MEASUREMENT AND SIMULATION OF SOLUTE TRANSPORT IN A HUMMOCKY LANDSCAPE

BY

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A Thesis
Submitted to the Faculty of Graduate Studies
at the University of Manitoba
in Partial Fulfilment of the Requirements
for the Degree of

DOCTOR OF PHILOSOPHY

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ABSTRACT

Olatuyi, Solomon Olalekan, Ph.D., The University of Manitoba, December 2010. Measurement and Simulation of Solute Transport in a Hummocky Landscape. Major Professor; Olalekan O. Akinremi.

Due to the complexity of nitrogen dynamics in the soil, tracer techniques are employed to estimate the fate and transport of nitrate-nitrogen in agricultural fields. This study was conducted to examine effects of N fertilization and landscape position on two-dimensional redistribution of bromide in a hummocky landscape, and to identify the landscape position with the greatest potential for solute loss using a dual application of Br^- and labelled ^{15}N. The field data on Br^- transport was also simulated using the HYDRUS models. The study was carried out near Brandon, Manitoba in 2007 and 2008, using two separate plots denoted as Site-2007 and Site-2008, respectively. The field plot was delineated into three landscape positions as upper (UPP), middle (MID) and lower (LOW) slope. The microplots demarcated at each landscape position received labelled ^{15}N fertilizer in form of KNO_3 at the rates of 0, 90 and 135 kg N ha^{-1}, and KBr at the rate of 200 kg Br^- ha^{-1}. Site-2007 was seeded to canola while Site-2008 had winter wheat. Soil samples were taken in the fall and the following spring and were analyzed for Br^-, NO_3-N, total N, and isotope N ratio. Nitrogen fertilization reduced the downward movement of Br^- in the soil profile, resulting in a greater lateral movement of Br^- compared to the unfertilized plots. The greatest vertical and lateral movement of Br^- occurred at the LOW slope compared to other landscape positions. The study confirmed the "Campbell hypothesis" which states that proper N fertilization reduces nitrate leaching. In the dual-tracer experiment, the smallest amounts of Br^-, ^{15}N, and NO_3-N were measured in the soil at the LOW slope, while the greatest amounts were at the MID slope; indicating that solute loss was: LOW > UPP > MID. Between fall and spring season, Br^- and ^{15}N had declined in the soil profile, but NO_3-N distribution remained unchanged. In the absence of crop uptake, Br^- transport was identical to that of ^{15}N. These findings have implications for precision farming management, as it will not be advisable to add excessive rates of N fertilizer to the lower slope position. The simulation study showed that HYDRUS-1D model was inadequate to describe solute transport in the landscape, as HYDRUS-2D/3D reproduced the field data better than HYDRUS-1D. However, the 2D model did not reflect effects of landscape position and N fertility on Br^- transport. Overall, the field experiment and model simulation both showed that downward movement is the main pathway of solute loss in the landscape. This study shows that it is possible to obtain a better understanding of factors controlling nitrate leaching in the sub-humid region of the Canadian prairie by using a model simulation to complement field data.

ACKNOWLEDGMENTS

My profound gratitude goes to my supervisor, Dr. Wole Akinremi, for providing me the opportunity to accomplish my scholarly ambitions as a Soil Scientist under his supervision. My appreciation also extends to members of my advisory committee: Drs. Don Flaten, David Lobb, and Martin Entz – for their advisory roles towards the successful completion of my program.

Many thanks to the Manitoba Rural Adaptation Council (MRAC) for sponsoring the field study. Thanks to the Manitoba Zero Tillage Research Association (MZTRA) for the opportunity to conduct the field experiment at their research farm. I also acknowledge the assistance received from Lindsay Coulthard, the Farm Manager at the MZTRA farm, and Dr. Al Moulin, Agriculture and Agri-Food Canada at Brandon. I acknowledge Dr. Francis Zvomuya, Department of Soil Science, for assistance with the statistical analyses, Dr. Sheng Li, for assisting with the landscape segmentation program, and Dr. Jirka Simunek, University of California Riverside, Riverside, CA, for technical assistance with the HYDRUS software.

Thanks to Dr. X. Hao and Clarence Gilbertson at the Agriculture and Agri-Food Canada at Lethbridge, for help with isotope analyses. I also acknowledge the following technicians in the Department of Soil Science (Tim Stem, Bo Pan, Rob Ellis, and Mulikat Bammeke) for technical supports in the field and in the laboratory. I appreciate the financial awards received from the Faculty of

Graduate Studies (International Graduate Students Supplementary Scholarships and Conference Travel Awards) and the Faculty of Agricultural and Food Science Endowment Scholarship and Bursary. Thanks to members of the Soil Chemodynamics Research Lab, the staff and graduate students at the Department of Soil Science, University of Manitoba.

My success became a reality because of the love, prayers and supports from my wife, Oluwa-Fisayo, and the wonderful gift of God in our lives, our son Oluwa-Daramola. My praises to the Lord Jesus, the Author and the Finisher of my Scripts.

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1. REVIEW OF LITERATURE

1.1 Introduction

On addition of nitrogen fertilizer or manures to agricultural soils, the N component undergoes various biological transformations initiated by microbial activities. Such transformations include mineralization, immobilization, nitrification, and denitrification (Schepers and Mosier 1991; Wilkinson et al. 2000). While mineralization generally increases the amount of available N in the soil, nitrification and immobilization tend to redistribute the speciation of N in the soil as nitrate-nitrogen (NO_3-N) and organic N forms, respectively. The plant-available or inorganic forms of N in soils are ammonium (NH_4^+) and nitrate (NO_3^-). In warm, well-drained arable soils, NO_3-N is more prevalent due to active nitrification processes (Jarvis 1996).

Nitrate is highly soluble and mobile in the soil (Nielsen et al. 1986). Therefore, NO_3-N can be easily leached from agricultural systems through soil water movement. Leaching is defined as a downward transport of soil solutes out of the root zone by water flow. Once the NO_3-N is leached below the root zone, it is difficult to recover and may subsequently reduce crop growth and nitrogen-use efficiency particularly if leaching occurs early in the growing season (Follett 1989; Follett and Walker 1989). Leaching losses of nitrate from agricultural systems also result in the contamination of groundwater and surface water bodies due to elevated concentrations of NO_3-N (Addiscott et al. 1991; Townsend et al. 2003).

Human consumption of nitrate-contaminated groundwater can be detrimental because of associated ailments such as methemoglobinemia in infants ("Blue Baby Syndrome") and intestinal cancer (Shuval and Gruener 1972). The maximum contaminant level (MCL) for drinking water from groundwater source is approximately 10 mg NO_3-N L^{-1}, according to the World Health Organization (WHO) regulation (European Communities 1980, 1991). A major part of rural Manitoba in Canada relies on groundwater as a source of domestic and industrial water supply. It is very important to protect this water resource from nitrate contamination because of the critical impacts of groundwater source on the physical well being of the people in the community.

Excessive amounts of nitrate in groundwater discharge to surface waters can also lead to eutrophication and adverse changes in ecological functioning and food webs in the aquatic systems (National Research Council 2000). In combination with phosphorus, NO_3-N concentrations as low as 0.3 mg L^{-1} can trigger the development of algal blooms in surface waters (Brooks et al. 1991). For instance, nitrate-nitrogen has been identified as the primary pollutant contributing to the growth of zones of hypoxia in the Louisiana continental shelf and the Mississippi River watershed of the Gulf of Mexico (NOAA'S National Ocean Service 2003). The resulting low oxygen status, due to NO_3-N loading into the watershed, has a negative effect on the economic, recreational and ecological importance of these water bodies.

At the global scale, the rates of leaching loss of nitrate from farmlands range from 10 to 15 kg N ha^{-1} yr^{-1} (Smil 1999). The average rate of nitrate leaching from Canadian agricultural systems is approximately 0.43 Tg N yr^{-1} (Janzen et al. 2003). Leaching losses of NO_3-N vary widely among regions. Under the prairie agricultural systems in western Canada, Janzen et al. (2003) reported that NO_3-N leaching is important in the sub-humid regions, while leaching may be minimal in the semi-arid regions because of high moisture deficit and evapotranspiration. However, previous studies have shown that significant leaching losses of NO_3-N may occur under irrigation (Chang and Janzen 1996) or from fallow fields (Campbell et al. 1984).

Soil physical properties and topographic features are the main factors controlling soil water flow and solute redistribution within agricultural landscapes (Bathke and Cassel 1991; Bathke et al. 1992; Afyuni et al. 1994; Olson and Cassel 1999). For leaching of nitrate to occur, a sufficient amount of water must be available for the transport of nitrate within the soil profile (Campbell et al. 1984). Although the possibility of nitrate leaching in agricultural landscapes depends on the soil moisture content, availability of soil moisture alone does not always translate into nitrate leaching.

Studies have shown that N fertilizer management practices and cropping systems also have a significant effect on nitrate leaching in agricultural soils, depending on the design of soil fertility program in place (Randall and Iragavarapu 1995). Under some management practices, Campbell et al. (1984, 1993) demonstrated that adequate N fertilization may reduce NO_3-N leaching because of improved root growth and crop water utilization.

Considering the factors mentioned above, it is intuitive to assert that nitrate can be leached from agricultural soil if available soil N exceeds plant requirements. Also, nitrate leaching is most likely to occur in the absence of vegetative growth that is capable of utilizing the soil NO_3-N before excess water moves it beyond the root zone (Follett 1989). To enhance our present understanding of nitrate leaching in agricultural landscapes, the fate and transport of nitrate should be investigated with respect to N fertility management and spatial variability across the landscape.

This research study was conducted: (i) to examine effects of N fertilization on two-dimensional redistribution of bromide within a hummocky landscape in fall and spring seasons; (ii) to investigate the vertical distribution of NO_3-N across this landscape using a dual application of bromide and labelled ^{15}N; (iii) to identify the topographical position with the greatest potential for NO_3-N leaching in a hummocky landscape; (iv) to simulate the field experiment on solute transport using mechanistic models such as HYDRUS-1D and HYDRUS-2D/3D (Sejna and Simunek 2007; Simunek et al. 1998, 1999, 2005, 2006).

The findings generated from this study are presented in three sections in order to capture the outcome of the objectives stated above. Briefly, the first section describes effects of N fertilization on two-dimensional redistribution of bromide in the landscape. In the second section, the dual application of bromide and labelled ^{15}N was employed to investigate the vertical movement of NO_3-N. The third section deals with the numerical modelling of water flow and solute transport in the landscape.

1.2 Mechanisms of Nitrate Transport in Agricultural Soils

Due to the soluble nature of nitrate, the transport of NO_3-N in soil is governed by water flow within the soil pore spaces (Nielsen et al. 1986). Therefore, the modes of nitrate transport in soil are predominantly by advection (mass flow) and dispersion (Jury et al. 1991; Leij and van Genuchten 1999; Vitousek et al. 2002).

Advection is defined as the flux of solute due to the movement of water containing the solute. It is expressed as the product of water flux ("Darcian flow") and the solute concentration (c). The mathematical definition for advection is as follows:

$$\left(\frac{\partial c}{\partial t}\right)_m = q_w c \qquad [1.1]$$

$$q_w = -K\,\nabla H \qquad [1.2]$$

where $(\partial c/\partial t)_m$ is the advective flux; q_w is the water flow per unit cross sectional area per unit time, also known as flux density; K is the soil hydraulic conductivity, which is a non-linear function of soil moisture content (θ) and matric potential (Φ_m); and ∇ is the gradient in x, y, and z directions; ∇H is the three-dimensional hydraulic gradient. In a case where the soil moisture content is constant, the water flux is expressed as:

$$q_w = v\theta \qquad [1.3]$$

where v is the pore water velocity.

Advective transport of solute occurs through the soil pore spaces, and therefore depends on matrix flow of soil water. Chemical transport by advection is based on the assumption that both solute and water travel through uniform straight-line flow paths in the soil pores, leading to piston flow (Vitousek et al. 2002). In reality, flow paths are usually irregular and tortuous. As such, the solute fronts are variably localized within the pore channels (Lal and Shukla 2004). Due to the irregular soil pore space geometry, some of the incoming solution may be ahead or behind relative to the resident solution.

The process of solute transport in soil also involves diffusion and dispersion. Diffusion is the spontaneous movement of solute along a concentration gradient based on random, thermal ("Brownian") motion of solute molecules. According to Fick's Law, the diffusive transport of solute in one-dimensional (x-coordinate) direction is described as:

$$\left(\frac{\partial c}{\partial t}\right)_d = -D_d\left(\frac{\partial c}{\partial x}\right) \qquad [1.4]$$

where $(\partial c/\partial t)_d$ is the diffusive flux; D_d is the diffusion coefficient in soil; and $(\partial c/\partial \mathrm{x})$ is the concentration gradient. The diffusion coefficient can be influenced by other factors such as the diffusion coefficient of the solute in pure water, D_o, the tortuosity of the flow paths in soil, τ, as well as the soil moisture content, θ:

$$D_d = D_o\left(\frac{\theta}{\tau}\right) \qquad [1.5]$$

Due to the inverse relationship between the tortuous flow paths and the diffusion coefficient in soil, D_d is usually less than D_o.

Dispersion is the spreading out of solute due to variations in water velocity within individual pores, across pores with differing sizes and shapes, and across interconnected pore pathways with different geometries (Mulla and Strock 2008). By mathematical definition, dispersive flux is expressed as:

$$\left(\frac{\partial c}{\partial t}\right)_h = -D_h\left(\frac{\partial c}{\partial x}\right) \qquad [1.6]$$

where $(\partial c/\partial t)_h$ is the solute flux due to dispersion; D_h is the dispersion coefficient, which increases linearly with increase in the velocity of pore water:

$$D_h = \lambda v \qquad [1.7]$$

where λ is the coefficient of proportionality known as dispersivity.

Diffusion and dispersion produce similar effects on solute transport in soil, i.e. they both tend to mix and eventually eliminate non-uniformity in solute concentration in the soil solution. However, the basic mechanisms by which they occur are different. While diffusion process is predominant at low to zero soil water flow velocity, hydrodynamic dispersion exceeds chemical diffusion at high flow velocity particularly in large pore spaces.

Advection, chemical diffusion, and hydrodynamic dispersion are the three basic mechanisms of solute transport in the soil. The general equation for one-dimensional solute transport with a steady water flow in a homogeneous porous medium is described by advection-dispersion equation:

$$\left(\frac{\partial c}{\partial t}\right)_s = D_{dh}\left(\frac{\partial^2 c}{\partial x^2}\right) - v\left(\frac{\partial c}{\partial x}\right) \qquad [1.8]$$

where $(\partial c/\partial t)_s$ is the advective-dispersive solute flux; D_{dh} is the diffusive-dispersive coefficient, which is defined as:

$$D_{dh} = D_d + D_h \qquad [1.9]$$

Equation 1.8 assumes that there is no interaction between the solute and the soil solid phase. Examples of such solute are bromide and chloride ions, which are usually employed in tracer studies to simulate soil water flow and nitrate leaching in soils (Jury and Horton 2004).

In well-structured field soils, solute transport does not always occur through the soil matrix. These soils contain large pores and channels characterized by high soil water flux. Therefore, water flow and transport of non-reactive solutes in structured soils can be very rapid and turbulent due to the presence of large pores. The turbulent, rapid transport mechanism is known as preferential flow.

Preferential flow is defined as a transport mechanism in which non-uniform, rapid flow of water and transport of dissolved solutes occur through preferred pathways within the soil profile to a certain point below the root zone (Strock et al. 2001; Mulla and Strock 2008). In the preferential flow process, the solute does not have sufficient time to interact with the soil matrix due to the rapid and turbulent pattern of flow in the soil pores. Therefore, the preferential flow phenomenon is also referred to as physical non-equilibrium transport.

Preferential flow is the principal mechanism responsible for accelerated movement of solutes such as NO_3-N, in many agricultural soils (Luxmoore 1991; Li and Ghodrati 1994). Three types of preferential flow exist in the soil system, namely: (i) bypass flow through the soil macropores (Watson and Luxmoore 1986); (ii) funnel flow; (iii) finger flow due to wetting front instability. Flow in macropores is the commonest type of preferential flow in agricultural soil. Macropores are described as relatively large and more or less continuous voids through which rapid flow can occur (Nielsen et al. 1986). The diameters of macropores can range from 0.03 to 30 mm (Beven and Germann 1982). However, the size of macropores is less important for preferential flow compared to the continuity of the pore channels (Bouma 1981).

1.3 Factors affecting Nitrate Leaching in Agricultural Landscape

The factors controlling nitrate leaching in agricultural soils are broadly categorized into those due to natural processes and those attributed to management practices. The natural factors are practically uncontrollable as they evolve from effects of soil heterogeneity, landscape attributes, and climatic conditions on soil water dynamics and N cycling. In contrast, farm management practices and cropping systems can be tailored to improve agronomic productivity and to minimize the environmental hazards associated with nitrate leaching. The mechanisms by which these factors affect nitrate leaching in soils are discussed in the subsequent sections.

1.3.1 Effects of Spatial Variability in Soil Physical Properties on Nitrate Leaching

Soil water flow and solute transport are directly influenced by soil physical and hydraulic properties such as the bulk density (Mapa and Pathmarajah 1995), soil structure and particle size distribution, soil water retention, hydraulic conductivity, and macroporosity (Bathke and Cassel 1991; Olson and Cassel 1999; Mohanty and Mousli 2000; Strock et al. 2001). The spatial variability and abrupt changes in soil physical properties within the vadose zone are part of the factors controlling nitrate leaching in agricultural soils (Power et al. 2001).

A three-year study was conducted by van Es et al. (2006) to quantify N losses from liquid manure added to clay loam and loamy sand soils, both seeded to maize (*Zea mays* L.). Mean drain water NO_3–N concentration from the loamy sand soil was 2.5 folds greater than that from the clay loam plots. This indicated that the loamy sand soils were more prone to NO_3–N leaching compared to the clay loam soils. These differences were attributed to greater hydraulic conductivity and smaller soil water retention in the coarse-textured loamy sand soils, compared to the clay loam. The greater potential for NO_3-N leaching in the loamy sand soil may also be due to the greater mineralization of N from the organic pools in well-drained soils (Magdoff 1978) and smaller denitrification potential (Sogbedji et al. 2001a, b), compared to the poorly-drained, fine-textured clay loam.

1.3.2 Effects of Topographic Features on Soil Hydrologic Processes and Nitrate Leaching

To understand the fate and transport of nitrate in an agricultural landscape, knowledge of the variability in soil water transmission and effects of topographic attributes on soil water flow are important (Afyuni et al. 1994; Mohanty and Mousli 2000). Topographic attributes are the principal factors controlling soil hydrologic processes such as dissipation, transmission and accumulation of water, as well as solute redistribution within the landscape, and ultimately, the magnitude of nitrate leaching (Bathke and Cassel 1991; Bathke et al. 1992; Afyuni et al. 1994; Farrell et al. 1996; Olson and Cassel 1999; Manning et al. 2001a, b).

Topographic attributes affecting soil water distribution within the landscape are slope length, slope gradient, and slope curvatures (Sinai et al. 1981). These landscape features determine the quantity and rates of lateral flow of water and the subsequent infiltration into the soil profile. Infiltration is the most important aspect of soil hydrologic processes in agricultural soils. It is described as the entry of water into the soil through the soil surface. The critical soil hydraulic properties controlling the rate of infiltration are the hydraulic conductivity of the soil profile, the antecedent soil moisture content and the matric potential gradient. Infiltration also depends on soil physical properties such as the soil structure and the textural composition (Hillel 1998; Lal and Shukla 2004).

Rockstrom et al. (1999) showed that the infiltration rate near the soil surface determines the soil water flux, i.e. volume of water per unit area per time and the redistribution of the associated solutes in the soil profile. This implies that infiltration has a significant effect on water supply into the soil, as well as the entry of soluble ions into the subsurface and groundwater bodies (Bathke and Cassel 1991; Bathke et al., 1992).

1.3.3 Effects of Topographic Features on Nitrogen Transformations and Nitrate Leaching

Across a typical landscape, there are significant variations in the potential for N mineralization, immobilization, denitrification, and NO_3-N leaching. These variations are partly due to large differences in soil properties that tend to modify soil moisture balance, soil N dynamics, and organic carbon content (Pennock et al. 1987; Manning et al. 2001a, b). Pennock and Vreeken (1986) found that soil organic carbon varied in a predictable manner within the landscape. It is expected that the variation in soil organic carbon will affect the water holding capacity of the soil (Gollany et al. 1992), as well as the potentially mineralizable nitrogen (Fiez et al. 1995).

Soil organic carbon also affects the magnitude of denitrification process, which is the alternative pathway for soil NO_3-N loss (Rice and Rogers 1993). Nitrate leaching and denitrification are the two major pathways of NO_3-N loss from the agricultural system, as both processes result in localized removal of NO_3-N from the rooting zone. Anaerobic conditions due to soil saturation or compaction and available soil organic carbon are the prerequisites for denitrification process. The reduction of nitrate leaching by denitrification process has been recognized as an important mechanism modifying nitrate dynamics in agricultural landscape (Steinheimer et al. 1998).

The lower slope position may be the most vulnerable segment of the landscape to nitrate leaching due to deposition of soil materials removed from upslope regions and deeper penetration of water into the soil profile compared to the upper slope (Manning et al. 2001a, b). On the contrary in poorly drained landscapes, the presence of high water table and accumulation of water dissipated from upper regions of the landscape have been shown to enhance the denitrification of nitrate at the lower slope position at the expense of leaching losses (Schepers and Mosier 1991; Farrell et al. 1996).

1.3.4 Effects of Factors due to Weather, Climate and Seasonal Variability

The quantity and intensity of precipitation affect the soil water flux and the available soil moisture. The interactions of precipitation and the ambient temperature subsequently affect plant water uptake, soil drainage condition, denitrification, mineralization and nitrification, and ultimately, crop N utilization and nitrate leaching. The individual phase of N transformations in the N cycle can be interrupted by abnormally high or low soil temperatures, and also by soil moisture deficit or excess water supplies (Myrold and Bottomley 2008). The potentials for mineralization and nitrification of soil organic N are also suppressed under low or freezing soil temperature regime.

Regional variability in climate patterns also influences the intensity of nitrate leaching in agricultural soils. This can be associated with variation in soil water consumption through evapotranspiration. Evans et al. (1994) hypothesized that the depth of water and NO_3-N penetration would be inversely related to mean potential evapotranspiration of the area. The study was conducted over a climate gradient composed of three sites. The field sites selected represent a potential evapotranspiration gradient from 1,000 to 1,900 mm yr^{-1}. Although their results were not consistent among experimental sites, the findings obtained from two out of the three study sites confirmed the hypothesis.

The characteristics of weather and climate pattern of a region have a strong influence on biomass production, which consequently affect nitrate availability through mineralization, crop uptake, and leaching potentials in the soil (Power et al. 2001). The soils in the more humid, warm climates are more prone to nitrate leaching, compared to those in the semi-arid regions. In addition to regional variability, nitrate leaching in agricultural systems is controlled by temporal variation in biomass production and net evapotranspiration. Stewart (1970) stated that in the Northern dryland agricultural systems, soil water and NO_3-N move below the root zone in early spring season shortly after snowmelt, when plant growth is minimal and evapotranspiration is low.

Timing of N fertilization also has a profound implication on NO_3-N leaching in soils. van Es et al. (2006) performed a time-dependent fertilization experiment by applying dairy manure to lysimeter plots seeded to maize (*Zea mays* L.) at various seasons as follows: early fall, late fall, early spring, and a split application in early and late spring. The order of mean NO_3–N concentrations in the drain water among application seasons was: early fall > late fall > early spring = split applications. The trend in NO_3–N accumulation among application seasons suggests that fall application of manure on maize had a high risk potential for NO_3–N leaching, particularly in well drained soils. The absence of vegetative growth and lower soil temperature regimes following the application of N in the fall season (Tiessen et al. 2006) may be responsible for the accumulation of NO_3-N from the added manure.

1.3.5 Effects due to Soil Management Practices and Cropping Systems

In dryland agroecosystems, the extent of nitrate leaching below the root zone can be intensified under stubble mulch surface conditions. Evans et al. (1994) attributed the increased potential for nitrate leaching under stubble management to two factors: (i) protection of the soil surface against raindrop impacts, which helps to maintain the surface soil structure, thereby increasing infiltration and reducing runoff; (ii) trapping and retention of snow by stubble and crop residue (Smika and Unger 1986). As such, soil surface conditions that enhance the retention and entry of water into the soil profile contribute to nitrate leaching.

Zero tillage is an effective method of conserving soil and water on the Canadian prairies, particularly in the agricultural zones where soil moisture deficit limits crop production (Campbell et al. 1984; Malhi et al. 1996). Under a zero tillage system, however, the presence of a myriad of continuous pores due to the undisturbed soil condition can provide access for rapid movement of NO_3-N via macropore flow into the region below the root zone (Addiscott et al. 1991).

It is common to assert that there is a minimal risk of NO_3-N leaching on the Canadian prairie, where moisture deficit is large. However, studies have shown that significant leaching of nitrate can occur from both cropped and summer-fallowed soils during years in which precipitation is above average and when there is a build-up of soil moisture (Campbell et al. 1984; Izaurralde et al. 1995).

In semi-arid regions of western Canada, Campbell et al. (1984) showed that summer fallow practices can result in groundwater contamination by NO_3-N leaching. Their results indicated that there was more NO_3-N accumulation both within and below the root zone in fallow soils than those that were cropped continuously. Therefore, incorporating a fallow system into a cropping sequence can markedly intensify NO_3-N leaching in the soil profile.

The type of crop rotation program adopted in a cropping system can influence the magnitude of NO_3-N leaching in soils. Campbell et al. (1992) showed that there was less NO_3-N leached below the root zone on an Orthic Brown Chernozemic silt loam under lentil (*Lens culinaris* medikus) and wheat (*Triticum aestivum* L.) rotations, compared to continuous wheat receiving nitrogen and phosphorus fertilizer. This was attributed to an enhanced synchronization between N supply from the legume and the demand for N by the cereal, as compared to the cereal-cereal rotation scheme.

The magnitude of NO_3-N leaching also depends on plants morphology and physiology. Campbell et al. (1984) showed that fall rye (*Secale cereale*) was more effective in reducing the amount of NO_3-N leached compared to continuous cropping with spring wheat (*Triticum aestivum* L.). The attenuation of nitrate leaching by rye was attributed to its deeper root system and vigorous growth pattern compared to spring wheat.

1.3.6 Effects of Nitrogen Fertilizer Management on Nitrate Leaching

In attempts to meet the demands of modern agricultural production, large amounts of N are often applied to the soils. Excessive N fertilization of agricultural crops can increase the potential for NO_3-N loading into groundwater and surface water bodies (Campbell et al. 1994), particularly if N application is followed by above normal precipitation (Liang and MacKenzie 1994). The negative implications of agricultural production on surface water and groundwater qualities have generated increasing interest and growing recognition of new methods for N fertilizer management that incorporate environmentally safe approaches (Bergstrom and Kirchmann 2004).

Application of N fertilizer at recommended rates has the potential to reduce excess NO_3-N that is susceptible to leaching. This is attributed to increased crop water utilization and N uptake at the recommended N rate as reported by Campbell et al. (1993). The authors compared effects of six rates of N fertilizer (0, 25, 50, 75, 100 and 125 kg N ha^{-1}) on NO_3-N leached in a plot seeded to spring wheat in the Brown soil zone of Canadian prairies. Their results showed that the greatest amount of NO_3-N was leached beyond the root zone at lower N rates (0, 25 kg N ha^{-1}), which was similar to that under the highest N rate (125 kg N ha^{-1}). The smallest amount of nitrate was leached at 100 kg N ha^{-1}, followed by that at the rates of 50 and 75 kg N ha^{-1}. As such, the moderate rates of N fertilizer reduced NO_3-N leaching compared to the lowest and highest N rates.

The large amount of NO_3-N leaching observed at the lower N rates was attributed to reduced crop water utilization due to poor tillering and root growth at these rates, while the leaching under the highest N rate was due to the presence of NO_3-N in excess of crop uptake (Campbell et al. 1993). The poor tillering and root growth at lower N rates resulted in reduced evapotranspiration, compared to that at higher N rates. Hence, there was more soil water available for inducing net N mineralization and NO_3-N leaching at the lower N rates.

Modifying the form and composition of N fertilizer can minimize the magnitude of NO_3-N leaching from the added N. One of the management practices for achieving this strategy was demonstrated by Guillard and Kopp (2004). The authors investigated the differences in the amount of NO_3–N leaching from lawn turf that received different formulations of N fertilizers. The treatments consisted of four fertilizer sources as: (i) ammonium nitrate (AN), containing all soluble N; (ii) polymer-coated sulphur-coated urea (PCSCU), approximately 15.1% slow-release N; (iii) organic product from composted poultry litter, with 0.2% water soluble N; (iv) a non-fertilized control treatment.

After correcting for losses from the control treatment, average annual NO_3–N leaching losses as percent of N applied were 16.8% for AN, 1.7% for PCSCU, and 0.6% for the organic source. Guillard and Kopp (2004) suggested that formulating N fertilizer with a large proportion of slow-release N is a key strategy for reducing NO_3–N leaching losses. However, slow-release N may not be effective for controlling nitrate leaching in coarse-textured soils under humid conditions. The bulk of N fertilizer can be lost to leaching due to high soil moisture flux prevalent under these conditions.

Nitrogen fertilization using green manures from cover crops is a possible way to reduce leaching of NO_3-N from agricultural soils (Owens et al. 1994). Green manures include legumes such as alfalfa (*Medicago sativa* L.), clover (*Trifolium repens* L.), and peas (*Pisum sativum* L.); and non-legumes such as ryegrass (*Lolium perenne* L.) and winter wheat (*Triticum aestivum* L.). However, production of sufficient N from green manures to supply crops demands during the active growing season remains an agronomic challenge, particularly in cold, temperate regions where the decomposition of organic residue is relatively slow. An environmental concern with the use of green manure is that, if the N release from the green manure occurs too late in the season, or after the growing season, it may leach through the soil in a magnitude that can trigger groundwater contamination (Macdonald et al. 1989).

1.3.7 Reducing Nitrate Leaching by Site-Specific Management Practices

Due to spatial variability in soil properties across the field, it has been suggested that site-specific farming methods that would manage individual regions of the field according to its needs, rather than treating the field as a unit, should be adopted (Power et al. 2001). Site-specific management practices are considered as an attractive and intuitive approach for reducing NO_3-N leaching losses, by modifying the fate of N in the soil in order to increase N fertilizer use efficiency (Sawyer 1994; Ferguson et al. 2002). Improving the NO_3-N fertilizer use efficiency of crops will ultimately enhance agronomic productivity and environmental sustainability in N fertilization program and crop production.

Some of the strategies employed in achieving an effective site-specific management practices are precision farming techniques such as variable rate fertilizer, split N application and soil N test (Beckie et al. 1997; Ferguson et al. 2002). Site-specific management practices involve delineating a cropped field into management units in an attempt to increase the efficiency of crop production and environmental sustainability (Power et al. 2001). As such, areas with similar productive potential are treated alike by applying optimum amount of inputs to obtain the best economic return. In traditional, non-precision farming systems, farmers generally treat the whole agricultural field as a single unit, by applying similar management practices over the entire area. The negative implication of such farming practice is that the management program being used may not be appropriate for many parts of the field.

Application of similar management practices over a field with large spatial variability often result in adverse effects on crop yields, economic profitability and water quality from those areas that require special treatment. Special tools such as Geographical Information System (GIS) have been employed to examine effects of soil variability on soil fertility status. For example, researchers (Blackmer and Schepers 1995; 1996) have used remote sensing techniques to assess and map the impact of spatial variability on soil and crop N status in an attempt to improve site-specific management practices over the area. Their results indicated that aerial photographs provided reliable information on soil and crop variability.

It is important to improve the current knowledge on site-specific management practices through field experimentation, model simulation and remote sensing technology. A combination of these strategies can effectively minimize the adverse effects of NO_3-N leaching on groundwater quality.

1.4 Field-Scale Methods of Monitoring Nitrate Leaching in Soil

Various methods have been employed to estimate NO_3-N leaching in the vadose zone of agricultural soils. Among these methods, field core lysimeters and soil coring techniques are more popular (Pampolino et al. 2000; Zotarelli et al. 2007). Lysimeters are containers with various shapes, sizes and materials, installed in the field for measuring nitrate flux and concentration in water that has percolated through the soil profile. The commonest types used in field studies of nitrate leaching are the suction lysimeters and the drainage lysimeters (Webster et al. 1993).

Once installed, lysimeters can allow for repeated measurements from the same location without disturbing the soil. A major disadvantage of lysimeters is that the sampling region is under a controlled boundary condition due to the physical barriers imposed on the flow domain. With suction cup lysimeters it is difficult to obtain mass balance of N at a single point in time, while the installation of drainage lysimeters may result in considerable soil disturbance.

The soil coring technique is simple, relatively cheap, widely used, and applicable to most soils. However, some drawbacks are associated with this method. Soil coring is time-consuming, labour-intensive, destructive, and strongly influenced by spatial variability. It is also impossible to obtain a direct measurement of solute flux with the coring method. Although soil coring can provide information on N distribution within the soil profile and N balance at a point in time (Mulla and Strock 2008), this method is more suitable for monitoring NO_3-N transport when combined with tracer techniques and model simulation

(Willian and Nielsen 1989; Zotarelli et al. 2007). In order to achieve an effective sampling strategy, issues with sampling depth, spatial arrangements of sampling points, sample volume, time of sampling, and the anticipated number of samples should be considered in soil coring technique (Wollenhaupt et al. 1997).

1.5 Quantification of Nitrate Leaching using Tracer Techniques

Due to the complexity of different sources of N in the soil and transformation of N into various pools, the labelled isotope ^{15}N has been employed for investigating the transport and recovery of N in hydrologic and agricultural research (Olson and Swallow 1984; Jensen 1991, Mulla and Strock 2008). This practice is generally referred to as a tracer technique. In field experimentation of N cycling and transport, ^{15}N enrichment or depletion method permits the study of N dynamics by measuring the changes in N isotope ratios (Mulla and Strock 2008). However, the stable ^{15}N is susceptible to microbial immobilization, denitrification losses, and uptake by plants, just as the stable isotope ^{14}N.

