of the field data, however, is complicated because the agrichemical facility was operational during the project period. The cores collected 19 and 98 days after the initial release were collected during the winter months of 2003-2004 when activities at the facility were minimal. However, the cores collected 137 and 208 days since the spill were collected when fertilizers were being applied to fields. Smaller, incidental spills of anhydrous ammonia could have added new ammonium to the study area. For example, when the 488-day cores were collected during the spring of 2005, anhydrous ammonia was being pumped from a semi-truck into the ammonia tank in the study area. When the hose was disconnected, anhydrous ammonia was released near the original spill, resulting in two small areas that were contaminated. Note the apparent increase in ammonium in the surface sample of the 488-day core (Fig. 5). Moreover, not all the nitrate nor ammonium may have been derived from the initial ammonia release. Dry fertilizers were stored in open stalls next to the spill area. The nitrogen-containing fertilizers were urea 46-0-0 ($\text{CO}(\text{NH}_2)_2$), and diammonium phosphate (DAP 18-46-0). The facility also stored potassium chloride (potash) and calcium phosphate (Triple Superphosphate). The dry fertilizers could have been blown or had swept off the loading pad on to the gravel area. Therefore, data interpretations must be made cautiously because the field site was an open system with potential undocumented inputs of nitrogen in addition to the initial release in November of 2003.

Water-soluble ammonium was detected in the extracts of core samples collected in nests 1, 2, and 3. Ammonium was detected throughout the entire profile in nest 1 (Fig. 9), whereas it appeared to be largely confined to the upper 25 cm in nests 2 (Fig. 10) and 3 (not shown). There was no obvious relationship between the ammonium concentrations and the time since the release. The lack of a relationship may have been the result of field-scale variability or the re-distribution of ammonium by rainfall and infiltration that had occurred at the site since the spill in November of 2003. If the average concentration of ammonium in the spill area and all three nests are plotted as a function of depth (Fig. 11), the spill area was the most contaminated, whereas nests 2 and 3 contained less

ammonium because the cores were collected outside the area of the original spill.

Nitrate was detected in the extracts of the samples collected in nests 1, 2, and 3. For example, nitrate concentrations were largest in the upper part of the profile in nest 2 and 3 (Figs. 12 and 13), which suggested nitrification, but the distribution of nitrate in the extracts did not appear to depend on when the core sample was collected. If the NH_4/NO_3 ratio for the 32-depth sample is plotted as a function of time for each nest (Fig. 14), the results suggested there had been little change in the proportion of nitrate relative to ammonium.

The relative persistence of ammonium with respect to nitrification may have been related to the pH of the fill-water system. The optimum pH range for *Nitrosomonas* to convert ammonium to nitrite is 7.8 to 8.0, and the optimum pH range for *Nitrobacter* to transform nitrite to nitrate is 7.3 to 7.5. Hence, the pH range of about 7.3 to 8 would be most favorable for nitrification. The pH of the extracts of samples collected in the spill area, however, were greater than this optimal range. For example, the pH of samples collected at a depth of about 32 cm were generally greater than 8.5 (Fig. 15) whereas the pH of extracts of samples collected from nests 2 and 3 were less than the mean pH of the spill area and nest 1. The greater pH values in the spill area and nest 1 matched the largest amounts of ammonium (see Fig. 16), whereas the mean pH of all the cores collected in nests 2 and 3 were comparable with that of the background pH.

It was expected that any nitrate produced by nitrification would be subject to downward leaching by infiltration. The fill materials were unsaturated during each sampling event. Roy et al. (1995) reported that the mean hydraulic conductivity of gravel fill at four agricultural facilities ranged from 1.4×10^{-4} to 1.7×10^{-5} cm/sec. In addition to ammonium and nitrate, the extracts contained relatively large amounts of chloride (Fig. 17). As shown, the mean chloride content in the extracts from the spill area and nests was greater than that for the off-site field that served as the control. The presence of relatively large amounts of chloride in the gravel fill and throughout the soil profile suggested that water infiltrated vertically through the gravel fill. Because chloride is a mobile and relatively unreactive anion, we anticipated smaller concentrations near the surface resulting from flushing

during rainfall events. However, as noted earlier, dry fertilizers could have been blown or swept off the loading pad on to the gravel area. For example, when the 251-day cores were collected, granular potassium chloride was present at the surface near core 16. Therefore, the persistence of chloride in the fill probably resulted from incidental releases of potassium chloride throughout the project period.

Estimated Half-Life of Ammonium in Groundwater

Buss et al. (2003) concluded that there are very few studies on the rate of biological nitrification in soil materials other than those for agricultural purposes. Based on studies of unsaturated subsoils and sand and gravel aquifers, they estimated that the half-life of ammonium under aerobic conditions was between 1 and 6 years. In an attempt to evaluate the relative persistence of ammonium in groundwater, groundwater monitoring data from three case studies were examined.

Case Study Number 1

In May of 1995, there was a catastrophic release of about 100,000 to 150,000 gallons of storm water containing 3 to 4% ammonium nitrate from an unlined basin at an agrichemical facility located in Illinois. It was estimated that about 25,000 to 50,000 pounds of nitrogen was introduced into the soil and groundwater. In 1996, the facility entered the Illinois EPA's Site Remediation Program. An attempt was made to remove as much of the nitrogen-enriched soil as possible. Nine groundwater monitoring wells were installed on-site, and groundwater samples had been collected four times a year for nine years.

Well MW-5 was located about 650 feet down gradient of the spill area. Relatively large concentrations of nitrate and ammonium were detected in the well samples from 1996 to about the end of 2000, 70 months after the spill (Fig. 18). Nitrite was detected sporadically during the same period, ranging from 0.18 to 7.20 mg/L. The pH of the water samples again indicated favorable conditions for nitrification. The ratio of ammonium to nitrate, however, showed no consistent trend. The ratio varied from 0.03 (69 months after the spill) to 0.82 (36 months after the spill). Therefore

it appeared that nitrification was a minor mechanism responsible for converting ammonium to nitrate. It appeared that the gradual decrease in concentration could be characterized as having a half-life of about 3 years. The relatively slow reduction in ammonium concentrations was likely the result of the gradual decrease in ammonium available to move from the initial spill, groundwater dilution of the ammonium, and nitrification as the groundwater recovered from the catastrophic release.

Case Study Number 2

A site assessment detected nitrogen-contaminated soil at an agrichemical facility in Wisconsin. A total of 1,398 m³ of soil were excavated from three areas from a depth ranging from 0.9 to 4.6 m. A soil cleanup objective of 150 mg N/kg was applied. Ten monitoring wells were then installed and shallow groundwater samples were collected from September of 1998 to November of 2002. Well MW-9 was about 61 m down gradient from the largest area of contamination. The concentration of ammonium decreased during the four-year period to concentrations that were less than 10 mg N/L (Table A2). The ratio of NH₄/NO₃ decreased from 6.7 to 0.09 from February 2000 to November 2002. Unlike Case Study 1, no nitrite or pH data were available, and the decrease in ammonium concentrations may have been the result of several factors in addition to nitrification such as the removal of the contaminated source areas. In any event, the reduction in ammonium yielded a half-life of about 0.7 year.

Case Study Number 3

In 1978, a leak occurred from a liquid fertilizer storage pit at an agrichemical facility in Illinois. The pit was about 2.7 m deep and the depth to groundwater was between 0.6 and 1.2 m. Water samples collected from 1994 to 2002 from groundwater monitoring wells contained relatively large amounts of ammonium and nitrate. For example, well MW-3 was about 46 m from the edge of the source pit. A groundwater sample collected 16 years after the leak contained 1,150 mg N/L ammonium, and 630 mg/L nitrate. A sample collected 7.8 years later contained 520 mg N/L ammonium and 110 mg N/L as nitrate. The ratio of NH₄/NO₃ varied from 1.8 to 4.7, and did not change in a consistent pattern to suggest nitrification. It appeared that the ammonium in the shallow groundwater was relatively persistent.

Conclusions

Default soil cleanup objectives for N as nitrate depends only on the type of groundwater present at the facility. A default soil cleanup objective for N as nitrate can be based on essentially only groundwater dilution. Using the water quality standard of 10 mg N/L, the default nitrate SCO is 38 mg N/kg for Class I groundwater, and 375 mgN/kg for Class II groundwater. When compared with soil cleanup objectives currently applied in other states (Table 1) an SCO of 38 mg N/kg for nitrate is comparatively conservative. The application of nitrogen fertilizers in Illinois from 2000 to 2003 averaged about 179 kg/ha (160 lbs per acre) (IASS, 2005) which is equivalent to 79.8 mg N/kg. The use of an agronomic-based SCO for Class I groundwater quality is relatively consistent in magnitude with other States. Note for example, that South Dakota applies 80 mg/kg as nitrate-N as a SCO (Table 1).

A default soil cleanup objective for N as ammonium can be based on the amount of ammonium sorbed in equilibrium with the amount in groundwater in addition to groundwater dilution. There has been a concern, however, about the possibility that ammonium in soil and groundwater will rapidly convert to nitrate and nitrite via nitrification which has prompted some water quality regulations to combine ammonium and nitrate together as total nitrogen. Such conversions occur in typical agricultural applications, but they may not occur as rapidly in spill scenarios. This study has shown that, 488 days after a major spill of anhydrous ammonia, the ammonium in aqueous extracts of fill samples collected in the spill area appeared to be relatively persistent. There was some evidence that nitrification occurred; the amount of water-soluble nitrate increased in some profiles. However, it also appeared that the relative persistence of ammonium in the soil may have been related to the alkaline conditions generated by the hydrolysis of ammonia. Areas adjacent to the spill also yielded relatively large amounts of nitrate, but there was no obvious relationship between nitrate concentrations and the time since the initial release of anhydrous ammonia.

Groundwater samples were not collected during this study, but an analysis of groundwater data collected at three other agricultural facilities suggested that ammonium in shallow groundwater can be relatively persistent. In one detailed study in which both nitrate and nitrite was measured in

groundwater samples collected downgradient from a major release of nitrogen-rich waste water, it appeared that the rate of ammonium dissipation could be described as having a half-life of about 3 years. These case studies, taken together with our spill study all suggest that ammonium does not quickly convert to nitrate in a spill scenario. It could be argued that there will always be a potential for some portion of the ammonium in soil to be transformed into nitrate. As discussed above, the LHAL of ammonium in drinking water is 30 mg N/L, which could be used as the groundwater objective. However, in order to be environmentally conservative to reflect the possibility of slow nitrification, we recommend that the default SCOs be based on one third of the LHAL. Default SCOs (Table 7) were calculated using the relationship between CEC and SCO. The cubic model and coefficients given in Table 5, and the midpoint of each range in CEC values given in Appendix A-3 were used to calculate a SCO for each CEC range, and for both Class I and Class II groundwater. These default objectives are protective of groundwater quality, and environmentally conservative because they consider the potential for long-term nitrification and the movement of ammonium in the environment.

When the proposed default SCOs for nitrate are applied to the spill facility, it appeared that more than 90% of the 251 sample yielded more water-soluble nitrate than the agronomic-based SCO of 80 mg N/kg. As shown in Fig. 19, the use of 80 mg N/kg yielded results that were comparable to the 38 mg N/kg SCO. If the 375 mg N/kg SCO is applied for Class II groundwater, then more than half of all the samples exceeded the SCO. Of the samples that exceeded the 80 mg/kg SCO, it appeared that excessive nitrate was detected in all of the areas studied (Fig. 20). The application of the 375 mg N/kg SCO revealed that the spill area and nest 2 contained the greatest amounts of water-soluble nitrate. The distribution of nitrate in excess of each SCO was more uniformly distributed to a depth of about 56 cm than ammonium. Therefore, the entire area evaluated in this study would need to be excavated to a least a depth of about 60 cm because of the nitrate content—not the ammonium content—of the samples.

Site-specific SCO for ammonium can be established by measuring ammonium sorption using the standard U.S. EPA method (Roy et al., 1992) with uncontaminated soil and fill samples collected

at a given agricultural facility. Soil samples could also be collected adjacent to the facility to avoid contaminated areas. An alternative to measuring ammonium sorption may be to measure CEC as a method to calculate site-specific SCOs. A summary of approximate CEC values for soils in Illinois is provided in the Appendix (Table A-3). This summary can be used as a guide in determining a site-specific SCO. For example, the CEC values of the fill samples (Table 6) collected at the spill site ranged from 2.0 to 23.8 cmol/kg at depths of 7 to 45 cm. The mean CEC is 10.7 and the median value is 11.0. If the anhydrous ammonia spill took place over Class I groundwater, the CEC values indicate that the default SCO is 1,000 mg N/kg (from Table 7). If the ammonium SCO of 1,000 mg N/kg is applied, then of the 251 soil samples collected during the 488-day study, 100 samples contained ammonium in excess of this SCO. Of these 100 samples, 63% of them were collected in the spill area, while 35 samples were collected in nest 1. The distribution of these 135 samples as a function of depth is shown in Fig. 21. The application of the 1,000 mg N/kg SCO implied that the spill area and nest 1 should be excavated to a depth of about 60 cm. It appeared that the areas corresponding to nests 2 and 3 would not require removal. If the CEC values of the soil material samples collected at our spill study site (given in Table 6) are used to calculate a site-specific ammonium SCO, and if the site is over Class I groundwater, the cubic model in Table 6 yielded the mean SCO would be 1,285 mg N/kg which is larger than the default SCO.

In general, if the CEC or clay content of a site soil is greater than 10 cmol/kg or 21.4% respectively and is underlain by Class I groundwater, it would likely be beneficial to the site owner to determine a site-specific SCO rather than to use the default SCO. If a site is underlain by Class II groundwater, a CEC and clay content greater than 8.7 cmol/kg and 21.8% respectively would likely result in a site-specific ammonium SCO that is greater than the default. Fig. 22 summarizes the process and options available for deriving soil cleanup objectives for nitrogen as nitrate or ammonium in soil materials that are protective of groundwater quality.

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Table 1. Summary of soil cleanup objectives for ten States (as of 2005).

State, contact, and rationale	Soil Cleanup Objectives
Illinois (T. Hornshaw) Based on the TCLP and the Class I Groundwater standard for nitrate (10 mg/L)	200 mg total ¹ N/kg for Class I groundwater (200 mg NO ₃ + 600 mg NH ₄) per kg for Class II groundwater
California (A. Terrell) Based on “professional judgement”	100 mg/kg total N for groundwater depth of 10 feet 250 mg/kg total N for groundwater depth of 30 to 50 feet “Can be site-specific”
Florida (R. Register) Based on risk assessment	No values available. “Site specific” Ammonia based on inhalation from soil surface
Minnesota (published) Based on “professional judgement”	150 to 200 mg nitrate-N/kg. 5,000 mg/kg Total Kjeldahl N for depth of 0 to 2 feet 1,000 mg/kg Total Kjeldahl N for depth greater than 2 feet
Missouri (C. Cady) Based on “professional judgement”	“No set numbers” Ammonia based on “direct exposure”
Nebraska (D. Miesback) Based on agronomic rates	100 mg/kg total N. “Can be site specific” Nothing for ammonium
North Carolina (D. Hardy)	No objectives have been proposed
South Dakota (K. Reitsmas) Based on “average soil tests”	80 mg nitrate-N/kg. Nothing for ammonium
Texas (A. Strahl) Based on risk assessment	19 to 1,900 mg nitrate-N/kg. Depends on the volume of contamination and groundwater type. Nothing for ammonium
Wisconsin (R. Graham) Based on agronomic rates plus “professional judgement”	100 mg/kg total N. “Can be increased or decreased” Nothing for ammonium

¹Nitrate, ammonium, and nitrite are converted to nitrogen, then added together as “total N.”

Table 2. Properties of the soil and gravel fill samples used in determining ammonium soil cleanup objectives.