It is also possible that a proportion of the added ^{15}N is incorporated into the soil organic N reserve with a subsequent release or mineralization of the native ^{14}N, thereby diluting the concentration of ^{15}N in the soil or converting part of the ^{15}N into organic forms (Jenkinson et al. 1985; Kessavalou et al. 1996). Therefore, the use of isotope technique for simulating N cycling and transport in agricultural soils should be conducted with caution.

To obtain detailed information on the influence of soil water flow on nitrate leaching, a combination of bromide (Br^-) and ^{15}N as dual tracers has been suggested for estimating NO_3-N distribution in soils (Kessavalou et al. 1996; Ottman et al. 2000; Ottman and Pope 2000). Bromide is used as a tracer for nitrate based on the following attributes: (i) Br^- is a conservative tracer that is not susceptible to microbial transformations and gaseous losses; (ii) Br^- has low background concentration in agricultural soils (Bowman, 1984); and (iii) similar to NO_3^-, Br^- is a monovalent anion. Therefore, Br^- and NO_3^- are generally assumed to undergo similar leaching patterns. The difference between the amount of Br^- applied and that recovered is calculated to estimate the amount of NO_3-N lost to leaching below the root zone (Smith and Davis 1974).

Nevertheless, there are differences between the behaviour of bromide and nitrate in the soil. For example, Kessavalou et al. (1996) showed that the magnitude of bromide leaching in the soil profile was greater than that of nitrate. The greater movement of bromide compared to NO_3-N can be attributed to the non-reactive behaviour of Br^- in the soil, unlike the reduction of nitrate movement due to microbial immobilization, gaseous losses and plant uptake of N (Kessavalou et al. 1996; Ottman and Pope 2000; Ottman et al. 2000). Therefore, in order to eliminate the drawback associated with each tracer, a better understanding of nitrogen dynamics can be obtained using both bromide and ^{15}N as dual-tracers.

1.6 Numerical Modelling of Nitrate Leaching in Agricultural Soils

Field-scale studies of water flow and chemical transport are time-consuming, expensive, and labour-intensive to conduct. To address these challenges, a variety of mechanistic models with varying degrees of complexity have been developed to simulate soil water flow and solute transport processes. Mechanistic models are developed based on the mathematical theories of flow and chemical transport in soil. The models have been applied to N cycling research in an attempt to synthesize and to improve the contemporary knowledge on NO_3-N leaching in agricultural soils (Addiscot and Wagenet 1985).

Due to the simplicity of the governing equations, mechanistic models based on numerical solution are increasingly used for predicting or analyzing water flow and chemical transport in soils compared to analytical models (Abbasi et al. 2004). Some of the important computational schemes in numerical models are calibration and validation. Calibration is a computational procedure for estimating the model parameters that cannot be easily measured or determined (Hanson et al. 1999). Validation involves using a set of independent experimental data for testing the performance of a calibrated model (Abbasi et al. 2004).

A few examples of mechanistic models commonly used for predicting nitrate and solute leaching in soils are LEACHMN (Hutson and Wagenet 1993), RZWQM (Hanson et al. 1999; Ahuja et al. 2000), NLEAP (Shaffer et al. 1991), and HYDRUS program (Simunek et al. 1998, 1999, 2005, 2006). However, many of these simulation models have not been widely tested in Canada; hence, their reliability under a variety of Canadian field conditions is not well-known (Akinremi et al. 2005).

HYDRUS-1D is a software package for simulating water flow and solute transport in variably saturated porous media (Simunek et al. 1998, 2005). The program is flexible with respect to the direction of flow; however, flow is only allowed in one dimension as vertical, horizontal, or inclined. HYDRUS-2D/3D model is an extended code of HYDRUS-1D in terms of its ability to analyze water flow and solute transport in two-dimensional horizontal plane, two-dimensional vertical plane, two-dimensional axisymmetrical vertical flow, or three-dimensional plane (Simunek et al. 2006; Sejna and Simunek 2007).

Major problems often encountered in mechanistic modelling are extensive efforts required for data preparation, numerical grid design, and graphical presentation of the outputs. These computational challenges have been eliminated in HYDRUS program. The HYDRUS models are built with the capacity to analyze water flow and solute transport using an interactive graphics-based user interface (GUI). The graphics-based interface is connected directly to the computational codes, which is compatible with the contemporary operating systems such as MS Windows 95, 98, NT, ME, Vista and XP environments.

HYDRUS-1D and HYDRUS-2D/3D models consider two forms of soil water flow: uniform flow (single porosity) and flow in a dual-porosity system. Both models describe uniform flow of water based on a modified form of the Richards' equation, which is solved numerically using the Galerkin-type linear finite element schemes (Simunek et al. 1998, 1999). The governing equation of transient, one-dimensional uniform water flow in HYDRUS is described as:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x}\left[K\left(\frac{\partial h}{\partial x} + \cos\alpha\right)\right] - S \qquad [1.10]$$

where h is the water pressure head; θ is the volumetric water content; t is time; x is the spatial coordinate; S is the sink term to account for water uptake by plant roots; α is the angle between the flow direction and the vertical axis (i.e., $\alpha = 0°$ for vertical flow, 90° for horizontal flow, and $0° < \alpha < 90°$ for inclined flow); and K is the unsaturated hydraulic conductivity function given by:

$$K(h, x) = K_{sat}(x) K_r(h, x) \qquad [1.11]$$

where K_r is the relative hydraulic conductivity; and K_{sat} is the saturated hydraulic conductivity.

To solve the Richards' equation, knowledge of soil hydraulic properties such as the soil moisture characteristic curve and unsaturated hydraulic conductivity function is required. In simulation models, these hydraulic properties are commonly expressed in form of analytical functions and often referred to as parametric models. A common approach for deriving the hydraulic parameters is by fitting the hydraulic functions to experimental water retention and conductivity data (Arya et al. 1999; Akinremi et al. 2005). Alternatively, the parameters can be directly estimated from experimental database for soil hydraulic properties known as the pedotransfer functions.

The pedotransfer functions approach correlates basic soil properties, such as the percent sand, silt, clay and organic carbon, and the water content held at certain hydraulic potentials (usually at -33 kPa and -1500 kPa) to estimate the hydraulic parameters. Five different analytical models are implemented in

HYDRUS program to generate hydraulic conductivity and retentivity functions. The models are developed according to Brooks and Corey (1964), van Genuchten (1980), Vogel and Cislerova (1988), Kosugi (1996), and Durner (1994).

The governing equation of solute transport in HYDRUS is based on the advection-dispersion model, which is also solved by the Galerkin finite element method. The equation for one-dimensional transport of a single ion is simplified as:

$$R\frac{\theta\partial c}{\partial t}=\theta D_w\left(\frac{\partial^2 c}{\partial x^2}\right)-q\left(\frac{\partial c}{\partial x}\right)+Fc+G \qquad [1.12]$$

where c is the solute concentration in the liquid phases; q is the volumetric water flux density; D_w is the dispersion coefficient for the liquid phase; R is the retardation factor which equals to unity for non-reactive tracers; F and G are coefficients defining the first-order decay term and zero-order decay term in the solute transport equation, respectively. The concept of two-region, dual-porosity type of solute transport (van Genuchten and Wierenga, 1976) is also implemented in both HYDRUS-1D and HYDRUS-2D models to allow for the simulation of physical non-equilibrium transport of solute. The application of HYDRUS models to water and solute movement in a hummocky landscape is presented in the subsequent section.

1.7 Summary

The fate of N supplied from fertilizer sources and the native N mineralized from the soil organic pool is influenced by the physical and biological ecosystem processes occurring in the N cycle. Among these processes, leaching has a prominent impact on crop utilization of N and water quality. Nitrate leaching in agricultural soils is a hydrologic-dependent process through which NO_3-N is lost into the groundwater. The mode of NO_3-N movement in agricultural soils is through advective-dispersive transport mechanisms. There is also a special case of preferential flow in well-structured heterogeneous soils. This review also addressed effects of soil properties, landscape features, seasonal variations, and various management practices on nitrate leaching.

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2. TWO-DIMENSIONAL REDISTRIBUTION OF BROMIDE AS INFLUENCED BY NITROGEN FERTILIZATION AND LANDSCAPE POSITION

2.1 Abstract

Bromide has been widely used in field studies to estimate nitrate leaching in agricultural soils. This study examined the impacts of N fertilization on the vertical and lateral redistribution of bromide, and the subsequent recovery of Br^- in the fall and spring seasons in a hummocky landscape. The study was carried out near Brandon, Manitoba in 2007 and 2008, using two separate plots denoted as Site-2007 and Site-2008, respectively. The plots were delineated into three landscape positions as upper (UPP), middle (MID) and lower (LOW) slope. A microplot demarcated at each landscape position received ^{15}N labelled fertilizer in form of KNO_3 at the rates of 0, 90 and 135 kg N ha^{-1}, and KBr at the rate of 200 kg Br^- ha^{-1}. Site-2007 was seeded to canola while Site-2008 was seeded to winter wheat. Soil samples were taken within the microplot to a depth of 120 cm to obtain vertical distribution, and up to 200 cm away from the microplot to obtain the lateral distribution of Br^- in the top 20 cm depth. Nitrogen fertilization reduced the downward movement of Br^- in the soil profile. This reduction resulted in accumulation of Br^- in the fertilized plots, and a greater lateral movement of Br^- with N fertilization compared to the unfertilized plots. The greatest vertical and

lateral movement of Br^- occurred at the LOW slope position. In the fall season following Br^- application in both site-years, 55 and 15% of the Br^- applied were recovered in the vertical and lateral components of the landscape, respectively, while the amount of Br^- in the plant tissue was less than 2%. In the spring season, the mass of Br^- in the vertical component had declined by 37% of that measured in the previous fall, and by 50% in the lateral component. Estimated loss of Br^- due to vertical and lateral movement, and crop uptake was 47% of the Br^- applied in the unfertilized treatment and 36% with N fertilization. The order of solute loss among landscape positions was: LOW (48%) > MID (40%) > UPP (37%). The two-dimensional redistribution of solute in the landscape was significantly influenced by the interaction of landscape position and N fertility. The study suggests that adequate N fertilization can reduce solute loss within the landscape, thereby providing an experimental verification of the "Campbell hypothesis" which states that proper N fertilization reduces nitrate leaching.

2.2 Introduction

Nitrate leaching from agricultural soils reduces crop utilization of added nitrogen, with a subsequent reduction in nitrogen-use efficiency of the crop (Follett and Walker, 1989). Once nitrate has leached below the root zone, it is difficult to recover and can result in degradation of groundwater quality due to elevated concentrations of NO_3-N (Addiscott et al. 1991; Vitousek et al. 2002; Townsend et al. 2003). To understand the fate and transport of nitrate in agricultural landscapes, knowledge of the variability in soil water transmission and effects of topographic attributes on soil water flow and solute redistribution are important (Bathke and Cassel 1991; Bathke et al. 1992; Olson and Cassel 1999; Mohanty and Mousli 2000).

Since nitrate leaching is a hydrologic-dependent transport process, a soluble and mobile ion such as bromide has been used as a tracer for soil water flow and nitrate movement (Kessavalou et al. 1996; Ottman et al. 2000; Mulla and Strock 2008; Whetter et al. 2008). Bromide is a conservative ion that is not susceptible to microbial transformations and gaseous losses. Bromide also has a low background concentration in agricultural soils (Bowman, 1984). Similar to NO_3^-, Br^- is a monovalent anion and, therefore, non-reactive with the negatively charged soil particles. Assuming no lateral movement, the difference between the mass of bromide applied and that recovered is calculated to estimate the amount of NO_3-N that can be potentially lost below the root zone (Smith and Davis 1974; Kessavalou et al. 1996).

Although nitrate leaching in agricultural landscapes depends on soil water flow and topographic attributes, certain farm management practices also influence the intensity of nitrate leaching. Studies have shown that N fertilizer management practices have a significant effect on the magnitude of nitrate leaching in agricultural soils (Randall and Iragavarapu 1995). Campbell et al. (1993) investigated the effect of N fertilization on nitrate leaching in the Brown Soil zone of the Canadian prairies. They compared effects of six rates of urea N fertilizer (0, 25, 50, 75, 100 and 125 kg N ha^{-1}) on NO_3-N leached in a plot seeded to spring wheat. The results showed that the greatest amount of NO_3-N leached beyond the root zone at lower N rates (0, 25 kg N ha^{-1}), which was similar to that under the highest N rate (125 kg N ha^{-1}). The smallest amount of nitrate was leached at 100 kg N ha^{-1}, followed by that at the rates of 50 and 75 kg N ha^{-1}.

As such, the moderate rates of N fertilizer reduced NO_3-N leaching compared to the lowest and highest N rates. This was attributed to improved root growth, nutrient and water uptake under optimum application of N (Campbell et al. 1984; 1993). Intuitively, the more nitrogen that is added to the soil the more nitrate leaching will be expected. Therefore, it is difficult to understand why nitrate leaching will be less with fertilizer N addition than without the addition of N fertilizer as reported by Campbell and his co-authors. To our knowledge, this hypothesis has not been independently reproduced, perhaps, as a result of the difficulty in directly measuring the amount of nitrate that is leached in the field.

It is also important to verify which dimension of solute transport is predominant in the landscape, vertical or lateral movement, and to quantify the relative magnitude of solute transport in both dimensions. In the landscape, nitrate can move vertically and pollute groundwater (Addiscott et al. 1991; Vitousek et al. 2002; Townsend et al. 2003) or laterally to pollute surface waters (Brooks et al. 1991). Several researchers (Bathke et al. 1992; Afyuni et al. 1994) have investigated the vertical and lateral transport of bromide in a dissected Piedmont of North Carolina. Afyuni et al. (1994) observed the greatest vertical and lateral transport of bromide at the footslope position. Their findings showed that the extents of vertical transport of bromide relative to the lateral movement were influenced by variations in soil profile characteristics, slope gradients, and hydrologic processes among landscape positions.

The study of Afyuni et al. (1994) has provided comprehensive information on the role of landscape position on two-dimensional redistribution of solute. However, this information may not be applicable to solute redistribution in hummocky landscapes due to non-uniform slope gradients and surface forms (Pennock et al. 1987). Also, unlike the wetter climate in North Carolina where the long-term annual precipitation is 1,160 mm, the total annual precipitation in the Canadian prairies is less than 500 mm.

Under Canadian prairies condition, Whetter et al. (2008) employed transport parameters known as vertical and lateral redistribution functions to estimate the relative strength of vertical downward and lateral downslope bromide distribution in a hummocky landscape. Following bromide application in the fall season, the lateral movement of solute at the crest position was smaller than at the midslope position and depression. The vertical movement of bromide was greatest at the crest position by the next spring, while the greatest vertical movement occurred at the depression in the subsequent fall season. Whetter et al. (2008) concluded that the variability in soil profile morphology and pedogenic properties among landscape positions affected the magnitude of bromide redistribution into the vertical or lateral component of the landscape.

Manning et al. (2001) reported that, within an undulating Manitoba landscape, there was a consistent ranking of lower > mid > upper elevation landform element complex with respect to the depth to carbonates, a measure of the extent of water penetration into the soil profile. A similar ranking was reported for the A horizon thickness, solum thickness and soil organic matter content. From this ranking, the lower slope position may be expected to possess the greatest potential for nitrate leaching. Within two hummocky landscapes in central Alberta, Thibodeau et al. (2008) also showed that the greatest intensity of bromide leaching was observed at the lower slope position. This was attributed to greater depth of clay accumulation and soil profile development compared to the top and middle slope positions.

While there is good documentation of effects of soil profile attributes on solute redistribution in the landscape, there is little information on how N fertilizer management affects the partitioning of solute into lateral or vertical component of the landscape, and the subsequent intensities of solute loss among landscape positions.

To contribute to our understanding of the fate and transport of nitrate, several hypotheses were tested in this study. First, whether improved biomass production due to N fertilization can reduce the leaching loss of mobile nutrients other than nitrate. Second, whether there is a significant interaction between effects of landscape and N fertility management on the vertical and lateral redistribution of solutes. Third, whether the lower landscape position, which is associated with the greatest soil water flux, is also the region with the greatest loss of solute due to vertical and/or lateral movement of bromide. Therefore, the objectives of this study were: (i) to examine the impacts of N fertilization on the downward and lateral movement of bromide in a hummocky landscape; (ii) to identify the landscape position with the greatest two-dimensional transport of solute; (iii) to quantify bromide recovery in the fall and spring seasons, following spring application.

2.3 Materials and Methods

2.3.1 Description of the Study Area

The study was conducted at the Manitoba Zero Tillage Research Association (MZTRA) farm, located 17.6 km north of the city of Brandon, Manitoba. The research farm covers a section of land (2.59 km^2), situated within the Newdale Plain subsection of the Assiniboine River Plain (Section 31-12-18 West of the Principal Meridian). This area is found within the Aspen Parkland of the Prairies Ecoregion of South-West Manitoba, latitude $49^{o}55'$N, and longitude $99^{o}57'$W. The land resources information for this area was documented and refined from a detailed soil survey database collected by aerial photographs, mapped at a scale of 1:5,000 using Geographical Information System (GIS) technique (Podolsky and Schindler 1993).

The soils at this site are predominantly clay loam formed over moderately to strongly calcareous glacial till materials derived from a mix of shale, limestone and granite. The soil type is described as the Newdale association (frigid Calcic Hapludolls), comprising of various soil subgroups occurring at different landscape positions with their characteristic internal drainage conditions.

At upslope position, the dominant soil types are the well drained Calcareous Black and Rego Black Chernozems comprising of the Cordova and Rufford series, respectively. The soils at the mid slope position are the well drained Orthic Black Chernozem known as the Newdale series, which covers the largest portion of the total area. The soils present at the lower slope to elevated depression are the imperfectly and poorly drained Gleyed Eluviated Black and Gleyed Rego Black Chernozems, represented by the series known as Angusville and Varcoe, respectively. The soils at the depressional areas are poorly drained and remain wet with prolonged inundation for a considerable period of the year. They are comprised of the series known as Drokan and Penrith belonging to the Rego Humic and Humic Luvic Gleysols. A detailed discussion of the classification and morphological description of these soils have been documented in the MZTRA Special Report (Podolsky and Schindler 1993).

The natural ecosystem at this site consists of native grass, shrubs and wetlands. The land has been cultivated for the past 100 years and converted to zero tillage in 1993 for cropping systems research, where the only soil disturbance occurs at seeding. In the past five years, the annual crops grown on the experimental plots specific for this study were peas (*Pisum sativum* L.), canola (*Brassica napus* L.), and wheat (*Triticum aestivum* L.) in rotation.

The surface relief of the study area is depicted as a hummocky landscape with a gently sloping terrain (2-5%). Hummocky landscape is a complex pattern of knolls and depressions, with seasonal wetlands (Podolsky and Schindler 1993). Elevation data for the area have been collected by MZTRA using Light Detection and Ranging (LIDAR) technique and were compiled in the form of a grid file. Elevations on the study area ranged from 495 m at the southeast corner to 510 m in the northwest corner. The digital elevation data specific for the experimental plots were extracted from the grid file for the area to generate the surface relief maps using Surfer® Version 8 (Golden Software, Golden Colorado).

The climate of the study area is moderately sub-humid, with short, cool summers and long, cold winters. Long-term records from weather stations in the area indicate a mean annual precipitation of 459 mm, of which rainfall accounts for 340 mm, and a mean annual air temperature of 1.5 ºC. Daily precipitation and air temperature for the period of study were obtained from two weather stations close to the site. The weather record was obtained from the Manitoba Ag-Weather Program operated by the Manitoba Agriculture, Food and Rural Initiatives (MAFRI). The MAFRI weather station is situated at the MZTRA farm where the study was conducted. Weather information was also obtained from the Environment Canada (ENVIR) meteorological station at Brandon Airport, 15 km south of the study area. The weather record was summarized and illustrated as cumulative monthly precipitation and mean monthly air temperature.

2.3.2 Field Layout and Experimental Design

The study was carried out during the growing seasons of 2007 and 2008, denoted as Site-2007 and Site-2008, respectively. The field experiment was conducted on a separate plot in each year. The two experimental sites were approximately 50 m apart, while the plot size was 40 m × 40 m. The study was established as a factorial experiment with a split plot design; landscape position was the main plot factor, and N fertilizer rate was the subplot factor (Fig. 2.1).

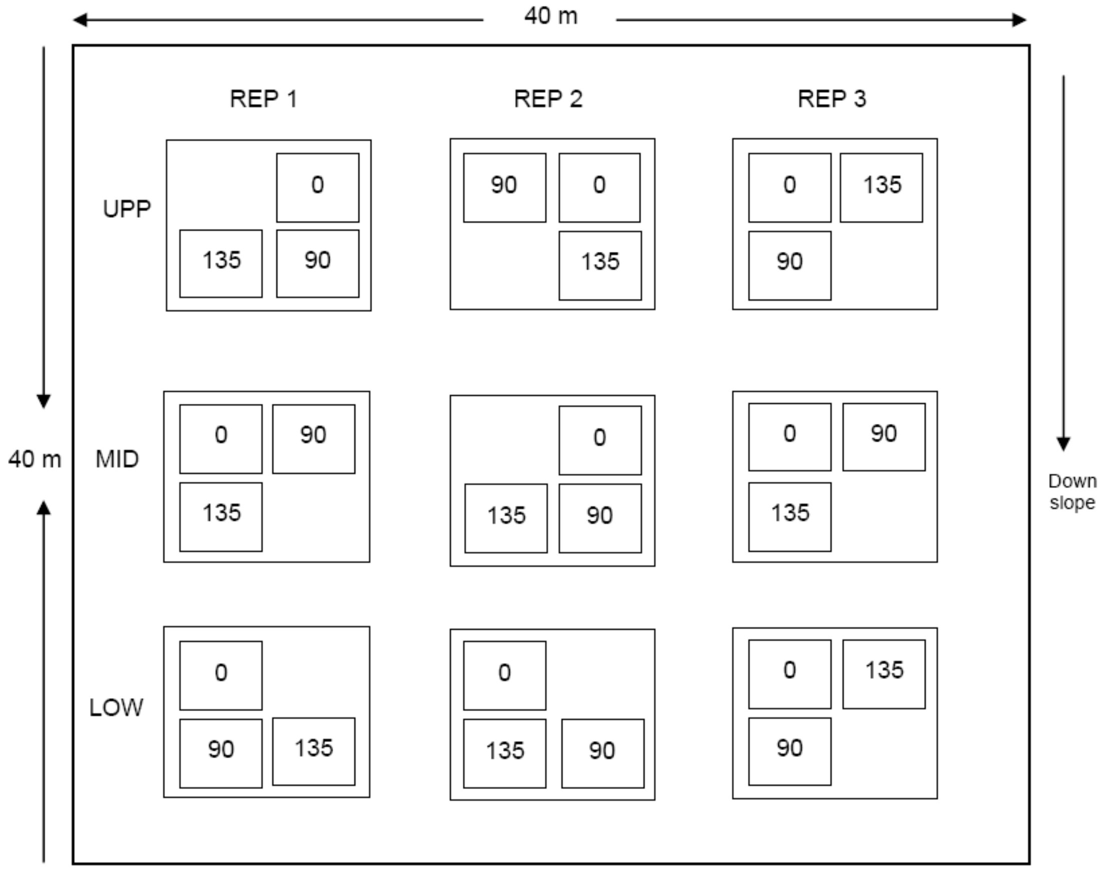

Fig. 2.1. The field layout with three replicates of mainplot at each landscape position. Each main plot contains three subplots for N rates.

In both experimental sites, there was visual evidence of hillslope features. Therefore, topographical segmentation of the experimental sites was based on simple delineation of transects measured from the crest position of the plot to the depressional areas.

Three replicates of pre-selected transects within the plot were delineated into three equal segments as upper (UPP), middle (MID) and lower (LOW) landscape positions. The contours and surface reliefs for both plots in Site-2007 and Site-2008 were captured as a single map and generated using the Surfer® program (Fig. 2.2). To better understand the impacts of landform attributes of the regions surrounding the plots on soil water distribution within the plots, the surface maps for both experimental sites were extended by 25 m distance in the south and north directions for Site-2007 and Site-2008, respectively (Fig. 2.2). The depressional areas of the landscape were poorly drained with visible water-logging conditions, hence, were not included in the landscape delineation.

In order to assess the visual delineation of the experimental sites, the LandMapR© software toolkit C++ version (MacMillan 2003) was used to segment the plots into topographical positions based on landform element complexes (LEC) developed by Pennock et al. (1987) and MacMillan and Pettapiece (1997). According to the LandMapR© program, the distribution of topographical positions within the plots (Fig. 2.3) was not consistent with the orientation and dimensions of the main plots in the plot layout (Fig. 2.1). The hillslope in Site-2007 (north-facing slope) was segmented mainly into upper and mid elevation LEC, while only a small portion of the site was classified as lower LEC at the south-east

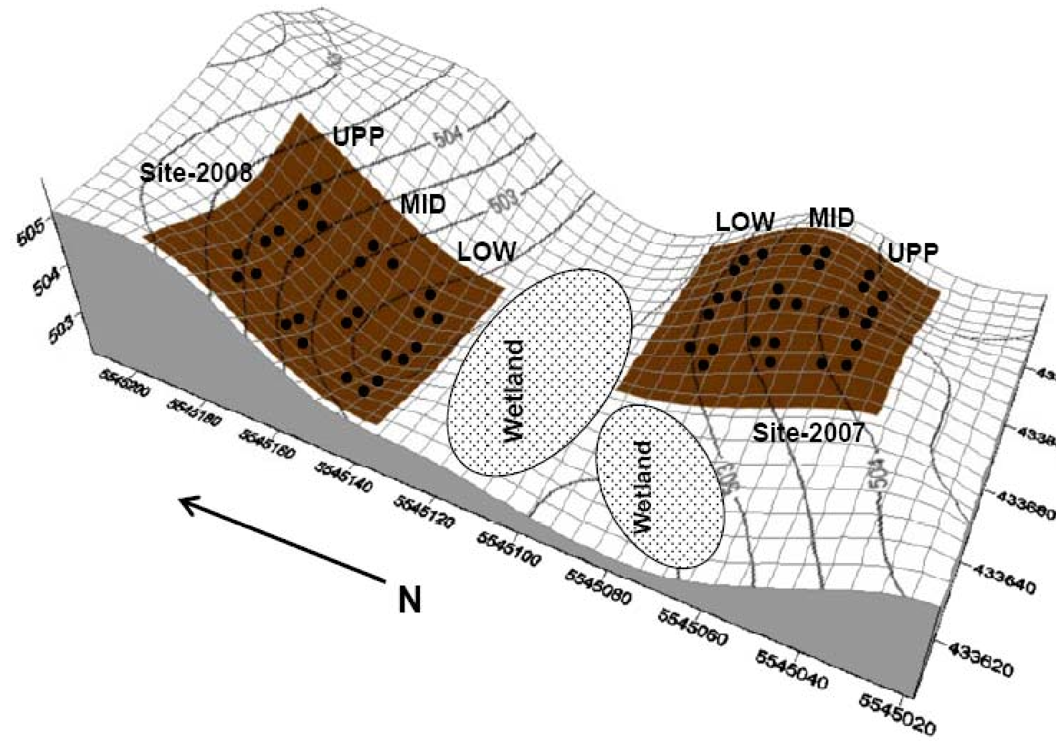

Fig. 2.2. Contours and surface relief maps of the experimental plots with extended layouts (25 m in the north-south direction). Sampling locations are shown across landscape positions in Site-2007 (north-facing slope) and Site-2008 (south-facing slope). Surface relief maps were generated using the Surfer® program.

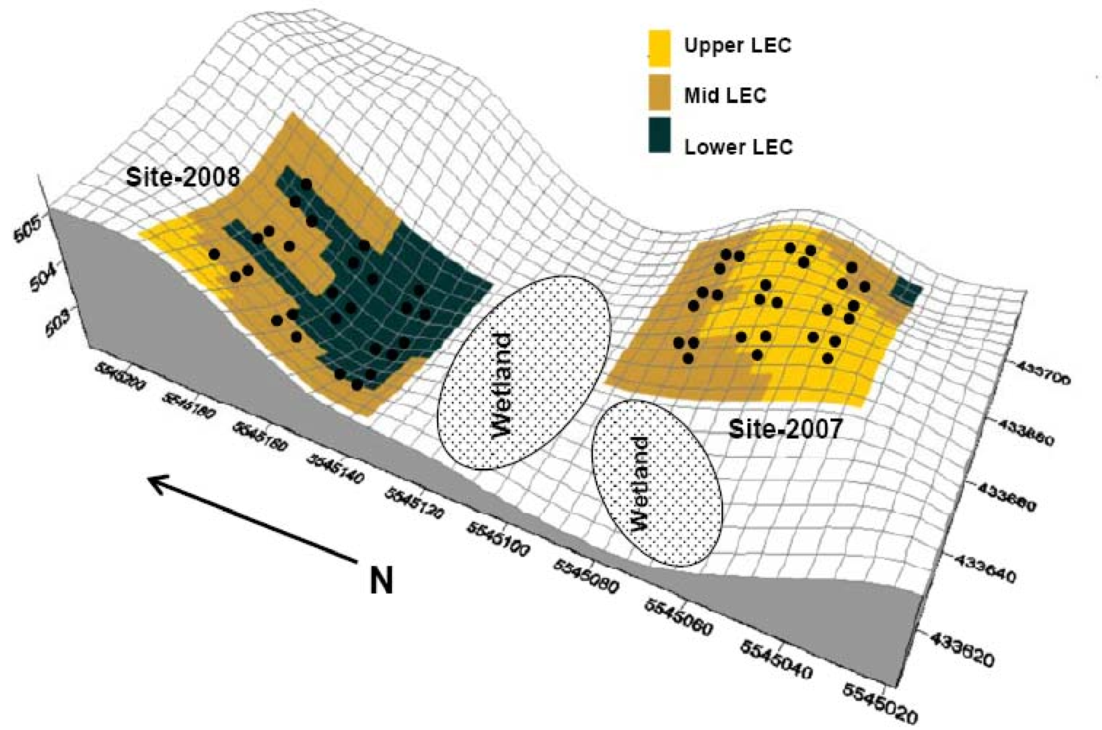

Fig. 2.3. Topographical segmentation of the experimental plots into upper, mid and lower elevation landform element complexes (LEC) using the LandMapR© program. The distributions of sampling locations are shown within the LEC in Site-2007 (north-facing slope) and Site-2008 (south-facing slope). Surface maps were generated using the Surfer® program.

corner of the plot (Fig. 2.3). Also in Site-2008 (south-facing slope), the smallest portion of the experimental site was classified as upper LEC, located at the north-west corner of the plot, while there was no definite boundary between the mid and lower LEC.

To impose the selected treatment design and plot layout on the experimental sites, the orientations of landscape positions in both site-years were based on simple delineation of the plots into three uniform segments as UPP, MID and LOW slope positions. A main plot (10 m × 10 m) was established at each landscape position and demarcated into three randomized subplots (4 m × 4 m) (Fig. 2.1). A microplot was imposed within each subplot to conduct the tracer study (Fig. 2.4). The size of the microplot strip was 40 cm long (parallel to the slope direction) by 200 cm wide (perpendicular to the slope direction).

Prior to treatment application, baseline measurements were conducted to characterize the background soil properties. Three intact soil cores were collected from the centre of all three landscape positions (a total of nine soil cores) to a depth of 120 cm. The soil cores were examined for soil profile morphology. The soil profile characterization was conducted according to the Canadian Soil Survey Committee (1998) guidelines. The soil profiles were characterized for A horizon depth, solum depth (depth of A and B horizons), and depth to carbonates. The colour of the soil horizon was used as the basic criterion for characterizing the depth of soil horizon. The depth to carbonates was indicated by the depth at which effervescence reaction occurs due to CO_2 release from carbonate minerals on addition of dilute hydrochloric acid.

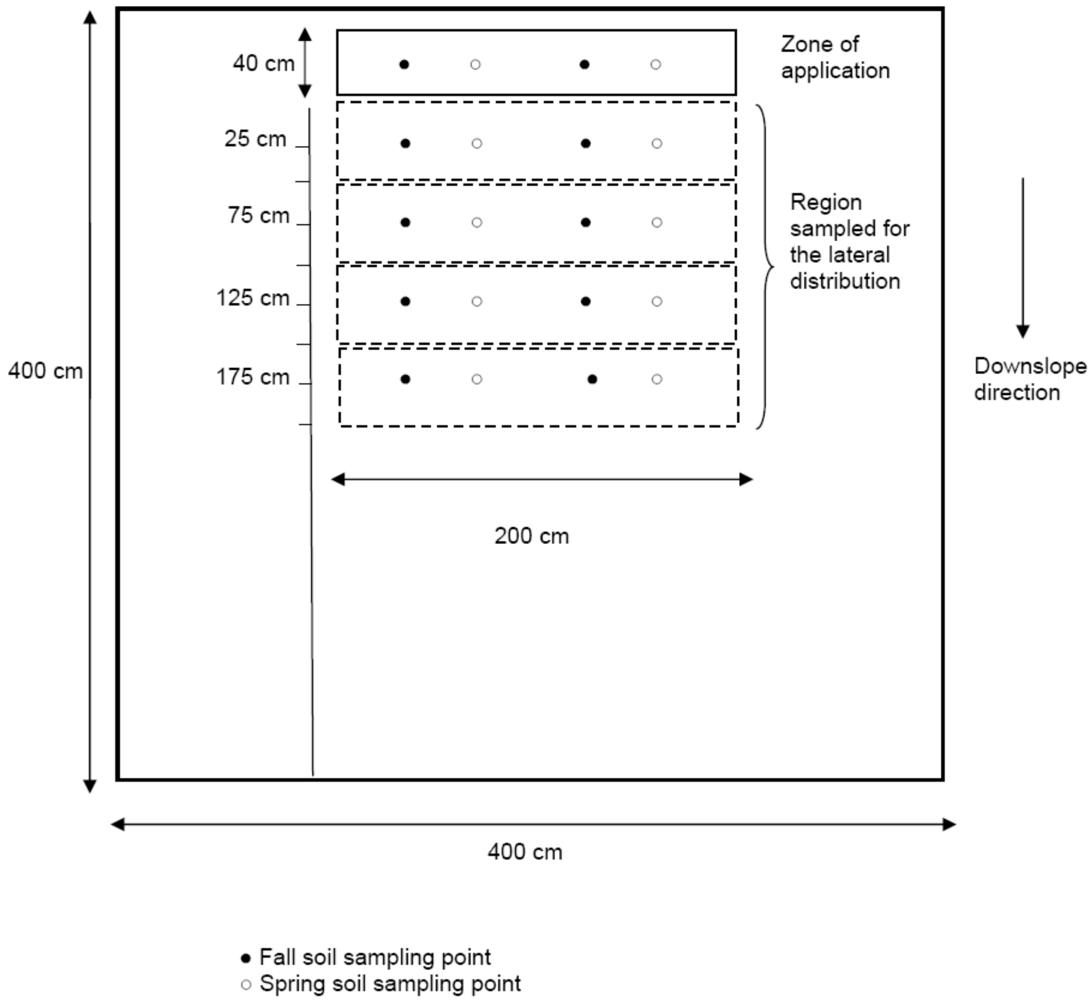

Fig. 2.4. A subplot containing a microplot with sampling points.