Soil	Sand (%)	Silt (%)	Clay (%)	Texture	pH	CEC (meq/100g)	Surface area (m ² /g)	Organic Carbon (%)
Catlin	11.0	69	21	Silt loam	6.5	14.9	14.8	4.22
Cisne	13.8	70.8	14.8	Silt loam	5.9	8.0	6.1	1.14
Drummer	17.9	49.5	31.6	Silty clay loam	6.2	22.8	22.1	2.20
Elsah	20.3	66.4	13.3	Silt loam	6.4	7.3	3.25	1.44
Pike County 1	0.3	77.4	22.3	Silt loam	5.8	6.6	13.0	1.23
Flanagan	5.4	65.2	29.4	Silty clay loam	6.0	17.5	12.6	2.62
Muscatune	1.0	74.7	24.3	Silt loam	6.3	12.9	4.52	2.53
Plainfield	94.0	4.62	1.44	Sand	4.8	0.9	0.91	0.47
Pike County 2	0.4	84.6	15.0	Silt loam	6.4	6.4	8.49	1.47
Tama	0.8	72.9	26.2	Silt loam	6.4	13.3	4.49	2.43
Fill 7-1	ND ¹	ND	ND	Sandy loam	8.2	0.4	2.2	1.69
Fill 28-4	N D	ND	ND	Sandy loam	8.5	0.04	2.2	0.49
Fill 38-1	ND	ND	ND	Loam	7.7	2.1	6.6	1.06

¹Not determined.

Table 3. Summary of isotherm constants¹ used to calculate the default soil cleanup objectives.

Soil or fill	K_f (L/kg)	1/n	r^2	K_L (L/mg)	M (mg/kg)	r^2
Catlin	21.3	0.667	0.986	-	-	-
Cisne	17.7	0.606	0.987	-	-	-
Drummer	132	0.546	0.994	-	-	-
Elsah	29.4	0.610	0.993	-	-	-
Pike County 1	-	-	-	0.015	1,250	0.996
Flanagan	63.3	0.595	0.993	-	-	-
Muscataune	72.9	0.523	0.993	-	-	-
Plainfield	-	-	-	0.018	98.0	0.989
Pike County 2	-	-	-	0.020	370	0.943
Tama	41.6	0.627	0.991	-	-	-
Fill 7-1	-	-	-	0.022	104	0.983
Fill 28-4	-	-	-	0.016	67.6	0.987
Fill 38-1	-	-	-	0.015	714	0.996

¹ K_f = the Freundlich constant, 1/n = the Freundlich exponent, r^2 = correlation coefficient, K_L = the Langmuir constant, and M = the Langmuir capacity.

Table 4. Default soil cleanup objectives (SCO) for nitrogen as ammonium based on the soil and fill samples, and the type of groundwater.

Soil or fill	SCO (mg N/kg)	
	<i>Class I</i>	<i>Class II</i>
Catlin silt loam	994	2,985
Cisne silt loam	599	1,676
Drummer silty clay loam	3,001	7,344
Elsah silt loam	989	2,733
Pike County 1 silt loam	1,060	1,384
Flanagan silty clay loam	1,921	5,099
Muscataune silt loam	1,473	3,528
Plainfield sand	120	282
Pike County 2 silt loam	317	546
Tama silt loam	1,521	4,266
Fill 7-1	128	288
Fill 28-4	94	252
Fill 38-1	624	871
Mean	987	2,404
Standard Deviation	840	2,195
Median	989	1,676

Table 5. Regression coefficients and coefficients for various models used to predict ammonium SCOs from various soil properties, and a groundwater objective of 10 mg N/L.

Soil Parameter	Class I Groundwater			Class II Groundwater		
	Regression Coefficient			Regression Coefficient		
	Model Coefficients			Model Coefficients		
	Linear Model	Quadratic Model	Cubic Model	Linear Model	Quadratic Model	Cubic Model
Cation Exchange Capacity	0.86* a=109, b=40.1	0.89* a=3.07, b=45.9, c=210	0.91* a=0.36, b=-8.93, c=143, d=103	0.91* a=293 b=-142	0.93* a=7.03, b=149, c=246	0.93* a=0.52, b=5.30, c=163, d=231
Clay Content	0.76* a=81.5, b=-425	0.89* a=3.36, b=-33.6, c=289	0.94* a=0.28, b=-10.3, c=142, d=-59.6	0.70* a=203, b=-1068	0.83* a=8.76, b=-96.8, c=794	0.89* a=0.80, b=-30.9, c=413, d=-216
Surface Area	0.56*	0.60*	0.67*			
pH	0.10	0.39	0.38			
Organic Carbon	0.27	0.50*	0.57*			
Multiple	0.94* SCO = 132 x CEC - 145 x pH - 7.21 x SA - 29.0 x OC + 26.4 x clay + 726			0.96* SCO = 466 x CEC - 212 x pH - 79.6 x SA - 57.4 x OC - 6.22 x clay + 1100		

*Correlation is significant at the 0.05 level

Table 6. Characterization of the soil-fill samples collected at the agrichemical facility where the ammonia spill occurred.

Sample	Location	Depth (cm)	Sand (%)	Silt (%)	Clay (%)	Texture	pH	CEC (cmol/kg)	Surface Area (m ² /g)	Organic Carbon (%)
G4-04-1	Spill Area	9.5	49.5	34.0	16.5	loam	8.89	11.0	9.59	1.16
G4-04-2	Spill Area	21.6	18.3	50.1	31.6	silty clay loam	8.66	18.1	16.9	1.71
G4-04-6	Spill Area	44.6	9.1	64.2	26.7	silt loam	8.51	23.8	8.3	1.76
G9-04-1	Nest 2	7.0	73.9	18.1	8.0	sandy loam	8.50	4.3	10.0	0.69
G9-04-2	Nest 2	16.6	24.8	45.1	30.1	clay loam	7.37	8.4	11.1	0.19
G9-04-6	Nest 2	36.6	21.9	49.9	28.2	clay loam	6.91	12.4	16.4	1.89
G10-04-1	Nest 3	5.0	79.0	14.7	6.3	loamy sand	9.07	2.4	6.8	0.70
G10-04-2	Nest 3	12.6	31.9	45.0	23.1	loam	7.58	11.6	9.7	2.84
G10-04-6	Nest 3	32.6	15.6	57.9	26.5	silt loam	7.45	15.6	13.7	3.09
G13-04-1	Nest 1	11.0	76.8	17.8	5.4	loamy sand	9.03	2.0	2.7	0.49
G13-04-2	Nest 1	24.6	40.6	39.4	20.1	loam	9.26	7.7	11.4	1.59
G13-04-6	Nest 1	44.6	14.4	58.6	27.0	silty clay loam	8.89	19.5	9.8	2.94

Table 7. Proposed default soil cleanup objectives for ammonium as nitrogen for Class I and Class II groundwater based on the cation exchange capacity (CEC) of the soil or fill.

CEC	Class I as mg N/kg	Class II as mg N/kg
< 8	500	1000
8 to 15	1,000	3,600
15 to 24	2,200	9,300
> 24	5,000	20,300

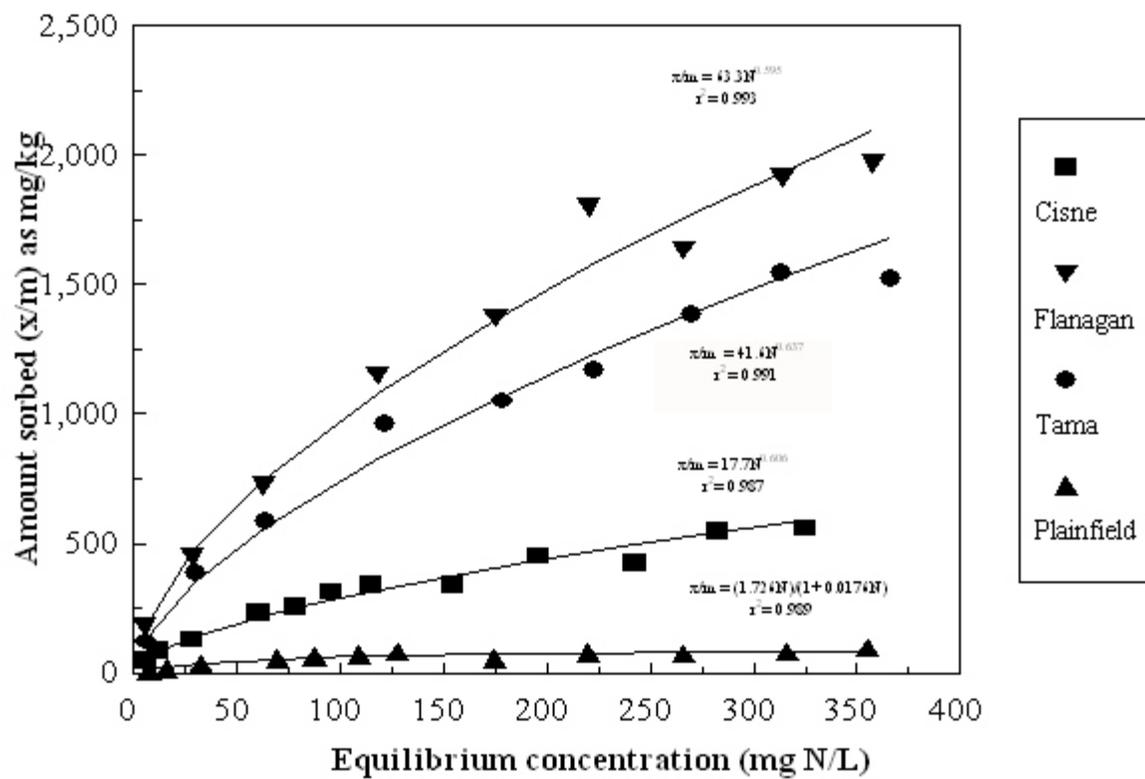


Figure 1. Ammonium sorption isotherms at 25° C for four of the soils in Table 2.

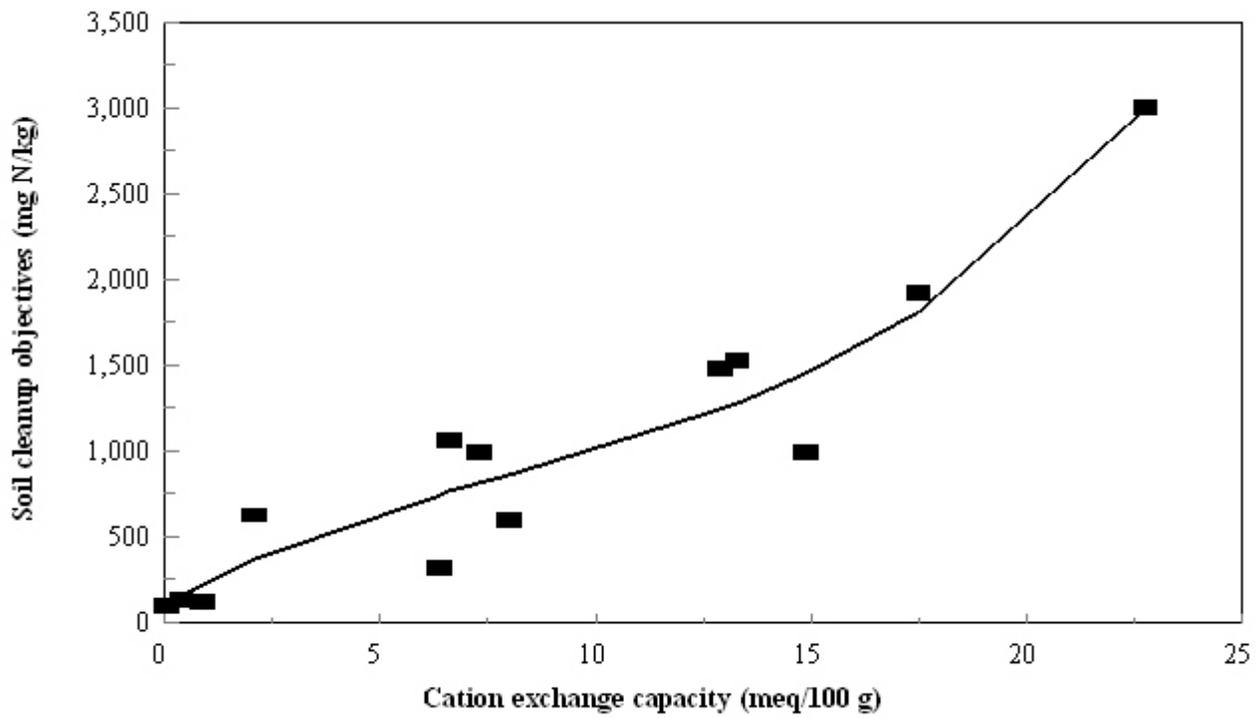


Figure 2. Relationship (cubic model) between the cation exchange capacity of the soil and fill samples in Table 2 and the soil cleanup objectives in Table 4.

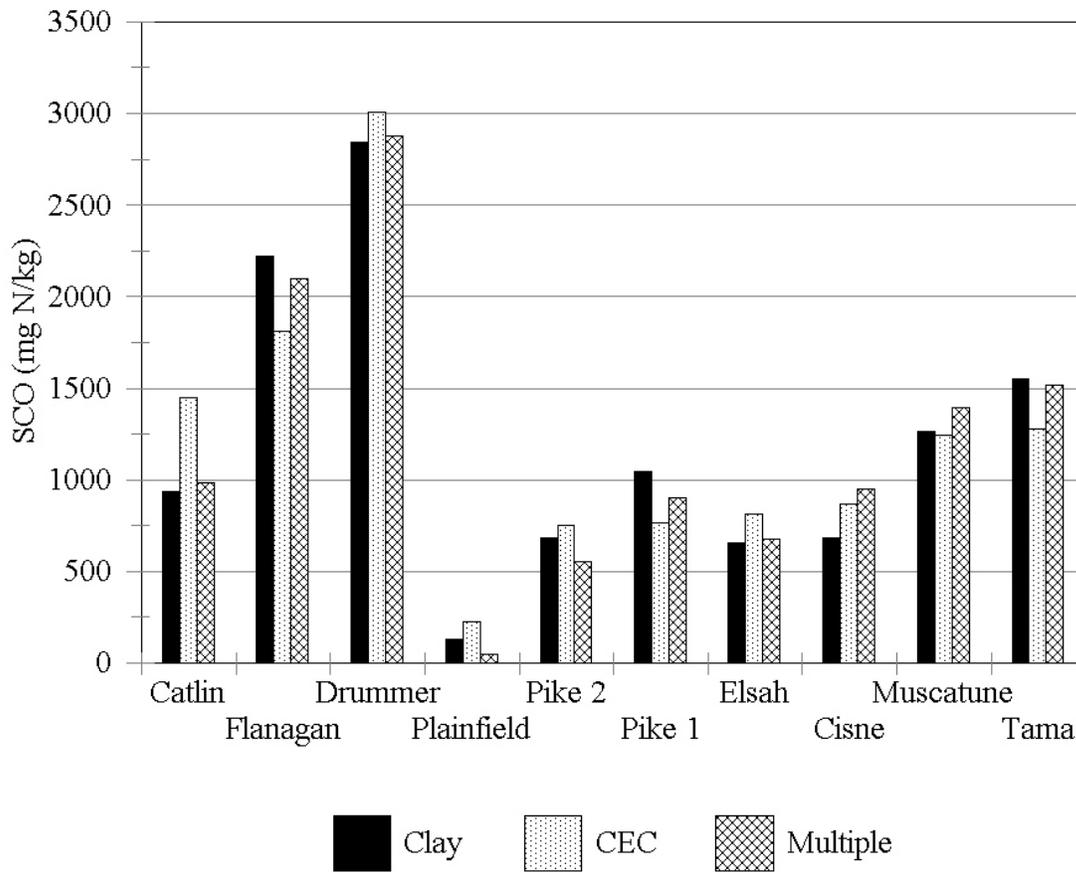


Figure 3. Ammonium soil cleanup objectives (SCO) for Class I groundwater predicted by the cubic model in Table 5 using either the clay content, CEC, or a combination of clay content, CEC, surface area, organic carbon, and pH (multiple model).