At the end of each growing season, the experimental plots were characterized for soil physical and hydraulic properties using standard procedures. The soil moisture content at field capacity was measured using the procedure described by Shaykewich et al. (1998). The soil bulk density was also measured at each landscape position. The field capacity and bulk density were determined from the same core (Shaykewich et al. 1998). Soil particle size analysis was determined using the pipette method (Loveland and Whalley 1991), while the field saturated hydraulic conductivity was measured at various depths (0-15; 15-30; 30-60; and 60-90 cm) using the Guelph Permeameter (Model 2800K1; Reynolds and Elrick 1985).

Three replications of soil cores were taken for the soil physical properties at each of the three landscape positions, for a total of nine soil cores at each site. The saturated hydraulic conductivity measurements were based on two cores per landscape position. However, there were difficulties in measurements in some cases due to the presence of stones and impermeable subsoil layer. Therefore, it was impossible to obtain realistic measurements at some depths in these cases, while only one core was taken per landscape position in some extreme cases.

2.3.3 Treatment Application and Tracer Studies

Routine field operations such as seeding, weed and pest control were handled by the MZTRA technicians according to the association's farm objectives. The first plot (Site-2007) was seeded to canola on 11 May 2007. At the time of seeding, 53.8 kg ha^{-1} of monoammonium phosphate (11-52-0) equivalent to 5.9 kg N ha^{-1} was applied to the entire field. However, application of tracers and N fertilizer to the microplots was delayed until crop emergence to allow for precise and uniform application of treatments.

The remaining region of the subplots (excluding the microplots) received a one-time application of unlabelled potassium nitrate fertilizer (13.5-0-46.2, Soluble NK Fertilizer, Haifa Chemicals) on 5 June 2007. The rates of N application were 0, 90 and 135 kg N ha^{-1}; denoted as TRT0, TRT90 and TRT135, respectively. The N fertilizer rates were randomly distributed over the three subplots in each mainplot. During fertilizer application to the subplot, the microplot was covered with plastic sheeting to prevent it from receiving unlabelled N fertilizer.

The microplot in each subplot received labelled N fertilizer in the form of $K^{15}NO_3$ (potassium nitrate-^{15}N, minimum 10 atom% ^{15}N, ISOTEC^{TM}, Sigma Aldrich) at the rate equivalent to that of the corresponding subplot. Reagent grade potassium bromide (KBr) was also applied to all microplots at a rate of 200 kg Br^- ha^{-1} (Kessavalou et al. 1996). The $K^{15}NO_3$ and KBr salts were mixed together and added to the microplots at the same time in form of a solution (Ottman and Pope 2000). The solution containing the two salts was uniformly applied to the row spacing within the microplots using an automatic dispenser calibrated to a discharge rate of 100 mL per minute. A similar method of treatment application was used by Jowkin and Schoenau (1998).

The second plot (Site-2008) was established in the spring of 2008. The field had been previously seeded to winter wheat on 12 September 2007. The seeding was also accompanied by an application of 76.2 kg ha^{-1} of 11-52-0 (equivalent to 8.4 kg N ha^{-1}). The treatment application similar to that in Site-2007 was repeated for Site-2008 on 9 June 2008. While the microplots served as the experimental units within each mainplot, the remaining portion of the subplots served as the buffer zones.

2.3.4 Harvesting and Soil Sampling

The above ground portion of canola plants in the microplots of Site-2007 was harvested by hand at maturity on 12 August 2007. The harvested plants were air-dried and weighed to obtain the total biomass as a measure of crop yield. The above ground portion of winter wheat in the microplots of Site-2008 was harvested on 13 August 2008. The wheat biomass was air-dried and threshed into grain and straw. Air-dry plant samples were ground to pass a 1-mm sieve using a Plant Tissue Mill (Thomas-Wiley Laboratory Mill, Model 4).

Soil samples were taken in the fall and spring seasons. Fall samples were taken on 15th to 16th of October 2007, 132-133 days after tracer application (DAT). All microplots were sampled for vertical distribution of solutes at two locations within the microplot (1 m apart) to a depth of 120 cm. The samples were obtained as intact soil cores using a tractor-mounted Gidding's drill fitted with a coring probe (5 cm internal diameter). A plastic sleeve previously inserted into the coring probe was used to store the intact soil core, which was later sectioned into the following segments: 0-5; 5-10; 10-20; 20-40; 40-60; 60-90; 90-120 cm.

Another set of soil samples was collected from the surrounding region within the subplots of TRT0 and TRT90 only. The soil samples were obtained to characterize the lateral movement of bromide in the downslope direction. The samples were taken from the top 20 cm depth at 10 cm intervals using a Dutch

Auger. The orientation of the sampling points was similar to that reported by Bathke and Cassel (1992) and Whetter et al. (2008). Four locations were sampled at a distance of 25, 75, 125, and 175 cm away from the edge of the microplot in the downslope direction (Fig. 2.4). The lateral samples were taken at two points, similar to what was done within the microplot (Fig. 2.4). The downslope direction was further sampled from 20 to 40 cm depth at 10 cm intervals in fall 2007. This was done to verify the magnitude of downward penetration of bromide below the top 20 cm soil layer in the lateral direction.

Soil samples were air-dried and ground to pass a 2-mm opening sieve. The spring soil sampling for Site-2007 was conducted on 13th to 14th of May 2008 (346-347 DAT) following the protocol used in the fall of 2007. Soil sampling protocols and seasons for Site-2008 were similar to those for Site-2007. Soil samples in fall of 2008 were taken on 13th to 14th of October 2008 (127-128 DAT), while the spring soil sampling was carried out on 11th to 12th of May 2009 (337-338 DAT). Six replications of sampling per landscape position were obtained for each N fertility treatment.

In each sampling season, a total of 54 intact soil cores were obtained for the vertical distribution of solute in the microplot (the zone of application), while a total of 144 sample cores were obtained for the lateral distribution of solute. For bromide and nitrate analyses all six samples were independently analyzed, while a composite sample per replicate was analyzed for total soil N and ^{15}N isotope ratio. At the end of each sampling exercise, holes were back-filled with soils excavated from areas outside the experimental plot but adjacent to the corresponding slope position.

2.3.5 Plant and Soil Analyses

Four hundred milligrams of plant tissue sample was digested using the modified wet oxidation method of Akinremi et al. (2003). The acid-digested samples were neutralized and analyzed for bromide. Thirty millilitres of distilled water was added to 10 g of the ground soil sample. The saturated soil-water mixture was extracted by shaking for 30 minutes at 120 strokes per minute on a reciprocating shaker. The mixture was centrifuged at 3466 × *g* (Accuspin™ 400, Fisher Scientific Ltd.). The clear solution was decanted and analyzed for bromide.

The bromide content in the plant tissue and soil sample was measured using a bromide selective electrode (Orion 9635) fitted to a dual channel pH/ion/conductivity meter (Accument Research AR50, Fisher Scientific). The bromide measurement was standardized by four-point calibration with a series of standard solutions diluted to the range of 1.0 to 20 mg L^{-1}, to capture the expected concentration in the samples. The four-point calibration also enabled the detection of bromide at low concentration (< 0.4 mg L^{-1}).

Reagent grade KBr salt was used to spike a known mass of pulverized industrial grade inert quartz sand (Unimin Silica Sand, Unimin Corporation, Le Sueur, MN) at the rate of 300 mg Br^- kg^{-1} soil and diluted to a series of concentrations. The spiked “soil” was extracted and analyzed as reference samples to check the accuracy of the electrode measurements. In order to adjust the ionic strength of the system, 5 *M* $NaNO_3$ was added to all the extracted soil samples and the standards at solution to sample ratio of 1:50 (Orion Bromide Electrode Manual 2003, Thermo Electron Corporation).

A simple linear regression of the estimated and measured concentration of the reference samples predicted > 99% of the variation in bromide concentrations of the spiked “soil” (Fig. 2.5). The accuracy of the ion selective electrode measurements was also compared with that of ion chromatography method(ICS-100 Ion Chromatograph System). There was a strong similarity between the two methods of bromide determination, as measured by the regression equation (R^2 = 0.97).

Bromide concentration in mg kg^{-1} soil was converted to kilograms per hectare, using the corresponding bulk density at the sampling point. As such, the mass of solute recovered per soil layer was expressed in kg ha^{-1} for each sampled depth. The vertical distribution of solute in the soil profile was illustrated by plotting the mass of bromide in kg ha^{-1} with the midpoint of each sampled depth. By expressing the mass of bromide in kg ha^{-1}, the contributions from lateral versus vertical dimensions relative to the amount of bromide added could be clearly illustrated.

2.3.6 Statistical Tests

Analysis of variance (ANOVA) tests were conducted on the measured variables using the PROC MIXED procedure of SAS® software for Windows (Version 9.1, SAS Institute, Inc., Cary, NC). Treatment effects due to N fertilizer, landscape, and their interactions on bromide transport were tested against the six replicates of soil samples [Rep (N fertilizer × landscape)]. The PROC MIXED procedure was treated as a split-plot design with a double repeated measure,

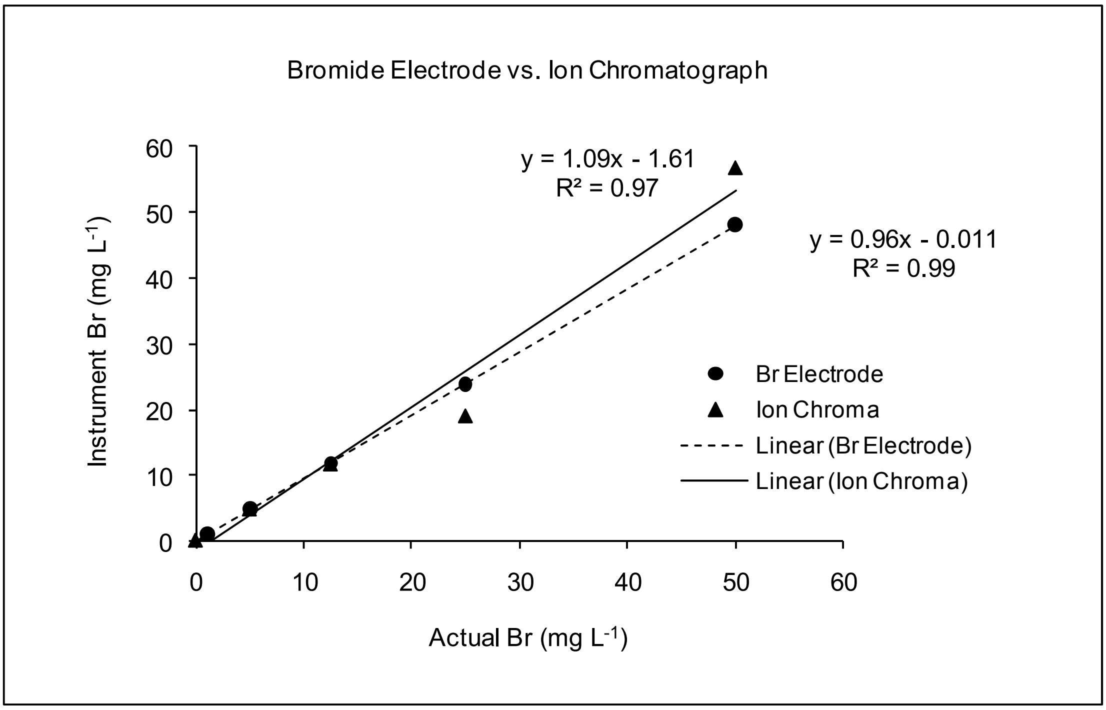

Fig. 2.5. Comparison of bromide ion determination using ion selective electrode and ion chromatograph method.

whereby the random effect was the replicate (Rep), while depth (or distance for the lateral distribution) and season were regarded as the repeated measures.

Various covariance structures were examined for the PROC MIXED procedure. The First-order Autoregressive [AR(1)] covariance structure consistently produced the smallest value of fit statistics based on the Akaike's Information Criterion (Littell et al. 1996). Therefore, AR(1) error structure was selected for all the double repeated analyses, thereby imposing homogeneous variances and correlations that decline exponentially with depth and season on the bromide data (Kincaid 2005).

The PROC UNIVARIATE tests for normality indicated that, in almost all cases, the data were significantly different from a normal distribution, based on the Shapiro-Wilk's normality test and skewness of the histograms distribution. As such, the raw dataset was log transformed to achieve homogeneity of variances commonly required for parametric statistics. However, the mass of bromide was plotted using the raw data to capture the spatial and temporal distribution of solute in the soil, while the back-transformed estimates of means were used to calculate bromide mass balance.

The Tukey-Kramer test was used to determine the differences in treatment means at $P < 0.1$. A probability level of 0.1 was chosen for identifying significantly different treatment means in this study. This is due to the large variability commonly associated with field experimentation of N dynamics at the landscape scale (Walley et al. 1996; Beckie and Brandt 1997; Zvomuya et al. 2003; Kutcher et al. 2005).

In this chapter, the vertical and lateral distribution of bromide, as well as the mass recovery of bromide among landscape positions were compared between the plots treated with 90 kg N ha^{-1} (TRT90) versus those with no N fertilizer added (TRT0) to illustrate the effect of fertilizer application on two-dimensional transport of bromide in the landscape.

2.4 Results and Discussion

2.4.1 Weather Characteristics of the Experimental Sites

According to the MAFRI weather record, the cumulative rainfall from May 2007 to September 2007 (prior to fall soil sampling in October 2007) was 248.2 mm (Fig. 2.6). The snowfall data were not available at the MAFRI weather station during the field experimentation. Therefore, the Environment Canada weather record (ENVIR) was used as a source of total precipitation between October 2007 and April 2008 (prior to spring soil sampling in May 2008), which was approximately 125.6 mm. The cumulative rainfall from May 2008 to September 2008 was 285.7 mm (MAFRI), followed by a total precipitation of 216.2 mm between October 2008 and April 2009 (ENVIR).

The cumulative rainfall (May to September) in Site-2008 was greater than in Site-2007 by 37.5 mm (Fig. 2.6). The ENVIR precipitation data was greater than the MAFRI record, particularly in 2008-2009. However, the trend in rainfall distribution was similar between the two weather stations.

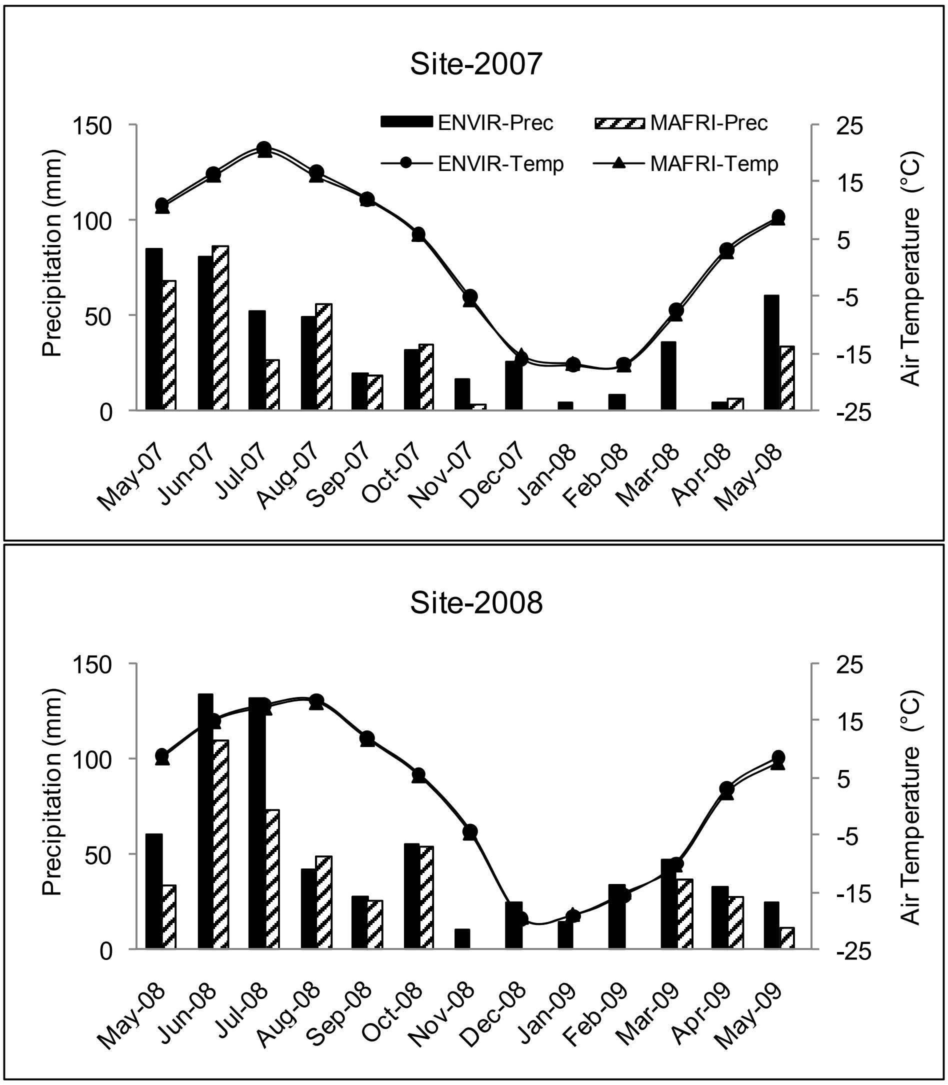

Fig. 2.6. Mean monthly air temperature and cumulative precipitation for the site-years. Weather record from Manitoba Ag-Weather Program for Forrest Station, Manitoba Agriculture, Food and Rural Initiatives (MAFRI) and Environment Canada weather record at Brandon Airport (ENVIR).

Air temperature patterns were generally similar between the two site-years (Fig. 2.6). However, the mean monthly temperature during the growing season (May 2007 – August 2007) in Site-2007 ranged from 11°C to 21°C while the same period was characterized by cooler temperatures in the range of 8°C to 18°C in Site-2008. The coldest period in both site-years was between December and February. This period was associated with an average monthly air temperature of −18°C.

2.4.2 Surface Relief and Landform Attributes of the Experimental Sites

The LandMapR© program was used to quantify the landform parameters which define the surface morphological attributes of the landscape (Table 2.1). These parameters are the slope gradient, profile curvature, and plan curvature (Pennock et al. 1987). The slope gradient is defined as the maximum rate of change of elevation. The profile curvature is the rate of change of the slope gradient in the downslope direction, while plan curvature refers to the shape of the terrain along a contour line across the slope.

Based on the criteria for classifying landform elements as described by Pennock et al. (1987), negative values for profile and plan curvatures indicate a concave element with convergent surface form and positive values for convex element with divergent surface form. The upslope contributing area is also an important landform parameter controlling soil water distribution within the landscape. The upslope contributing area is defined as the area above a certain length of contour that contributes flow across the contour, and often referred to as the upslope catchment area (Gallant and Wilson 2000). The size of an upslope

Table 2.1. Surface terrain attributes among landform element complexes (LEC) in Site-2007 and Site-2008 using the LandMapR© program.

LEC	Slope gradient (°)	Profile curvature (° $100m^{-1}$)	Plan curvature (° $100m^{-1}$)	Upslope catchment area (m^2)	Area covered (%)
Site-2007					
Upper	1.56	13.8	8.31	45	51
Mid	2.32	−4.15	−0.109	112	47
Lower	1.36	−17.6	−17.8	443	2
Mean^z	*1.94*	*4.83*	*4.10*	*78*	*NA^x*
Site-2008					
Upper	3.36	17.5	2.11	71	7
Mid	3.41	−2.85	−8.19	148	47
Lower	2.87	−8.23	−12.1	786	46
Mean^y	*3.14*	*−5.54*	*−10.2*	*467*	*NA^x*

Mean^{z,y}: mean values of topographical data in Site-2007 comprised of the upper and mid LEC only; mean values of topographical data in Site-2008 comprised of the mid and lower LEC only.

NA^x: Not applicable.

catchment area is directly related to the extent of water draining into the surrounding region of the catchment area.

The greatest slope gradient was at the mid landform element complex (LEC) and the smallest at the lower LEC in Site-2007 and Site-2008 (Table 2.1). The slope steepness in Site-2008 (3.2°) was greater than in Site-2007 (1.7°). This implies that Site-2008 is expected to shed more water than Site-2007. The trend in profile and plan curvatures across the landscape was similar for both site-years (Table 2.1). The divergent attribute of the landform decreased in the downslope direction, while the convergent feature increased accordingly. This is similar to the trend reported by Pennock et al. (1987) and Manning et al. (2001) in a hummocky landscape. The size of upslope catchment area also increased from the upper to lower LEC in both site-years, indicating that the lower elevation LEC would receive the greatest amount of dissipated water, particularly in Site-2008.

The experimental sites were segmented by the LandMapR© program mainly into upper and mid LEC in Site-2007, and into mid and lower LEC in Site-2008 (Fig. 2.3). The portion of the experimental site classified as mid LEC was 47% of the total area in both site-years (Table 2.1). At the mid elevation LEC, the values of profile and plan curvatures suggest that the surface form in Site-2007 is more convergent along the slope direction but less convergent across the slope, compared to Site-2008. In Site-2008, the upslope catchment area contributing flow to the mid LEC was greater than in Site-2007 by 32% (Table 2.1).

Mean values of the topographical data for the dominant landform element complexes were derived for each site-year. The mean topographical data were computed for the upper and mid LEC in Site-2007, and the mid and lower LEC in

Site-2008. The data showed that Site-2007 generally has divergent (convex) surface forms along and across the slope, while the surface forms in Site-2008 are convergent (concave) accordingly (Table 2.1). This suggests that the divergent attribute of the profile curvature in Site-2007 would enhance the downslope lateral flow of water dissipated from the upslope region, however, with subsequent reduction in net infiltration and downward transmission of water in the soil compared to Site-2008. In contrast, the convergent profile and plan curvatures in Site-2008 have a greater potential for water accumulation at the soil surface, which can enhance the entry and transmission of water in the soil profile than in Site-2007. The mean upslope catchment area for the dominant LEC in Site-2008 was greater than in Site-2007 by six-fold (Table 2.1). In addition to the presence of convergent landscape character, the greater upslope contributing area in Site-2008 indicates a greater intensity of hydrologic processes compared to Site-2007.

Although the greater slope gradient in Site-2008 implies a greater dissipation of water from upslope regions than in Site-2007, this is in contrast to the trends indicated by the profile and plan curvatures between the two site-years (Table 2.1). Manning et al. (2001) also showed that the slope gradient was inversely related to convergent landscape character in a hummocky landscape. As such, it is important to quantify these landform parameters for a better description of effects of landform attributes on soil water distribution. According to Manning et al. (2001), the extent of convergent landscape character coincided with the trends in indicators of internal drainage such as the thickness of A horizon, solum depth and depth to carbonates. The magnitudes of these soil profile features increased from the upper to lower LEC (Manning et al. 2001). Hence, the profile and plan curvatures are better indices of soil water distribution in the landscape compared to the slope gradient.

The differences in surface forms between Site-2007 and Site-2008 are expected to influence the extent of vertical and lateral distribution of solute in the soil. While the LandMapR© program was used to derive information on the landform attributes of the landscape, the bromide redistribution data and other parameters such as the soil properties and plant data are based on simple delineation of the experimental sites into relative positions in the landscape as UPP, MID and LOW slope positions.

2.4.3 Soil Profile Morphology, Physical and Hydraulic Properties

Due to technical error in data compilation for the depth to carbonates in Site-2007, depths of A horizon and solum were presented only for Site-2007. The soil profile morphology presented for Site-2008 comprised of depths of A horizon and solum, and the depth to carbonates.

In Site-2007, the mean depth of A horizon at the UPP slope position was 18 cm, and slightly greater at the MID slope which was 20 cm (Fig. 2.7a). The greatest depth of A horizon was at the LOW slope position, where the horizon thickness was 30 cm. The solum depth across the landscape was 34, 41 and 56 cm at the UPP, MID and LOW slope, respectively. In Site-2008, the depth of A horizon ranged from 12 cm at the UPP slope to 30 cm at the LOW slope position (Fig. 2.7b). While the smallest depth of solum was at the UPP slope, solum depth was similar for the MID and LOW slope positions in Site-2008. The trend in depth

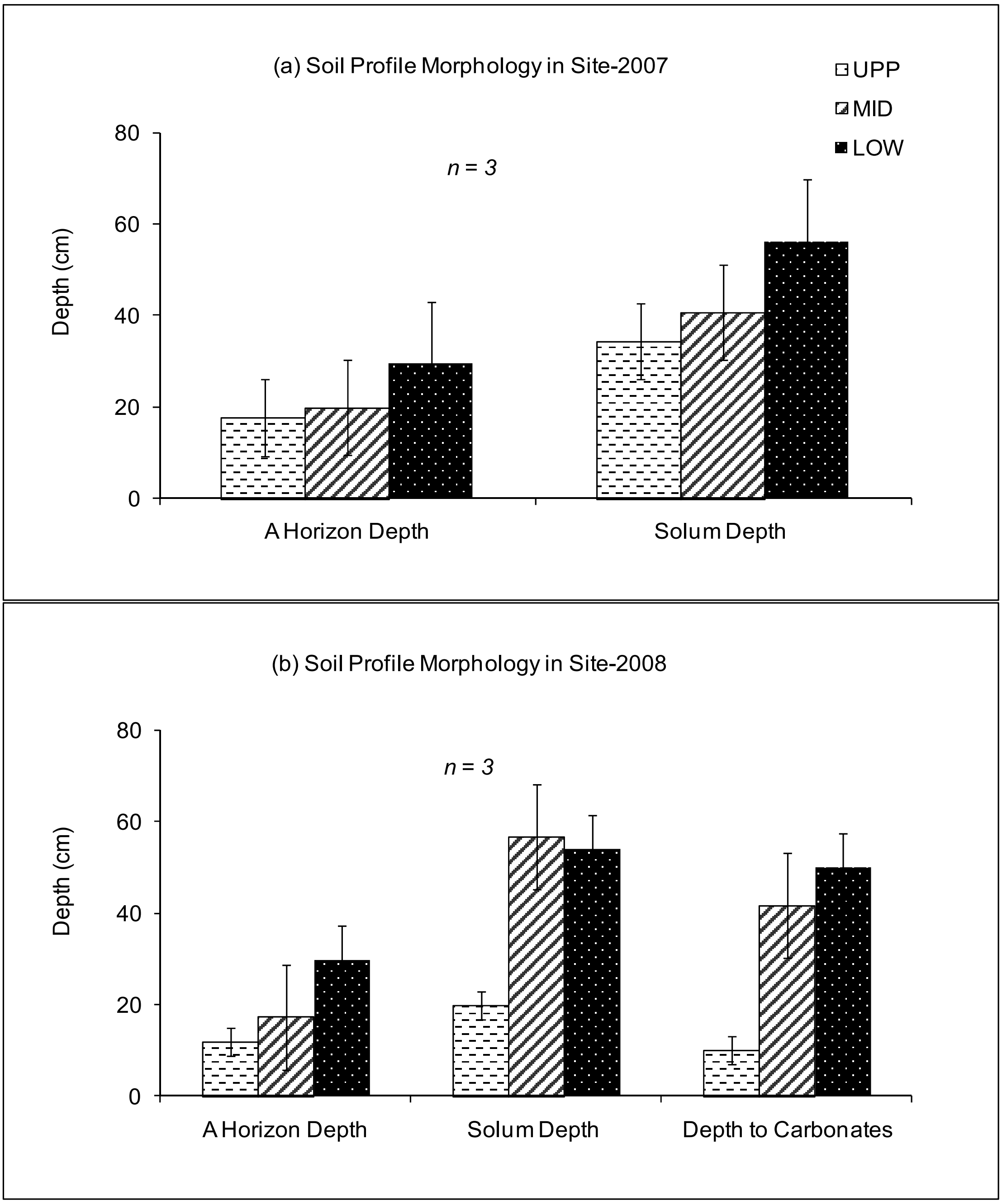

Fig. 2.7. Depths of A horizon and solum among landscape positions in (a) Site-2007 and depths of A horizon, solum and depth to carbonates among landscape positions in (b) Site-2008. *n* denotes the number of cores taken for each sample analysis. Error bars represent the standard error of the measurements.

to carbonates in Site-2008 was identical to that of solum depth, except that the depth to carbonate at the LOW slope was slightly greater than at the MID slope position.

The magnitudes of A horizon thickness and solum depth were similar for both site-years, particularly at the LOW slope position where the greatest depth of A horizon was measured. The trends in soil profile attributes across the landscape in both site-years were consistent with those documented in the soil survey for the study area (Podolsky and Schindler 1993). Soil profile morphological features have been used as indicators of internal drainage and hydrologic processes in hummocky landscapes (Manning et al. 2001; Whetter et al. 2008). In the present study, the trends in soil profile attributes across the landscape indicated that the potential for downward transmission of water in the soil profile increases in the downslope direction. At the UPP slope position where there were the smallest depths of A horizon and solum, soil water transmission is expected to encounter the greatest degree of partitioning into lateral flow due to less permeable subsoil layer (Arnett 1974; Bathke and Cassel 1991).

The greatest depths of A horizon and solum at the LOW slope position are expected to enhance net infiltration and downward transmission of precipitation and water dissipated from upslope regions (Pennock and de Jong 1990; Fiez et al. 1995; Pachepsky et al. 1999; Manning et al. 2001; Whetter et al. 2008). The thickness of A horizon at the LOW slope position can also result in large pore channels with interconnected pore geometry in the soil profile, thereby promoting lateral redistribution of water over vertical downward flow. The two-dimensional redistribution of bromide across landscape positions will provide a better understanding of the roles of soil profile morphology on soil water dynamics in the landscape.

The soil physical and hydraulic characteristics for both site-years (Site-2007 and Site-2008) are shown in Tables 2.2-2.3. The field capacity in Site-2007 ranged from 0.28 to 0.42 cm^3 cm^{-3} across the landscape, while the bulk density generally increased with depth (Table 2.2). Assuming that the top 10 cm depth represents the zone with the greatest potential for soil moisture flux, the soil properties measured at the 0-10 cm depth were used to illustrate variability among landscape positions. Within the top 10 cm depth in Site-2007, the greatest bulk density was at the UPP slope compared to other slope positions. The soil texture at the top 10 cm depth also varied across the landscape. The UPP slope had the greatest % clay and the MID slope had the smallest. In contrast, % silt was greatest at the MID slope compared to other slope positions while the sand fraction was generally similar at all slope positions.

The field capacity that was measured in Site-2008 was smaller than in Site-2007, ranging from 0.22 to 0.38 cm^3 cm^{-3} (Table 2.3). Similar to the trend in Site-2007, the greatest bulk density was at the UPP slope compared to other slope positions in Site-2008. The bulk density measured in Site-2008 was generally greater than in Site-2007. The greatest bulk density in Site-2008 was approximately 1.8 g cm^{-3}, which was observed at 40-60 cm depth at the UPP slope. While the bulk density also increased with depth in Site-2008, the greatest value was at the 40-60 cm depth at all slope positions.

Table 2.2. Soil physical properties of Site-2007 experimental plot.

Landscape position	Depth (cm)	Field capacity ($cm^3\ cm^{-3}$)	Bulk density ($g\ cm^{-3}$)	Sand	Silt	Clay
				(%)		
		n = 3				
UPP	0-10	0.29 (0.01)	0.98 (0.04)	30 (3.3)	32 (0.3)	38 (3.5)
	10-20	0.34 (0.05)	1.2 (0.13)	31 (3.3)	37 (3.3)	32 (0.0)
	20-40	0.32 (0.04)	1.4 (0.09)	29 (2.9)	39 (3.6)	32 (0.7)
	40-60	0.42 (0.03)	1.6 (0.06)	30 (3.3)	38 (3.3)	32 (0.0)
	60-90	0.37 (0.07)	1.5 (0.14)	31 (4.1)	34 (1.5)	36 (5.7)
	90-120	0.37 (0.04)	1.5 (0.03)	29 (3.2)	36 (1.1)	34 (2.1)
MID	0-10	0.29 (0.04)	0.88 (0.17)	31 (3.1)	42 (1.7)	27 (1.4)
	10-20	0.42 (0.05)	1.4 (0.08)	33 (3.9)	36 (3.3)	31 (0.7)
	20-40	0.38 (0.04)	1.4 (0.04)	33 (3.5)	36 (2.1)	31 (1.4)
	40-60	0.34 (0.02)	1.40 (0.05)	34 (2.1)	35 (1.4)	31 (0.7)
	60-90	0.36 (0.04)	1.50 (0.07)	33 (2.2)	39 (2.9)	28 (0.7)
	90-120	0.38 (0.05)	1.7 (0.04)	37 (3.9)	35 (4.6)	28 (0.7)
LOW	0-10	0.28 (0.03)	0.86 (0.05)	34 (3.6)	36 (2.2)	31 (1.4)
	10-20	0.36 (0.09)	1.2 (0.26)	30 (2.9)	37 (0.7)	33 (3.5)
	20-40	0.37 (0.04)	1.1 (0.11)	35 (3.2)	36 (0.4)	29 (3.5)
	40-60	0.33 (0.07)	1.4 (0.04)	38 (1.2)	34 (1.2)	28 (0.0)
	60-90	0.39 (0.11)	1.7 (0.12)	31 (2.9)	41 (2.9)	28 (0.0)
	90-120	0.33 (0.03)	1.7 (0.13)	36 (2.5)	38 (5.3)	26 (2.8)

n = number of cores taken for sample analysis at each landscape position.
Values in parentheses represent standard deviation from the sample mean.

Table 2.3. Soil physical properties of Site-2008 experimental plot.

Landscape position	Depth (cm)	Field capacity ($cm^3\ cm^{-3}$)	Bulk density ($g\ cm^{-3}$)	Sand	Silt	Clay
				——	(%)	——
		n = 3				
UPP	0-5	0.28 (0.00)	0.88 (0.00)	38 (2.3)	32 (5.2)	29 (2.8)
	5-10	0.35 (0.04)	1.2 (0.01)	38 (3.3)	30 (8.9)	32 (5.7)
	10-20	0.30 (0.07)	1.4 (0.05)	35 (6.2)	31 (3.4)	34 (2.8)
	20-40	0.30 (0.03)	1.4 (0.12)	37 (8.1)	31 (8.1)	32 (0.0)
	40-60	0.38 (0.02)	1.8 (0.03)	39 (7.4)	37 (7.4)	24 (0.0)
	60-90	0.32 (0.01)	1.7 (0.02)	36 (0.6)	34 (3.4)	30 (2.8)
	90-120	0.27 (0.02)	1.5 (0.02)	38 (1.3)	38 (1.3)	24 (0.0)
MID	0-5	0.22 (0.02)	0.69 (0.02)	41 (1.8)	29 (1.1)	30 (2.8)
	5-10	0.35 (0.02)	1.3 (0.07)	41 (0.9)	32 (0.9)	28 (0.0)
	10-20	0.35 (0.02)	1.4 (0.15)	36 (3.1)	34 (0.3)	30 (2.8)
	20-40	0.29 (0.02)	1.4 (0.14)	38 (4.1)	33 (6.9)	30 (2.8)
	40-60	0.31 (0.09)	1.7 (0.04)	33 (9.6)	37 (8.1)	30 (8.5)
	60-90	0.28 (0.06)	1.5 (0.04)	35 (6.2)	37 (0.5)	28 (5.7)
	90-120	0.28 (0.01)	1.5 (0.02)	30 (0.5)	45 (3.3)	26 (2.8)
LOW	0-5	0.24 (0.02)	0.76 (0.02)	41 (1.6)	37 (1.2)	22 (2.8)
	5-10	0.26 (0.04)	0.90 (0.08)	41 (0.1)	32 (0.1)	28 (0.0)
	10-20	0.35 (0.07)	1.2 (0.07)	38 (2.7)	35 (3.0)	28 (5.7)
	20-40	0.33 (0.00)	1.4 (0.13)	42 (8.7)	27 (0.4)	32 (9.3)
	40-60	0.36 (0.08)	1.7 (0.06)	47 (9.7)	27 (9.2)	26 (8.5)
	60-90	0.33 (0.02)	1.6 (0.01)	37 (5.8)	37 (3.0)	26 (2.8)
	90-120	0.32 (0.00)	1.6 (0.03)	34 (2.6)	45 (9.1)	22 (8.5)

n = number of cores taken for sample analysis at each landscape position.
Values in parentheses represent standard deviation from the sample mean.

The percent sand in the top 10 cm depth in Site-2008 was similar for all slope positions. The greatest silt fraction was at the LOW slope but with the smallest % clay compared to the UPP and MID slope positions. The soil textural compositions were somewhat different between the two site-years based on the % sand and clay. Within the 0-10 cm depth, the sand fraction in Site-2008 was greater than in Site-2007 by 7-10% across the landscape. The corresponding clay fraction in Site-2008 was smaller than in Site-2007, except at the MID slope where the % clay was similar between the two site-years.

The greatest saturated hydraulic conductivity (K_{sat}) was in the top 15 cm depth at the LOW slope position in Site-2007, while the smallest values were measured at the UPP slope position (Table 2.4). The K_{sat} generally declined with depth, and it was impossible to obtain realistic values at 60-90 cm depth at the MID and LOW slope positions in Site-2007. The trend in K_{sat} with depth and among slope positions in Site-2008 was similar to that in Site-2007. In both site-years the K_{sat} decreased with depth while the bulk density increased accordingly.

Bathke and Cassel (1991) also observed an inverse relationship between K_{sat} and bulk density within the soil profile in a convex, gently-sloping landscape. The increase in bulk density and the decline in K_{sat} with depth indicated that soil permeability decreased with depth at this landscape, which may consequently enhance subsurface lateral flow (Arnett 1974; Bathke and Cassel 1991).

Table 2.4. Saturated soil hydraulic conductivity (K_{sat} in cm hr^{-1}) of the experimental sites.

Landscape	Depth (cm)	Site-2007	Site-2008
		n = 1 or 2	*n = 1 or 2*
UPP	0-15	0.588	1.74
	15-30	1.99	1.12
	30-60	1.02	1.05
	60-90	0.042	0.989
MID	0-15	2.60	2.05
	15-30	3.07	1.04
	30-60	1.23	0.272
	60-90	–[z]	0.204
LOW	0-15	3. 79	3.09
	15-30	2.19	2.16
	30-60	1.92	1.28
	60-90	–[y]	0.659

n = number of cores taken for sample analysis at each landscape position.
[z,y] It was impossible to obtain realistic measurements at 60-90 cm depths at the MID and LOW landscape positions in Site-2007.

2.4.4 Vertical Distribution of Bromide in Site-2007

In addition to treatment effects of N fertilizer and landscape, the ANOVA test generated model effects of depth and season as sources of variation on the vertical distribution of bromide (Table 2.5). In both site-years, there was a 4-way interaction of N fertilizer, landscape position, soil depth, and sampling seasons on the vertical distribution of bromide. Therefore, the differences in the vertical distribution of bromide between the N fertility treatments were compared among landscape positions in the fall and spring seasons.

In the fall season in Site-2007, the mass of bromide recovered at the soil surface (the top 5 cm depth) at the UPP slope position was approximately 10 kg ha^{-1} (Fig. 2.8a). This amount of bromide was equivalent to 5% of the bromide applied. The mass of bromide in the unfertilized treatment (TRT0) increased with depth from the soil surface, reaching a peak of 33 kg ha^{-1} (17% of Br^- applied) at 10-20 cm depth. Thereafter, bromide declined to 15 kg ha^{-1} at 20-40 cm depth. The mass of bromide remained at 15 kg ha^{-1} throughout the remaining portion of the soil profile down to 120 cm depth.

In the fertilized treatment (TRT90) at the UPP slope, the bromide peak in the fall season was 52 kg ha^{-1} (26% of Br^- applied) at 20-40 cm depth (Fig. 2.8a). Bromide also declined in TRT90 to 12 kg ha^{-1} at 40-60 cm depth and subsequently moved downward into the soil profile in a similar pattern to TRT0. At all points within the soil profile, amounts of bromide measured in both fertility treatments at the UPP slope position were greater than the background amounts.

Table 2.5. Effects of N fertilization and landscape position on the vertical distribution of bromide in Site-2007 and Site-2008.

Model effect	d.f.	Site-2007[z]	Site-2008[y]
		n = 504	*n = 504*
Fertilizer	1	0.2853	0.0034
Landscape	2	<0.0001	<0.0001
Fertilizer × Landscape	2	0.0185	0.4875
Depth	6	<0.0001	<0.0001
Depth × Fertilizer	6	0.0339	<0.0001
Depth × Landscape	12	<0.0001	<0.0001
Depth × Fertilizer × Landscape	12	<0.0001	0.0321
Season	1	<0.0001	<0.0001
Season × Depth × Fertilizer	7	0.0082	0.3363
Season × Depth × Landscape	14	0.0044	0.2816
Season × Depth × Fertilizer × Landscape	14	0.0007	<0.0001
SEM_{fall}[x]		*0.1381*	*0.1655*
SEM_{spring}[w]		*0.1134*	*0.1400*

[z,y] Probability value is significant at $P < 0.1$.
[x]SEM_{fall} = standard error of the mean for bromide distribution in the fall season.
[w]SEM_{spring} = standard error of the mean for bromide distribution in the spring season.
n = number of samples taken for observation.

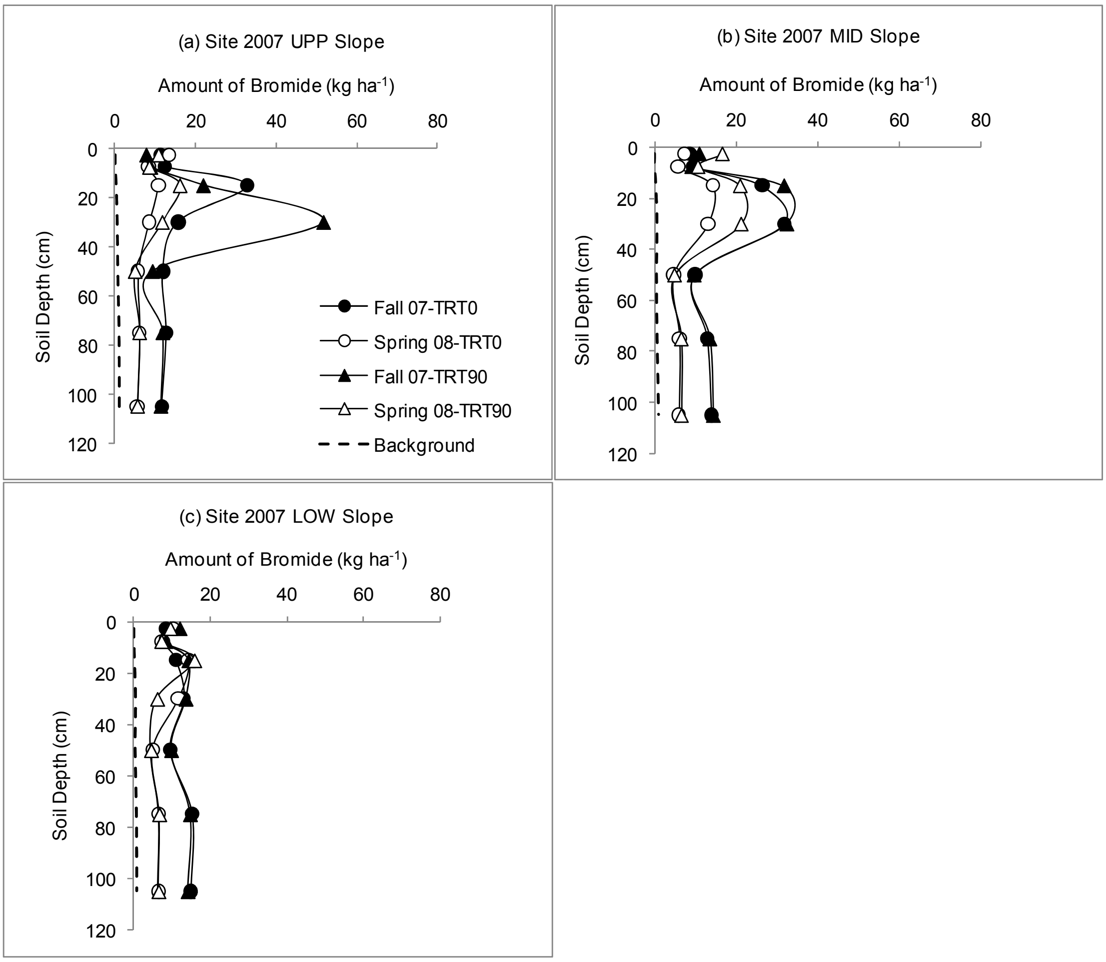

Fig. 2.8. Vertical distribution of bromide between fertility treatments and among landscape positions in fall 2007 and spring 2008 in Site-2007.

In the fall of 2007, the vertical movement of bromide at the UPP slope position was reduced by N fertilization, as more bromide was recovered from TRT90 than from TRT0 (Fig. 2.8a). Although the bromide peak occurred at the 20-40 cm depth in TRT90 compared to 10-20 cm depth in TRT0, inferring the extent of bromide movement based on the position of the centre of mass alone could lead to an erroneous conclusion that the downward movement of bromide was greater in TRT90 than in TRT0. According to amounts of bromide remaining in all sampled depths in the soil profile, it is clear that there was a greater loss of bromide in TRT0 compared to TRT90.

By spring 2008, amounts of bromide had declined for both N fertility treatments at the UPP slope position due to the movement of bromide out of the sampled region through leaching and possibly by lateral flow (Fig. 2.8a). The maximum mass of bromide in the soil profile was at 10-20 cm depth, in the amounts of 10 and 16 kg ha^{-1} for TRT0 and TRT90, respectively. Amounts of bromide recovered in TRT90 were similar or greater than in TRT0, implying that more bromide was lost from the TRT0 plots than from the TRT90 plots. The bromide peak that was observed in both fertility treatments in the fall season had disappeared by spring 2008 as the solute moved out of the soil profile after spring snowmelt.

The reduction in bromide movement with N application at the UPP slope may be due to depletion of soil water in TRT90, as a result of improved root growth and biomass production (Campbell et al. 1984, 1993, Ottman and Pope 2000). In contrast, the smaller amount of bromide in the soil profile of TRT0 suggested that solute movement was enhanced by excess water in the soil probably due to reduced crop water utilization as a result of reduced soil fertility.

The pattern of bromide distribution at the MID slope position in fall 2007 was similar for both N fertility treatments (Fig. 2.8b). The maximum amount of bromide in the soil profile was approximately 30 kg ha^{-1} (15% of Br^- applied), and was centered within 10 to 40 cm depth. Unlike in the fall season when the two fertility treatments had similar amounts of bromide, by the next spring, the amounts of bromide in the top 40 cm depth in TRT90 were significantly greater than in TRT0 (P < 0.02 at each sampled depth within the top 40 cm). At the region below 60 cm depth, bromide distribution was similar for the two fertility treatments as the mass remained as 7 kg ha^{-1} down to 120 cm depth.

The greater amounts of bromide in the top 40 cm depth in TRT90 compared to TRT0 by spring 2008 showed that N fertilization reduced the vertical movement of bromide at the MID slope position. This may be due to greater biomass production and water use by the fertilized crop in the previous growing season, resulting in more upward flow and smaller amount of antecedent water in the soil profile of TRT90 compared to TRT0 by spring 2008. The smaller amounts of bromide in the spring of 2008 compared to fall 2007, especially at depth, inferred that bromide was leached out of the soil profile between fall and spring (Fig. 2.8b).

At the LOW slope position, the mass of bromide in fall 2007 was 15 kg ha^{-1} or less at any point in the soil profile, with little or no fertility effect. There was also only a small difference in amounts of bromide between fall and spring (Fig. 2.8c). The data suggest that a combination of water dissipated from upslope regions, and water flow through pore channels resulted in intense leaching of bromide that eliminated the differences in bromide distribution between the two fertility treatments and also between sampling seasons at the lower slope position in Site-2007.

2.4.5 Vertical Distribution of Bromide in Site-2008

At the UPP slope in Site-2008, amounts of bromide in fall 2008 increased with depth from the soil surface and peaked as 52 kg Br^- ha^{-1} (26% of Br^- applied) at the 40-60 cm depth in TRT0 (Fig. 2.9a). The downward movement of bromide was reduced in TRT90, where the centre of mass of bromide was at 20-40 cm depth, with a bromide peak of 60 kg ha^{-1} (30% of Br^- applied). As such, the centre of mass of bromide travelled further with TRT0 than with TRT90 at the UPP slope position in Site-2008. At the region below the bromide peak in both fertility treatments, bromide declined markedly with depth and a similar amount (6 kg Br^- ha^{-1}) was detected at 90-120 cm depth for both TRT0 and TRT90.

According to the vertical distribution of solute in the spring of 2009, there was a marked reduction in the amount of bromide remaining in the soil profile, compared to the amounts measured in the previous fall at the UPP slope position (Fig. 2.9a). This may be due to further downward transport of bromide following spring snowmelt. Nitrogen fertilization reduced bromide penetration into the lower region of the soil profile at the UPP slope in spring 2009. This was reflected in the greater amount of bromide at 60-90 cm depth in TRT0 compared to TRT90 (P = 0.0004), while there was a greater accumulation of bromide in the top 60 cm depth in TRT90 (Fig. 2.9a).

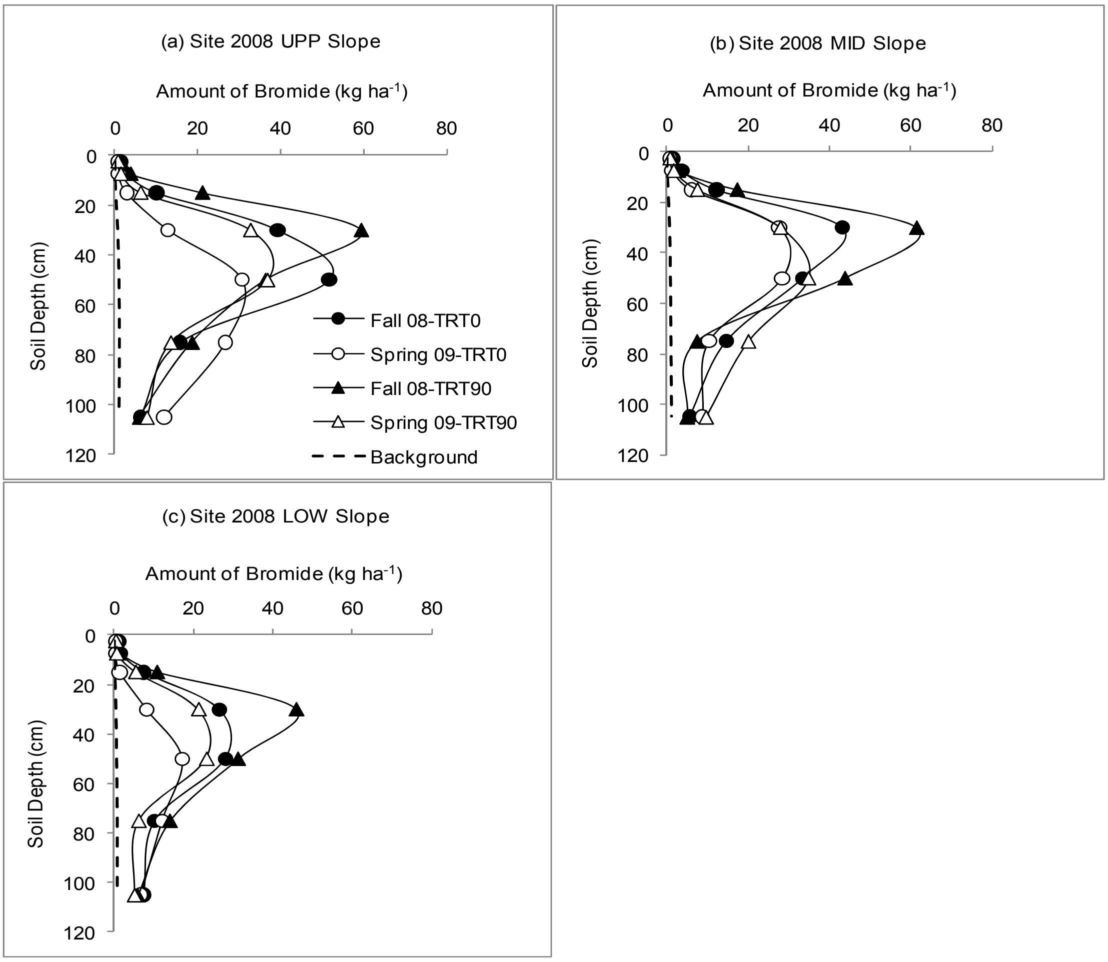

Fig. 2.9. Vertical distribution of bromide between fertility treatments and among landscape positions in fall 2008 and spring 2009 in Site-2008.

At the MID slope position in fall 2008, the mass of bromide also increased with depth from the soil surface in both fertility treatments, resulting in bromide peaks of 43 and 61 kg Br^{-} ha^{-1} at 20-40 cm depth for TRT0 and TRT90, respectively (Fig. 2.9b). The smaller bromide peak in TRT0 compared to TRT90 (P = 0.0143) at the MID slope position indicated a greater movement of solute in the unfertilized plot was probably due to enhanced downward movement by the excess soil water percolating through the root zone. By spring 2009 at the MID slope, the vertical distribution of bromide was similar for the two fertility treatments but there was a greater mass of bromide with depth in TRT90 compared to TRT0 (P = 0.0039 at 60-90 cm depth).

Nitrogen fertilization affected bromide transport at the LOW slope in Site-2008 (Fig. 2.9c). In fall 2008, a bromide peak of 46 kg ha^{-1} occurred at 20-40 cm depth in TRT90. The corresponding value in TRT0 was 27 kg Br^{-} ha^{-1} at the same depth. The bromide peaks for both fertility treatments at the LOW slope position were smaller than those measured at the UPP and MID slope positions; indicating greater vertical transport of bromide at the LOW slope compared to other landscape positions in fall 2008 (Fig. 2.9c). The vertical distribution of bromide at the LOW slope in fall 2008 confirmed the results in fall 2007. The vertical distribution of bromide in both site-years suggests that the greatest leaching loss was at the LOW slope position compared to other landscape segments.

Unlike in Site-2007 where bromide distribution at the LOW slope was similar between fall and spring seasons, the mass of bromide at the LOW slope in Site-2008 had declined by the spring season relative to the amounts measured in the previous fall (Fig. 2.9c). Nevertheless, the results in both years showed that significantly large amount of solute added can disappear from the soil profile at the lower landscape position. Therefore, farmers need to be aware of the greater leaching potential for solutes at the lower slope position and avoid any management practice that will leave residual soil nitrate in this portion of the landscape.

The vertical movement of bromide was reduced by N fertilization compared to the unfertilized treatment in Site-2007 and Site-2008. Campbell et al. (1984, 1993) showed that N fertilization at recommended rates increased plant N uptake and water use in the Brown Soil zone of the Canadian prairies, thereby reducing nitrate leaching compared to the unfertilized treatments or fertilizer application at lower rates. The large amount of nitrate leaching observed at the lower and zero N rates according to Campbell et al. (1984, 1993) was attributed to reduction in crop water utilization due to poor tillering and root growth. As such, there was more soil water available for nitrate leaching at the lower N rates. Since bromide is a tracer for soil water distribution, the differences in the vertical movement of bromide between the two fertility treatments in this study confirmed the results reported by Campbell et al. (1984, 1993).

The differences in vertical transport of bromide among landscape positions can be explained by how topographical attributes influence soil water dynamics. Water has a greater tendency to move laterally at upslope regions of a hillslope due to large slope steepness, compared to the relatively flat lower regions (Hanna et al. 1982; Afyuni et al. 1994; Bathke and Cassel 1991; Rockstrom et al. 1999; Olson and Cassel 1999). The decrease in slope gradient and increase in convergent landscape character in the downslope direction (Table 2.1) clearly indicate effects of runoff and lateral flow on dissipation of water from upslope areas and accumulation at the LOW slope position. As such, the reduced downward movement of bromide at the UPP and MID slope positions compared to the LOW slope was probably due to shedding of incoming precipitation at upslope positions, and its concomitant accumulation at the LOW slope position.

The variability in drainage classification within the landscape is an evidence of long-term effects of topography on soil water flow and hydrologic process. The soils at the upper to mid slope positions are known to be well-drained, while those at the lower slope to depressional areas are generally imperfectly to poorly drained, depending on the proximity of the water table to the soil surface (Podolsky and Schindler 1993; Manning et al. 2001; Whetter et al. 2008). While soil permeability depends on the textural composition, as soil water flow is rapid in coarse-textured soils but slower in fine-textured soils, the soil drainage class strongly depends on the relative position of the soil within the landscape. Therefore, a coarse-textured soil at the lower landscape position may not be necessarily well-drained.

In this study, however, the greatest depth of A horizon (Fig. 2.7) and saturated hydraulic conductivity (K_{sat}) (Table 2.4) were measured at the LOW slope compared to other landscape positions. Other studies conducted on hummocky landscapes in Manitoba (Podolsky and Schindler 1993; Manning et al. 2001; Whetter et al. 2008) have also shown that certain soils with imperfect drainage conditions, such as the Angusville and Penrith series which are commonly found at the lower slope to elevated depression, are also associated with the greatest leaching potential. This is based on their diagnostic soil profile attributes such as the thickness of A horizon and solum, depth to carbonates and the presence of eluviated horizons (Manning et al. 2001; Whetter et al. 2008).

While the soils at the LOW slope position in the present study may be imperfectly-drained, the deposition of topsoil and water removed from the UPP and MID slope positions at the LOW slope over a long period of time, resulted in the greatest thickness of soil depth at the LOW slope position. The characteristically thicker A horizon and greater soil organic matter at the lower landscape are known to enhance the formation of continuous large pore channels in the soil profile, compared to upslope regions (Pennock and de Jong 1990; Fiez et al. 1995; Pachepsky et al. 1999; Manning et al. 2001; Whetter et al. 2008). Therefore, the greatest vertical movement of bromide at the LOW slope may be due to the greatest depth of A horizon and K_{sat} at this landscape position. The soil profile attributes and the greatest K_{sat} at the LOW slope are expected to enhance net infiltration of water dissipated from upslope regions and water received directly from precipitation. The high intensity of soil hydrologic processes resulting from deposition and transmission of water at the LOW slope position would ultimately promote solute leaching at this region of the landscape.

The factors affecting vertical transport of bromide in this landscape are similar to those reported by past authors. Afyuni et al. (1994) showed that the differences in solute transport among landscape positions in the North Carolina Piedmont were due to variations in textural compositions, bulk density, the slope gradient and hydrologic processes. Under the prairie conditions in western Canada, Whetter et al. (2008) also observed that bromide movement through preferential flow paths created by the pore channels was an important transport process at the depression position in the hummocky landscape. Whetter et al. (2008) showed that after bromide application in the fall season, the vertical movement of bromide was greatest at the crest position by the next spring, while the greatest vertical movement occurred at the depression in the subsequent fall season. In the present study, following bromide application in the spring of 2007 and 2008 to Site-2007 and Site-2008, respectively, the greatest vertical movement was at the LOW slope position by the next fall and spring seasons.

A bromide tracer study was conducted at the MZTRA site by Cavers et al. (2002) as part of efforts to assess the leaching potential of various soils in Manitoba. Their results showed that a greater leaching occurred at the lower slope position compared to the upland region during the growing season. The greatest leaching was observed at the elevated depressional area, while no bromide remained in the soil profile (120 cm depth) at the slough area of the landscape. Cavers et al. (2002) reported that the disappearance of bromide at the depressional and slough areas was due to leaching into the water table which was less than 200 cm from the soil surface during the growing season.

The consistently significant downward movement of bromide at the LOW slope position in the present study suggests that producers should avoid excessive amounts of N fertilizer application at the vulnerable lower slope position in hummocky landscapes.

2.4.6 Lateral Distribution of Bromide in Site-2007

The lateral distribution of bromide in Site-2007 was influenced by 3-way interaction of N fertilizer × landscape position × distance, while season and distance were the only main effects that were significant. In Site-2008, however, there was a significant 4-way interaction of N fertilizer, landscape position, distance, and sampling seasons on the lateral distribution of bromide (Table 2.6). Since the main effect of season was significant in both site-years, the lateral distribution of bromide was described separately by graphical illustration for each sampling season in Site-2007 and Site-2008.

The lateral distribution of bromide was derived based on the total mass of bromide in the top 20 cm depth at each of the four locations sampled in the downslope direction outside of the microplot. The mass of bromide at these four locations was compared with that in the microplot to illustrate the lateral movement of bromide away from the point of solute application.

At the UPP slope in fall 2007, the amount of bromide in the top 20 cm depth in the microplot (0-cm lateral distance) was 57 kg ha^{-1} in TRT0, and 40 kg ha^{-1} in TRT90 (Fig. 2.10a). At a distance of 25 cm from the microplot, the mass of bromide was 13 kg ha^{-1} in the top 20 cm in TRT0. Bromide declined slightly as the movement continued over the entire sampled distance in the downslope direction. Measurable amounts of bromide (greater than the background amounts) were obtained at a distance of 175 cm from the zone of application (Fig. 2.10a).

Amounts of bromide in the lateral direction at the UPP slope position were greater in TRT90 than in TRT0, indicating a greater lateral movement and/or less

Table 2.6. Effects of N fertilization and landscape position on the lateral distribution of bromide in Site-2007 and Site-2008.

Model effect	d.f.	Site-2007[z]	Site-2008[y]
		n = 288	*n = 288*
Fertilizer	1	0.0986	<0.0001
Landscape	2	0.3516	<0.0001
Fertilizer × Landscape	2	0.0011	0.0018
Distance	3	<0.0001	<0.0001
Distance × Fertilizer	3	0.1058	0.6806
Distance × Landscape	6	0.0425	<0.0001
Distance × Fertilizer × Landscape	6	0.0005	0.0007
Season	1	<0.0001	<0.0001
Season × Distance × Fertilizer	4	0.8552	0.0757
Season × Distance × Landscape	8	0.6433	<0.0001
Season × Distance × Fertilizer × Landscape	8	0.4359	<0.0001
SEM_{fall}[x]		*0.2843*	*0.0598*
SEM_{spring}[w]		*0.3058*	*0.0886*

[z,y] Probability value is significant at $P < 0.1$.
[x]SEM_{fall} = standard error of the mean for bromide distribution in the fall season.
[w]SEM_{spring} = standard error of the mean for bromide distribution in the spring season.
n = number of samples taken for observation.

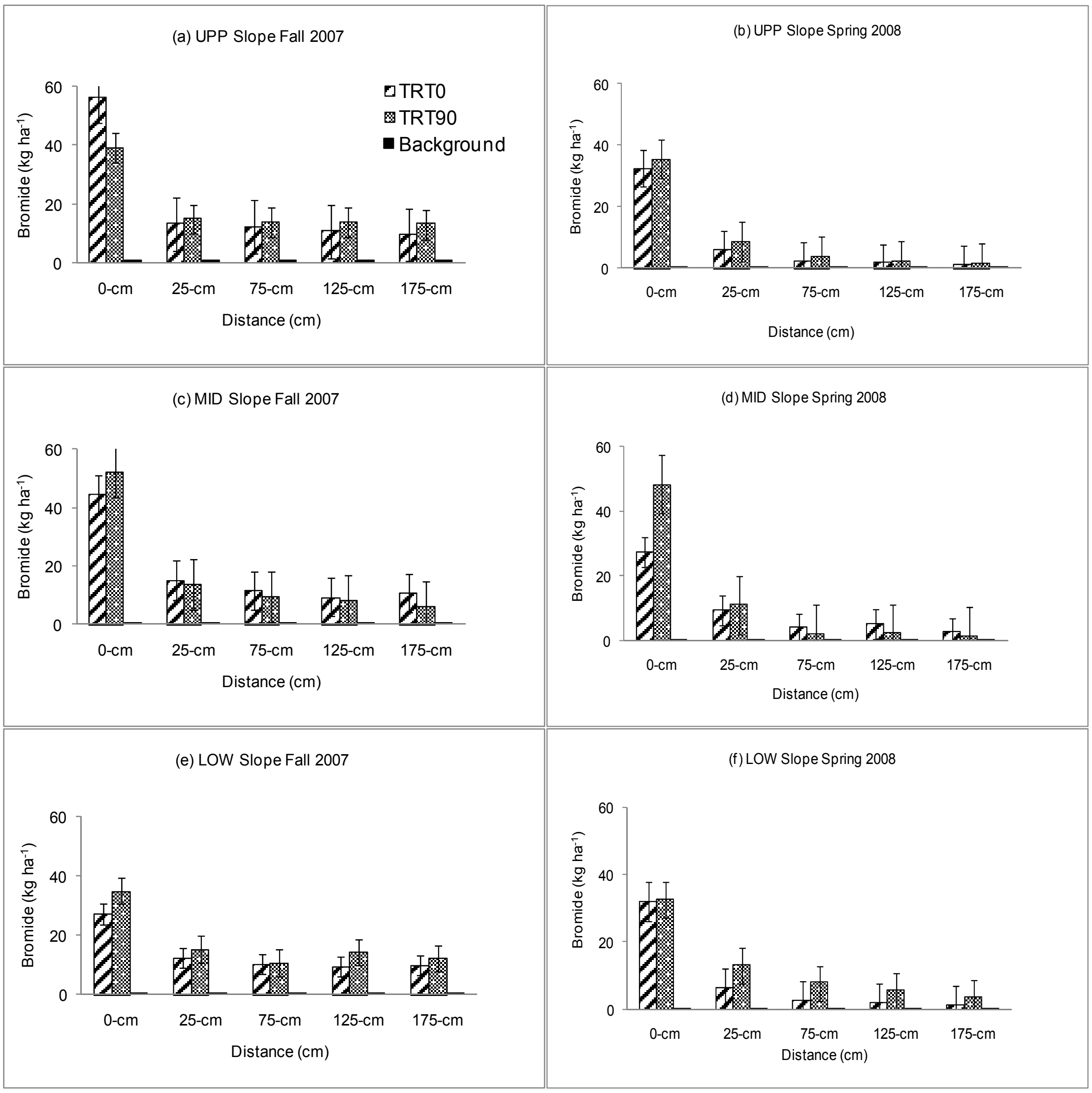

Fig. 2.10. Lateral distribution of bromide in the top 20 cm depth between fertility treatments and among landscape positions in fall 2007 and spring 2008 in Site-2007.

vertical movement of bromide in TRT90 (Fig. 2.10a). This may be due to the reduced vertical movement of bromide in TRT90 which caused a greater accumulation of bromide near the soil surface, thereby permitting more movement in the lateral direction. By spring 2008, amounts of bromide in the lateral direction had declined with distance in both fertility treatments compared to fall 2007, with smaller amounts in TRT0 compared to TRT90 (Fig. 2.10b). The reduction in mass of bromide in the lateral direction by spring 2008 could be more due to downward movement below the 20 cm depth that was sampled, than to further lateral movement.