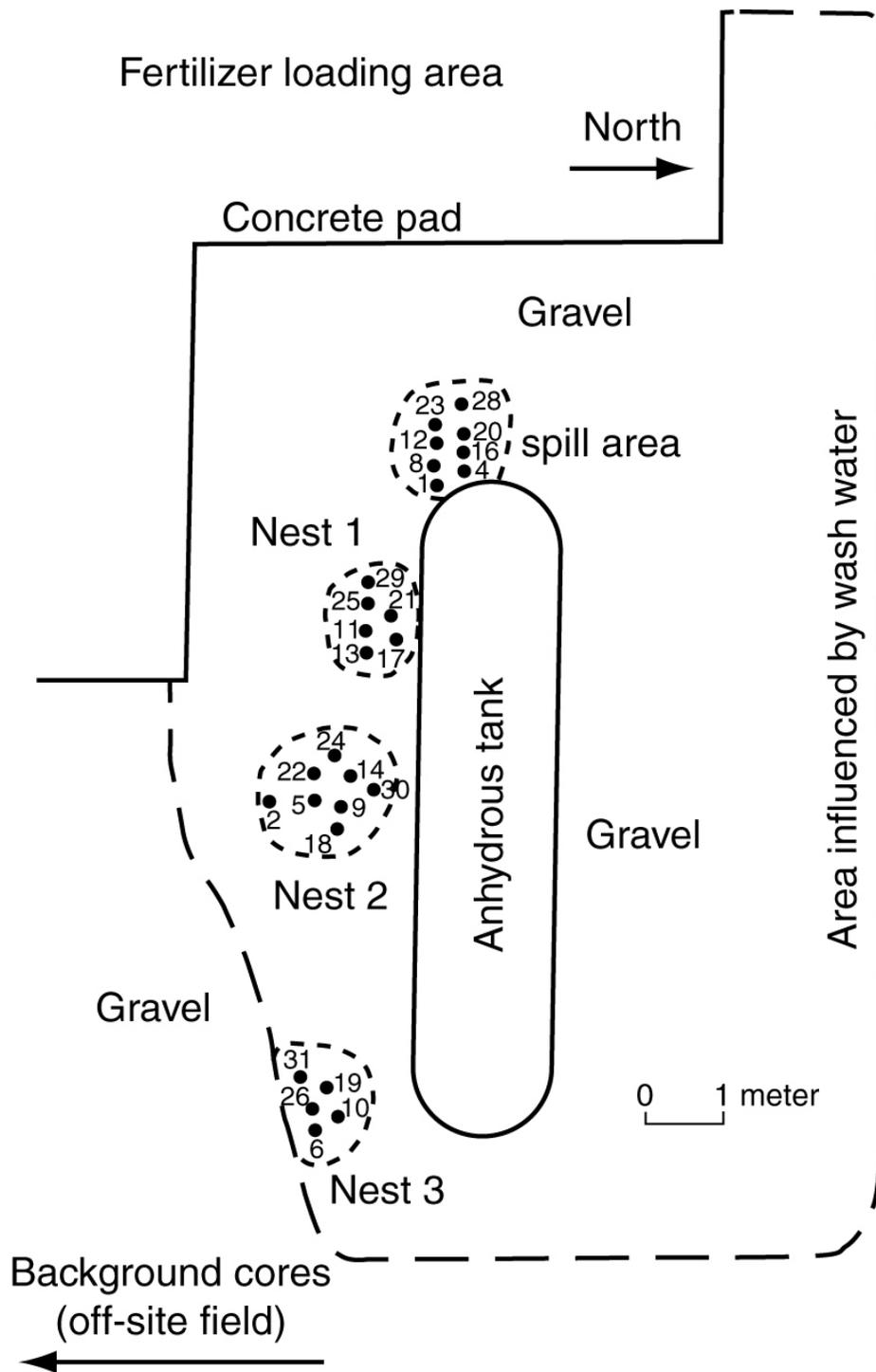


Figure 4. Schematic map of the spill area and the location of the soil cores. Numbers in each area indicate corresponding core number.

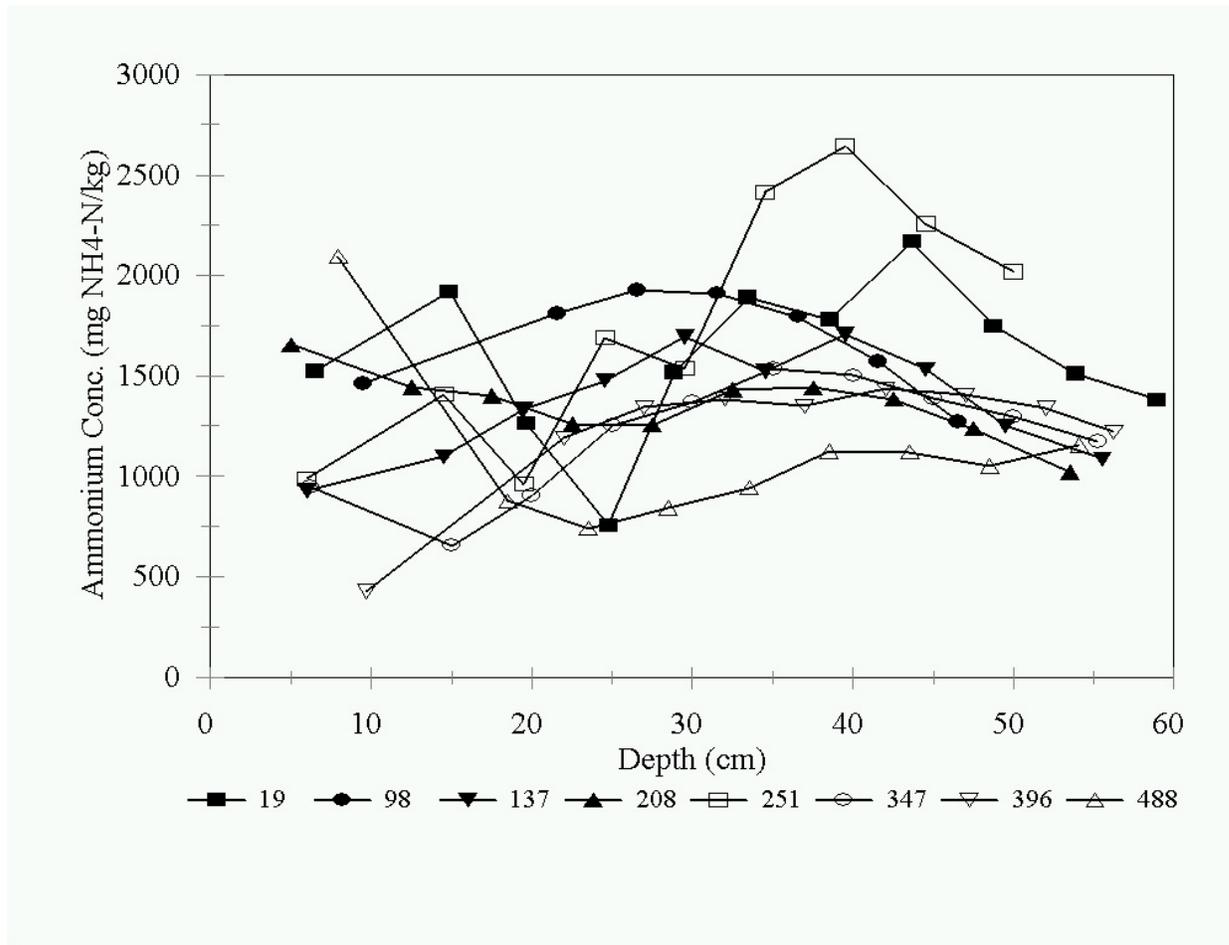


Figure 5. Distribution of ammonium in the spill area as a function of depth and time (in days) since the initial release of ammonia.

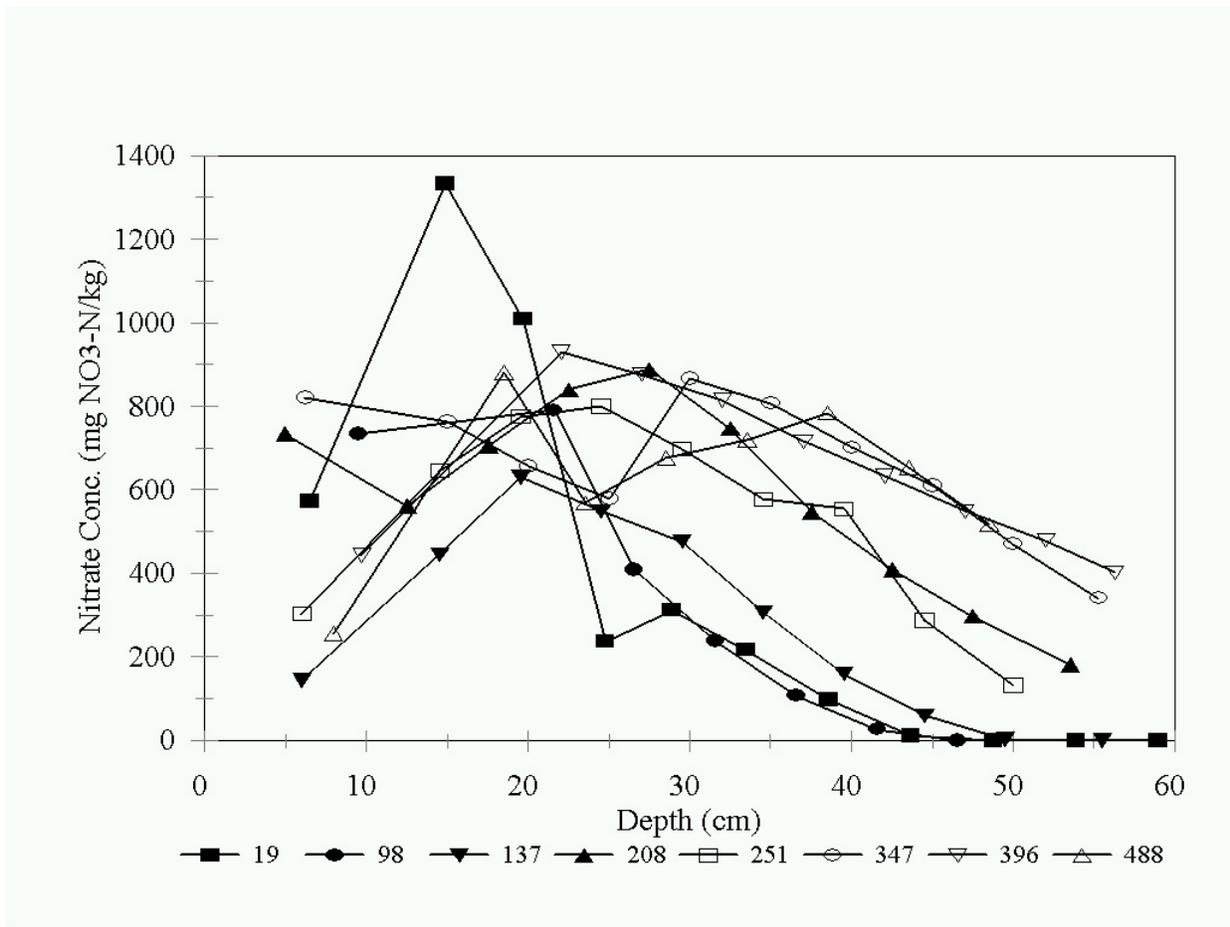


Figure 6. Distribution of nitrate in the spill area as a function of depth and time (in days) since the initial release of ammonia.

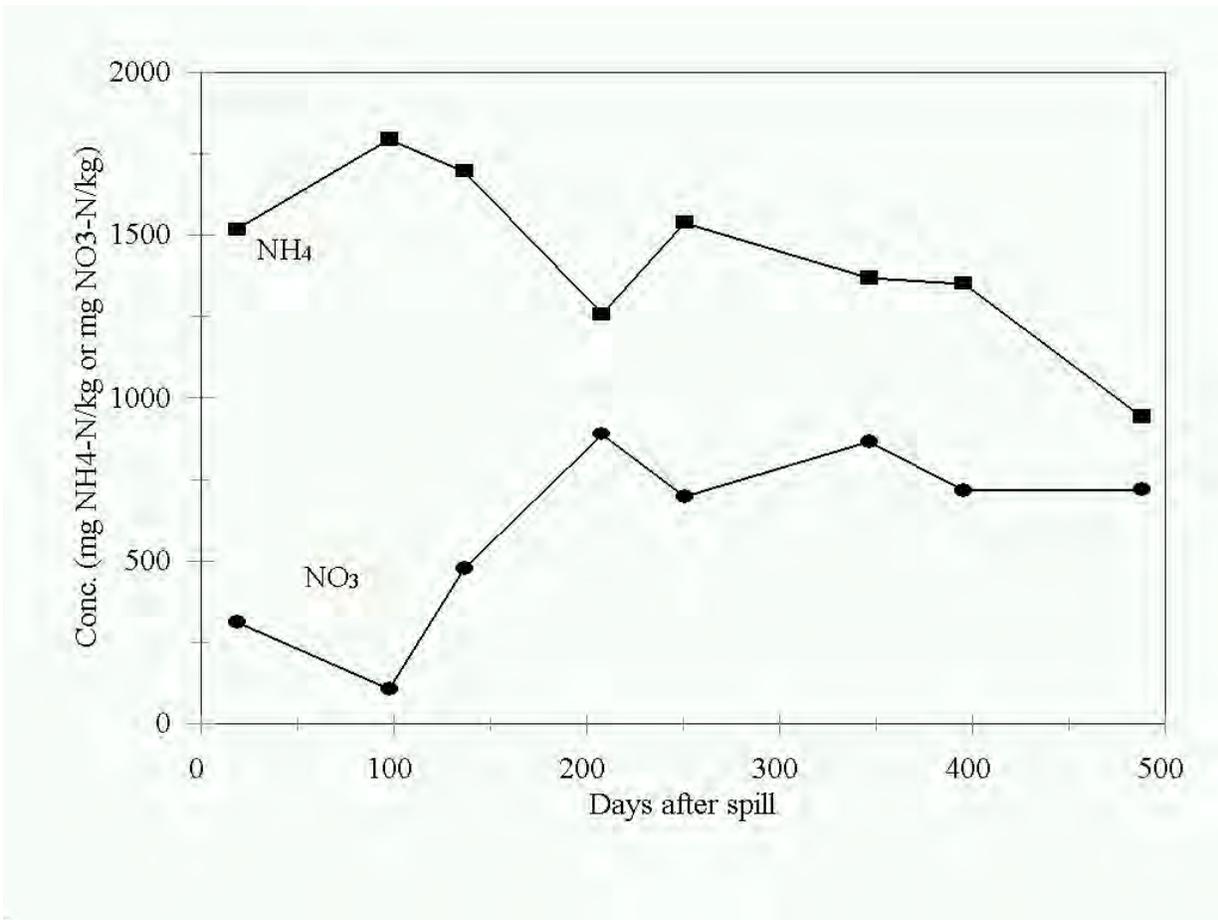


Figure 7. Concentration of ammonium and nitrate at a depth of about 32 cm in the spill area as a function of time.

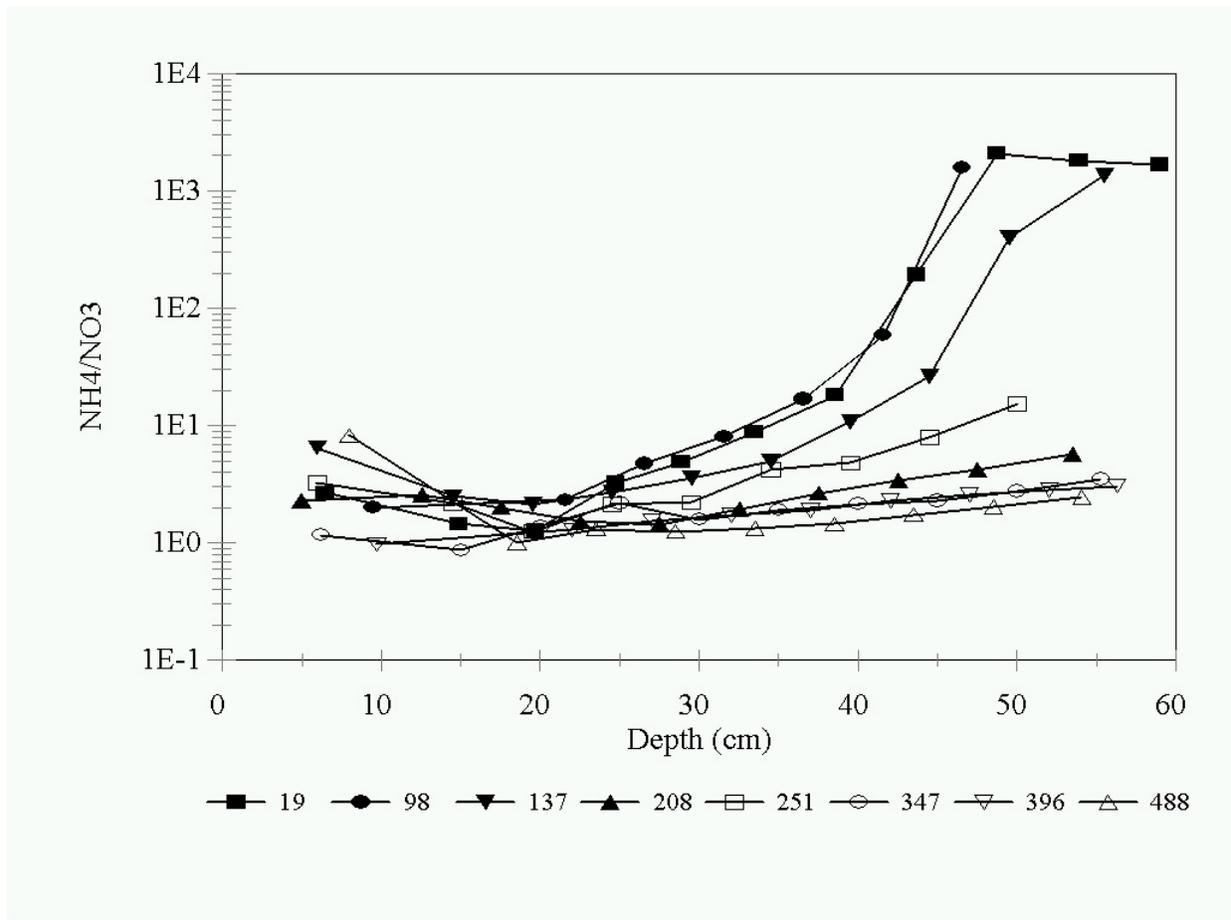


Figure 8. Ratio of ammonium to nitrate in the spill area as a function of depth and time (in days) since the initial release of ammonia.