At the MID slope position, the mass of bromide was 15 kg ha^{-1} at 25 cm distance from the microplot in both fertility treatments in fall 2007 (Fig. 2.10c). At greater distance from the zone of application, the amount of bromide recovered in TRT0 was slightly greater than in TRT90. Overall, the mass of bromide declined from 15 kg ha^{-1} at 25 cm to approximately 5 kg ha^{-1} at 175 cm away from the microplot at the MID slope, particularly in the spring of 2008 (Fig 2.10d).

At the LOW slope position in fall 2007 (Fig. 2.10e), the lateral distribution of bromide indicated that a greater amount of bromide moved downslope from the microplot with TRT90 compared to TRT0, similar to the trend observed at the UPP slope position (Fig. 2.10a). By spring of 2008, amounts of bromide in TRT90 at the LOW slope were still greater than in TRT0 (Fig. 2.10f), but the mass of bromide recovered in both fertility treatments had declined compared to the previous fall perhaps, due to downward movement below the sampled 20 cm depth. The greatest downward and lateral movement of bromide at the LOW slope may be attributed to accumulation and transmission of water dissipated from higher elevations.

The lateral distribution of bromide within 20 to 40 cm depth in the fall of 2007 is presented for TRT0 only (Fig. 2.11). Due to small amounts of bromide in the 20 to 40 cm soil layer, a smaller concentration scale was used to illustrate bromide distribution at this depth compared to the scale used for the lateral movement in the top 20 cm depth. At 25 cm distance from the microplot, the mass of bromide in the 20 to 40 cm soil layer ranged from 3.5 kg ha^{-1} at the UPP slope to 1.0 kg ha^{-1} at the LOW slope (Fig. 2.11).

The mass of bromide declined in an irregular pattern with distance in the downslope direction, while the amount of bromide at 175 cm distance was below the detection limit (Fig. 2.11). At 25 cm distance where the greatest amount of bromide was detected in the 20 to 40 cm layer, the mean mass of bromide was seven times smaller than in the corresponding top 20 cm depth. As a result of the relatively small amounts of bromide recovered in the 20 to 40 cm depth, the depth of sampling for lateral distribution of bromide was subsequently restricted to the top 20 cm.

2.4.7 Lateral Distribution of Bromide in Site-2008

At the UPP slope position in fall 2008, the mass of bromide in TRT90 declined from 26 kg ha^{-1} in the microplot to 10 kg ha^{-1} at 25 cm distance away from the microplot (Fig. 2.12a). Approximately 4 kg Br^{-} ha^{-1} was detected at 175 cm distance in TRT90. At every point sampled in the downslope direction, smaller amounts of bromide were in TRT0 compared to TRT90. By spring 2009, bromide

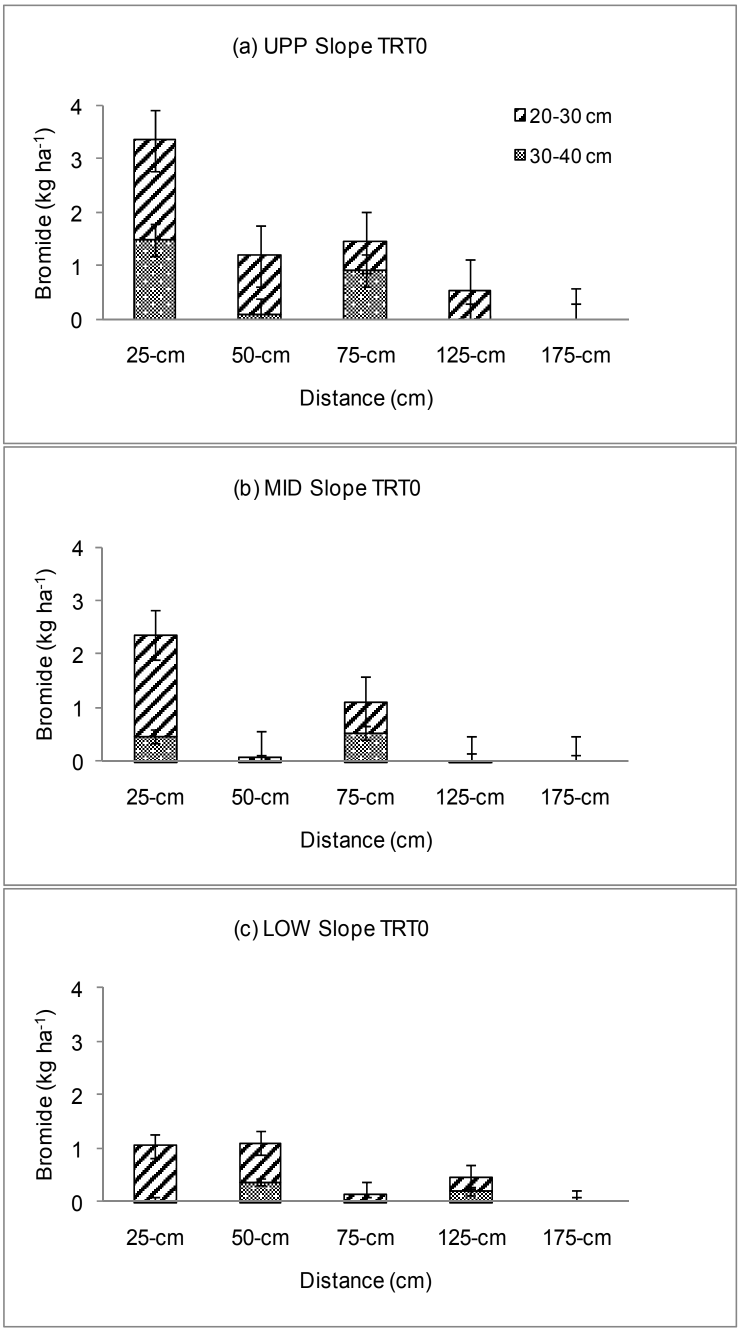

Fig. 2.11. Lateral distribution of bromide within 20 to 40 cm depth among landscape positions with TRT0 in fall 2007.

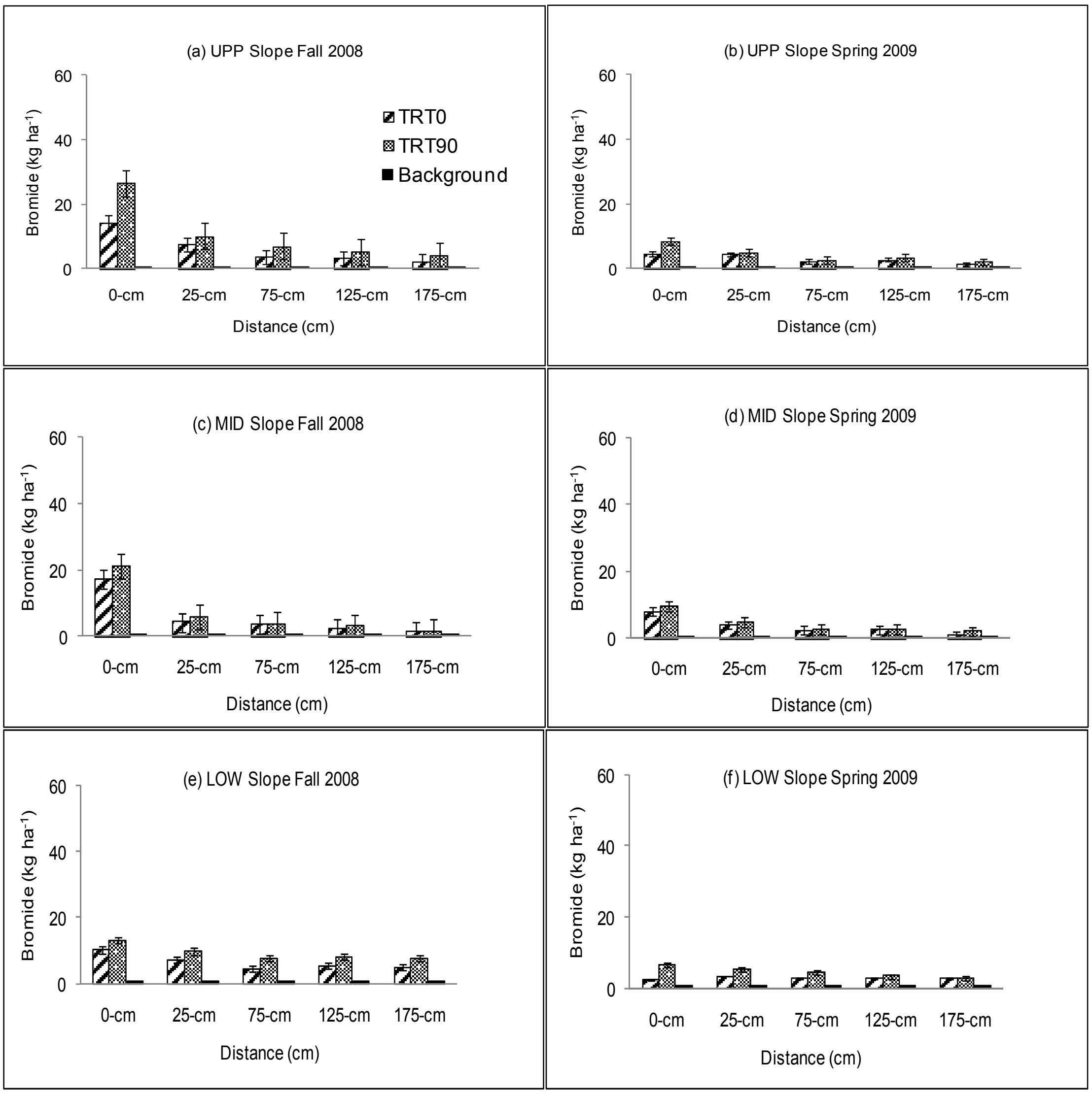

Fig. 2.12. Lateral distribution of bromide in the top 20 cm depth between fertility treatments and among landscape positions in fall 2008 and spring 2009 in Site-2008.

mass had declined compared to fall 2008, while the lateral distribution of bromide was similar for the two fertility treatments (Fig. 2.12b).

There were no marked differences in the lateral distribution of bromide between the two fertility treatments in both sampling seasons at the MID slope position (Figs. 2.12c-d). The data also showed that there was less lateral and/or vertical movement of bromide at the MID slope in Site-2008. At the LOW slope position in fall 2008, greater amounts of bromide were measured in the lateral direction in TRT90 compared to TRT0 (Fig. 2.12e). The greatest lateral and/or vertical movement was at the LOW slope compared to the UPP and MID slope positions. By spring of 2009, the mass of bromide had declined significantly at all sampled points at the LOW slope due to continued lateral and/or vertical movement of bromide at these positions with time (Fig. 2.12f).

The results obtained in Site-2008 showed that the MID slope had the least lateral and/or vertical movement compared to the UPP and LOW slope positions. A similar result was obtained in Site-2007 with the least lateral and/or vertical movement at the MID slope, except that the main effect of landscape on lateral distribution of bromide was significant in Site-2008 and was not in Site-2007 (Table 2.6).

There are several pathways to the fate of bromide within the landscape: bromide can move downward into the soil profile; by lateral movement downslope or across the slope; and by plant uptake which is not expected to be significant since bromide is a conservative tracer. This implies that the mass of bromide that was not recovered in the soil profile was due to downward movement below the 120 cm depth and lateral movement across the slope, in addition to downslope lateral movement. Therefore, it is possible that the characteristic surface curvature within the MID slope area resulted in spreading of solute into the regions perpendicular to the downslope direction, thereby reducing the downslope lateral movement of bromide at the MID slope position.

Overall, N fertilization increased the lateral movement of bromide in both site-years; particularly at the UPP and LOW slope positions. The lateral transport of bromide at the UPP slope may be due to water flow deflected into the horizontal direction as the water encountered a less permeable subsoil layer (Arnett 1974; Bathke and Cassel 1991; Bathke et al. 1992). The marked decrease in soil permeability with depth at the UPP slope was reflected in the smallest magnitude of K_{sat} (Table 2.4) and depths of A horizon and solum (Fig. 2.7) at this landscape region; thereby confirming the significant lateral movement of bromide at the UPP slope position.

The greatest lateral and vertical movement of bromide was at the LOW slope compared to the UPP and MID slope positions in Site-2007 and Site-2008. The greatest lateral movement at the LOW slope can be attributed to accumulation and passage of water transported from higher elevations. The distribution of bromide suggests that water dissipated from upslope regions moved a large amount of bromide both vertically and laterally at the LOW slope position. Afyuni et al. (1994) also observed the greatest vertical and lateral transport of bromide at the footslope position in the Piedmont of North Carolina, which they attributed to the smallest clay content and accumulation of water from higher elevations.

It is also possible that the presence of vegetation in the region surrounding the wetland contributed to more lateral movement at the LOW slope. This is particularly important after harvest and in the early spring when there was no crop uptake in the field area. As the wetland plants transpire, the plant roots may be drawing water away from the cropped field area within the LOW landscape region, thereby enhancing the lateral movement of bromide in the downslope direction.

A detectable amount of bromide was measured at a distance of 175 cm from the zone of application in all landscape positions in both site-years. In a wetter condition (mean annual precipitation of 1,160 mm) in the Piedmont of North Carolina, Afyuni et al. (1994) reported an average downslope lateral transport of 225 cm for bromide. They also showed that the maximum bromide concentration remained in the top 30 cm depth with an application rate of 300 kg Br^- ha^{-1}. With an application rate of 4,030 kg Br^- ha^{-1} in a hummocky landscape of the Canadian prairies, Whetter (2004) observed a maximum downslope movement of 240 cm within the top 20 cm depth in the fall season. The present study confirmed that solutes can be transported to a considerable distance downslope away from the point of application in hummocky landscapes.

Seasonal variability and freeze-thaw conditions are important factors controlling the vertical and lateral redistribution of water and solute in the prairies of western Canada (Whetter et al. 2008; Thibodeau et al. 2008). Due to freezing soil conditions and accumulation of snow on the soil surface during winter period, water and solute movement in the soil cease to continue. As the temperature begins to rise by early spring, the snow layer at the soil surface would start thawing while the subsoil remains frozen.

Since the frozen subsoil remains impermeable in the early stage of snowmelt, there is a greater redistribution of drainage flux into the lateral direction compared to downward vertical flow. As such, a considerable amount of the bromide remaining in the soil, particularly close to the soil surface, would be partitioned into the lateral component of the landscape, thereby enhancing the lateral movement of bromide. This may explain the significant decline in amounts of bromide in the lateral direction by the spring season. However, as the frozen subsoil begins to thaw, the direction of water flow and bromide movement would shift downward into the soil profile, thereby enhancing net infiltration and bromide penetration at the expense of lateral movement. This may also explain the sharp decline in mass of bromide with distance in the lateral direction by the spring season, indicating a vertical redistribution of bromide as opposed to further lateral movement.

2.4.8 Comparison of Bromide Distribution between Site-2007 and Site-2008

There were notable differences in the vertical distribution of bromide between the two site-years. While the vertical distribution of bromide in Site-2007 showed a bulge of bromide near the soil surface with a long tail of 7-15 kg Br^- ha^{-1} at depth, the vertical distribution in Site-2008 has a bell shape or the shape of a normal distribution curve. Amounts of bromide in the top 20 cm depth and at the 90 to 120 cm region of the soil profile in Site-2008 were smaller than in Site-2007.

However, a greater amount of bromide remained within 20 to 60 cm depth in Site-2008 compared to Site-2007. Amounts of bromide measured in the lateral direction in Site-2008 were smaller than in Site-2007. These patterns of bromide distribution indicated that the vertical and lateral movement of bromide in Site-2007 were greater than in Site-2008.

The differences in bromide distribution between Site-2007 and Site-2008 may be attributed to differences in amounts of precipitation. According to the ENVIR weather record, the total precipitation in Site-2008 was 164 mm greater than in Site-2007; it was therefore surprising that both vertical and lateral movements of bromide in Site-2008 were smaller than in Site-2007. The reduced vertical and lateral transport of bromide in Site-2008 may be due to the type of crop seeded to this plot. Winter wheat that was seeded on the plot in 2008 has a different phenology and water use pattern compared to an annual crop such as canola that was planted on the plot in 2007. Unlike a spring crop such as canola, winter wheat utilizes soil water early in the season (Campbell et al. 1984). The depletion of soil moisture in early spring by the winter wheat probably resulted in a drier soil profile with an increased capacity to retain water, which consequently reduced the downward penetration and lateral movement of bromide in Site-2008, compared to Site-2007.

In semi-arid regions of the Canadian prairies, Campbell et al. (1984) showed that winter wheat has a deeper root system and a growth habit similar to fall rye. The authors suggested that the inclusion of winter wheat in a crop rotation program is expected to reduce nitrate leaching. Campbell et al. (2006) later showed that fall-seeded crop removed NO_3-N and water stored in the soil in the fall and early spring season, thereby reducing the potential for nitrate leaching. The findings reported by Campbell et al. (1984, 2006) were confirmed in the results obtained in Site-2008, where winter wheat reduced the downward movement of bromide considerably, in spite of the greater precipitation than in 2007.

The amount of soil-available water in a hillslope depends on the slope aspect (Hanna et al. 1982). Slope aspect is described as the orientation of a hillslope relative to the direction of solar radiation. The surface of a south-facing slope receives a greater intensity of solar radiation, which results in a higher soil temperature and evapotranspiration during the growing season, compared to the north-facing slope. According to Fanning and Fanning (1989), the soil temperatures in the northern hemisphere is approximately 1 to 3°C warmer on south-facing slopes compared to north-facing slopes.

Hanna et al. (1982) showed that the available soil water at planting and throughout the year in the north-facing slope was greater than in the south-facing slope by 20%. Therefore, the smaller magnitude of bromide movement in Site-2008 plot, which was a south-facing slope, may also be due to higher evapotranspiration and soil water depletion resulting from the direct effect of solar radiation on the soil surface temperature, as compared to Site-2007 which was north-facing. However, elevation and slope steepness of the landform at this site may not be large enough to induce such differences in soil water depletion due to different slope aspects.

Past studies (Pennock et al. 1987; Zebarth and de Jong 1989; Manning et al. 2001) have shown the influence of slope curvature on soil water distribution within hummocky landscapes. They reported that soil water accumulation and downward transmission within the landscape with convergent surface form were greater than at the corresponding landscape with divergent surface form.

According to the landform attributes of the experimental sites (Table 2.1), the convergent surface forms in Site-2008 were expected to enhance net infiltration and downward movement of water and bromide in the soil profile while the divergent surface forms in Site-2007 should promote lateral movement of bromide. However, accumulation of bromide in the soil profile, with subsequent smaller amounts of bromide in the lateral component of the landscape in Site-2008 indicated that the extents of vertical and lateral movement of solute in Site-2008 were smaller than in Site-2007. This suggests that effects of crop type on bromide transport in the landscape override the impacts of landform attributes.

There was more clay fraction and less sand fraction in Site-2007 compared to Site-2008 (Tables 2.2-2.3). However, the K_{sat} data (Table 2.4) did not indicate significant differences in soil permeability between the two site-years. Due to the discrepancies between the soil textural composition and the K_{sat} data, it is difficult to associate the variability in soil permeability with the differences in bromide distribution between Site-2007 and Site-2008.

Overall, these results indicated that the extent of solute transport in Site-2007 was greater than in Site-2008, in spite of more precipitation in 2008 compared to 2007. These findings suggest that large amount of precipitation may not directly translate into solute leaching, as bromide transport was greater in Site-2007 than in Site-2008 probably due to effects of crop water uptake on soil water depletion.

2.4.9 Treatment Effects on Crop Yield and Bromide Uptake

Effects of N fertilization and landscape on crop yield and bromide uptake are summarized in Tables 2.7-2.8. Note that both landscape and fertility effects on crop yield and bromide uptake are significant at $P < 0.1$ in this study.

The biomass yield of canola in Site-2007 was significantly greater ($P = 0.0581$) with N fertilization (TRT90) compared to the unfertilized treatment (TRT0), but there was no significant difference in yield among landscape positions (Table 2.7). However, there was a significant interaction of landscape and N fertility treatments on canola yield in Site-2007 ($P = 0.0512$). In the plots without N fertilizer (TRT0), canola biomass yield at the UPP slope was smaller than at other landscape positions. In the fertilized plots (TRT90), however, the yield was similar among landscape positions.

The addition of N fertilizer (TRT90) significantly increased the wheat grain yield ($P = 0.0506$) and dry matter yield of wheat straw ($P = 0.0401$) compared to TRT0 in Site-2008 (Table 2.7). Overall, the grain yield and wheat straw also increased from the UPP slope to the LOW slope position. However, there was a significant fertilizer × landscape interaction ($P = 0.0227$) on straw yield in Site-2008, because the differences in straw yield among landscape positions were greater for TRT0 than for TRT90.

The results showed that the variability in fertility among landscape positions can significantly influence yield response without fertilizer addition. In contrast, N fertilization can reduce differences in yield among landscape positions. However, if excessive amounts of N are applied to the LOW slope

Table 2.7. Effects of N fertilization and landscape position on dry matter yields of canola in Site-2007 and winter wheat in Site-2008.

Fertilizer	Landscape	Site-2007	Site-2008	
		Canola biomass	Wheat grain	Wheat straw
		kg ha^{-1}		
		n = 3		
TRT0	UPP	4298*b*	3605	5373*c*
	MID	6461*a*	4167	7967*b*
	LOW	6405*a*	6162	10479*a*
TRT90	UPP	9369*a*	5065	8833*b*
	MID	8907*a*	6010	11116*a*
	LOW	5283*a*	6968	11707*a*
Fert means				
TRT0		5624	4524*b*	7655
TRT90		7611	5964*a*	10475
Landsc means				
UPP		6346	4273*b*	6889
MID		7586	5005*b*	9411
LOW		5817	6553*a*	11076
Model effect	*d.f.*	*P* value^{z}		
Fertilizer	1	0.0581	0.0506	0.0401
Landscape	2	0.3371	0.0014	0.0147
Fert × Landsc	2	0.0512	0.2655	0.0227
SEM		*0.1769*	*0.0936*	*0.0635*

a – *b*: means with different letter(s) within the column for each treatment effect are significantly different at $P < 0.1$ according to Tukey-Kramer test.

^{z} Probability value is significant at $P < 0.1$.

SEM = standard error of the mean.

n = number of samples taken for observation.

Table 2.8. Effects of N fertilization and landscape position on bromide uptake by canola in Site-2007 and winter wheat in Site-2008.

Fertilizer	Landscape	Site-2007	Site-2008	
		Canola biomass	Wheat grain	Wheat straw
		kg ha^{-1}		
		n = 3		
TRT0	UPP	0.917*b*	0.416	1.11
	MID	2.35*ab*	0.557	0.952
	LOW	7.61*a*	0.616	1.89
TRT90	UPP	6.79*a*	0.861	1.34
	MID	7.83*a*	1.13	1.61
	LOW	4.42*a*	0.962	1.82
Fert means				
TRT0		3.63	0.523*b*	1.26
TRT90		6.35	0.978*a*	1.58
Landsc means				
UPP		3.85	0.598	1.22
MID		5.09	0.793	1.24
LOW		6.02	0.770	1.86
Model effect	*d.f.*	*P* value^{z}		
Fertilizer	1	0.1129	0.0514	0.2361
Landscape	2	0.5545	0.6770	0.1343
Fert × Landsc	2	0.0768	0.8865	0.4595
SEM		*1.850*	*0.3275*	*0.2199*

a – *b*: means with different letter(s) within the column for each treatment effect are significantly different at P < 0.1 according to Tukey-Kramer test.

^{z} Probability value is significant at P < 0.1.

SEM = standard error of the mean.

n = number of samples taken for observation.

position, which has the greatest yield potential particularly in Site-2008, this can result in adverse environmental impacts, as the LOW slope is also associated with the greatest solute loss compared to other landscape positions. The practical recommendation is to apply an optimum rate of N fertilizer based on soil N test and hydrologic condition of each landscape position. Overall, the data in both years showed that N fertilization increased crop yield, thereby inferring the greater soil water consumption with subsequent reduction in downward movement of bromide compared to the unfertilized treatment.

There was a significant fertilizer × landscape interaction (P = 0.0768) on bromide uptake by canola in Site-2007 (Table 2.8). This was due to greater uptake of bromide at the LOW slope than at the UPP slope position with TRT0, while the uptake was similar for all slope positions with TRT90. In Site-2008, there was no statistically significant effect of N fertilizer or landscape on the amount of bromide in the wheat straw. Conversely, N fertilization increased the amount of bromide in the wheat grain; an indication of greater water uptake with fertilizer application. A greater amount of bromide was recovered in the wheat straw compared to the grain, similar to what was observed by Ottman et al. (2000). Bromide uptake by winter wheat was smaller than the uptake by canola.

A detectable amount of bromide was recovered in the plant tissue (1-8 kg ha^{-1}), but this quantity was very small relative to the mass of bromide added (200 kg ha^{-1}). The maximum quantities of bromide taken up by canola and winter wheat were approximately 4 and 2% of the bromide applied, respectively. The proportion of bromide taken up by crops has been reported to be as large as 30% in a grass pasture (Owens et al. 1985), 53% in potato (Kung 1990), and 27% in corn plants (Kessavalou et al. 1996). In contrast, other studies showed that plant uptake of bromide was smaller in irrigated wheat (Ottman et al. 2000; Ottman and Pope 2000) or negligible in a laboratory column study (Gish and Jury 1982) compared to that of nitrogen. Since plant uptake of N is known to be greater than that of bromide, it is important to investigate the crop uptake of the labelled ^{15}N and the anticipated reduction of the extent of nitrate movement in the soil.

2.4.10 Mass Balance and Recovery of Bromide

The total mass of bromide recovered in different components of the soil (vertical and lateral) and plant was obtained to generate a mass balance for bromide, and to estimate the amount of bromide lost from the system (Tables 2.9-2.10). It should be noted that the geometric means of the measured bromide were grouped according to the sampling seasons in each site-year. However, in cases where there is a 3-way interaction of season × N fertility × landscape, all 12 means in the fall and spring seasons were compared as a group. Also note that effects of landscape and N fertility on bromide recovery and loss are regarded as significant at $P < 0.1$.

In the fall of 2007, the total mass of bromide recovered in the vertical section of the soil within the zone of treatment application (microplot) ranged from 40 to 62% of the bromide applied (Table 2.9). The amount of bromide recovered in the vertical component of the landscape with TRT90 was greater than with TRT0, particularly at the UPP slope where the difference was significant between the fertility treatments. The smallest amounts of bromide were measured in the vertical component at the LOW slope compared to other landscape positions.

Table 2.9. Geometric means for bromide mass balance in Site-2007.

Landscape	Fertilizer	Vertical	Lateral	Plant	Total	Loss[z]	Recov[y]
		kg ha^{-1}					%
Fall 2007							
UPP	TRT0	108*b*	40.6	0.917*b*	150	50.3*ef*	74.9
	TRT90	124*a*	46.5	6.79*a*	177	23.4*g*	88.3
MID	TRT0	113*ab*	31.5	2.35*ab*	146	53.7*efg*	73.2
	TRT90	122*a*	29.7	7.83*a*	159	40.7*fg*	79.7
LOW	TRT0	79.7*cd*	38.6	7.61*a*	124	75.8*cd*	62.1
	TRT90	86.5*bc*	48.9	4.42*a*	139	61.0*de*	69.5
Spring 2008							
UPP	TRT0	56.8*e*	10.6	–	68.3	132*a*	34.2
	TRT90	63.8*de*	16.0	–	86.5	114*ab*	43.2
MID	TRT0	56.9*e*	19.9	–	78.8	121*a*	39.4
	TRT90	87.0*bc*	15.3	–	110	90.2*b*	54.9
LOW	TRT0	61.6*e*	10.1	–	77.6	122*a*	38.8
	TRT90	56.2*e*	29.6	–	89.4	111*ab*	44.7
Fertilizer means							
TRT0		76.1	19.4	3.63		92.5	
TRT90		86.1	26.0	6.35		73.2	
Landscape means							
UPP		83.5	22.5	3.85		79.7	
MID		90.9	20.7	5.09		76.4	
LOW		69.9	24.5	6.02		92.4	
Season means							
Fall		106	39.3*a*	–		50.8	
spring		63.7	16.9*b*	–		115	
Fert × Landsc means							
UPP-0		78.4	18.6*ab*			91.0	
UPP-90		88.8	27.3*ab*			68.5	
MID-0		80.2	22.4*ab*			87.5	
MID-90		103	19.0*ab*			65.5	
LOW-0		70.1	17.6*b*			99.1	
LOW-90		69.8	34.0*a*			85.8	
Model effect	*d.f.*	*P* value					
Fertilizer	1	0.0955	0.0151	0.1129		0.0647	
Landscape	2	<0.0001	0.7283	0.5545		0.0115	
Fert × Landsc	2	0.0091	0.0187	0.0768		0.0938	
Season	1	<0.0001	<0.0001	–		<0.0001	
Seas × Fert	1	0.4010	0.1098	–		0.0793	
Seas × Landsc	2	0.0019	0.1678	–		0.0079	
Seas × Fert × Landsc	2	0.0069	0.1800	–		0.0600	
SEM		*0.0623*	*0.2293*	*1.850*		*0.2380*	

a – c: means with different letter(s) within the column are significantly different at $P < 0.1$ according to Tukey-Kramer test. Probability value is significant at $P < 0.1$.

[z] Apparent loss of bromide; Loss = Mass of bromide applied (200 kg ha^{-1}) - Total bromide

[y] Recovery = $\dfrac{\text{Total bromide measured from all pools}}{\text{Bromide applied (200 kg ha}^{-1})} \times 100$

Bromide uptake was accounted for in the Total for spring 2008.

SEM = standard error of the mean.

Table 2.10. Geometric means for bromide mass balance in Site-2008.

Landscape	Fertilizer	Vertical	Lateral	Plant	Total	Loss[z]	Recov[y]
		kg ha^{-1}					%
Fall 2008							
UPP	TRT0	126	16.2*c*	1.53	144	56.4	71.8
	TRT90	142	26.3*b*	2.31	171	29.4	85.3
MID	TRT0	112	12.2*de*	1.60	126	74.2	62.9
	TRT90	136	13.6*cd*	2.78	153	47.3	76.4
LOW	TRT0	84.3	23.1*b*	2.57	110	90.0	55.0
	TRT90	113	33.6*a*	2.91	149	50.6	74.7
Spring 2009							
UPP	TRT0	86.7	9.9*e*	–	98.1	102	49.1
	TRT90	98.2	12.6*d*	–	113	86.9	56.6
MID	TRT0	81.7	10.1*e*	–	93.4	107	46.7
	TRT90	99.8	11.7*de*	–	114	85.7	57.2
LOW	TRT0	47.7	11.9*de*	–	62.2	138	31.1
	TRT90	64.5	15.8*c*	–	83.2	117	41.6
Fertilizer means							
TRT0		85.4	13.3	1.90*b*		94.5*a*	
TRT90		106	17.4	2.67*a*		69.5*b*	
Landscape means							
UPP		112	15.2	1.90		68.7*c*	
MID		107	11.8	2.20		78.5*b*	
LOW		71.3	19.6	2.75		98.8*a*	
Season means							
Fall		119*a*	20.8	–		58.0*b*	
spring		79.8*b*	12.0	–		106*a*	
Fert × Landsc means							
UPP-0		105*bc*	12.7			79.2	
UPP-90		120*a*	18.2			58.2	
MID-0		96.9*c*	11.1			90.4	
MID-90		118*ab*	12.6			66.5	
LOW-0		61.4*e*	16.6			114	
LOW-90		82.7*d*	23.1			83.7	
Model effect	*d.f.*	*P* value					
Fertilizer	1	<0.0001	<0.0001	0.0658		0.0001	
Landscape	2	<0.0001	<0.0001	0.2211		<0.0001	
Fert × Landsc	2	0.0740	0.0009	0.5635		0.2761	
Season	1	<0.0001	<0.0001	–		<0.0001	
Seas × Fert	1	0.9860	0.0542	–		<0.0001	
Seas × Landsc	2	0.0015	<0.0001	–		0.0007	
Seas × Fert × Landsc	2	0.9951	0.0756	–		0.1756	
SEM		*0.0508*	*0.0478*	*0.2099*		*0.0938*	

a – c: means with different letter(s) within the column are significantly different at $P < 0.1$ according to Tukey-Kramer test. Probability value is significant at $P < 0.1$.

[z] Apparent loss of bromide; Loss = Mass of bromide applied (200 kg ha^{-1}) - Total bromide

[y] Recovery = $\dfrac{\text{Total bromide measured from all pools}}{\text{Bromide applied (200 kg ha}^{-1}\text{)}} \times 100$

Bromide uptake was accounted for in the Total for spring 2009.

SEM = standard error of the mean.

By spring of 2008, the mass of bromide in the vertical component had declined for both TRT0 and TRT90 at all slope positions (Table 2.9). Unlike in the fall season when the effect of N fertility on bromide recovery in the vertical component was significant at the UPP slope position, the amount of bromide was significantly greater with TRT90 than with TRT0 at the MID slope in spring 2008. This explains the significant interaction of season × N fertility × landscape (P = 0.0069) on bromide recovery in the vertical component in Site-2007.

At the LOW slope position, the mass of bromide recovered in the vertical component had declined to approximately 80 kg ha^{-1} by fall 2007. Following the significant reduction in mass of bromide at the LOW slope position by the fall season, the amount of bromide remaining in the vertical component by spring 2008 was 30% of the bromide applied. Across landscape positions and between fertility treatments, the mass of bromide remaining in the vertical component by spring 2008 was equivalent to 60% of the amount recovered in the previous fall, indicating a further loss of 40% of the remaining bromide between fall 2007 and spring 2008.

In the lateral component of the landscape, the significant interaction of landscape position and N fertility treatment (P = 0.0187) indicated that addition of N fertilizer increased the mass of bromide only at the LOW slope position in both sampling seasons (Table 2.9). Approximately 20% of the bromide applied was in the lateral component in fall 2007, which had declined by 57% of that measured in the fall season by spring 2008. While there was a significant interaction of landscape by N fertility (P = 0.0768) on the amount of bromide in the canola tissue, N fertilization increased bromide uptake only at the UPP slope position

(Table 2.9). The average amount of bromide recovered from the canola tissue was 2.5% of the bromide added.

An apparent loss of bromide was estimated as the missing mass of bromide after accounting for the total recovery, using a simple arithmetic approach (Table 2.9). According to the estimated bromide loss in fall 2007 and spring 2008, addition of N fertilizer (TRT90) consistently reduced the magnitude of bromide lost, compared to the unfertilized treatment (TRT0). However, the degree of bromide loss was affected by the 3-way interaction of season × N fertility × landscape (P = 0.0600). Nitrogen fertilization significantly reduced bromide loss at the UPP and MID slope positions in fall 2007 and spring 2008, respectively. Between fall 2007 and spring 2008, a significant amount of bromide was lost from the soil components at all three landscape positions and in both fertility treatments. As much as 88% of the bromide applied was recovered from all components (soil and plant) in fall 2007. However, only 43% of bromide applied was recovered in spring 2008 (Table 2.9).

In the fall season in Site-2008 (Table 2.10), amounts of bromide recovered in the vertical component of the landscape were significantly influenced by interactions between landscape and N fertility (P = 0.0740) and between landscape and season. The results showed that a greater amount of bromide was recovered in the vertical component at all three landscape positions due to N fertilization, while the smallest amount of bromide was recovered at the LOW slope position. The mass of bromide recovered in the vertical component in fall 2008, relative to the amount of bromide applied, ranged from 42% in TRT0 at the LOW slope to 71% in TRT90 at the UPP slope. Between fall 2008 and spring

2009, the mass of bromide in the vertical component had declined significantly by 33% of that measured in fall 2008.

There was a significant interaction of season × N fertility × landscape (P = 0.0756) on amounts of bromide recovered in the lateral component in Site-2008 (Table 2.10). In both sampling seasons, the amount of bromide measured in TRT90 was consistently greater than in TRT0 at the UPP and LOW slope positions. While the greatest amount of bromide was measured at the LOW slope position in TRT90, the smallest amount was at the MID slope, where there were no significant differences in mass of bromide between the fertility treatments. Approximately 10% of the bromide applied was recovered in the lateral component in fall 2008. By spring 2009, the amount of bromide in the lateral component had declined by 42% of that in the previous fall season.

Nitrogen fertilization increased the mass of bromide recovered in winter wheat (P = 0.0658), but the amount of bromide in the plant tissue was 1.2% of the bromide added (Table 2.10). Overall, the apparent loss of bromide in Site-2008 confirmed that N fertilization reduced bromide loss (P = 0.0001), while the loss among landscape positions was in the order of: LOW > MID > UPP (P < 0.0001). Between fall 2008 and spring 2009, the magnitude of bromide loss increased by 50%.

The significant interaction between landscape position and N fertility was consistent for bromide recovery in the vertical and lateral components of the landscape. The missing mass of bromide in the soil profile confirmed the greater vertical movement of solute in the unfertilized treatment (TRT0) compared to N addition (TRT90), and the greatest vertical movement at the LOW slope compared to other landscape positions. Conversely, in cases where there was accumulation of bromide in the lateral component of the landscape, the data confirmed the greater lateral movement of bromide with N fertilization and at the LOW slope position.

These results also infer that the vertical distribution of solute in the hummocky landscape is greater than the lateral distribution, as more bromide was partitioned into the vertical component than into the lateral component. In the fall season following bromide application, 50-60% of the bromide added was recovered in the vertical component, compared to 10-20% in the lateral component of the landscape. Overall, these findings suggest that N fertilizer management practices and variations in soil water distribution among landscape positions are critical to viable crop production and environmental sustainability in agricultural landscapes, as N fertility treatments and landscape positions interact to impact the two-dimensional redistribution of bromide within the hummocky landscape.

2.5 Summary and Conclusions

Effects of N fertilization on the vertical and lateral redistribution of bromide were examined in a hummocky landscape in the fall and spring seasons. Nitrogen fertilization decreased losses of bromide, presumably due to reduced downward movement of bromide in the soil profile. This reduction resulted in accumulation of bromide in the soil profile, resulting in a greater lateral movement with N fertilization, compared to the unfertilized plots. The greatest vertical and lateral movement of bromide occurred at the lower slope position, compared to other landscape positions.

The vertical and lateral transport of bromide in Site-2008 was smaller than in Site-2007. The differences in bromide redistribution between the two site-years were attributed to differences in crop type and water use. A greater amount of the bromide applied was partitioned into the vertical component compared to the lateral component of the landscape, suggesting that the vertical downward transport of solute is more important than the lateral downslope movement. The proportion of bromide in the plant tissue was negligible compared to the amount applied. Between fall and spring season, amounts of bromide in the soil had declined significantly.

The study showed that there was a significant interaction of landscape and N fertility on the two-dimensional redistribution of solute within the hummocky landscape. These findings were able to verify the “Campbell hypothesis” which states that proper nitrogen fertilization reduces nitrate leaching. The findings also suggest that precision farming techniques based on soil N test and site-specific hydrologic condition, should be strongly considered as vital components of best management practices for reducing the leaching loss of residual soil nitrate, particularly at the vulnerable lower slope position.

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3. VERTICAL AND SEASONAL REDISTRIBUTION OF BROMIDE AND ^{15}N-LABELLED NITRATE IN A HUMMOCKY LANDSCAPE

3.1 Abstract

Bromide is used as a tracer to estimate nitrate distribution in the soil. An alternative is the use of labelled ^{15}N to provide a direct estimate of nitrate lost by leaching. A dual application of Br^- and ^{15}N was employed to estimate NO_3-N leaching in a hummocky landscape. The study was carried out near Brandon, Manitoba in 2007 and 2008, using two separate plots denoted as Site-2007 and Site-2008, respectively. The plots were delineated into three landscape positions as upper (UPP), middle (MID) and lower (LOW) slope. A microplot at each landscape received KBr at the rate of 200 kg Br^- ha^{-1}, and ^{15}N fertilizer as KNO_3 at the rates of 0, 90 and 135 kg N ha^{-1}. Site-2007 was seeded to canola while Site-2008 had winter wheat. The harvested plant tissue was analyzed for Br^-, total N and isotope N ratio. Soil samples were taken in the fall and spring seasons. The soil samples for treatments 90 and 135 kg N ha^{-1} were analyzed for vertical distribution of Br^-, ^{15}N, and NO_3-N. The smallest amounts of Br^-, ^{15}N, and NO_3-N were measured in the soil profile at the LOW slope for both rates of N fertilizer addition, while the greatest amounts were at the MID slope, indicating that solute loss followed the order of: LOW > UPP > MID. The vertical depth to

which 50% of the solute had leached (Q50) by the fall season showed that Br^- transport overestimated nitrate leaching by twice the estimate with ^{15}N. Crop uptake of ^{15}N was 35% and 63% of total N applied in canola and winter wheat, respectively, while Br^- uptake was negligible. In the absence of crop uptake, Br^- transport was identical to that of ^{15}N as 38% of both solutes were lost between fall and spring in Site-2007 and 33% in Site-2008, but NO_3-N distribution remained unchanged. By using the dual tracer technique, the results point to the limitations of using soil profile nitrate-N alone as an indicator of nitrate leaching. These findings have implications for precision farming practices, as it will be advisable to minimize fertilizer application at the lower slope position while the middle slope position can receive more N fertilizer since it is the region with the least leaching potential.

3.2 Introduction

Nitrate can be leached from agricultural systems if available soil nitrogen exceeds plant requirements (Campbell et al. 1994). Nitrate leaching can also occur in the absence of plant growth that is capable of utilizing soil NO_3-N before excess water moves it beyond the root zone (Follett 1989). In agricultural landscapes, soil physical properties and topographic features are the principal factors controlling soil water transmission and solute redistribution in the vadose zone (Bathke and Cassel 1991; Bathke et al. 1992; Afyuni et al. 1994; Farrell et al. 1996; Olson and Cassel 1999; Mohanty and Mousli 2000). Previous studies have shown that N fertilizer management practices also have a significant effect on the magnitude of nitrate leaching in agricultural soils (Randall and Iragavarapu 1995), particularly in semi-arid regions of western Canada (Campbell et al. 1984, 1993).

The labelled isotope ^{15}N is widely used for investigating the transport and recovery of N in hydrologic and agricultural research (Olson and Swallow 1984; Jensen 1991). The ^{15}N enrichment method permits the study of N dynamics by measuring the changes in N isotope ratios (Mulla and Strock 2008). However, the labelled ^{15}N is susceptible to microbial immobilization, denitrification losses, and plant uptake (Kessavalou et al. 1996; Ottman et al. 2000; Ottman and Pope 2000). It is also possible that some of the added ^{15}N may be incorporated into the soil organic N reserve with a subsequent mineralization of the native ^{14}N, thereby diluting the concentration of ^{15}N in the soil (Jenkinson et al. 1985; Kessavalou et al. 1996). Therefore, accurate estimation of nitrate distribution in agricultural soils based on ^{15}N tracer alone is limited by the biological factors affecting N speciation.

Bromide is used as an alternative tracer for estimating nitrate distribution in the soil because of the similar properties of the two anions (Smith and Davis 1974; Bowman 1984). The amount of bromide leached represents the maximum nitrate leaching potential of the soil (Smith and Davis 1974; Kessavalou et al. 1996). However, Kessavalou et al. (1996) showed that the amount of bromide leached from the soil was greater than that of nitrate, thereby exaggerating the magnitude of nitrate leaching. This was attributed to the non-reactive behaviour of bromide in the soil, unlike the reduction of nitrate movement due to microbial immobilization, gaseous losses and plant uptake.

It is clear from the foregoing that there are limitations with the use of bromide or ^{15}N tracer alone for estimating nitrate distribution in the soil. As such, the fate and movement of nitrate should be investigated using the dual application of both tracers. While both Br^- and ^{15}N have been applied to estimate NO_3-N leaching in the soil (Kessavalou et al. 1996; Ottman et al. 2000; Ottman and Pope 2000), these investigations were conducted under irrigation studies, and there was no information on landscape effects on nitrate redistribution.

In the prairie region of western Canada, several researchers (Woods et al. 2006; Whetter et al. 2008; Thibodeau et al. 2008) have investigated the spatial redistribution of chloride and bromide in hummocky landscapes. Nevertheless, none of these studies examined nitrate movement using labelled ^{15}N in conjunction with the bromide tracer. Other researchers (Jowkin and Schoenau 1998; Malhi et al. 2004; Soon and Malhi 2005) have evaluated effects of slope positions on soil N dynamics in hummocky landscape using labelled ^{15}N experiments. The findings reported from these studies were limited to N availability and recovery, but did not describe nitrate transport. It appears there has not been a comparative evaluation of solute movement among landscape positions using a dual application of Br^- and ^{15}N on the Canadian prairies.

To contribute to the understanding of the fate and transport of nitrate, various hypotheses were tested in this study. First, whether the earlier findings reported under irrigation system, that bromide transport overestimated nitrate leaching potential (Kessavalou et al. 1996; Ottman et al. 2000; Ottman and Pope

2000) can be verified under natural rainfall and landscape variability. Second, whether there is a significant interaction between rates of N fertilizer application and landscape positions on the vertical distribution of solutes. Third, it was hypothesized that the transport of Br^- and ^{15}N would follow a similar trend across landscape positions, and between fall and spring season in the absence of crop uptake. Therefore, the objectives of this study were: (i) to investigate the vertical movement of nitrate within a hummocky landscape using a dual application of bromide and labelled ^{15}N; (ii) to identify the landscape position with the greatest potential for leaching losses of nitrate; (iii) to quantify the redistribution and recovery of solutes at the end of the growing season in fall and shortly after snowmelt in spring.

3.3 Materials and Methods

The description of the study area and the field layout has been presented in Chapter 2. Other aspects of the research methodology such as treatment application, harvesting and soil sampling have also been covered in the previous chapter, and therefore, will not be repeated in this section.

3.3.1 Plant and Soil Analyses

Four hundred milligrams of the plant tissue sample was digested using the modified wet oxidation method of Akinremi et al. (2003). The acid-digested samples were neutralized and analyzed for bromide. Approximately 10 g of the ground soil sample was mixed with 30 mL of distilled water and mechanically shaken for 30 minutes at 120 strokes per minute on a reciprocating shaker. The saturated soil-water mixture was centrifuged at 3466 × *g* (Accuspin™ 400, Fisher Scientific Ltd.). The clear solution was decanted and analyzed for bromide.

The bromide content in plant tissue and soil samples was measured by bromide selective electrode (Orion 9635) fitted to a dual channel pH/Ion/Conductivity meter (Accument Research AR50, Fisher Scientific). The soil samples were obtained from the microplots and analyzed for the vertical distribution of NO_3-N in the soil profile. The sample was extracted with 2*N* KCl (1:5 soil to solution ratio) and NO_3-N content was measured by the Automated Cadmium Reduction method (Clesceri et al., 1998) using the Technicon Autoanalyzer.

A portion of the ground plant tissue and the soil sample was pulverized. Fifty milligrams of the pulverized soil sample and 6.5 mg of the plant tissue were taken to determine the total N and isotope N ratio. The total N and the ^{15}N atom% in the plant and soil samples were determined by the Dumas combustion technique using a Combustion Analyzer (Carlo Erba NA1500, Carlo Erba, Milan, Italy) interfaced with Optima Mass Spectrometer (V.G. Isotech, Middlewich Cheshire, United Kingdom) at the Agriculture and Agri-Food Canada research facility at Lethbridge, AB.

Solute concentration in mg kg^{-1} soil was converted to kilograms per hectare, using the corresponding bulk density at the sampling point. The vertical distribution of solute in the soil profile was illustrated by plotting the mass of solute recovered per soil layer in kg ha^{-1} with the midpoint of each sampled depth.

The percent N derived from the labelled fertilizer (%NDFF) was converted to kg ha^{-1} basis using the equations of Hauck and Bremner (1976) as follows:

$$\%\text{NDFF in plant or soil} = \frac{{}^{15}\text{N atom\% in sample} - \text{background } {}^{15}\text{N atom\%}}{{}^{15}\text{N atom\% in fertilizer} - \text{background } {}^{15}\text{N atom\%}} \times 100$$

where, the background ^{15}N atom% = 0.3663

NDFF (kg ha^{-1}) = %NDFF in sample × % total N × yield or soil mass in kg ha^{-1}

3.3.2 Statistical Tests and Data Analysis

Analysis of variance (ANOVA) tests were conducted on the measured variables using the PROC MIXED procedure of SAS® software for Windows (Version 9.1, SAS Institute, Inc., Cary, NC). The treatment effects due to N fertilizer rate, landscape, and their interactions on solute transport were tested against six replicates of soil samples for bromide and nitrate, while the three replicates of composite samples were tested for ^{15}N. The PROC MIXED procedure was treated as a split-plot design with a double repeated measure whereby the random effect was the replicate (Rep), while depth and season were regarded as the repeated measures.

Various covariance structures were examined for the PROC MIXED procedure. The First-order Autoregressive [AR(1)] covariance structure consistently produced the smallest value of fit statistics based on the Akaike's Information Criterion (Littell et al. 1996). Therefore, AR(1) error structure was selected for all the double repeated analyses, thereby imposing homogeneous variances and correlations that decline exponentially with depth and season on the observed data (Kincaid 2005).

The PROC UNIVARIATE tests for normality indicated that, in almost all cases, the data were significantly different from a normal distribution, based on the Shapiro-Wilk's normality test and skewness of the histograms distribution. As such, the raw dataset was transformed using natural log transformation to achieve homogeneity of variances commonly required for parametric statistics.

The mass of solute was plotted using the raw data to in order to capture the spatial and temporal distribution of solute in the soil profile, while the back-transformed estimates of the transformed means were used to calculate solute mass balance. The Tukey-Kramer test was used to determine the differences in treatment means at $P < 0.1$. A probability level of 0.1 was chosen for comparing treatment means in this study because of the large variability commonly associated with field experimentation of N cycling at the landscape scale (Walley et al. 1996; Beckie and Brandt 1997; Zvomuya et al. 2003; Kutcher et al. 2005).

The vertical depth to the centre of mass of solutes was calculated to estimate the extent of downward movement of individual solute across the landscape in the fall seasons of 2007 and 2008. The centre of mass was derived by first dividing the cumulative mass of solute recovered in the soil profile by two (Bathke et al. 1992; Olson and Cassel 1999; Whetter 2004). Then, the incremental cumulative mass of solute with depth was plotted and the distribution curve was fitted using an exponential function. The depth at which the centre of mass occurred was estimated by interpolation as the vertical depth to which 50% of the solute had leached, which was denoted as Q50.

In this chapter, the vertical distribution of Br^-, nitrogen derived from the labelled ^{15}N fertilizer (NDFF), and NO_3-N was compared between the N fertilized treatments as 90 kg N ha^{-1} (TRT90) versus 135 kg N ha^{-1} (TRT135). This was to illustrate effects of N fertilizer rate on solute distribution among landscape positions.

3.4 Results and Discussion

3.4.1 Treatment Effects on the Vertical Distribution of Solutes

In addition to treatment effects of N fertilizer rate and landscape, the ANOVA test generated model effects of depth and season as sources of variation on the vertical distribution of solutes (Tables 3.1-3.2). There was a significant 4-way interaction of N fertilizer rate, landscape position, soil depth, and sampling season ($P < 0.0001$) on the vertical distribution of bromide in Site-2007 and Site-2008 (Tables 3.1-3.2). Unlike the results obtained for bromide, there was a significant N fertilizer rate × landscape × depth × season interaction on the vertical distribution of ^{15}N only in Site-2008 ($P = 0.0823$).

There were significant effects of N fertilizer rates and landscape positions ($P < 0.1$) on the vertical distribution of nitrate in Site-2007 (Table 3.1). In Site-2008 (Table 3.2), however, there was a significant interaction of depth × landscape × N fertilizer rate on the vertical distribution of nitrate ($P = 0.0009$). The main effect of season on the vertical distribution of nitrate was not significant in both site-years.

Table 3.1. Treatment effects and their interactions on the vertical distribution of solutes in Site-2007.

Model Effect	d.f.	Bromide[z]	Soil ^{15}N[y]	Nitrate[x]
			n = 504	
Rate	1	0.8699	0.1421	0.0416
Landscape	2	<0.0001	0.0126	0.0001
Rate × Landscape	2	0.5774	0.0221	0.2161
Depth	6	<0.0001	<0.0001	<0.0001
Depth × Rate	6	0.0419	0.0105	<0.0001
Depth × Landscape	12	<0.0001	<0.0001	<0.0001
Depth × Rate × Landscape	12	0.0126	0.0138	0.1378
Season	1	<0.0001	<0.0001	0.2950
Season × Depth × Rate	7	0.4392	0.0426	0.4227
Season × Depth × Landscape	14	<0.0001	0.0289	0.1450
Season × Depth × Rate × Landscape	14	<0.0001	0.7405	0.9829
SEM_{fall}^{w}		*0.1125*	*0.2547*	*0.1591*
SEM_{spring}^{v}		*0.1317*	*0.2315*	*0.1185*

[z,y,x] Probability value is significant at $P < 0.1$.
[w]SEM_{fall} = standard error of the mean for solute distribution in the fall season.
[v]SEM_{spring} = standard error of the mean for solute distribution in the spring season.
n = number of samples taken for observation.

Table 3.2. Treatment effects and their interactions on the vertical distribution of solutes in Site-2008.

Model Effect	d.f.	Bromide[z]	Soil ^{15}N[y]	Nitrate[x]
			n = 504	
Rate	1	0.0082	0.0002	0.1311
Landscape	2	<0.0001	0.0058	<0.0001
Rate × Landscape	2	0.4363	0.6890	0.0155
Depth	6	<0.0001	<0.0001	<0.0001
Depth × Rate	6	0.0128	0.0661	0.3912
Depth × Landscape	12	<0.0001	0.0022	<0.0001
Depth × Rate × Landscape	12	0.0556	0.7069	0.0009
Season	1	<0.0001	0.0003	0.5422
Season × Depth × Rate	7	<0.0001	0.3975	0.0743
Season × Depth × Landscape	14	<0.0001	0.3069	0.0005
Season × Depth × Rate × Landscape	14	0.0208	0.0823	0.5611
SEM_{fall}[w]		*0.1576*	*0.2324*	*0.1515*
SEM_{spring}[v]		*0.1460*	*0.2289*	*0.1167*

[z,y,x] Probability value is significant at $P < 0.1$.
[w]SEM_{fall} = standard error of the mean for solute distribution in the fall season.
[v]SEM_{spring} = standard error of the mean for solute distribution in the spring season.
n = number of samples taken for observation.

3.4.2 Vertical Distribution of Bromide in the Soil in Site-2007

In fall 2007, the amount of bromide in the top 10 cm depth in TRT90 was 20 kg ha^{-1} (10% of bromide applied), and was similar for all landscape positions (Fig. 3.1a). At the LOW slope position, amounts of bromide were between 10 and 15 kg ha^{-1} throughout the 120 cm depth. In the top 60 cm depth, the smallest amounts of solute were measured at the LOW slope compared to the UPP and MID slope positions, indicating that a significant amount of bromide had moved out of the 120 cm depth at the LOW slope position.

At the MID slope position in fall 2007, the mass of bromide in TRT90 increased with depth from the soil surface (0-5 cm depth), reaching a peak of 32 kg ha^{-1} (16% of Br^- applied), within the 20 to 40 cm depth (Fig. 3.1a). Bromide also increased with depth at the UPP slope position, but with a greater peak of 52 kg ha^{-1} (26% of Br^- applied) at 20-40 cm depth compared to the MID slope position. At 40-60 cm depth, the amount of bromide in the soil profile was similar for all slope positions down to 120 cm depth in fall 2007. The vertical distribution of bromide among landscape positions by fall 2007 in TRT135 (Fig 3.1b) were similar to those in TRT90; indicating that the extent of bromide movement was similar for both rates of N fertilizer. The pattern of bromide distribution in both N fertilizer rates showed that the maximum vertical movement was at the LOW slope position in fall 2007, compared to the UPP and MID slope positions.

By spring 2008, amounts of bromide in the soil profiles had declined at all landscape positions for both N fertilizer rates, compared to the previous fall (Fig. 3.1c-d). The mass of bromide in the region below 40 cm had declined by 50% between fall and spring, showing that at least one-half of the bromide measured

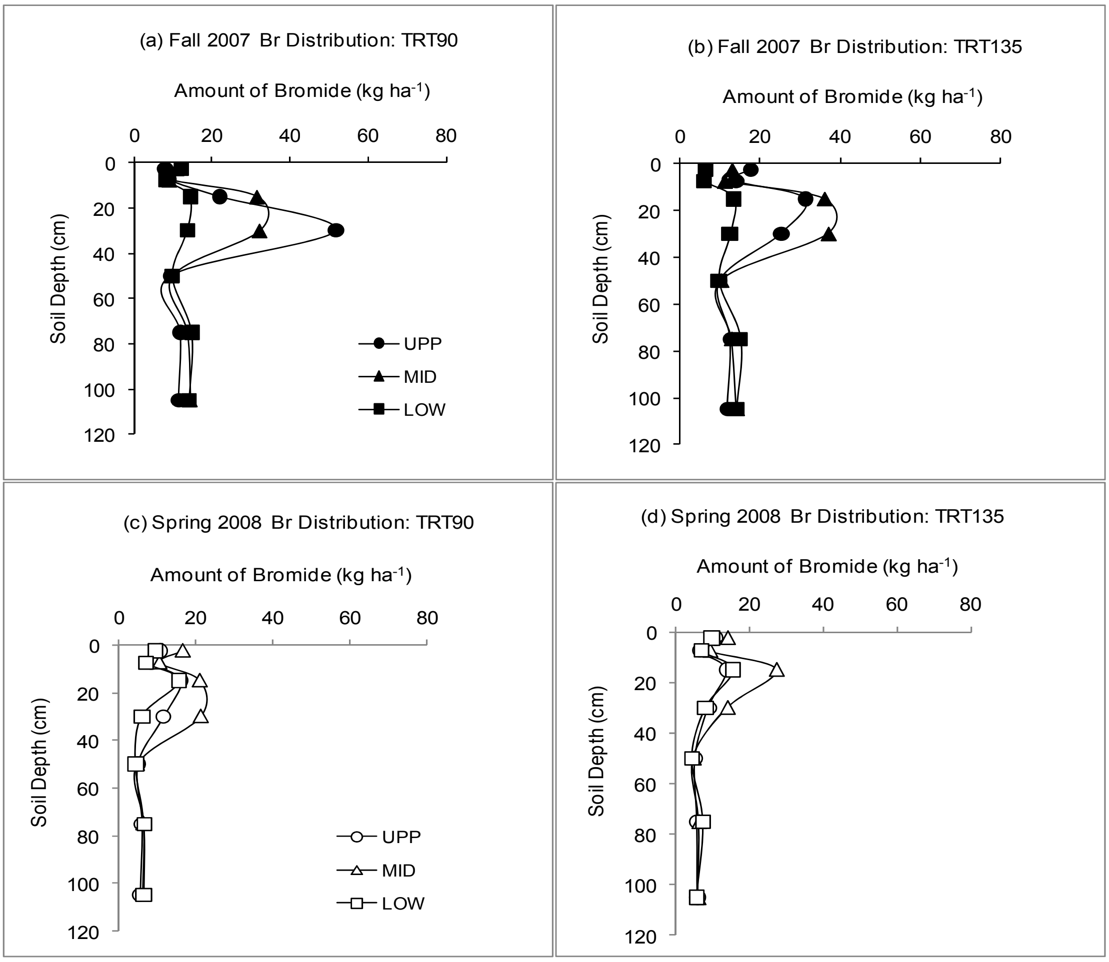

Fig. 3.1. Vertical and seasonal distribution of bromide among slope positions and between N fertilizer rates in Site-2007.

in the fall had moved out of the sampling zone by the following spring. All peaks of bromide in fall 2007 had declined significantly by spring 2008, particularly at the UPP slope position. The smallest amounts of bromide with depth at the LOW slope in the spring season confirmed the largest magnitude of solute loss at this landscape position as observed in the previous fall.

3.4.3 Vertical Distribution of Bromide in the Soil in Site-2008

In Site-2008, the centre of mass of bromide was situated at 20-40 cm depth in all slope positions for both N fertilizer treatments in fall 2008 (Fig. 3.2a-b). In TRT90, the vertical movement of bromide at the LOW slope resulted in a peak of 45 kg Br^- ha^{-1} (23% of Br^- applied) at 20-40 cm depth, while the maximum amount of bromide at the UPP and MID slope positions at this depth was 60 kg ha^{-1} (30% of Br^- applied) (Fig. 3.2a). Below 40 cm depth, the mass of bromide subsequently declined to 5 kg ha^{-1} at 90-120 cm depth in all slope positions. The vertical distribution of bromide in TRT135 (Fig. 3.2b) was similar to that in TRT90, except that the bromide peak at the UPP slope was slightly greater in TRT135 compared to TRT90. Nevertheless, the greatest movement of bromide occurred at the LOW slope compared to other landscape positions according to the bromide peaks in both fertilizer rates (Fig. 3.2a-b).

The bromide data in fall 2008 confirmed the trend in bromide distribution among landscape positions in fall 2007. The vertical distribution of bromide in fall 2008 was similar to that reported by Ottman and Pope (2000) in a clay loam soil subjected to irrigation water. The authors showed that bromide concentration

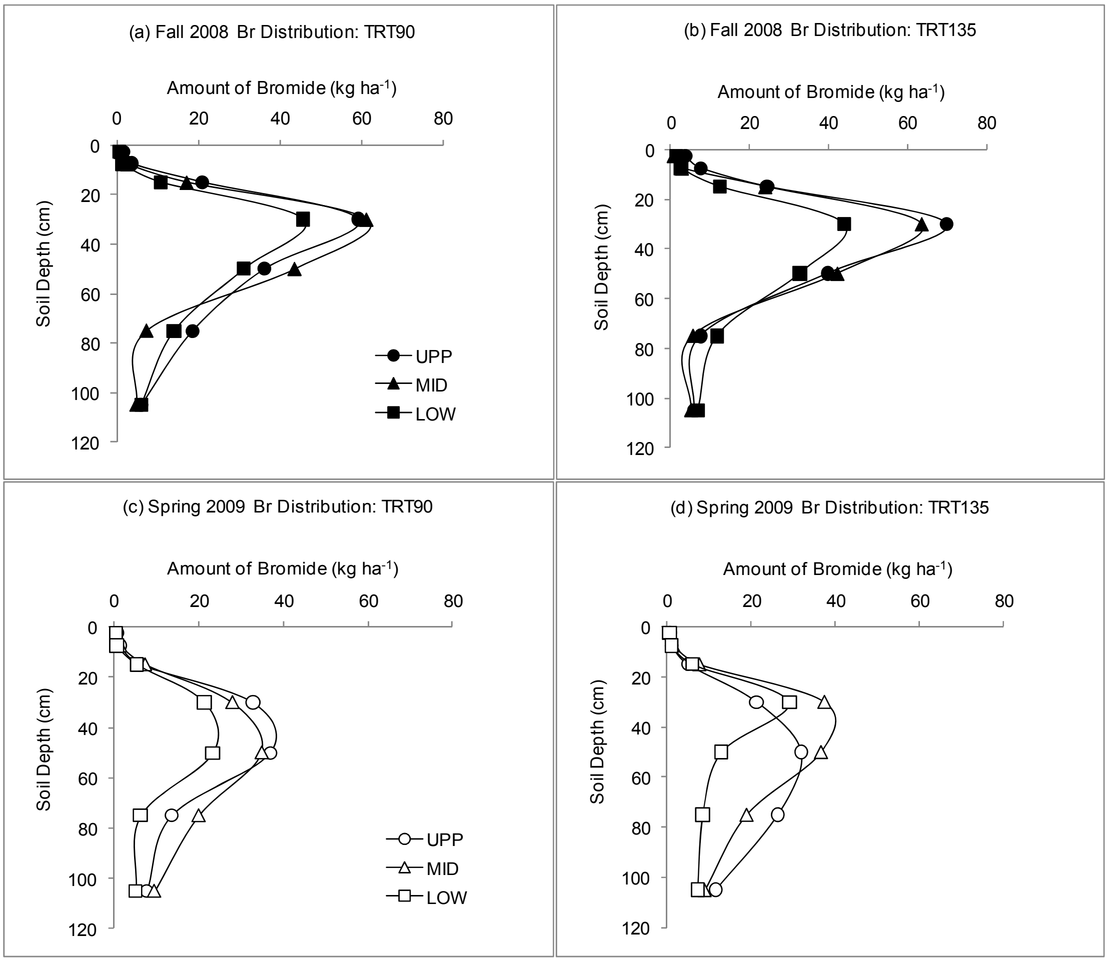

Fig. 3.2. Vertical and seasonal distribution of bromide among slope positions and between N fertilizer rates in Site-2008.

increased with depth from the soil surface, while the vertical distribution of bromide was characterized by a single peak.

In spring 2009, the bromide peaks observed at 20-40 cm depth in the fall season had been redistributed following spring snowmelt for both N fertilizer rates (Fig. 3.2c-d). This broadened the bromide peak, resulting in a smooth wide curve extending within the 20 to 60 cm depth of the soil profile, particularly in TRT90 (Fig. 3.2c). This feature was common to all three slope positions. The greatest loss of bromide, hence, the greatest movement of solute occurred at the LOW slope compared to other landscape positions.

The data for the two rates of N fertilizer in Site-2007 and Site-2008 clearly showed that the greatest loss of bromide occurred at the LOW slope and the least movement was at the MID slope position. The results also showed that following bromide application in the previous spring season, there was a significant decline in bromide by the next fall and also between fall and spring in all slope positions. Other studies have shown seasonal variability of bromide transport in hummocky landscape following fall application of bromide (Whetter et al. 2008; Thibodeau et al. 2008).

Whetter et al. (2008) showed that the greatest vertical movement of bromide was at the crest position by the next spring after the fall application of bromide, while the greatest vertical movement occurred at the depression in the subsequent fall season. The differences in bromide transport between the present study and Whetter et al. (2008) suggest that the timing of fertilizer application may affect the redistribution of solute among landscape positions. The consistently greatest downward movement of bromide at the LOW slope in the present study also suggests that precision farming techniques based on site-specific soil N tests and hydrologic conditions should be considered within the landscape. This is an important step towards the control of nitrate leaching at the vulnerable lower slope position of the landscape.

3.4.4 Vertical Distribution of ^{15}N in the Soil in Site-2007

The proportion of ^{15}N recovered in the soil profile was expressed relative to the mass of nitrogen derived from the labelled ^{15}N fertilizer (NDFF). It should be noted that the ^{15}N enrichment in the labelled fertilizer was 10 atom%.

In fall 2007, approximately 6 kg ha^{-1} of ^{15}N (7% of total ^{15}N applied) was retained at the soil surface (0-5 cm depth) at the UPP and MID slope positions in TRT90, while a smaller amount, 4.5 kg ha^{-1} (5% of total ^{15}N applied), remained at the same depth at the LOW slope (Fig. 3.3a). The mass of ^{15}N declined to 2 kg ha^{-1} in all slope positions, indicating that detectable amounts of ^{15}N moved beyond the 120 cm depth. The pattern of ^{15}N penetration in the soil profile was similar among landscape positions with TRT90 (Fig. 3.3a).