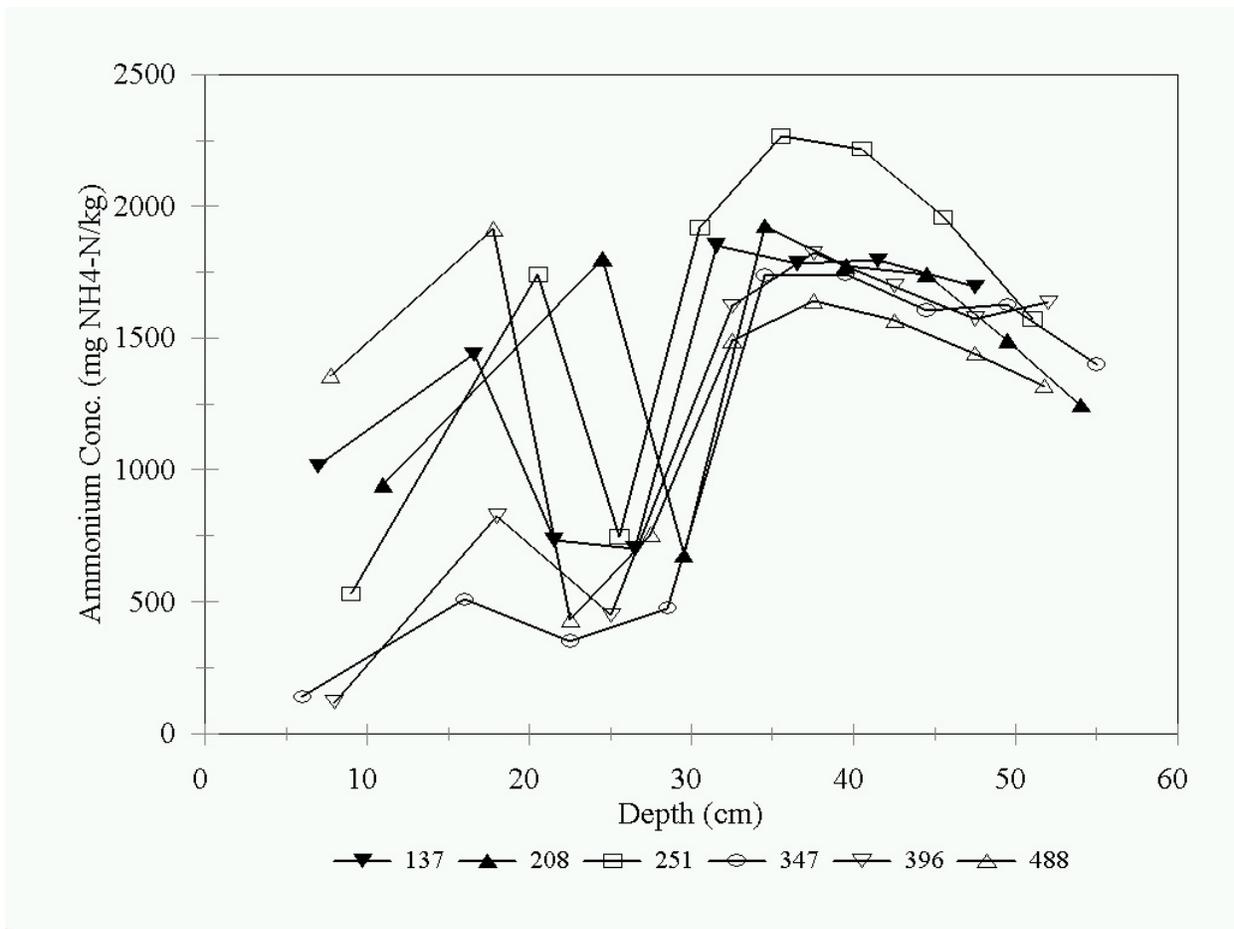


Figure 9. Distribution of ammonium in nest 1 as a function of depth and time (in days) since the initial release of ammonia.

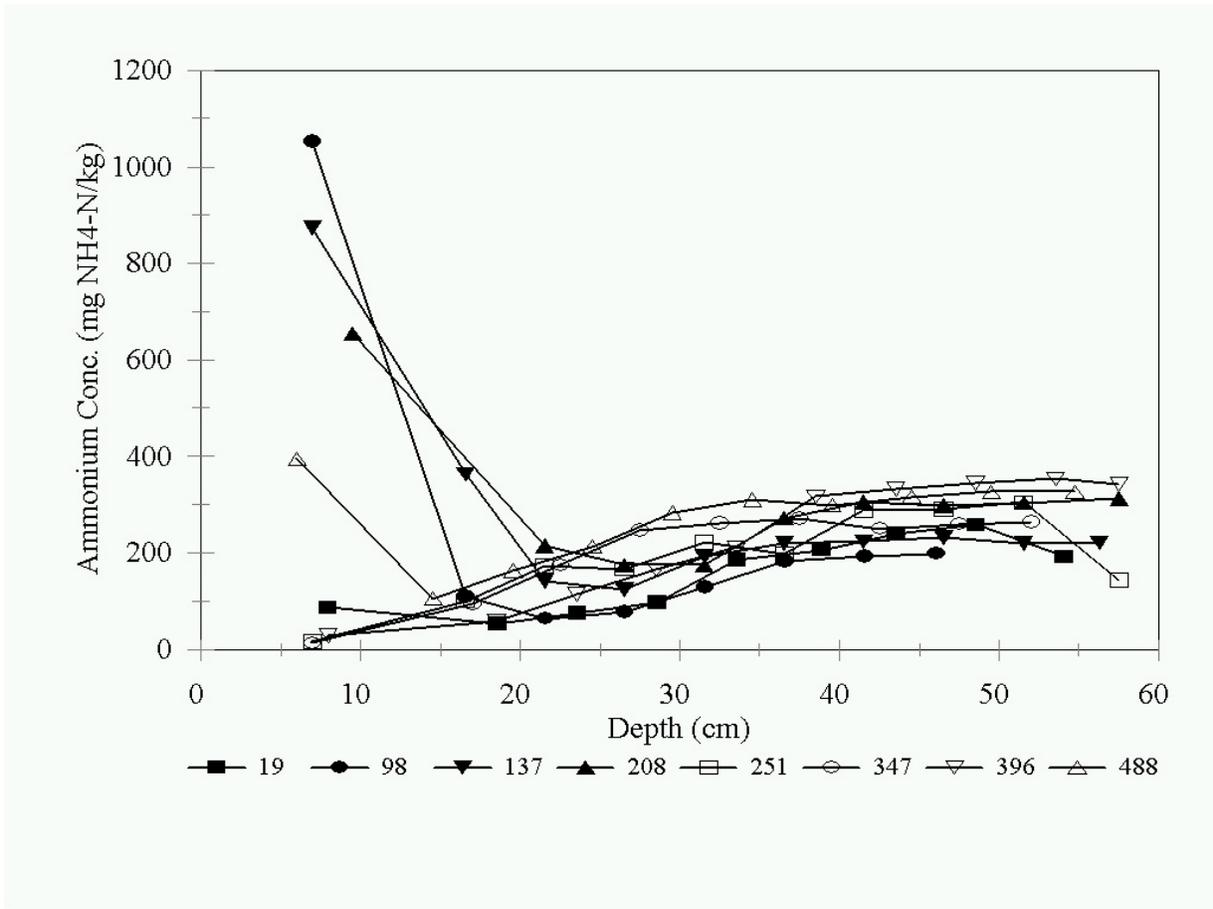


Figure 10. Distribution of ammonium in nest 2 as a function of depth and time (in days) since the initial release of ammonia.

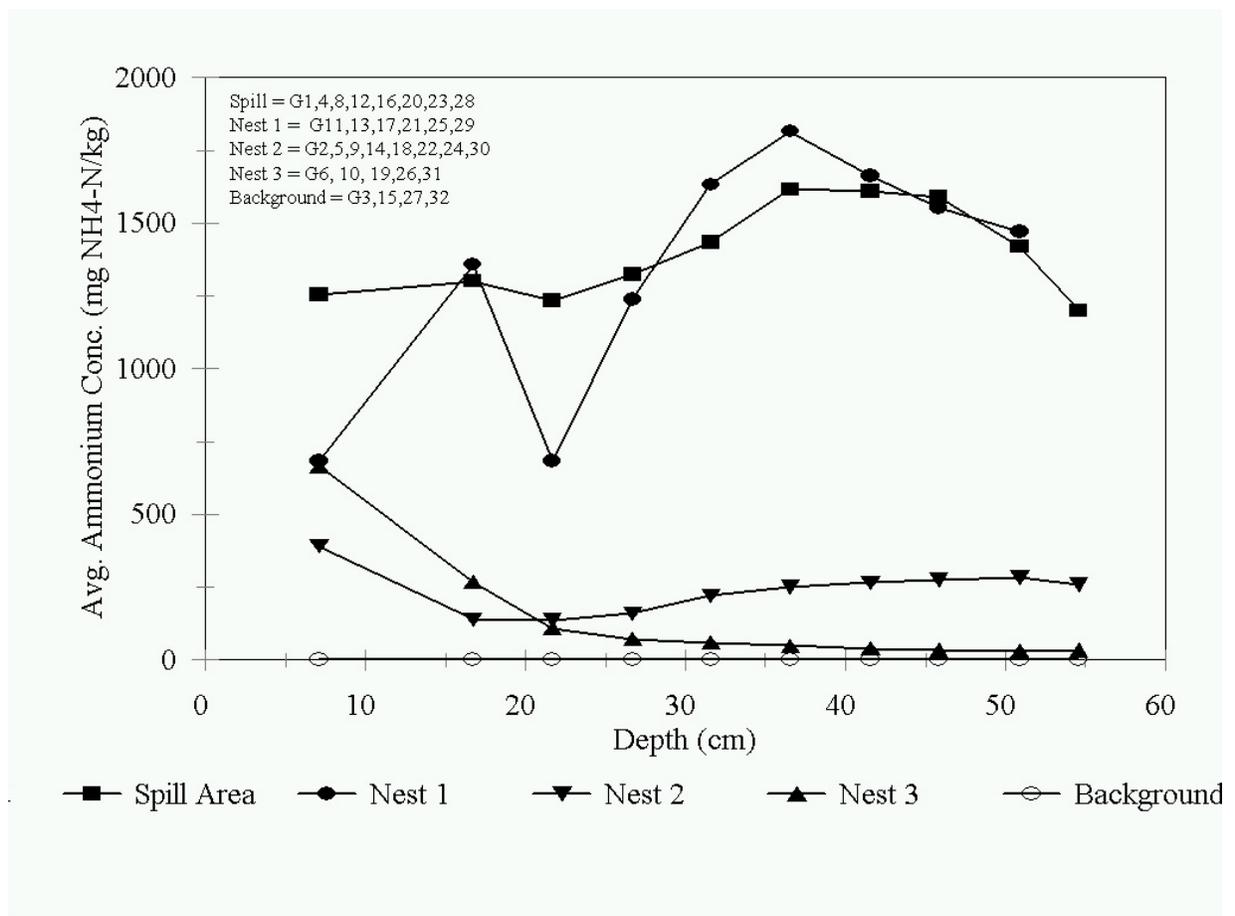


Figure 11. Mean concentration of ammonium in each sampling area plotted as a function of depth.

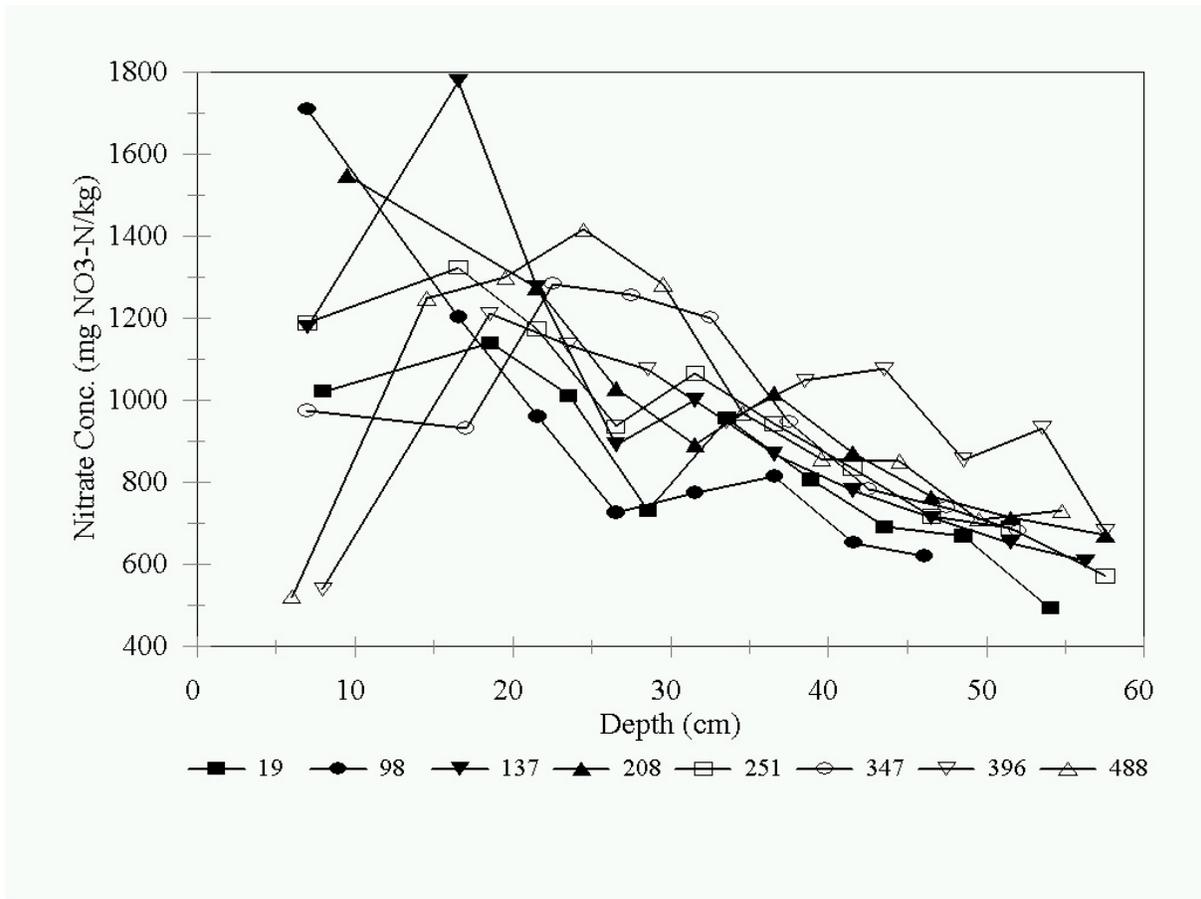


Figure 12. Distribution of nitrate in nest 2 as a function of depth and time (in days) since the initial release of ammonia.

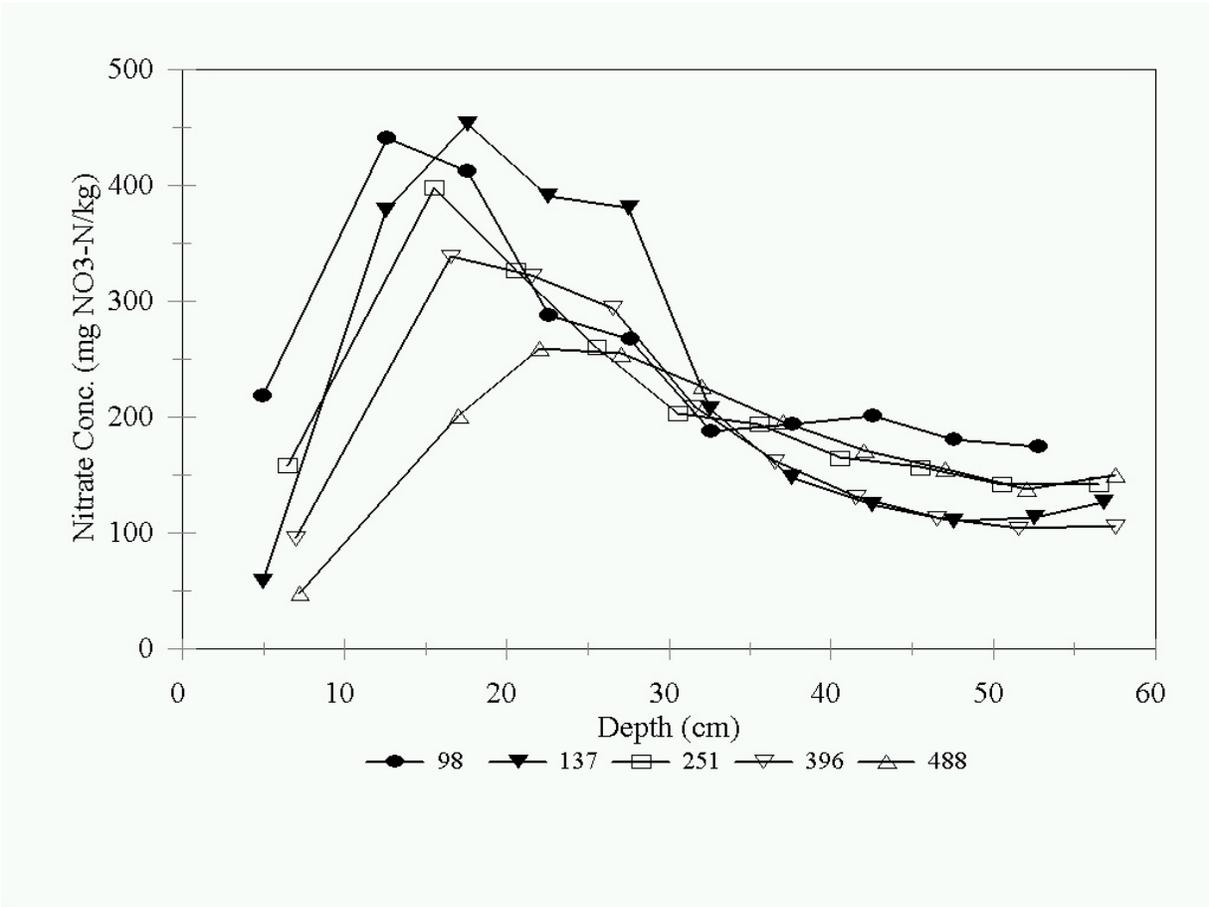


Figure 13. Distribution of nitrate in nest 3 as a function of depth and time (in days) since the initial release of ammonia.

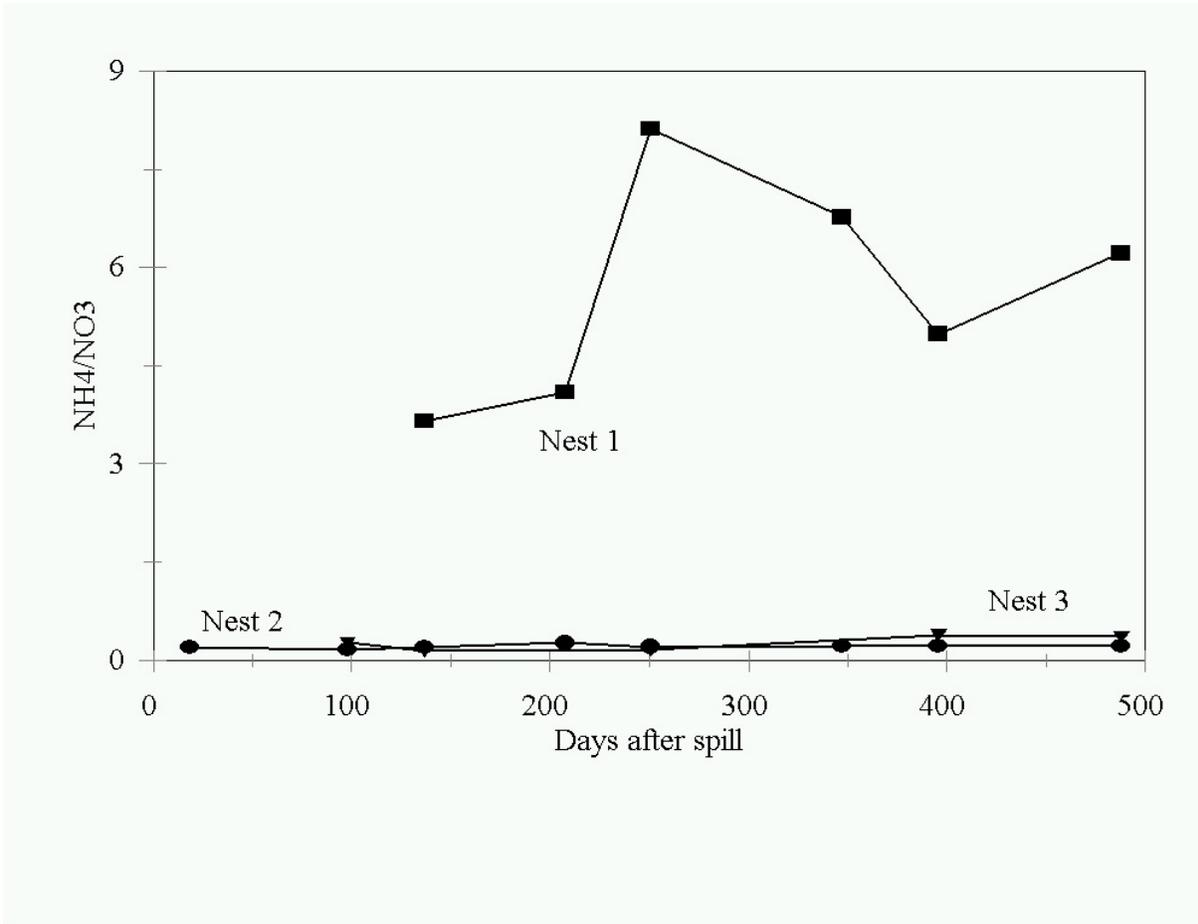


Figure 14. Ratio of ammonium to nitrate at a depth of about 34 cm in each sampling nest as a function of time since the initial release of ammonia.

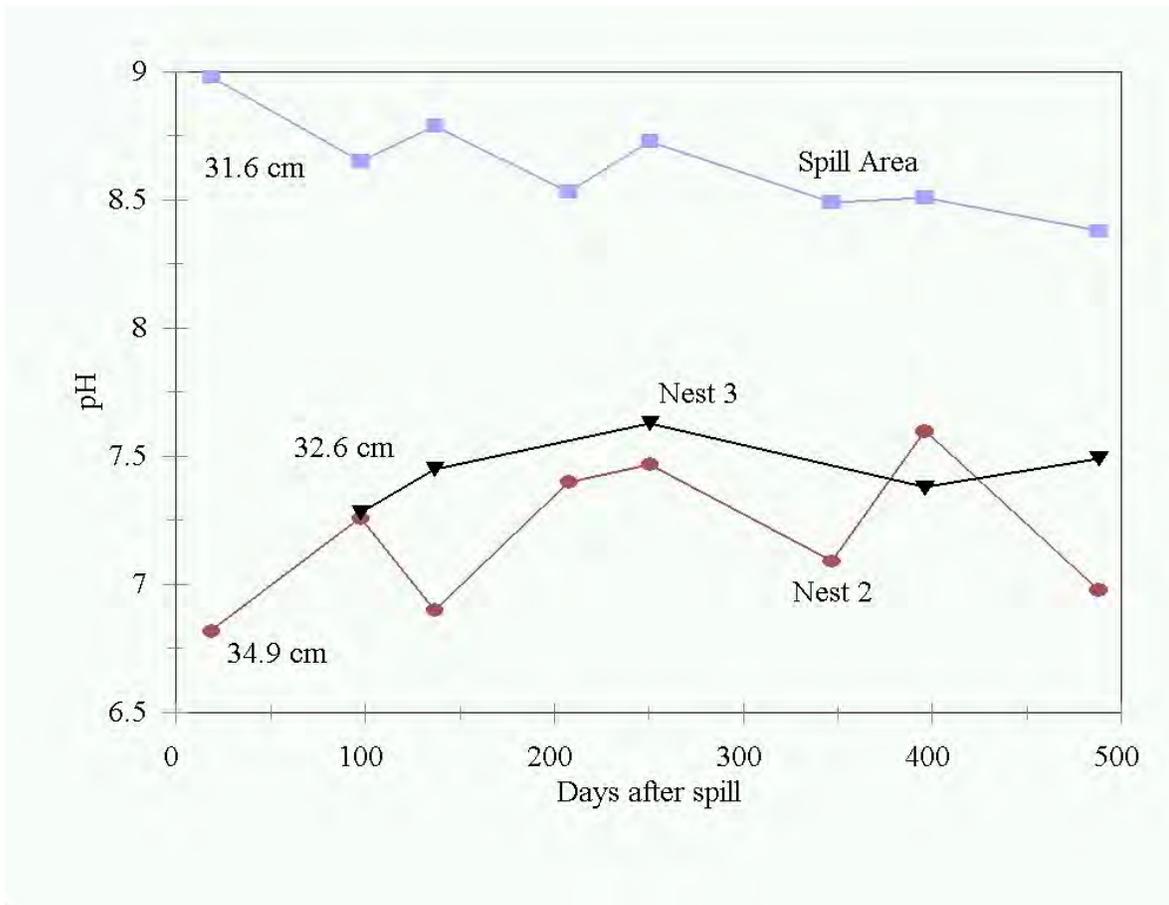


Figure 15. Distribution of pH at a depth of about 33 cm in the spill area, and nests 2 and 3 as a function of time.

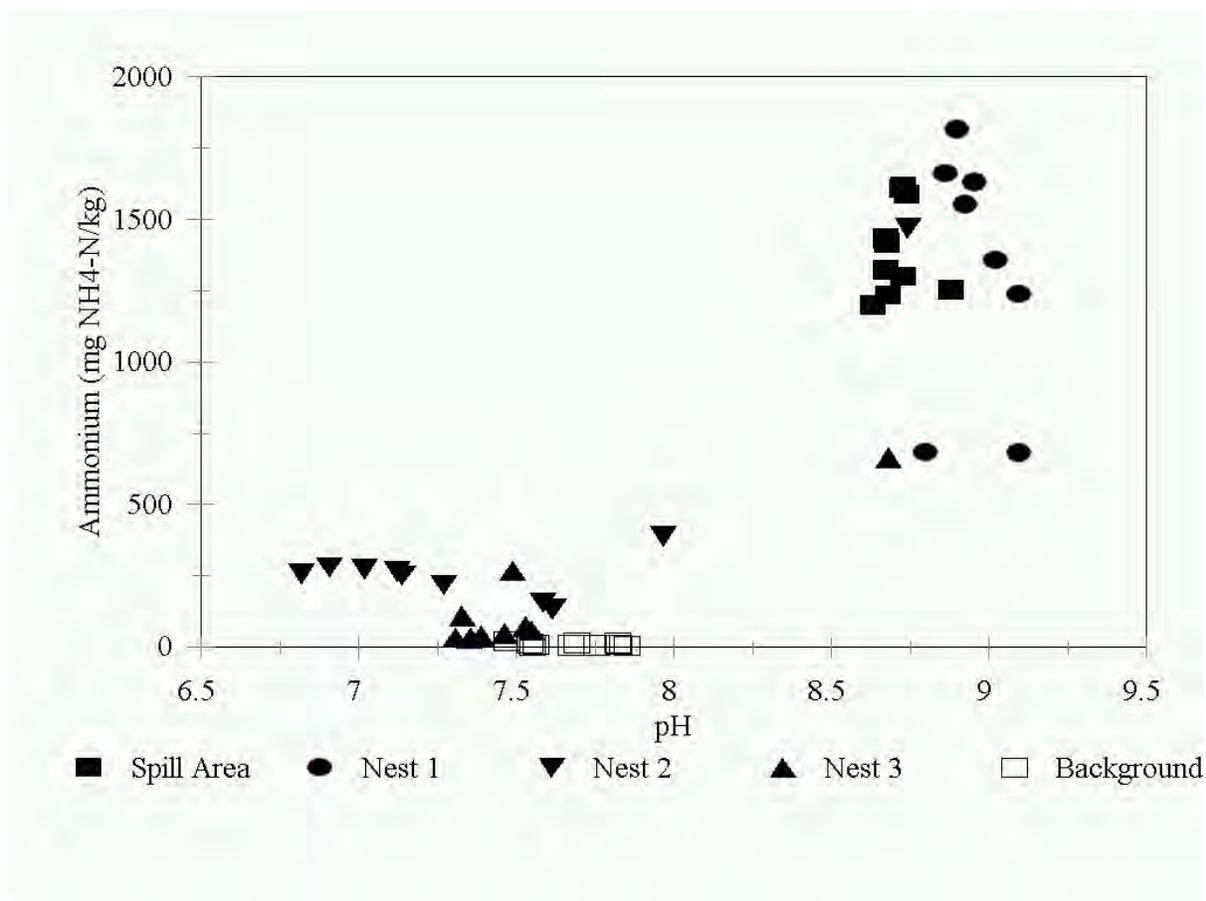


Figure 16. Mean pH of all samples collected in each sampling area.

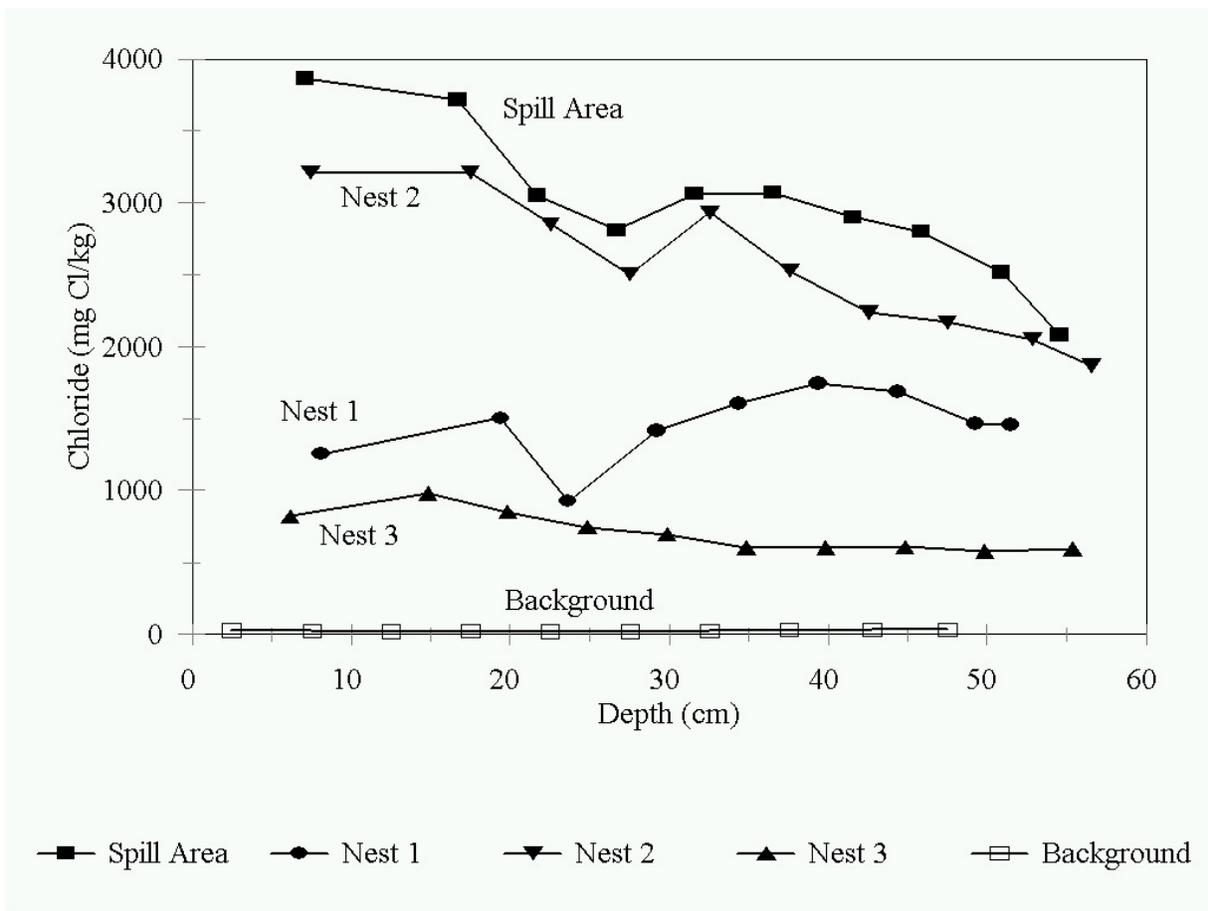


Figure 17. Mean concentration of chloride in each sampling area plotted as a function of depth.