Due to the addition of more nitrogen in TRT135 (Fig. 3.3b), there were more pronounced differences in the vertical distribution of ^{15}N in the soil profile among landscape positions than in TRT90. The mass of ^{15}N remaining at the soil surface was 7.5% of the total N applied at the MID slope, followed by 5% at the UPP slope, and the smallest amount was 3.5% at the LOW slope. The vertical distribution of ^{15}N in

TRT135 clearly showed that the smallest amounts of ^{15}N were measured at the LOW slope by the end of the growing season in 2007, while the greatest

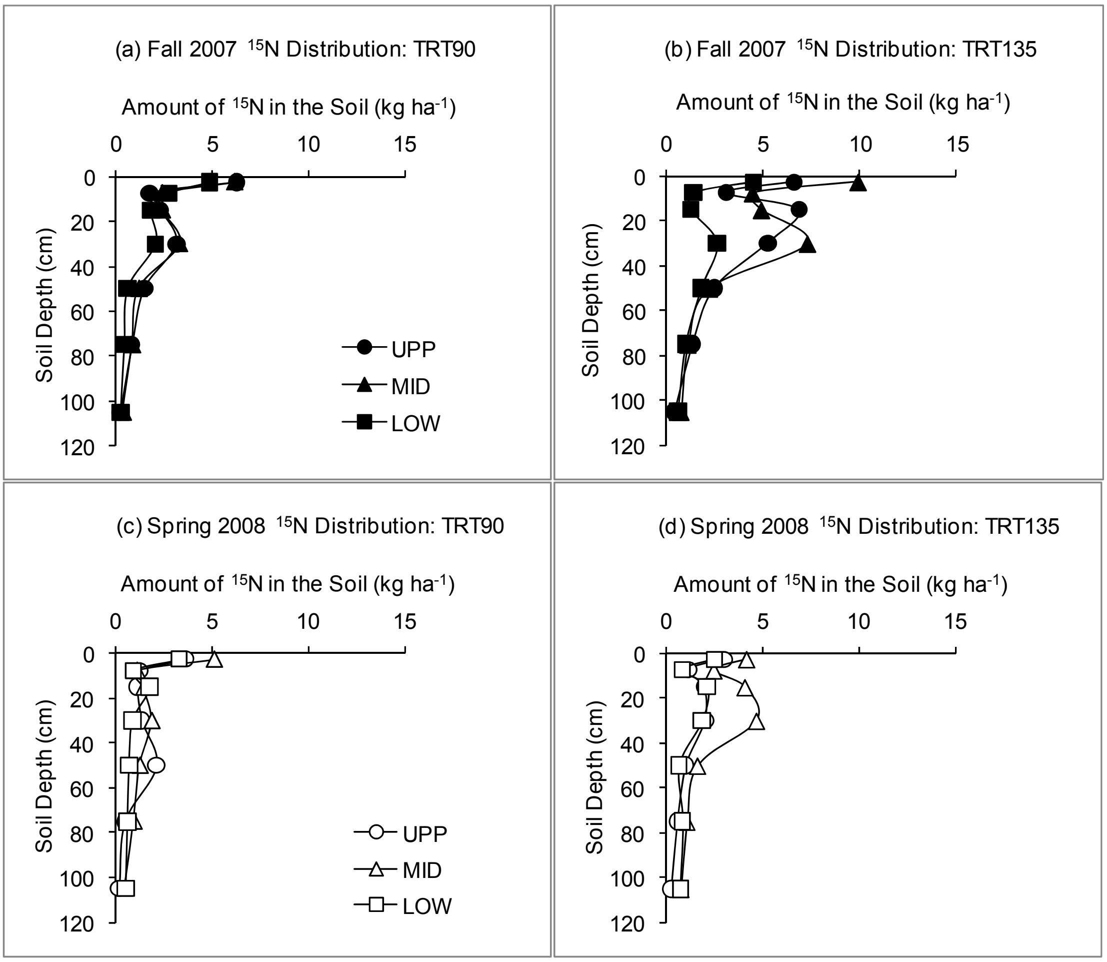

Fig. 3.3. Vertical and seasonal distribution of soil ^{15}N among slope positions and between N fertilizer rates in Site-2007.

amounts were measured at the MID slope position (Fig. 3.3b). The TRT135 data also showed that the trends in vertical distribution of ^{15}N among slope positions in fall 2007 were similar to those for bromide (Fig. 3.1b), suggesting that the transport of both tracers was similar within the landscape.

Although the largest amount of ^{15}N remained at the soil surface by spring 2008 in TRT90, the mass of ^{15}N in the soil profile had declined in all landscape positions between fall and spring for both N fertilizer rates (Fig. 3.3c-d). While the vertical distribution of ^{15}N in spring 2008 was similar for all landscape positions in TRT90 (Fig. 3.3c), the greatest amount of ^{15}N was measured at the MID slope in TRT135 compared to the UPP and LOW slope positions (Fig. 3.3d). In fall 2007, the vertical distribution of bromide and ^{15}N with TRT135 at the MID slope reflected the smallest potential for solute leaching at this portion of the landscape, compared to other slope positions.

3.4.5 Vertical Distribution of ^{15}N in the Soil in Site-2008

In the fall of 2008, the vertical distribution of ^{15}N in TRT90 was similar for the UPP and MID slope positions, with similar peaks of ^{15}N at 20-40 cm depth (Fig. 3.4a). In contrast, the amount of ^{15}N at the 20-40 cm depth at the LOW slope was smaller than at the UPP or MID slope position. A greater amount of ^{15}N was measured in TRT135 compared to TRT90. At 20-40 cm depth in TRT135 the mass of ^{15}N relative to the total ^{15}N applied ranged from 4% at the LOW slope to 10% at the MID slope (Fig. 3.4b). As such, there were marked differences in the vertical distribution of ^{15}N among all three landscape positions with TRT135, thereby confirming the trend in ^{15}N distribution among landscape positions in fall

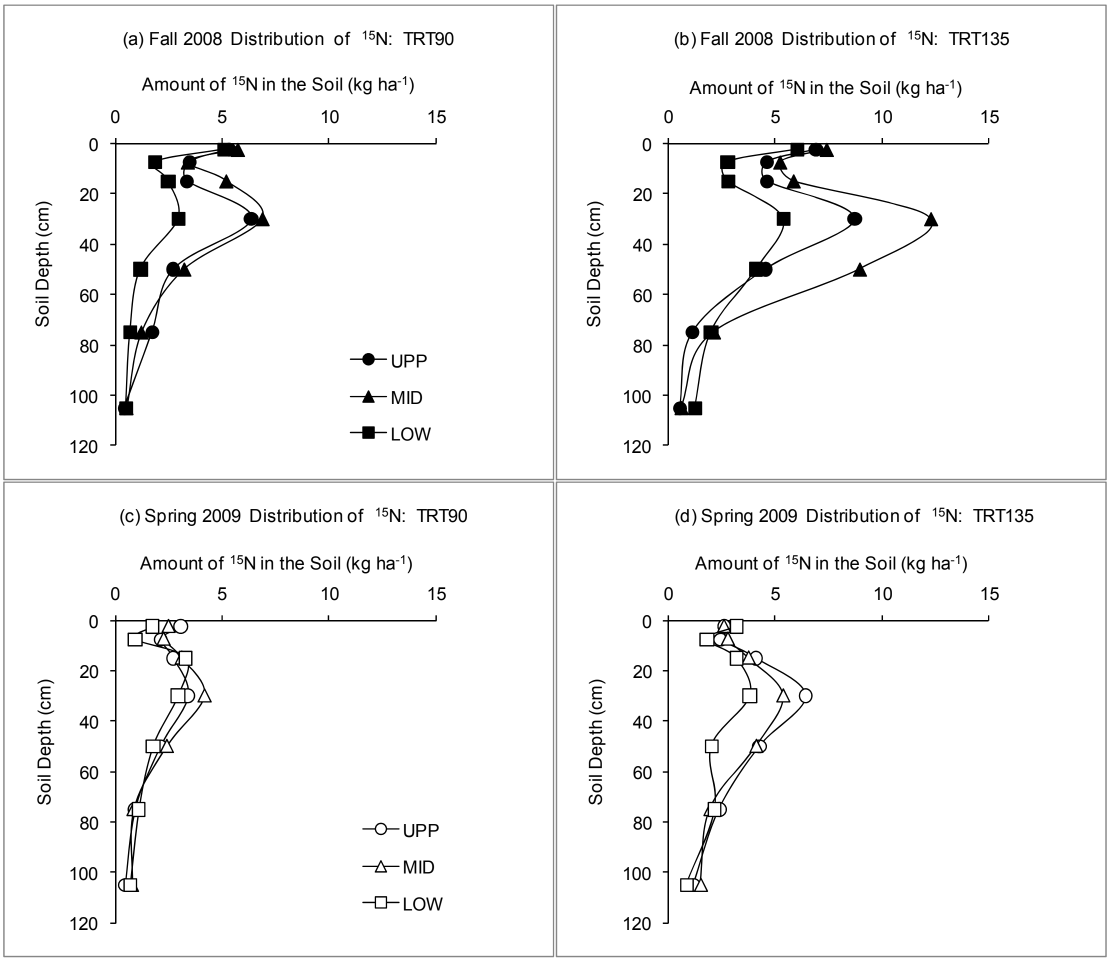

Fig. 3.4. Vertical and seasonal distribution of soil ^{15}N among slope positions and between N fertilizer rates in Site-2008.

2007. By spring 2009, ^{15}N had decreased compared to the amounts measured in fall 2008 while the differences in ^{15}N distribution among landscape positions were minimized due to solute loss following spring snowmelt (Fig. 3.4c-d).

While more ^{15}N was measured in TRT135 than in TRT90 in both site-years, the trends in ^{15}N distribution among landscape positions were similar for both rates of N fertilizer. The vertical distribution of ^{15}N in Site-2007 and Site-2008 showed that the extent of ^{15}N loss was as follows: LOW > UPP > MID. These results were consistent with the pattern of bromide transport among landscape positions, suggesting that the trend in soil water transmission within the landscape exerts similar effects on the vertical distribution of both tracers. The results further infer that in the absence of crop uptake, leaching loss had a greater influence on ^{15}N distribution as opposed to denitrification.

Following the dual application of bromide and ^{15}N in the previous spring, there was a significant loss of solute at the lower slope position, and between fall and spring in the absence of crop uptake. These findings have implications for variable rate fertilizer management within the landscape. According to the losses of bromide and ^{15}N, it is important to avoid excessive rates of N fertilizer applications to the lower slope position, particularly in the fall season, while the middle slope position can receive more N fertilizer since it was the region with the least leaching potential.

3.4.6 Vertical Distribution of Nitrate in the Soil in Site-2007 and Site-2008

The mass of NO_3-N that is presented in this section includes soil available nitrate and the NO_3-N from added (^{15}N labelled) nitrogen fertilizer. The amount of native soil nitrate was not subtracted from the control N treatment to capture the distribution of nitrate that could be attributed to N fertilizer alone. The vertical distribution of nitrate in the unfertilized plots in the fall season is shown separately for Site-2007 and Site-2008 (Fig. 3.5).

Amounts of native soil NO_3-N were similar across the landscape in Site-2007 and Site-2008 (Fig. 3.5a-b). The amount of NO_3-N in Site-2007 ranged from 5 to 15 kg ha^{-1} with a slight increase in from the soil surface to 120 cm depth. In Site-2008, a greater amount of NO_3-N was recovered in the top 40 cm depth compared to Site-2007, but the mass of NO_3-N declined slightly with depth and remained unchanged within 60 to 120 cm depth (Fig. 3.5b). This showed that Site-2008 had slightly greater native-N fertility than Site-2007.

In the 90 kg N ha^{-1} treatment (TRT90), the mass of NO_3-N increased with depth from the soil surface at all three slope positions in fall 2007 (Fig. 3.6a). The steepest increase in NO_3-N occurred at the MID slope, compared to the UPP and LOW slope positions. Below 10 cm depth, the smallest amount of NO_3-N was at the LOW slope throughout the soil profile, and the greatest amount was at the MID slope position (Fig. 3.6a). The vertical distribution of nitrate in TRT135 (Fig. 3.6b) was similar to that in TRT90, but a greater amount of NO_3-N was in the soil profile with TRT135 due to a greater addition of fertilizer N compared to TRT90.

The vertical distribution of nitrate in spring 2008 was similar to that in the previous fall, except at the 90 to 120 cm depth at the UPP and MID slope positions where the mass of NO_3-N had declined slightly, indicating the movement of nitrate-nitrogen beyond this region of the soil between fall and spring (Fig. 3.6c-d). While bromide and ^{15}N were lost from the soil profile by the

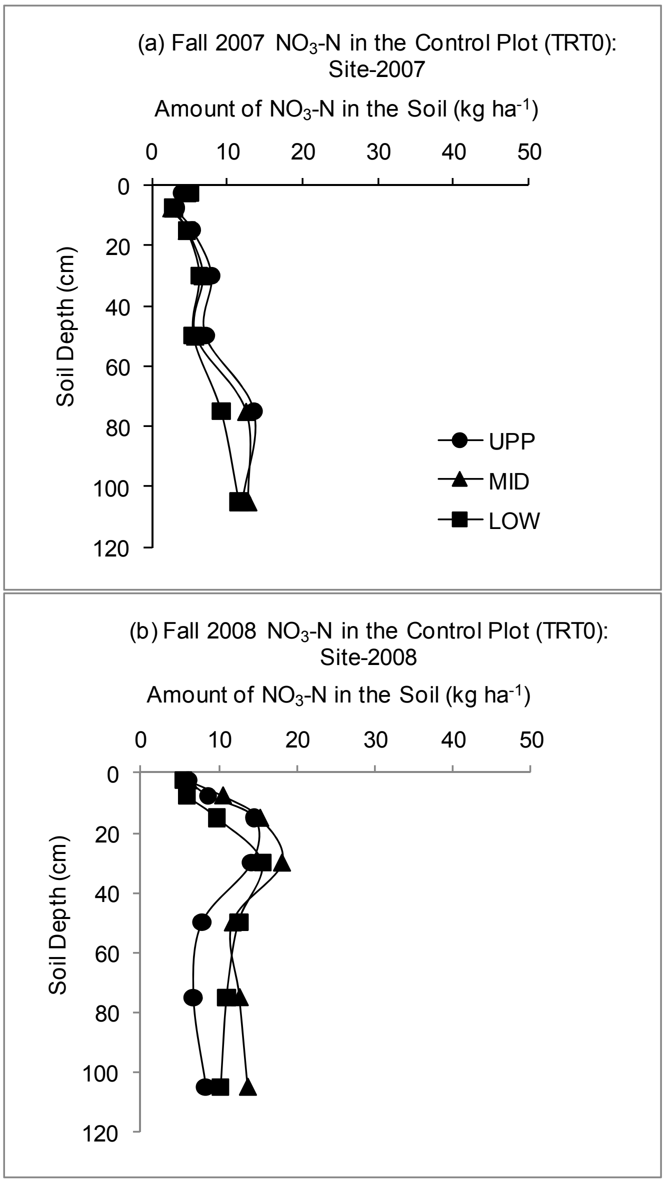

Fig. 3.5. Vertical distribution of soil nitrate in the control plot (TRT0) in the fall season in Site-2007 and Site-2008.

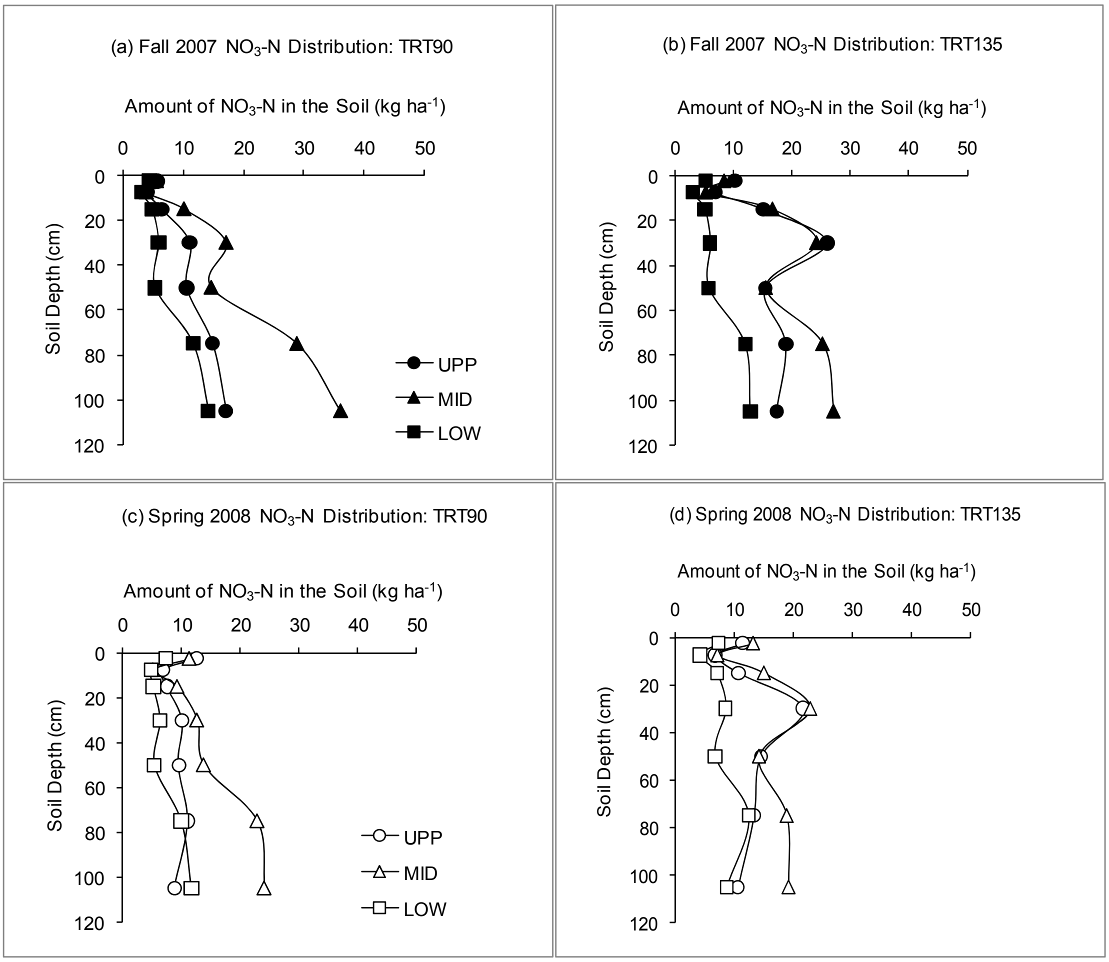

Fig. 3.6. Vertical and seasonal distribution of soil nitrate among slope positions and between N fertilizer rates in Site-2007.

spring season, the amount of NO_3-N in the soil profile did not decline significantly between fall and spring. It is possible that the nitrate-nitrogen lost at depth was replenished by the mineralization of soil organic N between fall and spring. These results point to the limitations of using soil profile nitrate-N alone as an indicator of nitrate leaching.

In Site-2008, there was a sharp increase in NO_3-N with depth in the top 40 cm in fall 2008 (Fig. 3.7a-b) while the mass of nitrate declined at the regions below 40 cm depth in all slope positions. Within the 20 to 120 cm depth of the soil profile, the greatest amounts of NO_3-N were measured at the MID slope, while the smallest amounts were at the LOW slope position. By spring 2009, the vertical distribution of NO_3-N in the soil profile was similar to the previous fall season (Fig. 3.7c-d). The similar pattern of nitrate distribution in fall 2008 and spring 2009 confirmed the effect of soil N replenishments on nitrate distribution between fall 2007 and spring 2008.

The trend in vertical distribution of NO_3-N among landscape positions in Site-2008 was consistent with that in Site-2007. The time invariant distribution of NO_3-N between fall and spring in both site-years suggests that mineralization of N from soil organic pool, and its subsequent nitrification buffered nitrate loss early in the spring season. The identical trend in vertical distribution of bromide and ^{15}N among landscape positions and significant loss of solute between fall and spring season clearly showed the importance of dual application of bromide and ^{15}N on field experimentation of nitrate leaching. Overall, the smallest amounts of bromide, ^{15}N, and NO_3-N were recovered at the LOW slope position, suggesting

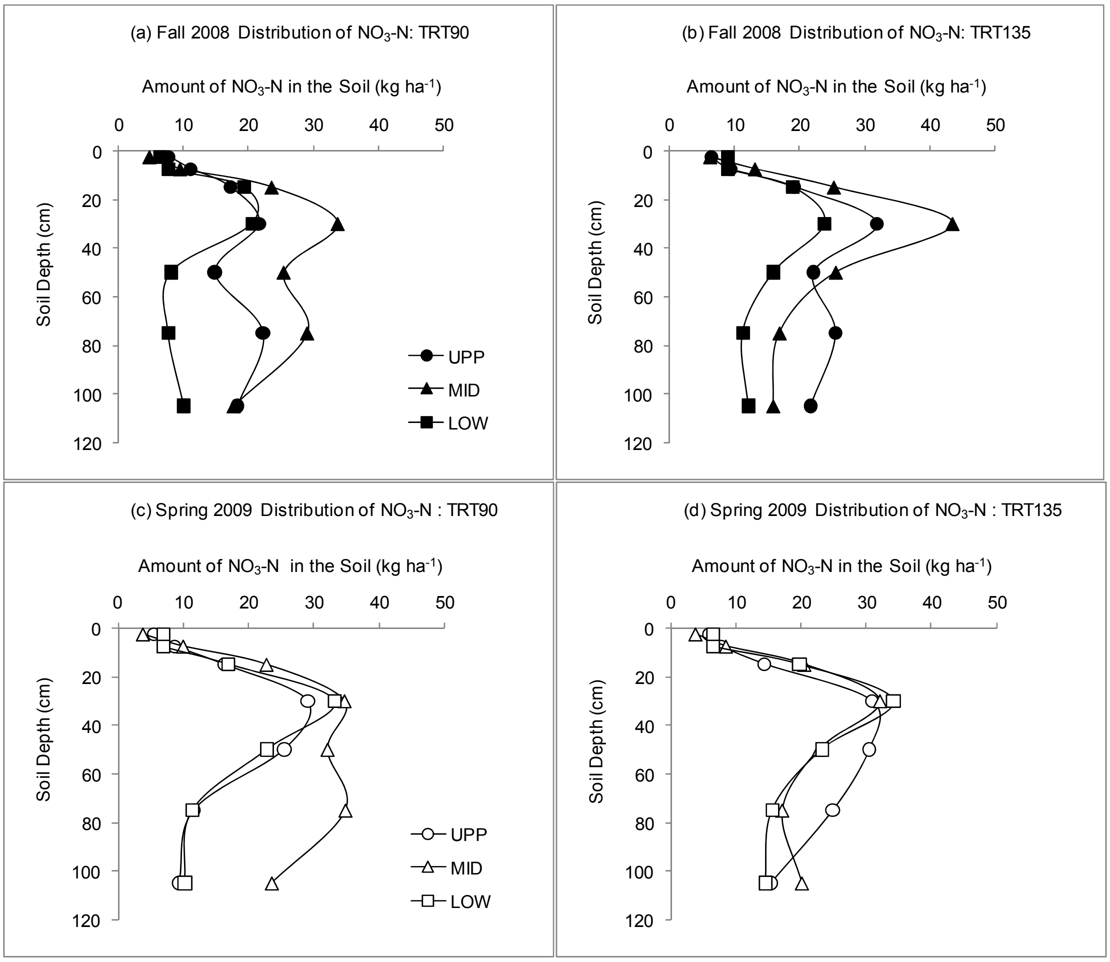

Fig. 3.7. Vertical and seasonal distribution of soil nitrate among slope positions and between N fertilizer rates in Site-2008.

that the disappearance of solute in the soil profile at the lower slope position was predominantly due to leaching losses.

3.4.7 Vertical Distribution of Solutes across the Landscape

The differences in vertical distribution of bromide among landscape positions can be attributed to factors discussed in the previous chapter. Briefly, the greatest transport of bromide at the LOW slope position was attributed to the greatest magnitude of saturated hydraulic conductivity and depth of A horizon at this slope position, thereby enhancing soil water transmission and solute movement compared to other landscape positions. Accumulation of water dissipated from upslope regions is also regarded as one of the factors promoting bromide movement at the LOW slope position (Afyuni et al. 1994; Rockstrom et al. 1999; Olson and Cassel 1999).

Since the movement of nitrate-nitrogen in the soil depends on soil water flow, the differences in the vertical distribution of ^{15}N among landscape positions can be attributed partly to the same factors governing bromide movement at this site. However, in addition to lateral and vertical movement, it is expected that ^{15}N distribution in the soil profile would be influenced by crop uptake and biological transformations (Kessavalou et al. 1996; Ottman et al. 2000; Ottman and Pope 2000).

For example, under a poor drainage condition and soluble organic carbon (Fiez et al. 1995; Malhi et al. 2004), the lower slope positions may have high soil water content, thereby promoting denitrification as opposed to leaching losses of NO_3-^{15}N (Farrell et al. 1996). This is unlike the more conservative bromide whose transport depends entirely on soil water flow. Since denitrification was not measured in the present study, the vertical distribution and recovery of ^{15}N in the soil profile was described in view of the crop uptake acting as a major sink for NO_3-^{15}N at the expense of leaching losses. Therefore, crop uptake of ^{15}N within the landscape is discussed in the subsequent section to complement the vertical distribution of ^{15}N in the soil profile.

The results also showed that accumulation of nitrate within the soil profile cannot be used solely to determine the extent of nitrate leaching amongst imposed treatments. According to the results in the fall season in Site-2007 and Site-2008, the greatest amount of nitrate was measured at the MID slope position especially at depth. This was followed by the UPP slope position and the smallest amount was measured at the LOW slope position.

If the extent of leaching amongst slope positions is based on the conventional soil profile sampling and analysis for nitrate alone, as it has been done in many studies in the literature, it will lead to the erroneous conclusion that the MID slope position experienced the greatest nitrate leaching, with the least leaching at the LOW slope position. However, this will contrast the results of bromide and ^{15}N (solutes that could not be replenished within the soil profile) which showed that the smallest amount of solute at the LOW slope position was largely due to significant leaching out of the soil profile. As such, the greater the amount of solute remaining in the soil profile, the smaller was the extent of solute leaching. By using bromide and ^{15}N tracers, it was possible to reach the correct conclusion regarding the impact of landscape position on nitrate leaching.

It is also possible to infer erroneous conclusions regarding the fate and temporal distribution of nitrate as was shown in the study. For example, the vertical distribution of bromide and ^{15}N indicated that between fall and spring in both site-years, there was a significant loss of solute from the soil profile. Conversely, there was a negligible change in the amount and vertical distribution of nitrate between fall and spring. This was due to the fact that the nitrate that was lost at depth by leaching between fall and spring was replenished by soil N mineralization and nitrification at the top of the soil profile.

Again, if the differences between amounts of nitrate from fall to spring were used to indicate leaching, it would lead to the erroneous conclusion that there was no leaching loss of nitrate between fall and spring in both site-years. This would have contradicted the results obtained from the vertical distribution of bromide and ^{15}N, both of which were not generated at the soil surface. Therefore, the amount of nitrate in the soil profile or its relative change in time cannot be used solely to measure nitrate leaching. Experimental findings on spatial distribution of nitrate or its temporal change as indices of nitrate leaching should be interpreted with caution. The information generated in this study is a significant contribution to the knowledge of fate and transport of nitrate in agricultural landscapes.

3.4.8 The Centre of Mass of Solute Leaching

The vertical depth to the centre of mass of solute, denoted as Q50, is defined as the depth to which 50% of the solute added had leached (Olson and Cassel 1999). This parameter is used for estimating the leaching depths of solute in the soil profile. The Q50 assumes that solute transport occurs strictly by matrix flow, and does not consider preferential flow.

The Q50 parameter was estimated for bromide, ^{15}N and NO_3-N based on the vertical distribution of these solutes in the fall season. Effects of N fertilizer rate, landscape and solute on the Q50 parameter were examined in Site-2007 and Site-2008 (Table 3.3). In all cases, there was no statistical difference in Q50 among landscape positions and between N rates. However, the Q50 was significantly different (P < 0.0001) among the solutes in both site-years.

The mean depths to the centre of mass of bromide in Site-2007 and Site-2008 were 25 and 30 cm, respectively (Table 3.3). The Q50 for bromide suggests that the depth of bromide penetration in Site-2008 was greater than in Site-2007. This is in contrast to the measured vertical distribution of bromide presented earlier, which showed that the downward movement of bromide in Site-2007 was greater than in Site-2008.

The discrepancy between the Q50 estimates and the measured vertical distribution of bromide for both site-years may be attributed to the assumption of bromide transport by matrix flow in the Q50 parameter. This is unlike the vertical distribution of bromide which is a result of both matrix and macropore flow. It should also be noted that the Q50 value is directly related to the amount of solute present in the soil profile, as the parameter assumes that solute accumulation with depth is an indicator of transport. Since there was a greater accumulation of bromide in the soil profile in Site-2008, a larger value of Q50 was estimated for bromide transport in Site-2008 compared to Site-2007.

Table 3.3. Depth to the centre of mass (Q50) for the vertical distribution of solutes in the fall season.

Landscape	N Rate	Br Q50	^{15}N Q50	NO_3-N Q50
	kg ha^{-1}	—— cm ——		
Site-2007		*n = 6*	*n = 3*	*n = 6*
UPP	90	25.2	9.42	37.6
	135	20.1	17.8	38.6
MID	90	24.1	9.41	43.7
	135	22.8	12.7	35.4
LOW	90	26.8	7.80	40.3
	135	30.5	12.9	40.5
Site-2008				
UPP	90	29.7	15.4	34.6
	135	26.9	15.6	32.7
MID	90	29.3	15.5	31.6
	135	28.4	18.7	29.4
LOW	90	31.6	11.1	26.1
	135	31.5	17.5	25.2
Rate means		*Site-2007*^{z}	*Site-2008*^{y}	
90		24.9	24.7	
135		25.7	25.4	
Landscape means				
UPP		24.8	25.7	
MID		24.7	25.7	
LOW		26.5	23.8	
Solute means				
Br Q50		24.9*b*	29.6*a*	
^{15}N Q50		11.7*c*	15.6*b*	
NO_3-N Q50		39.3*a*	29.9*a*	
Model effect	*d.f.*	*Site-2007*^{x}	*Site-2008*^{w}	
Rate	1	0.7836	0.5352	
Landscape	2	0.9127	0.5738	
Solute	2	<0.0001	<0.0001	
Rate × Landsc	2	0.6691	0.6355	
Solute × Rate	2	0.3606	0.3126	
Solute × Landsc	4	0.7546	0.5605	
Solute × Rate × Landsc	4	0.8652	0.6937	
SEM		*5.6591*	*2.7572*	

a – c: means with different letter(s) within the column are significantly different at $P < 0.1$ according to Tukey-Kramer test.

^{z,y} Treatment group means; ^{x,w} Probability value is significant at $P < 0.1$.

SEM = standard error of the mean.

n = number of samples taken for observation.

The Q50 data for NO_3-N resulted in unrealistically large or negative values in some cases, after accounting for the native soil nitrate. Therefore, the Q50 for NO_3-N was based on uncorrected mass of nitrate which includes the available soil nitrate, plus recently mineralized N, plus the added N. This may explain why the NO_3-N Q50 was greater than bromide Q50.

Nevertheless, according to the bromide and ^{15}N data in both site-years, the Q50 for bromide was greater than for ^{15}N, which represents added fertilizer N only. The Q50 data showed that bromide overestimated the downward transport of added nitrate as Q50 for ^{15}N was one-half of that for bromide in both years. This quantitatively confirmed that bromide transport exaggerates nitrate leaching potential, as reported in past studies conducted under irrigation systems (Kessavalou et al. 1996; Ottman et al. 2000). The smaller transport of ^{15}N compared to bromide was probably due to crop uptake of ^{15}N during the growing season.

3.4.9 Treatment Effects on Crop Yield and Solute Uptake

Effects of landscape and N fertilizer rate on crop yield and solute uptake are summarized in Tables 3.4-3.8. The plant data for TRT0 were included to illustrate the differences in yield and solute uptake between the fertilized and unfertilized plots. It should be noted that effects of landscape and N fertilizer rate on crop yield and solute uptake are significant at $P < 0.1$.

Table 3.4. Effects of N fertilizer rate and landscape position on dry matter yields of canola in Site-2007 and winter wheat in Site-2008.

Rate	Landscape	Site-2007	Site-2008	
		Canola biomass	Wheat grain	Wheat straw
		kg ha^{-1}		
		n = 3		
TRT0	UPP	4299*b*	3606	5374*d*
	MID	6461*a*	4168	7968*c*
	LOW	6406*a*	6162	10480*ab*
TRT90	UPP	9369*a*	5065	8834*bc*
	MID	8908*a*	6010	11116*a*
	LOW	5283a	6969	11707*a*
TRT135	UPP	7585*a*	5894	9706*abc*
	MID	6646*a*	5807	9639*abc*
	LOW	5343*a*	6638	11585*ab*
Rate means				
TRT0		5625	4524*b*	7656
TRT90		7607	5964*ab*	10476
TRT135		6458	6102*a*	10272
Landsc means				
UPP		6731	4757*b*	7723
MID		7259	5259*b*	9487
LOW		5655	6581*a*	11244
Model effect	*d.f.*	*P* value[z]		
Rate	2	0.1236	0.0652	0.0203
Landscape	2	0.2125	0.0010	<0.0001
Rate × Landsc	4	0.0938	0.1665	0.0004
SEM		*0.1707*	*0.1023*	*0.0708*

a – *b*: means with different letter(s) within the column for each treatment effect are significantly different at $P < 0.1$ according to Tukey-Kramer test.

[z] Probability value is significant at $P < 0.1$.

SEM = standard error of the mean.

n = number of samples taken for observation.

Table 3.5. Effects of N fertilizer rate and landscape position on bromide uptake by canola in Site-2007 and winter wheat in Site-2008.

Rate	Landscape	Site-2007	Site-2008	
		Canola biomass	Wheat grain	Wheat straw
		kg ha^{-1}		
		n = 3		
TRT0	UPP	0.914*b*	0.416	1.11
	MID	1.99*ab*	0.557	0.952
	LOW	5.90*a*	0.616	1.89
TRT90	UPP	6.64*a*	0.861	1.34
	MID	7.51*a*	1.13	1.61
	LOW	3.58*ab*	0.962	1.82
TRT135	UPP	4.06*ab*	0.737	1.63
	MID	4.72*ab*	1.15	1.72
	LOW	3.76*ab*	1.30	2.06
Rate means				
TRT0		2.21	0.523*b*	1.26
TRT90		5.63	0.978*a*	1.58
TRT135		4.16	1.03*a*	1.79
Landsc means				
UPP		2.91	0.641	1.34
MID		4.13	0.897	1.38
LOW		4.29	0.917	1.93
Model effect	*d.f.*	*P* value[z]		
Rate	2	0.0432	0.0306	0.1892
Landscape	2	0.4171	0.3257	0.1265
Rate × Landsc	4	0.0201	0.9635	0.7508
SEM		*0.3281*	*0.3178*	*0.2281*

a – *b*: means with different letter(s) within the column for each treatment effect are significantly different at $P < 0.1$ according to Tukey-Kramer test.