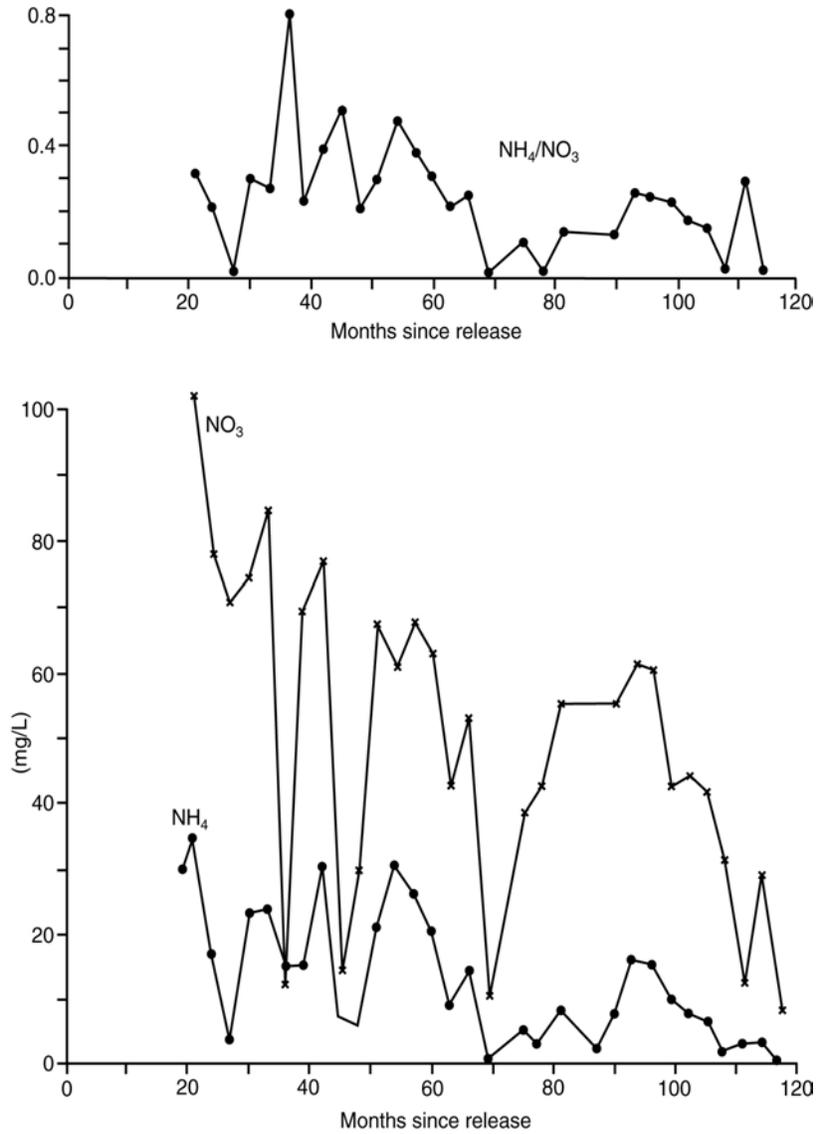


Figure 18. Ratio of ammonium to nitrate (upper figure), and concentrations of nitrate and ammonium in well MW-5 as a function of time since a catastrophic release of storm water containing 3 to 4% ammonium nitrate from an unlined basin.

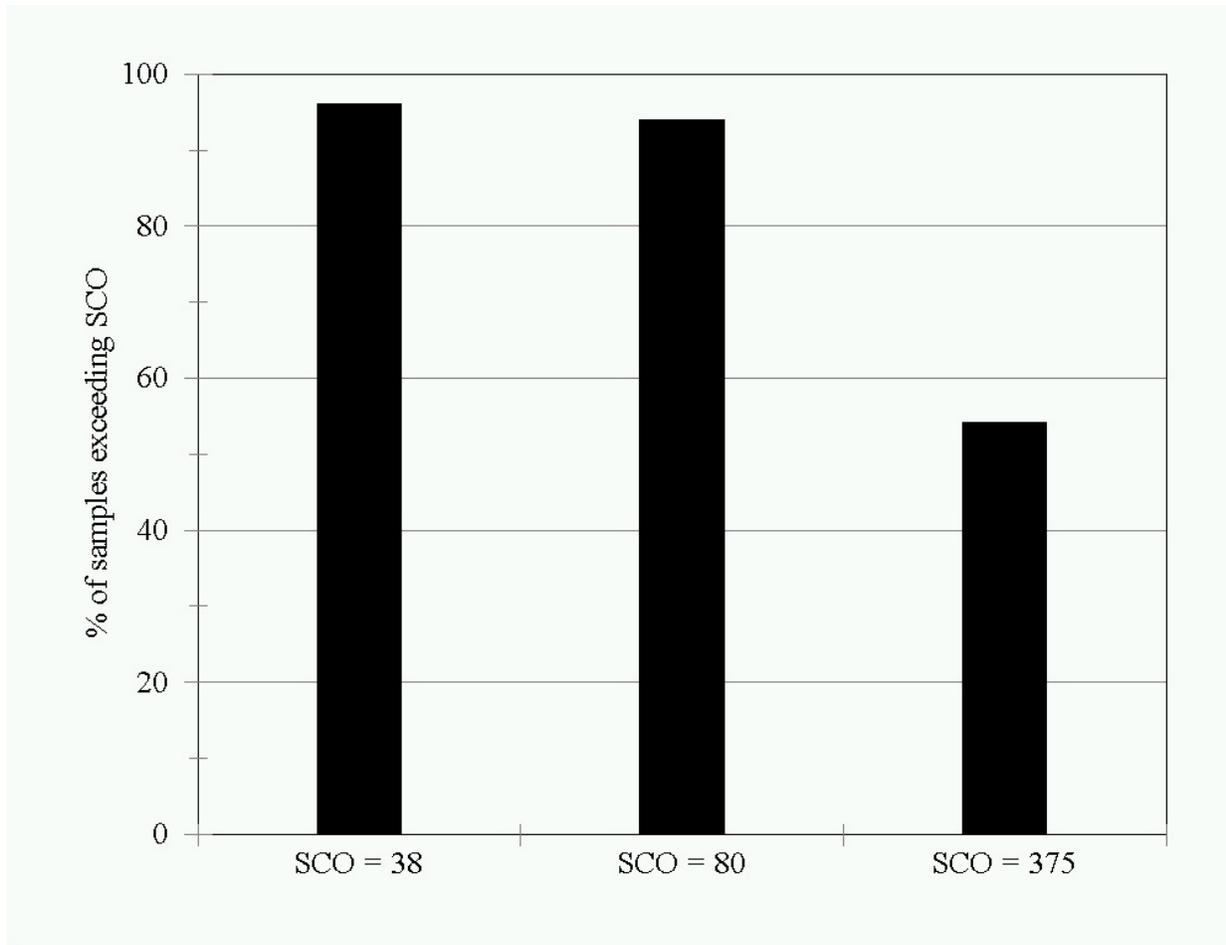


Figure 19. Distribution of all fill samples collected in the spill site that yielded nitrate in excess of a SCO of 38, 80, and 375 mg N/kg.

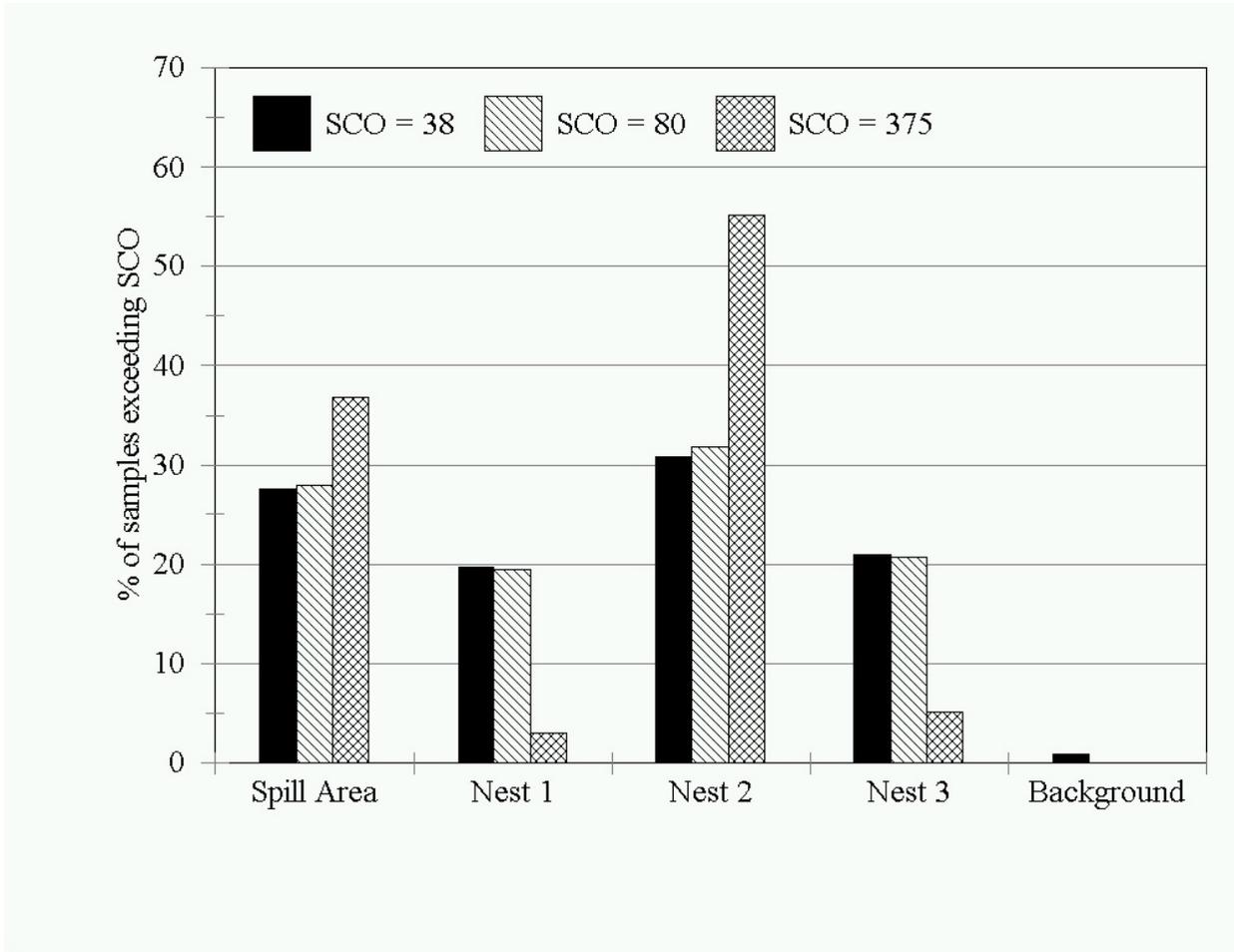


Figure 20. Distribution of the fill samples sorted by sampling area that yielded nitrate in excess of a SCO of 38, 80, and 375 mg/Nkg.

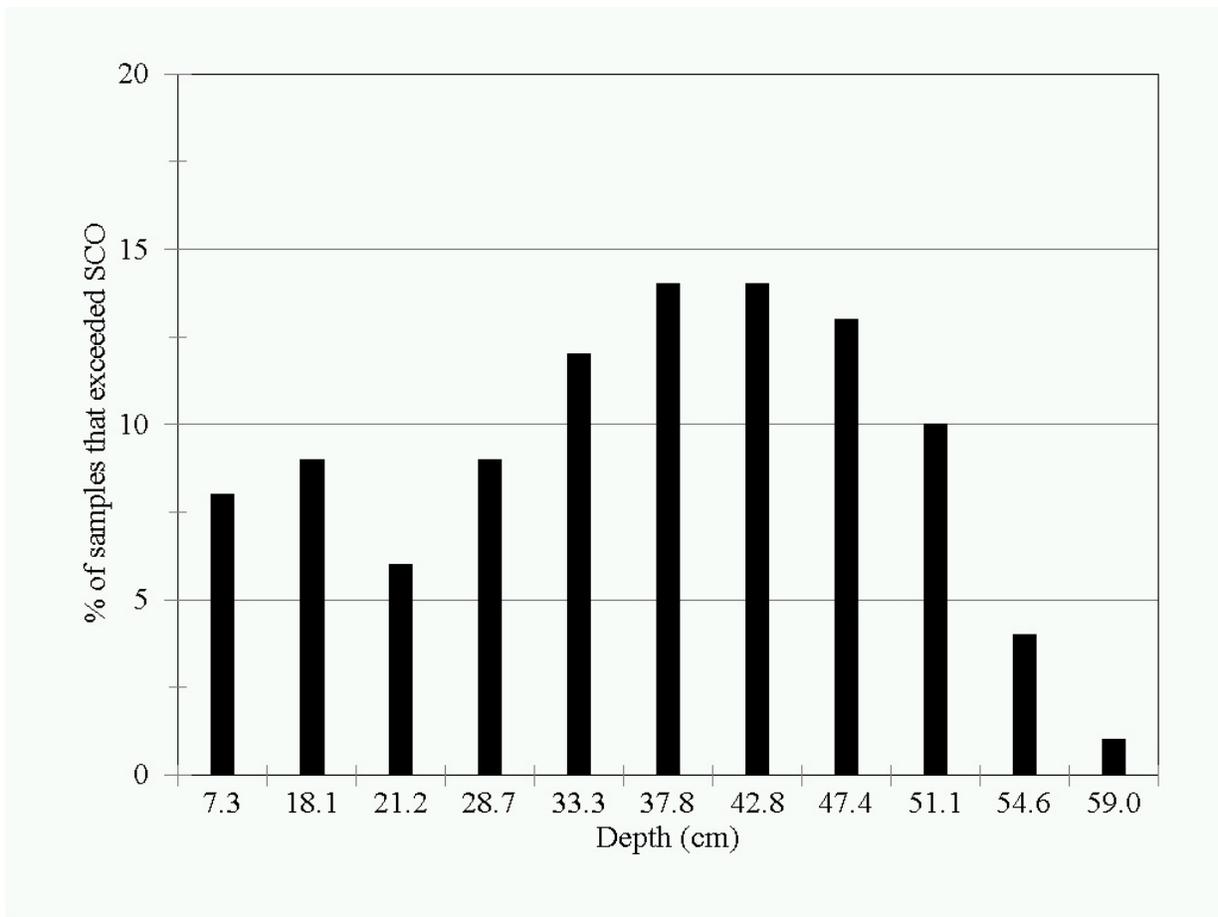


Figure 21. Distribution of fill samples as a function of depth that yielded ammonium in excess of the SCO of 1,000 mg N/kg.

Determining Nitrogen SCOs

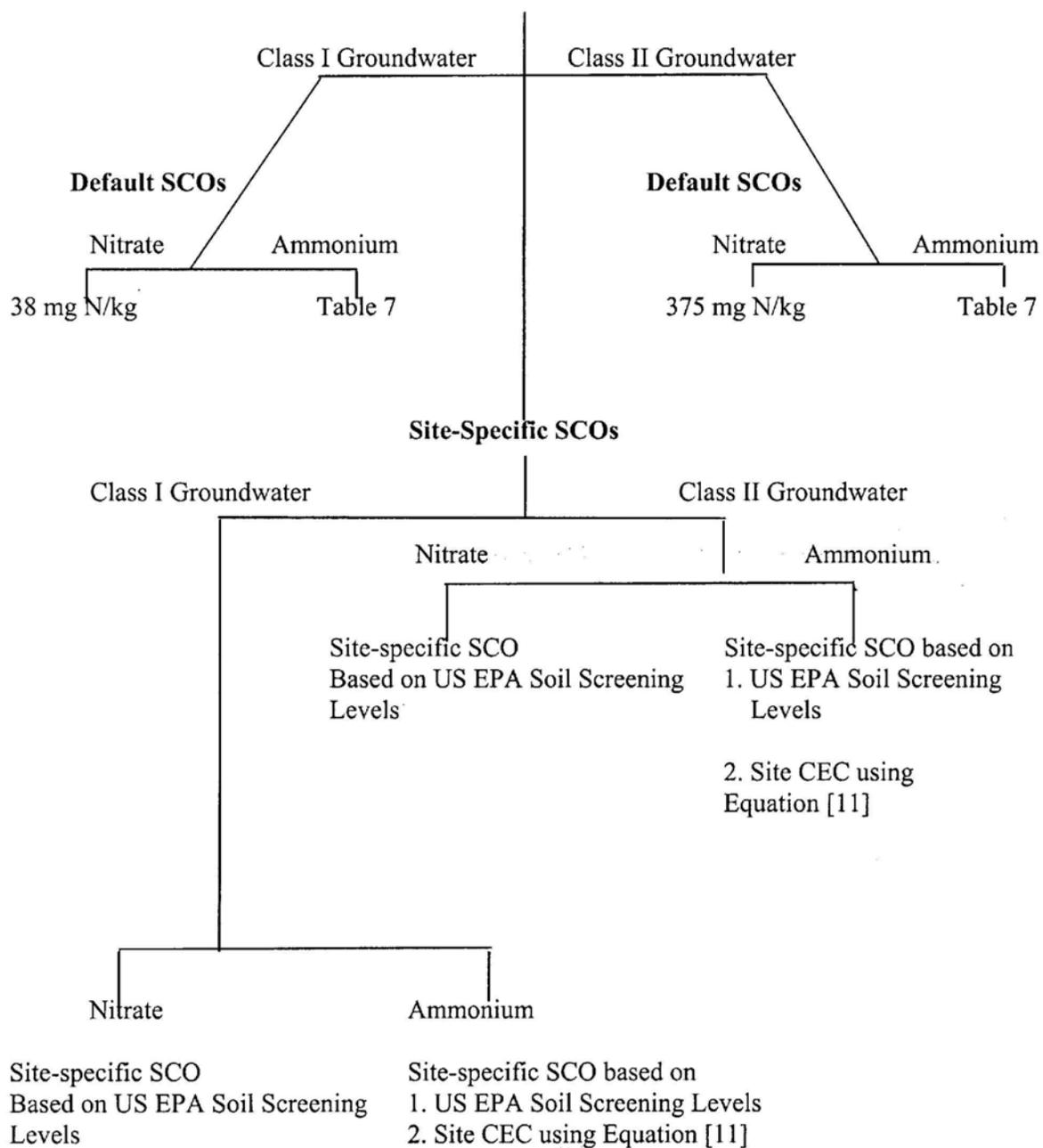


Figure 22. Flow chart showing the process and options of deriving a soil cleanup objective for nitrogen in soil materials.

Appendix

Table A-1. Summary of ammonium, nitrate, and nitrite concentrations (as mg N/L), and pH of groundwater samples collected from well MW-5. ND = no data (unpublished data from the Illinois Environmental Protection Agency).

	Ammonium	Nitrate	Nitrite	pH
22-Nov-96	ND	94	ND	ND
10-Dec-96	29.5	ND	ND	ND
13-Feb-97	34	102	2.1	7.16
15-May-97	17	78	5.7	8.1
13-Aug-97	2.9	70	<0.05	7.9
11-Nov-97	23	74	0.18	7.1
12-Feb-98	23.5	84.4	0.3	7.0
21-May-98	13.4	16.4	<0.10	6.5
25-Aug-98	15	68.7	<0.05	7.4
18-Nov-98	30.1	76.6	<0.05	7.5
18-Feb-99	6.9	13.2	<0.05	6.5
27-May-99	5.93	29.1	<0.05	7.60
31-Aug-99	20.9	66.8	<0.05	7.05
17-Nov-99	29.5	60.5	<0.05	7.09
22-Feb-00	25.8	67.6	<0.05	6.89
23-May-00	20.2	62.6	7.2	6.34
24-Aug-00	8.85	42.1	1.23	8.46
27-Nov-00	14	53	<0.15	6.85
12-Feb-01	0.34	9.9	<0.15	7.33
27-Apr-01	<0.05	36	<0.15	6.95
17-Aug-01	5.3	38	<0.15	6.85
21-Aug-01	<0.10	40.9	ND	ND
6-Nov-01	2.9	42	<0.15	6.91
1-Feb-02	8.1	55	<0.15	6.87
17-May-02	<0.05	8.9	<0.15	7.12
9-Aug-02	2.1	ND	ND	6.82
16-Aug-02	NS	46	<0.15	7.24
1-Nov-02	7.6	55	<0.15	6.58
24-Jan-03	16	61	<0.15	7.61
21-Apr-03	15	60	<0.15	6.72
8-Aug-03	10	42	<0.15	6.8
31-Oct-03	7.7	44	<0.15	6.55
6-Feb-04	6.4	42	<0.15	6.29
14-May-04	2.2	31	<0.15	7.07
30-Jul-04	3.6	12	<0.15	7.65
4-Nov-04	3.6	39	<0.15	6.5
28-Jan-05	<0.10	8	<0.15	7.59

Table A-2. Summary of ammonium and nitrate concentrations collected from well MW-9 (unpublished data from BT² Inc.).

Date	Nitrate (mg N/L)	Ammonium (mg N/L)
9-2-98	79	105
1-13-99	64	283
9-21-99	55	170
2-24-00	18	120
9-11-00	90	120
5-7-01	47	18
11-28-01	50	10
5-30-02	130	<3
11-8-02	16	1.5

Table A-3. Soil name and approximate cation exchange capacity (CEC). Provided by Prof. R. G. Hoelt, Department of Crop Sciences, University of Illinois at Urbana-Champaign.

<i>Soil name</i>	<i>CEC</i> <i>(meq/100g)</i>
Ade loamy fine sand	<8
Adrian muck	>100
Aholt silty clay	>24
Alford silt loam	8-15
Alford-Baxter complex	8-15
Alford-Bold complex	8-15
Alford-Hickory complex	8-15
Alford-Hurst silty clay loams	8-15
Alford-Ursa silt loams	8-15
Alford-Wellston silt loams	8-15
Alford-Westmore silt loams	8-15
Alganssee fine sandy loam	<8
Allison silty clay loam	>24
Alvin fine sandy loam	<8
Alvin-Lamont complex	<8
Ambraw clay loam	15-24
Ambraw silty clay loam, sandy substratum	15-24
Ambraw-Ceresco-Sarpy complex	8-15
Andres silt loam	15-24
Appleriver silt loam	8-15
Aptakistic silt loam	8-15
Aptakistic and Nappanee silt loams	8-15
Arenzville silt loam	8-15
Argyle silt loam	15-24
Armiesburg silty clay loam	15-24
Ashdale silt loam	15-24
Ashkum silty clay loam	>24
Assumption silt loam	15-24
Atkinson loam	8-15
Atlas silt loam	8-15
Atlas-Grantfork complex	8-15
Atlas-Grantfork variant complex	8-15
Atterberry silt loam	8-15
Aurelius muck	>100
Aurelius muck, sandy substratum	>100
Ava silt loam	8-15

Ava-Blair complex	8-15
Ava-Hickory complex	8-15
Aviston silt loam	15-24
Ayr sandy loam	<8
Backbone loamy sand	8-15
Banlic silt loam	8-15
Barony silt loam	15-24
Barrington silt loam	15-24
Barrington and Varna silt loams	15-24
Bartelso silt loam	15-24
Batavia silt loam	8-15
Baxter cherty silt loam	8-15
Baylis silt loam	8-15
Beardstown loam	8-15
Beasley silt loam	8-15
Beaucoup silty clay loam	>24
Beavercreek loam	<8
Bedford silt loam	8-15
Beecher silt loam	8-15
Belknap silt loam	8-15
Berks loam	8-15
Bertrand silt loam	8-15
Bethalto silt loam	15-24
Biddle silt loam	15-24
Biggsville silt loam	>24
Biggsville-Mannon silt loams	15-24
Billett sandy loam	<8
Binghampton sandy loam	8-15
Birds silt loam	8-15
Birkbeck silt loam	8-15
Birkbeck-Miami silt loams	8-15
Blackoar silt loam	15-24
Blair silt loam	8-15
Blair-Atlas silt loams	8-15
Blair-Grantfork complex	8-15
Blair-Ursa silt loams	8-15
Blake silty clay loam	15-24
Blake-Beaucoup complex	15-24
Bloomfield fine sand	<8
Blount silt loam	8-15
Bluford silt loam	8-15
Bluford silt loam, bench	15-24
Bluford-Darmstadt complex	8-15

Blyton silt loam	<8
Bold silt loam	8-15
Bonfield silt loam	8-15
Bonnie silt loam	8-15
Booker silty clay	>24
Boone loamy fine sand	<8
Bowdre silty clay	>24
Bowes silt loam	8-15
Boyer sandy loam	<8
Brandon and Saffell soils	8-15
Breeds silty clay loam	15-24
Brenton silt loam	15-24
Broadwell silt loam	15-24
Brooklyn silt loam	15-24
Brookside stony silty clay loam	15-24
Brouillett silt loam	15-24
Bryce silty clay	>24
Bryce-Calamine variant complex	15-24
Bunkum silty clay loam	15-24
Bunkum-Atlas silty clay loams	8-15
Bunkum-Coulterville silty clay loams	8-15
Burkhardt-Saude complex	8-15
Burksville silt loam	8-15
Burnside silt loam	8-15
Cairo silty clay	>24
Calamine silt loam	15-24
Calco silty clay loam	>24
Camden silt loam	8-15
Camden silt loam, sandy substratum	8-15
Canisteo silt loam	>24
Canisteo silt loam, sandy substratum	>24
Cape silty clay loam	15-24
Caprell silt loam	8-15
Carmi sandy loam	8-15
Casco silt loam	8-15
Casco-Fox complex	8-15
Caseyville silt loam	8-15
Catlin silt loam	15-24
Catlin-Saybrook complex	15-24
Ceresco loam	15-24
Channahon silt loam	15-24
Chaseburg silt loam	15-24
Chatsworth silt loam	8-15

Chauncey silt loam	8-15
Chautauqua silty clay loam	15-24
Chelsea loamy fine sand	<8
Chenoa silt loam	>24
Chute fine sand	<8
Cisne silt loam	8-15
Cisne silt loam, bench	8-15
Cisne-Huey complex	8-15
Cisne-Piasa complex	8-15
Clare silt loam	15-24
Clarence silty clay loam	>24
Clarksdale silt loam	8-15
Clarksville cherty silt loam	8-15
Clinton silt loam	8-15
Clinton-El Dara complex	<8
Clyde clay loam	>24
Coatsburg silt loam	15-24
Coffeen silt loam	15-24
Cohoctah loam	15-24
Colo silty clay loam	>24
Coloma silt loam	<8
Colp silt loam	8-15
Comfrey clay loam	>24
Coot loam	8-15
Copperas silty clay loam	15-24
Corwin silt loam	15-24
Coulterville silt loam	<8
Coulterville-Darmstadt complex	<8
Coulterville-Grantfork silty clay loams	<8
Coulterville-Hoyleton-Darmstadt complex	<8
Coulterville-Oconee silt loams	<8
Cowden silt loam	8-15
Cowden-Piasa complex	8-15
Coyne fine sandy loam	8-15
Craigmile sandy loam	<8
Crane silt loam	15-24
Crawleyville fine sandy loam	<8
Creal silt loam	8-15
Crider silt loam	8-15
Dakota silt loam	8-15
Dana silt loam	15-24
Danabrook silt loam	15-24
Darmstadt silt loam	8-15

Darmstadt-Grantfork complex	8-15
Darmstadt-Oconee silt loams	8-15
Darroch silt loam	15-24
Darwin silty clay	>24
Del Rey silt loam	8-15
Denny silt loam	8-15
Denrock silt loam	15-24
Derinda silt loam	8-15
Dickinson sandy loam	8-15
Dickinson sandy loam, loamy substratum	8-15
Dickinson-Hamburg complex	8-15
Dickinson-Onarga complex	8-15
Disco sandy loam	8-15
Dockery silt loam	8-15
Dodge silt loam	8-15
Dodgeville silt loam	15-24
Dorchester silt loam	8-15
Dorchester silt loam, cobbly substratum	8-15
Douglas silt loam	15-24
Dowagiac silt loam	8-15
Downs silt loam	8-15
Downsouth silt loam	15-24
Dresden silt loam	15-24
Drummer silty clay loam	>24
Drummer silty clay loam, gravelly substratum	>24
Drummer silty clay loam, till substratum	>24
Drury silt loam	8-15
Du Page silt loam	15-24
Dubuque silt loam	8-15
Dubuque and Dunbarton soils	8-15
Dubuque and Palsgrove soils	8-15
Dubuque-Orthents- Fayette Complex	8-15
Dunbarton silt loam	8-15
Dunbarton silt loam, cherty variant	8-15
Dunbarton-Dubuque complex	8-15
Dunham silty clay loam	>24
Dupo silt loam	8-15
Durand silt loam	15-24
Ebbert silt loam	15-24
Eden silty clay loam	>24
Edgington silt loam	15-24
Edinburg silty clay loam	>24

Edmund silt loam	15-24
Edwards muck	>100
Edwardsville silt loam	>24
El Dara sandy loam	<8
Elburn silt loam	15-24
Elburn silt loam, gravelly substratum	15-24
Elco silt loam	8-15
Elco-Atlas silt loams	8-15
Elco-Ursa silt loams	8-15
Eleroy silt loam	8-15
Eleroy and Derinda soils	8-15
Eleva sandy loam	<8
Elizabeth silt loam	15-24
Elkhart silt loam	15-24
Elliott silt loam	15-24
Elpaso silty clay loam	>24
Elsah cherty silt loam	8-15
Elvers silt loam	8-15
Emery silt loam	15-24
Emma silty clay loam	15-24
Evansville silt loam	8-15
Faxon clay loam	15-24
Faxon-Ripon complex	15-24
Fayette silt loam	8-15
Fayette silty clay loam, karst	<8
Fayette silt loam, sandy substratum	15-24
Fayette silt loam, till substratum	8-15
Fayette-Clarksville complex	8-15
Fayette-Hickory complex	8-15
Fayette-Westville complex	8-15
Fella silty clay loam	>24
Fieldon silt loam	15-24
Fincastle silt loam	8-15
Fishhook silt loam	8-15
Fishhook-Atlas complex	8-15
Flagg silt loam	8-15
Flagler sandy loam	8-15
Flanagan silt loam	15-24
Floraville silt loam	8-15
Fosterburg silt loam	>24
Fox silt loam	8-15
Frankfort silt loam	8-15
Frankville silt loam	15-24

Friesland sandy loam	8-15
Frondorf loam	8-15
Fulfs silty clay	>24
Gale silt loam	8-15
Geff silt loam	8-15
Genesee silt loam	8-15
Geryune silt loam	15-24
Gilford fine sandy loam	8-15
Ginat silt loam	8-15
Gorham silty clay loam	>24
Gosport silt loam	8-15
Goss gravelly silt loam	8-15
Goss-Alford complex	8-15
Granby loamy sand	<8
Grantfork silty clay loam	8-15
Grantsburg silt loam	8-15
Graymont silt loam	15-24
Grays silt loam	8-15
Grays and Markham silt loams	8-15
Greenbush silt loam	15-24
Grellton sandy loam	<8
Griswold loam	8-15
Grundelein silt loam	>24
Hamburg silt loam	8-15
Harco silt loam	15-24
Harpster silty clay loam	>24
Harrison silt loam	15-24
Hartsburg silty clay loam	>24
Harvard silt loam	8-15
Hayfield loam	8-15
Haymond silt loam	8-15
Haynie silt loam	8-15
Hennepin loam	<8
Hennepin and Miami soils	<8
Hennepin-Casco complex	<8
Hennepin-Vanmeter complex	8-15
Henshaw silt loam	8-15
Herbert silt loam	8-15
Herrick silt loam	15-24
Herrick-Biddle-Piasa silt loams	15-24
Herrick-Piasa complex	15-24
Hesch fine sandy loam	8-15
Hesch fine sandy loam, gray subsoil	8-15

variant	
Hesch loamy sand, shallow variant	8-15
Hickory loam	8-15
Hickory and Hennepin soils	8-15
Hickory and Sylvan soils	8-15
Hickory-Atlas complex	8-15
Hickory-Clinton complex	8-15
Hickory-Gosport complex	8-15
Hickory-Hennepin complex	8-15
Hickory-High Gap silt loams	8-15
Hickory-Hosmer silt loams	8-15
Hickory-Kell silt loams	8-15
Hickory-Negley complex	8-15
Hickory-Sylvan complex	8-15
Hickory-Sylvan-Fayette silt loams	8-15
Hickory-Wellston silt loams	8-15
High Gap loam	8-15
Hitt silt loam	15-24
Holly silt loam	15-24
Holton silt loam	8-15
Homen silt loam	15-24
Homer silt loam	8-15
Hononegah loamy coarse sand	<8
Hoopeston sandy loam	8-15
Hooppole loam	15-24
Hosmer silt loam	8-15
Hosmer-Lax silt loams	8-15
Hosmer-Ursa silt loams	8-15
Houghton muck	>100
Houghton peat	>100
Hoyleton silt loam	8-15
Hoyleton silt loam, bench	8-15
Hoyleton-Darmstadt complex	8-15
Hoyleton-Tamalco complex	8-15
Huey silt loam	8-15
Huntington silt loam	15-24
Huntsville silt loam	15-24
Hurst silt loam	8-15
Hurst silt loam, sandy substratum	8-15
Iona silt loam	8-15
Ipava silt loam	15-24
Ipava-Sable complex	15-24
Ipava-Tama complex	15-24

Iva silt loam	8-15
Jacob clay	15-24
Jasper silt loam	15-24
Jasper silt loam, sandy substratum	15-24
Joliet silty clay loam	>24
Joslin silt loam	15-24
Joy silt loam	15-24
Joyce silt loam	15-24
Joy silt loam, sandy substratum	15-24
Jules silt loam	8-15
Juneau silt loam	8-15
Kane silt loam	15-24
Kaneville silt loam	15-24
Kankakee fine sandy loam	8-15
Karnak silty clay	15-24
Kell silt loam	8-15
Keller silt loam	15-24
Keller-Coatsburg complex	15-24
Keltner silt loam	15-24
Kendall silt loam	8-15
Kendall silt loam, sandy substratum	8-15
Keomah silt loam	8-15
Kernan silt loam	8-15
Keswick loam	>24
Kidami silt loam	8-15
Kidder silt loam	<8
Kish loam	>24
Kishwaukee silt loam	15-24
Knight silt loam	15-24
La Hogue loam	8-15
La Rose silt loam	8-15
Lacrescent cobbly silty clay loam	15-24
Lahoguess loam	15-24
Lakaskia silt loam	15-24
Lamoille silt loam	15-24
Lamont fine sandy loam	<8
Lamont, Tell, and Bloomfield soils	<8
Landes fine sandy loam	8-15
Lanier fine sandy loam	8-15
Lawler loam	8-15
Lawndale silt loam	15-24
Lawson silt loam	15-24
Lax silt loam	8-15

Lena muck	>100
Lenzburg silt loam	15-24
Lenzburg silt loam, acid substratum	15-24
Lenzlo silty clay loam	15-24
Lenzwheel silty clay loam	15-24
Lindley loam	8-15
Lisbon silt loam	15-24
Lismod silt loam	15-24
Littleton silt loam	15-24
Lomax loam	8-15
Loran silt loam	15-24
Lorenzo loam	8-15
Mannon silt loam	8-15
Marbletown silt loam	15-24
Marine silt loam	8-15
Marissa silt loam	8-15
Markham silt loam	8-15
Markland silt loam	8-15
Marseilles silt loam	8-15
Marseilles silt loam, gravelly substratum	8-15
Marseilles silt loam, moderately wet	15-24
Marseilles-Atlas complex	8-15
Marseilles-Hickory complex	8-15
Marshan loam	8-15
Marshan loam, sandy substratum	15-24
Martinsville silt loam	8-15
Martinton silt loam	15-24
Mascoutah silty clay loam	>24
Massbach silt loam	8-15
Matherton silt loam	8-15
Maumee fine sandy loam	<8
Mayville silt loam	15-24
McFain silty clay	>24
McGary silt loam	8-15
McHenry silt loam	8-15
Meadowbank silt loam	15-24
Medary silty clay loam	8-15
Medway silty clay loam	>24
Menfro silt loam	8-15
Metea silt loam	<8
Miami fine sandy loam	<8
Miami silt loam	8-15
Miami-Casco complex	8-15

Miami-Hennepin complex	8-15
Miami-Russell silt loams	8-15
Middletown silt loam	8-15
Milford silty clay loam	>24
Millbrook silt loam	8-15
Millington loam	8-15
Millsdale silty clay loam	>24
Millstadt silt loam	>24
Millstream silt loam	15-24
Minneiska loam	15-24
Mokena silt loam	15-24
Mona silt loam	15-24
Monee silt loam	15-24
Monterey silty clay loam	>24
Montgomery silty clay loam	>24
Montmorenci silt loam	8-15
Morley silt loam	8-15
Morocco fine sand	<8
Morristown silt loam	8-15
Moundprairie silty clay loam	15-24
Mt. Carroll silt loam	8-15
Mudhen clay loam	>24
Mundelein silt loam	15-24
Mundelein and Elliott soils	15-24
Muren silt loam	8-15
Muscatine silt loam	15-24
Muscatune silt loam	15-24
Muskego muck	>100
Muskego silty clay loam, overwash	>24
Muskego and Houghton mucks	>100
Muskego and Peotone soils, ponded	>100
Muskingum stony silt loam	<8
Muskingum and Berks soils	<8
Myrtle silt loam	8-15
Nachusa silt loam	15-24
Nameoki silty clay	15-24
Nappanee silt loam	8-15
Nasset silt loam	15-24
Navlys silty clay loam	15-24
Negley loam	8-15
Neotoma-Wellston complex	8-15
Newark silty clay loam	15-24
Newberry silt loam	8-15

NewGlarus-Lamoille complex	8-15
NewGlarus-Palsgrove silt loams	8-15
Newhaven loam	8-15
Niota silt loam	8-15
Niota silty clay loam, clayey subsurface	15-24
variant	
Nolin silty clay loam	8-15
Normal silt loam	15-24
Normandy silt loam	15-24
Oakville fine sand	<8
Oakville-Tell complex	<8
Ockley silt loam	8-15
Oconee silt loam	8-15
Oconee-Coulterville-Darmstadt silt loams	8-15
Oconee-Darmstadt-Coulterville silt loams	8-15
Oconee-Tamalco complex	8-15
Octagon silt loam	8-15
Odell silt loam	15-24
Ogle silt loam	15-24
Ogle silt loam, silt loam subsoil variant	15-24
Okaw silt loam	8-15
Okaw silty clay loam	15-24
Onarga sandy loam	8-15
Onarga fine sandy loam, till substratum	8-15
Oneco silt loam	8-15
Orio sandy loam	<8
Orion silt loam	8-15
Osceola silt loam	15-24
Osco silt loam	15-24
Otter silt loam	>24
Ozaukee silt loam	15-24
Palms muck	>100
Palms silty clay loam, overwash	>24
Palsgrove silt loam	8-15
Palsgrove and Woodbine soils	8-15
Pana silt loam	15-24
Papineau fine sandy loam	8-15
Parke silt loam	8-15
Parkville silty clay	8-15
Parmod silt loam	15-24
Parr fine sandy loam	8-15
Parr silt loam	8-15
Passport silt loam	8-15

Passport-Grantfork complex	8-15
Patton silty clay loam	>24
Paxico silt loam	8-15
Pecatonica silt loam	8-15
Pella silty clay loam	>24
Peotone silty clay loam	>24
Peotone mucky silty clay loam, marl	>100
substratum	
Petrolia silty clay loam	15-24
Piasa silt loam	15-24
Pierron silt loam	8-15
Pike silt loam	8-15
Pillot silt loam	15-24
Piopolis silty clay loam	15-24
Piscasaw silt loam	15-24
Plainfield sand	<8
Plano silt loam	15-24
Plattville silt loam	15-24
Plumfield silty clay loam	15-24
Port Byron silt loam	15-24
Port Byron silt loam, sandy substratum	15-24
Prairieville silt loam	15-24
Princeton fine sandy loam	8-15
Proctor silt loam	15-24
Proctor silt loam, sandy substratum	15-24
Prophetstown silt loam	15-24
Quiver silty clay loam	15-24
Racoon silt loam	8-15
Raddle silt loam	15-24
Raddle-Sparta complex	8-15
Radford silt loam	15-24
Rantoul silty clay	>24
Rapatee silty clay loam	>24
Raub silt loam	15-24
Raveenwash silty clay loam	8-15
Redbud silt loam	8-15
Redbud-Colp silty clay loams	8-15
Redbud-Hurst silty clay loams	8-15
Reddick silty clay loam	>24
Reesville silt loam	8-15
Rend silt loam	8-15
Richview silt loam	8-15
Richwood silt loam	15-24

Ridgeville fine sandy loam	8-15
Ridgway silt loam	8-15
Ridott silt loam	8-15
Riley silty clay loam	>24
Ringwood silt loam	15-24
Ripon silt loam	8-15
Ritchey silt loam	8-15
Robbs silt loam	8-15
Roby fine sandy loam	<8
Rocher loam	8-15
Rockton loam	8-15
Rockton and Dodgeville soils	8-15
Rodman loam	8-15
Rodman gravelly loam	<8
Rodman-Casco complex	8-15
Rodman-Fox complex	8-15
Rodman-Warsaw complex	8-15
Romeo silt loam	15-24
Rooks silt loam	>24
Ross loam	8-15
Rossburg loam	>24
Rowe silty clay	>24
Rozetta silt loam	8-15
Ruark fine sandy loam	<8
Rubio silt loam	15-24
Ruma silt loam	15-24
Ruma-Ursa silty clay loams	8-15
Rush silt loam	8-15
Rushville silt loam	8-15
Rushville-Huey silt loams	8-15
Russell silt loam	8-15
Rutland silt loam	15-24
Sabina silt loam	15-24
Sable silty clay loam	>24
Saffell gravelly sandy loam	8-15
Sarpy sand	<8
Saude loam	15-24
Sawmill silty clay loam	>24
Sawmill-Lawson complex	>24
Saybrook silt loam	15-24
Saylesville silt loam	8-15
Schapville silt loam	15-24
Schuline silt loam	8-15

Sciotoville silt loam	8-15
Seaton silt loam	8-15
Seaton silt loam, sandy substratum	8-15
Seaton-Goss complex	8-15
Seaton-Hickory complex	8-15
Seaton-Oakville complex	8-15
Seaton-Timula silt loams	8-15
Selma loam	15-24
Selmass loam	15-24
Selma loam, bedrock substratum	15-24
Senachwine silt loam	8-15
Sepo silty clay loam	15-24
Sexton silt loam	8-15
Shadeland silt loam	8-15
Shaffton loam	15-24
Sharon silt loam	8-15
Shiloh silty clay loam	>24
Shoals silt loam	8-15
Shullsburg silt loam	15-24
Sidell silt loam	8-15
Skelton fine sandy loam	8-15
Slacwater silt loam	8-15
Sogn silt loam	15-24
Somonauk silt loam	8-15
Sonsac very cobbly silt loam	8-15
Sparta loamy sand	<8
Sparta loamy sand, loamy substratum	<8
Springerton loam	15-24
Starks silt loam	8-15
Stockland loam	8-15
Stonelick fine sandy loam	<8
Stookey silt loam	15-24
Stookey and Timula soils	8-15
Stookey-Bodine complex	15-24
Stoy silt loam	8-15
Strawn silt loam	8-15
Strawn-Chute complex	8-15
Strawn-Hennepin loams	8-15
Streator silty clay loam	>24
Stronghurst silt loam	8-15
St. Charles silt loam	8-15
St. Charles silt loam, sandy substratum	15-24
St. Clair silt loam	8-15

Sunbury silt loam	15-24
Swanwick silt loam	8-15
Swygert silty clay loam	>24
Sylvan silt loam	8-15
Sylvan-Bold complex	8-15
Symerton loam	15-24
Symerton silt loam	15-24
Tallula silt loam	15-24
Tallulabold silt loams	15-24
Tama silt loam	15-24
Tamalco silt loam	8-15
Tama silt loam, sandy substratum	15-24
Tell silt loam	8-15
Tell-Lamont complex	8-15
Terril loam	15-24
Thebes silt loam	8-15
Thorp silt loam	15-24
Tice silty clay loam	>24
Timewell silt loam	15-24
Timewell and Ipava soils	15-24
Timula silt loam	8-15
Timula-Hickory complex	8-15
Timula-Miami complex	8-15
Titus silty clay loam	>24
Toronto silt loam	15-24
Torox silt loam	15-24
Traer silt loam	8-15
Trempealeau silt loam	15-24
Troxel silt loam	15-24
Tuscola loam	8-15
Twomile silt loam	8-15
Udolpho loam	15-24
Udolpho loam, sandy substratum	15-24
Uniontown silt loam	8-15
Ursa silt loam	8-15
Ursa-Atlas complex	8-15
Ursa-Hickory complex	8-15
Vanmeter silty clay loam	15-24
Vanpetten loam	15-24
Varna silt loam	15-24
Velma loam	8-15
Velma-Coatsburg silt loams	15-24
Velma-Walshville complex	8-15

Vesser silt loam	15-24
Virden silty clay loam	>24
Virden-Fosterburg silt loams	>24
Virden-Piasa silt loams	>24
Virgil silt loam	8-15
Wabash silty clay	>24
Wagner silt loam	8-15
Wakeland silt loam	8-15
Wakenda silt loam	15-24
Wallkill silty clay loam	>24
Wallkill silt loam	15-24
Ware silt loam	15-24
Warsaw silt loam	15-24
Washtenaw silt loam	15-24
Watseka loamy fine sand	<8
Wauconda silt loam	8-15
Wauconda and Beecher silt loams	8-15
Wauconda and Frankfort silt loams	8-15
Waukee loam	8-15
Waukegan silt loam	15-24
Waupecan silt loam	15-24
Wea silt loam	15-24
Weinbach silt loam	8-15
Weir silt loam	8-15
Wellston silt loam	8-15
Wellston-Berks complex	8-15
Wenona silt loam	15-24
Wenona silt loam, loamy substratum	15-24
Wesley fine sandy loam	8-15
Westland clay loam	15-24
Westmore silt loam	8-15
Westmore-Neotoma complex	8-15
Westville silt loam	8-15
Whalan loam	8-15
Whalan and NewGlarus silt loams	8-15
Wheeling silt loam	8-15
Whitaker silt loam	8-15
Whitaker variant loam	8-15
Whitson silt loam	8-15
Wilbur silt loam	8-15
Will silty clay loam	>24
Windere silt loam	15-24
Winfield silt loam	8-15

Wingate silt loam	8-15
Winnebago silt loam	15-24
Wirt silt loam	8-15
Woodbine silt loam	8-15
Worthen silt loam	15-24
Wyanet silt loam	8-15
Wynoose silt loam	8-15
Wynoose silt loam, bench	8-15
Wynoose-Huey complex	8-15
Xenia silt loam	8-15
Zanesville silt loam	8-15
Zanesville-Westmore silt loams	8-15
Zipp silty clay loam	15-24
Zook silty clay	>24
Zumbro sandy loam	8-15
Zurich silt loam	15-24
Zurich and Morley silt loams	15-24
Zurich and Nappanee silt loams	15-24
Zwingle silt loam	8-15