[z] Probability value is significant at $P < 0.1$.

SEM = standard error of the mean.

n = number of samples taken for observation.

Table 3.6. Effects of N fertilizer rate and landscape position on %NDFF in canola in Site-2007 and winter wheat in Site-2008.

Rate	Landscape	Site-2007	Site-2008	
		Canola biomass	Wheat grain	Wheat straw
		%NDFF		
		n = 3		
TRT90	UPP	33.4	23.5	25.8
	MID	24.4	18.9	20.9
	LOW	27.7	18.4	22.3
TRT135	UPP	22.0	26.6	33.6
	MID	24.3	28.9	33.3
	LOW	28.7	30.9	31.3
Rate means				
TRT90		28.3	20.1*b*	22.9*b*
TRT135		24.8	28.7*a*	32.7*a*
Landsc means				
UPP		27.1	25.0	29.5
MID		24.3	23.3	26.4
LOW		28.2	23.9	26.4
Model effect	*d.f.*	*P* value[z]		
Rate	1	0.2791	0.0217	0.0009
Landscape	2	0.5645	0.7706	0.4514
Rate × Landsc	2	0.2422	0.1698	0.4395
SEM		*0.1391*	*0.1053*	*0.0834*

a – b: means with different letter(s) within the column for each treatment effect are significantly different at $P < 0.1$ according to Tukey-Kramer test.

[z] Probability value is significant at $P < 0.1$.

SEM = standard error of the mean.

n = number of samples taken for observation.

Table 3.7. Effects of N fertilizer rate and landscape position on yield-dependent NDFF in canola in Site-2007 and winter wheat in Site-2008.

Rate	Landscape	Site-2007	Site-2008	
		Canola biomass	Wheat grain	Wheat straw
		kg ha^{-1}		
		n = 3		
TRT90	UPP	52.7	42.2	23.1
	MID	50.6	41.8	29.7
	LOW	26.4	45.3	30.2
TRT135	UPP	23.6	53.1	37.3
	MID	28.9	56.3	38.9
	LOW	31.6	61.6	37.3
Rate means				
TRT90		41.3	43.1*b*	27.5*b*
TRT135		27.9	56.9*a*	37.9*a*
Landsc means				
UPP		35.3	47.4	29.4
MID		38.3	48.5	33.9
LOW		28.8	52.8	33.6
Model effect	*d.f.*	*P* value[z]		
Rate	1	0.1804	0.0937	0.0004
Landscape	2	0.7392	0.7572	0.2246
Rate × Landsc	2	0.3399	0.9616	0.1208
SEM		*0.3427*	*0.1526*	*0.0706*

a – *b*: means with different letter(s) within the column for each treatment effect are significantly different at $P < 0.1$ according to Tukey-Kramer test.

[z] Probability value is significant at $P < 0.1$.

SEM = standard error of the mean.

n = number of samples taken for observation.

Table 3.8. Effects of N fertilizer rate and landscape position on total N accumulation in canola in Site-2007 and winter wheat in Site-2008.

Rate	Landscape	Site-2007	Site-2008	
		Canola biomass	Wheat grain	Wheat straw
		kg ha^{-1}		
		n = 3		
TRT0	UPP	79.4	97.7*b*	49.0*b*
	MID	85.6	129*ab*	85.4*a*
	LOW	82.1	177*a*	98.8*a*
TRT90	UPP	158	147*a*	74.5*ab*
	MID	208	173*a*	113*a*
	LOW	95.2	192*a*	109*a*
TRT135	UPP	107	167*a*	96.1*a*
	MID	120	165*a*	101*a*
	LOW	110	171*a*	102*a*
Rate means				
TRT0		82.3*b*	131	74.5
TRT90		146*a*	170	97.5
TRT135		112*ab*	168	99.8
Landsc means				
UPP		110	134	70.5
MID		129	155	99.2
LOW		95.2	179	104
Model effect	*d.f.*	*P* value^{z}		
Rate	2	0.0338	0.0020	0.0842
Landscape	2	0.3443	0.0697	0.0004
Rate × Landsc	4	0.5563	0.0416	0.0372
SEM		*0.2450*	*0.0947*	*0.1111*

a – *b*: means with different letter(s) within the column for each treatment effect are significantly different at *P* < 0.1 according to Tukey-Kramer test.

^{z} Probability value is significant at *P* < 0.1.

SEM = standard error of the mean.

n = number of samples taken for observation.

There was a significant interaction between landscape positions and N fertilizer rates (P = 0.0938) on canola biomass in Site-2007 (Table 3.4). Canola biomass significantly increased from the UPP slope to the LOW slope position in the unfertilized treatment (TRT0), while the yield was similar among landscape positions in the fertilized treatments (TRT90 and TRT135). The results showed N response was greater at the upslope than at the lower landscape position.

In Site-2008, fertilizer and landscape significantly affected the winter wheat grain yield. The grain yield ranged from 4,757 kg ha^{-1} at the UPP slope to 6,581 kg ha^{-1} at the LOW slope, while the range among N fertilizer treatments was 4,524 kg ha^{-1} with TRT0 and 6,102 kg ha^{-1} with TRT135. The grain yield from adjacent fields harvested using a combine ranged from 1,470 to 5,520 kg ha^{-1}. There was a significant interaction of N fertilizer rate by landscape (P = 0.0004) on the wheat straw yield, as the straw yield increased significantly from the UPP to the LOW slope position in TRT0 and TRT90, but the yield was similar across landscape positions with the highest rate of N (TRT135).

In the unfertilized treatment, the data showed that differences in residual fertility among landscape positions can significantly influence yield response within the landscape. In contrast, N fertilization, particularly at high rates can reduce differences in fertility among landscape positions. The agronomic and environmental implications of landscape by N fertility interactions on crop yield have been discussed in the previous chapter.

There was a significant interaction of N fertilizer rate by landscape (P = 0.0201) on bromide uptake by canola in Site-2007 (Table 3.5). In the unfertilized treatment, the smallest uptake was at the UPP slope while the greatest was at the LOW slope. In the fertilized treatments, however, bromide uptake by canola was similar among landscape positions.

The amount of bromide in the wheat grain was affected only by N fertilizer rate (P = 0.0306), as bromide uptake was 50% greater with N fertilization compared to the unfertilized treatment (Table 3.5). Bromide uptake in the wheat straw was similar among N fertilizer rates and landscape positions. The data showed that the straw tissue was the major sink for bromide in winter wheat. The bromide retained in the wheat straw was twice the amount in the grain, similar to that reported by Ottman et al. (2000), while the amount of bromide recovered in the winter wheat (straw plus grain) was smaller than that in canola.

The plant tissue nitrogen derived from the labelled ^{15}N fertilizer (NDFF) was expressed in two ways (Hauck and Bremner 1976). The first is a yield-independent form of NDFF, which is quantified as the percent nitrogen derived from fertilizer (%NDFF) relative to the proportion of ^{15}N enrichment in the labelled N fertilizer. The second type is yield-dependent, expressed in kg ha^{-1}, i.e. ^{15}N recovery is estimated relative to the crop yield and total N in the plant tissue (Malhi et al. 2004, 2009).

The NDFF in TRT0 was negligible (data not shown). Therefore, the statistical effects of landscape and N rate on NDFF were presented for TRT90 and TRT135 only (Tables 3.6-3.7). Due to large variability within the landscape, there was no statistical difference in %NDFF and the yield-dependent NDFF of canola between TRT90 and TRT135 and among landscape positions in Site-2007 (Tables 3.6-3.7). In Site-2008, the %NDFF and NDFF of wheat grain and straw with TRT135 were significantly greater than TRT90 (*P* < 0.1), while %NDFF and NDFF values were similar among landscape positions (Tables 3.6-3.7).

The addition of N fertilizer enhanced total N uptake by canola compared to the unfertilized treatment, but there was no significant difference in uptake between the two rates of N fertilizer application (Table 3.8). There were no differences in total N uptake by canola among landscape positions in Site-2007. Unlike the results obtained for canola, there were statistically significant differences in total N uptake by winter wheat grain among N fertilizer rates (P = 0.0020) in Site-2008, while the total N uptake generally increased downslope (P = 0.0697). However, the significant interaction between landscape position and N fertilizer rate resulted in greater response of N uptake by wheat grain to fertilizer at the UPP slope than at other slope positions. A similar interaction was observed for N accumulation in winter wheat straw in Site-2008.

3.4.10 Mass Recovery of Bromide and ^{15}N in Soil and Plant Tissue

The total amount of solute in the soil was obtained by adding the ionic mass over the entire soil profile. The amounts of solute in the plant tissue were also summed together with the soil fraction to derive the total mass of ions recovered from both soil and plant. The mass balance of bromide and nitrogen derived from the labelled ^{15}N fertilizer (NDFF) were presented for TRT90 and TRT135 only (Tables 3.9-3.10). Note that the geometric means of the solute mass balance were grouped according to the time of the year they were sampled, while effects of landscape and rates of N fertilizer on solute recovery are

Table 3.9. Geometric means of mass recovery for bromide and ^{15}N in Site-2007.

Landscape	Rate	Br^-				^{15}N			
		Soil Br^- ($kg\ ha^{-1}$)	%Δ Soil Br^-	Plant Br^- ($kg\ ha^{-1}$)	%Rec Br^{-z}	Soil NDFF ($kg\ ha^{-1}$)	%Δ Soil NDFF	Plant NDFF ($kg\ ha^{-1}$)	%Rec $NDFF^y$
Fall 2007									
UPP	90	124	−	6.28	64.9	16.2	−	52.7	76.5
	135	123	−	4.06	63.3	24.4	−	23.6	35.6
MID	90	122	−	8.02	65.1	17.0	−	50.6	75.1
	135	135	−	4.63	69.9	29.8	−	28.9	43.5
LOW	90	86.5	−	3.54	45.0	12.9	−	26.4	43.6
	135	77.3	−	3.83	40.6	13.9	−	31.6	33.7
Spring 2008									
UPP	90	63.8	48.3	−	35.1	10.2	37.3	−	69.8
	135	56.9	53.6	−	30.5	9.76	60.1	−	24.7
MID	90	87.0	28.8	−	47.5	12.2	28.0	−	69.8
	135	81.9	39.4	−	43.2	17.4	41.6	−	34.3
LOW	90	56.2	35.0	−	29.9	8.89	31.1	−	39.2
	135	57.4	25.8	−	30.6	9.55	31.3	−	30.5
Rate means									
TRT90		86.1		5.63		12.6		41.3	
TRT135		83.6		4.16		15.8		27.9	
Landscape means									
UPP		86.1		5.19		14.1		35.3	
MID		104		5.95		18.1		38.3	
LOW		68.2		3.67		10.9		28.8	
Season means									
Fall		111		−		19.0		−	
spring		67.2		−		11.3		−	
Seas×Landsc means									
Fall-UPP		123*a*		−		19.9*ab*		−	
Fall-MID		128*a*		−		22.5*a*		−	
Fall-LOW		81.8*b*		−		13.1*cd*		−	
Spring-UPP		60.3*c*		−		10.0*de*		−	
Spring-MID		84.4*b*		−		14.6*bc*		−	
Spring-LOW		56.8*c*		−		9.21*e*		−	

Table 3.9. Cont'd.

Landscape	Rate	Br^-		^{15}N	
Model effect	*d.f.*	*Soil Br^-*	*Plant Br^-*	*Soil NDFF*	*Plant NDFF*
Rate	1	0.5976	0.2812	0.2608	0.1804
Landscape	2	0.0003	0.2273	0.0139	0.7392
Rate × Land	2	0.5695	0.5355	0.1407	0.3399
Season	1	<0.0001	–	<0.0001	–
Seas × Rate	1	0.4648	–	0.1063	–
Seas × Landsc	2	0.0011	–	0.0797	–
Seas × Rate × Landsc	2	0.1513	–	0.2539	–
SEM		*0.0601*	*0.2751*	*0.1609*	*0.3427*

a – c: means with different letter(s) within the column are significantly different at *P* < 0.1 according to Tukey-Kramer test.

%Δ denotes % loss of bromide and ^{15}N in the soil profile between fall and spring season;

$$\%\Delta = \frac{\text{fall - spring}}{\text{fall}} \times 100$$

[z,y] %Rec denotes % recovery = $\frac{\text{Total amount of solute from soil + plant}}{\text{Amount of solute applied (kg ha}^{-1}\text{)}} \times 100$

Note that solute uptake was accounted for in % Recovery for spring 2008.

SEM = standard error of the mean.

Table 3.10. Geometric means of mass recovery for bromide and ^{15}N in Site-2008.

Landscape	Rate	Br^-				^{15}N			
		Soil Br^- (kg ha^{-1})	%Δ Soil Br^-	Plant Br^- (kg ha^{-1})	%Rec Br^- [z]	Soil NDFF (kg ha^{-1})	%Δ Soil NDFF	Plant NDFF (kg ha^{-1})	%Rec NDFF[y]
Fall 2008									
UPP	90	142	–	2.27	72.2	23.4	–	54.8	86.9
	135	153	–	2.45	78.0	31.1	–	78.4	81.1
MID	90	136	–	2.76	69.6	26.3	–	57.3	92.8
	135	143	–	2.95	72.8	42.5	–	83.9	93.7
LOW	90	113	–	2.86	57.9	14.8	–	60.8	84.0
	135	114	–	3.46	58.7	24.3	–	84.9	81.0
Spring 2009									
UPP	90	98.2	30.8	–	50.3	14.8	36.8	–	77.3
	135	96.9	36.8	–	49.7	23.4	24.8	–	75.4
MID	90	99.8	26.8	–	51.3	16.2	38.3	–	81.6
	135	110	23.1	–	56.4	22.0	48.2	–	78.5
LOW	90	64.5	42.9	–	33.7	12.5	15.8	–	81.4
	135	66.8	41.3	–	35.2	17.0	30.2	–	75.5
Rate means									
TRT90		106		2.63		18.0*b*		57.6*b*	
TRT135		110		2.95		26.7*a*		82.4*a*	
Landscape means									
UPP		122		2.36		23.2		66.6	
MID		123		2.86		26.8		70.6	
LOW		83.8		3.16		17.2		72.8	
Season means									
Fall		134		–		27.1		–	
spring		89.4		–		17.6		–	
Seas×Landsc means									
Fall-UPP		152*a*		–		27.3*b*		–	
Fall-MID		141*a*		–		34.4*a*		–	
Fall-LOW		110*b*		–		19.6*c*		–	
Spring-UPP		98.1*b*		–		19.1*cd*		–	
Spring-MID		106*b*		–		19.1*cd*		–	
Spring-LOW		63.8*c*		–		14.7*d*		–	

Table 3.10. Cont'd.

Landscape	Rate	Br^-		^{15}N	
Model effect	*d.f.*	*Soil Br^-*	*Plant Br^-*	*Soil NDFF*	*Plant NDFF*
Rate	1	0.1679	0.5179	<0.0001	0.0040
Landscape	2	<0.0001	0.3812	0.0004	0.7580
Rate × Land	2	0.6408	0.9462	0.9882	0.9696
Season	1	<0.0001	–	<0.0001	–
Seas × Rate	1	0.9718	–	0.4562	–
Seas × Landsc	2	<0.0001	–	0.0485	–
Seas × Rate × Landsc	2	0.2866	–	0.1703	–
SEM		*0.0450*	*0.2023*	*1.883*	*7.591*

a – c: means with different letter(s) within the column are significantly different at $P < 0.1$ according to Tukey-Kramer test.

%Δ denotes % loss of bromide and ^{15}N in the soil profile between fall and spring season;

$$\%\Delta = \frac{\text{fall - spring}}{\text{fall}} \times 100$$

[z,y] %Rec denotes % recovery = $\frac{\text{Total amount of solute from soil + plant}}{\text{Amount of solute applied (kg ha}^{-1}\text{)}} \times 100$

Note that solute uptake was accounted for in %Recovery for spring 2009.

SEM = standard error of the mean.

significant at $P < 0.1$. Also note that the mass of solute in the plant tissue was included in the percent recovery of bromide and ^{15}N in the spring season.

In Site-2007, the total mass of bromide in all sampled layers in the soil profile was similar for the two rates of N fertilizer (TRT90 and TRT135) (Table 3.9). Amongst landscape positions, the mass of bromide recovered in the soil was in the following order: MID > UPP > LOW ($P = 0.0003$). However, the significant interaction of season and landscape on bromide recovery in the soil ($P = 0.0011$) indicated that same amounts of bromide were measured at the UPP and MID slope positions in the fall season, while bromide recovery at the UPP slope was similar to that at the LOW slope in the spring season. Approximately 39% of the soil bromide measured in fall 2007 was lost by spring 2008. The amount of bromide recovered in canola tissue was only 2-3% of the bromide applied.

The mass of ^{15}N recovered in the soil in Site-2007 was also affected by the interaction of season and landscape ($P = 0.0797$), but there was no significant effect of rate of N fertilizer on soil ^{15}N recovery (Table 3.9). While the amount of soil ^{15}N recovered at the LOW slope in the fall season was smaller than at the UPP or MID slope position, there was no significant difference in mass of ^{15}N between the UPP and LOW slope positions in the spring. The trend in soil ^{15}N recovery among landscape positions and between sampling seasons is similar to that observed for bromide. By the spring of 2008, the soil ^{15}N had also declined by 38% of that recovered in the previous fall season. The amount of ^{15}N recovered in the canola tissue was 46% of the total N applied for TRT90 and 21% for TRT135. This is unlike the negligible bromide uptake by canola. Relative to the total mass of N applied, a greater amount of ^{15}N was recovered in the canola tissue compared to that measured in the soil.

The statistical effects of landscape position and sampling season on soil bromide recovery in Site-2008 (Table 3.10) were similar to that obtained in Site-2007, while there were no effects of N fertilizer rates on bromide recovery in Site-2008. The mass of bromide recovered in the soil at the LOW slope in Site-2008 was consistently smaller than at the UPP or MID slope position in the fall and spring seasons. The mean mass of soil bromide in fall 2008 was 67% of the bromide applied, which subsequently declined by 34% between fall 2008 and spring 2009. The mass of bromide recovered in the winter wheat was less than 1.5% of the bromide applied.

In Site-2008, a greater amount of soil ^{15}N was recovered in TRT135 compared to TRT90 ($P < 0.0001$) (Table 3.10). In the fall of 2008, the smallest mass of ^{15}N was measured in the soil profile at the LOW slope position while the greatest amount was recovered at the MID slope position. By the spring of 2009, similar amounts of soil ^{15}N were measured at all three landscape positions. Between fall 2008 and spring 2009, the mass of ^{15}N in the soil had declined by 32% relative to the amount in the fall season, a proportion that was similar to that of bromide.

The mass of ^{15}N recovered in winter wheat in Site-2008 was influenced by the rate of N fertilizer ($P = 0.0040$), as amounts of ^{15}N in the plant tissue were 64 and 61% of the total N applied in TRT90 and TRT135, respectively (Table 3.10). A greater proportion of the ^{15}N applied was retained in the wheat tissue compared to the amount of ^{15}N in the soil profile. This is similar to the partitioning of ^{15}N between the soil profile and canola tissue in Site-2007.

In the fall season in both site-years, the smallest amount of soil ^{15}N was recovered at the LOW slope compared to other landscape positions. In some cases in the spring season, the amount of soil ^{15}N measured at the LOW slope was similar to other landscape position. This may infer that differences in soil ^{15}N recovery among landscape positions were due to crop uptake. However, there was no significant effect of landscape position on ^{15}N uptake according to the plant data in both years. Perhaps, the variability in soil water transmission among landscape positions in the fall season resulted in differences in soil ^{15}N recovery. However, when there was a greater soil water flux and solute movement in the spring season following spring snowmelt, the differences in soil ^{15}N among landscape positions were eliminated. More so, the pattern of ^{15}N recovery among landscape positions was similar to that of bromide, particularly in Site-2007 (Table 3.10).

Overall, the magnitude of solute loss between fall and spring was similar for bromide and ^{15}N in each site-year. The data showed that the mass of solute measured in fall 2007 had diminished by approximately 39% for bromide and 38% for ^{15}N by spring 2008 (Table 3.9). Also in Site-2008, the magnitude of solute loss between fall 2008 and spring 2009 was equivalent to 34% of that in fall 2008 for bromide and 32% for ^{15}N (Table 3.10). These results indicate that both tracers behaved alike in the absence of crop uptake, and that significant quantity of solute was lost between fall and spring season.

A greater proportion of ^{15}N was recovered in the plant tissue compared to the amount in the soil profile, while the amount of bromide in the plant tissue was very small relative to the bromide applied. The plant data showed that ^{15}N uptake by canola was 35% of total N applied, while the uptake by winter wheat was 63%. These results confirmed that the loss of ^{15}N in the soil profile during the growing season was predominantly due to plant uptake and leaching as opposed to losses through denitrification. Between the fall and spring season, however, the identical proportional change in mass of solute for bromide and ^{15}N tracers during this period indicates that the missing mass of ^{15}N was mainly due to losses by vertical and lateral movement in the absence of crop uptake.

3.5 Summary and Conclusions

The vertical movement of nitrate in the fall and spring seasons within a hummocky landscape was estimated using the dual application of bromide and labelled ^{15}N at 90 and 135 kg ha^{-1} rates of N fertilizer application. The smallest amounts of Br^-, ^{15}N, and NO_3-N were in the soil profile at the lower slope for both rates of N fertilizer addition, while the greatest amounts were at the middle slope. The trend in solute recovery among landscape positions indicated that solute transport followed the order of: lower > upper > middle slope position.

Between fall and spring season, the mass of bromide and ^{15}N in the soil profile declined significantly, while NO_3-N distribution remained unchanged due to soil N replenishment. In the absence of crop uptake, bromide and ^{15}N behaved alike in the soil profile as the magnitude of solute loss was similar for both tracers, suggesting that the missing mass of solute between fall and spring was predominantly due to leaching and lateral movement. As such, bromide is an ideal tracer for nitrate-nitrogen in the absence of plant uptake. With plant uptake, the Q50 parameter (the vertical depth to which 50% of solute in the soil profile had leached) showed that bromide overestimated the downward transport of the nitrate added as the Q50 of ^{15}N was only one-half of that of bromide in both years. This was reflected in the crop uptake of ^{15}N by canola and winter wheat as 35 and 63% of total N applied, respectively, compared to the negligible bromide uptake.

Estimation of nitrate movement using a dual application of Br^- and ^{15}N in this study has contributed to a better understanding of nitrate leaching in the hummocky landscape, particularly at the vulnerable lower slope position. This is also important for precision farming techniques such as site-specific management and variable rate N fertilizer application within the landscape. These findings suggest that it will not be advisable to apply high rates of N fertilizer to the lower slope position, while the middle slope position can receive more N fertilizer since it is the region with the least leaching potential.

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4. NUMERICAL MODELLING OF BROMIDE REDISTRIBUTION IN A HUMMOCKY LANDSCAPE

4.1 Abstract

Numerical models are used to analyze water and solute movement in the vadose zone in an attempt to understand the fate of nutrients in agricultural soils. The objectives of this study were to simulate the downward movement (1D) of water and bromide, and two-dimensional (2D) distribution of Br^- in a hummocky landscape using HYDRUS-1D and HYDRUS-2D/3D software. The field study investigated effects of landscape and nitrogen fertilization on Br^- transport in two site-years denoted as Site-2007 and Site-2008. The soil hydraulic parameters of van Genuchten-Mualem equations (θ_r, θ_s, α, n, K_{sat}, and l) were estimated from field-measured soil properties using the pedotransfer functions in the HYDRUS program. The application of Br^- was represented as surface flux of solute in the infiltrating water to provide the field application rate of 200 kg ha^{-1}. The final time of model calculation corresponds to the duration of field experiment in each site-year. HYDRUS-1D model showed a temporal variation in Br^- distribution, but the total mass of Br^- in the soil profile was time invariant in spite of estimated cumulative outflow of 10 cm of water. HYDRUS-2D/3D reproduced the field study better than HYDRUS-1D. The 2D model simulated the decline in mass of Br^-

between fall and spring season, while the estimated mass balance of Br^- compared well with some of the field data. The 2D simulation and field experiment both showed that vertical downward movement is the main pathway of solute loss in the landscape, thereby reflecting a high risk of nitrate leaching with above normal precipitation. HYDRUS-2D/3D showed that the differences in simulated transport of Br^- between the two site-years were due to precipitation and slope steepness. However, the 2D model did not reflect effects of landscape position, crop type, snow accumulation, and N fertility on Br^- transport. The study suggests that effects of crop fertilization on water uptake and its resultant effect on solute leaching should be incorporated into numerical models.

4.2 Introduction

Field-scale studies of water and solute movement are time-consuming, expensive, and labour-intensive to conduct. To address these challenges, a variety of mechanistic models have been developed to simulate soil water flow and solute transport processes. Mechanistic models are formulated based on the mathematical theories of water flow and chemical transport in soil. The models have been applied to N cycling and transport in an attempt to synthesize and improve the contemporary knowledge on NO_3-N leaching in agricultural soils (Addiscot and Wagenet 1985). However, many of these simulation models have not been widely tested in the Canadian prairies; hence, their reliability under a variety of Canadian field conditions is not well-known (Akinremi et al. 2005).

Akinremi et al. (2005) modified the LEACHMN model (Hutson and Wagenet 1993) to simulate the vertical movement of water and solute using data obtained from a field lysimeter study in Saskatchewan, Canada. The authors were able to validate the model for the Canadian prairies. Their preliminary evaluation of LEACHMN showed that the retentivity and conductivity functions used in the model were not appropriate for prairie soils. The program was subsequently modified by incorporating the van Genuchten retentivity function, after which LEACHMN was able to reproduce changes in water and chloride concentration in field core lysimeters.

Nevertheless, there is little information on numerical modelling of solute transport at the landscape-scale in the Canadian prairies. Also, one-dimensional (1D) models are inadequate to predict soil water flow in a field with sloping terrain, due to partitioning of drainage flux into vertical and lateral components of the landscape (Rassam and Littleboy 2003). Therefore, it is important to investigate two-dimensional (2D) distribution of solute in agricultural landscape using a numerical model with the capacity to simulate the process. However, numerical models with 2D capacity are very few due to the complex mathematics of the governing equations of water flow and chemical transport.

HYDRUS-2D program is a Windows-based software package for simulating water flow and solute transport in variably saturated porous media (Simunek et al. 1999, 2006). The model has the capacity to analyze two-dimensional horizontal plane, two-dimensional vertical plane, or two-dimensional axisymmetrical vertical flow. The advanced version (HYDRUS-2D/3D) includes flow in three-dimensional plane (Sejna and Simunek 2007; Simunek et al. 2008).

HYDRUS-2D model is an extended code of the basic HYDRUS-1D program (Simunek et al. 1998, 2005). HYDRUS-1D program is also flexible with respect to the direction of flow, however, flow is only allowed in one dimension as, vertical, horizontal, or inclined.

HYDRUS-2D has been widely applied to analyze two-dimensional water flow and solute transport under various field scenarios such as irrigation studies (Abbasi et al. 2004; Gardenas et al. 2005) and tillage systems (Coquet et al. 2005). The model has also been used to predict lateral and vertical components of drainage flux in hillslopes (Rassam and Littleboy 2003). However, HYDRUS-2D model has not been well tested for solute transport in agricultural landscapes.

A field study was conducted to investigate the vertical and lateral distribution of bromide in a hummocky landscape. The data generated from the field study were used to configure a numerical modelling of water and bromide movement in the landscape. The objectives of this study were: (i) to verify whether the HYDRUS-1D model (Version 4.xx) can reproduce the field data on downward movement of water and bromide in the landscape based on the direct estimation of soil hydraulic parameters from field-measured soil properties; (ii) to simulate two-dimensional distribution of water and bromide across the landscape using the HYDRUS-2D/3D model (Version 1.xx); (iii) to verify that the HYDRUS-2D/3D model can reproduce effects of factors affecting bromide transport in the field study. A successful application of these models to field situations will be a means of extending field data beyond experimental sites.

4.3 Materials and Methods

4.3.1 Field Study and Data Source

Detailed descriptions of the study area and field experimentation, as well as the data source for meteorological records and soil physical properties, have been discussed previously in Chapter 2. The study was carried out near Brandon, Manitoba during the growing seasons of 2007 and 2008, denoted as Site-2007 and Site-2008, respectively. Site-2007 was seeded to canola while Site-2008 had winter wheat.

The plot was delineated into three landscape positions as upper (UPP), middle (MID) and lower (LOW) slope. A microplot demarcated at each landscape position received ^{15}N labelled fertilizer in form of KNO_3 at the rates of 0, 90 and 135 kg N ha^{-1}, and Br^- (KBr) at the rate of 200 kg Br^- ha^{-1}. Soil samples were collected in the fall and spring seasons in both site-years and were analyzed for bromide. Soil samples were taken within the microplot to a depth of 120 cm to obtain vertical distribution and in the top 20 cm of soil layer up to 200 cm away from the microplot to obtain the lateral distribution of bromide.

Soil water content was determined during baseline measurements to compute the initial conditions for water flow. The soil water content was also measured during and after the growing season in Site-2007 only. The soil moisture measurements were carried out using the coring technique (Topp 1993) on the following dates in year 2007: 8th of May; 28th of June; 13th of July; 24th of July; 8th of August; and 9th of November. The gravimetric water content obtained from the soil core was converted to volumetric basis using the field-measured bulk density. The vertical distribution of bromide in all the nitrogen fertility treatments (0, 90, and 135 kg N ha^{-1} denoted as TRT0, TRT90, and TRT135) in the fall and spring seasons were compared with the simulated 1D solute transport in Site-2007 and Site-2008. The mass balance of bromide in the vertical and lateral components of the landscape was compared with that in the 2D model.

4.3.2 Theory of Water Flow and Solute Transport

HYDRUS-1D and HYDRUS-2D/3D models describe uniform flow of water in soil based on a modified form of Richards' equation (Simunek et al. 2005, 2006). The flow equations are solved numerically using the Galerkin-type linear finite element schemes. Ignoring any effect due to hysteresis, the governing equation of one-dimensional uniform (single-porosity) water flow in HYDRUS-1D is described as:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left[K\left(\frac{\partial h}{\partial z} + \cos\,\alpha\right)\right] - S \qquad [4.1]$$

where h is the water pressure head (cm); θ is the volumetric water content (cm^3 cm^{-3}); t is time (d); z is the spatial coordinate in vertical dimension (cm); S is the sink term (d^{-1}) to account for water uptake by plant roots; α is the angle between the flow direction and the vertical axis (i.e., α = 0° for vertical flow, 90° for horizontal flow, and 0° < α < 90° for inclined flow); and K is the unsaturated hydraulic conductivity, which is estimated from the saturated hydraulic conductivity using the following equation:

$$K(h,z) = K_{sat}(z)K_r(h,z) \qquad [4.2]$$

where K_{sat} is the saturated hydraulic conductivity (cm d^{-1}); and K_r is the relative hydraulic conductivity (dimensionless).

The Richards' equation for a uniform two-dimensional Darcian flow in HYDRUS-2D is modified as:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x_i}\left[K\left(K^A_{ij}\frac{\partial h}{\partial x_j} + K^A_{iz}\right)\right] - S \qquad [4.3]$$

where x_i (i = 1,2) are the spatial coordinates representing the horizontal and vertical dimensions; K_{ij}^A are components of a dimensionless anisotropy tensor K^A, and K is the unsaturated hydraulic conductivity function given as:

$$K(h,x,z) = K_{sat}(x,z)K_r(h,x,z) \qquad [4.4]$$

where x is the horizontal coordinate (cm); and z is the vertical coordinate (cm) which is positive upward.

The governing equation of solute transport in HYDRUS is based on the Fickian advection-dispersion model. In the present study, the spatial discretization for the solute transport was solved using the Galerkin finite element method, while the Crank-Nicholson method was used in the time-weighting scheme. The governing equation for one-dimensional vertical transport of a non-reactive solute (Simunek et al. 1999; Gardenas et al. 2005) is given as:

$$\frac{\theta \partial c}{\partial t} = \theta D_w\left(\frac{\partial^2 c}{\partial z^2}\right) - q\left(\frac{\partial c}{\partial z}\right) - NU(c,z,t) \qquad [4.5]$$

where c is the solute concentration in the liquid phases (mg cm^{-3}); q is the volumetric flux density (cm d^{-1}); D_w is the dispersion coefficient in the liquid phase (cm^2 d^{-1}); NU is the solute uptake by plant roots (mg cm^{-3} d^{-1}), which is a function of spatial coordinates and time. The *NU* term was set to zero for modelling bromide transport in the present study. The advection-dispersion equation of a non-reactive solute in HYDRUS-2D accounts for transport in multi-dimensions as:

$$\frac{\theta \partial c}{\partial t} = \frac{\partial}{\partial x_i}\left(\theta D_{ij}^{w} \frac{\partial c}{\partial x_j}\right) - \left(q_i \frac{\partial c}{\partial x_i}\right) - NU(c,i,j,t) \qquad [4.6]$$

where D_{ij}^{w} is the dispersion coefficient tensor for the liquid phase (cm^2 d^{-1}); q_i is the components of the Darcian fluid flux density (cm d^{-1}).

4.3.3 Domain Definition and Spatial Discretization

The simulated one-dimensional flow and transport domain was a simple vertical profile (Fig. 4.1). The profile depth was 120 cm, which was discretized into 121 nodes, each of which was 1 cm thick (*dz*). The simulated soil profile was made up of seven layers, which were used to specify the parameters for soil hydraulic properties and solute transport. The 1D domain was also partitioned into seven sub-regions for mass balance calculations. The distribution of the soil materials and profile sub-regions in HYDRUS-1D corresponds to the depth segments of the soil core: 0-5, 5-10, 10-20, 20-40, 40-60, 60-90, and 90-120 cm.

The conceptual domain for two-dimensional flow and transport across the landscape is shown in Figure 4.2. Based on the elevation data and the horizontal slope length (5,000 cm in the x-direction), the slope gradient in Site-2007 plot was 1.7° while that in Site-2008 was 3.2°. The actual flow domain was represented by a trapezoid with vertices ABCD, and the maximum depth in the

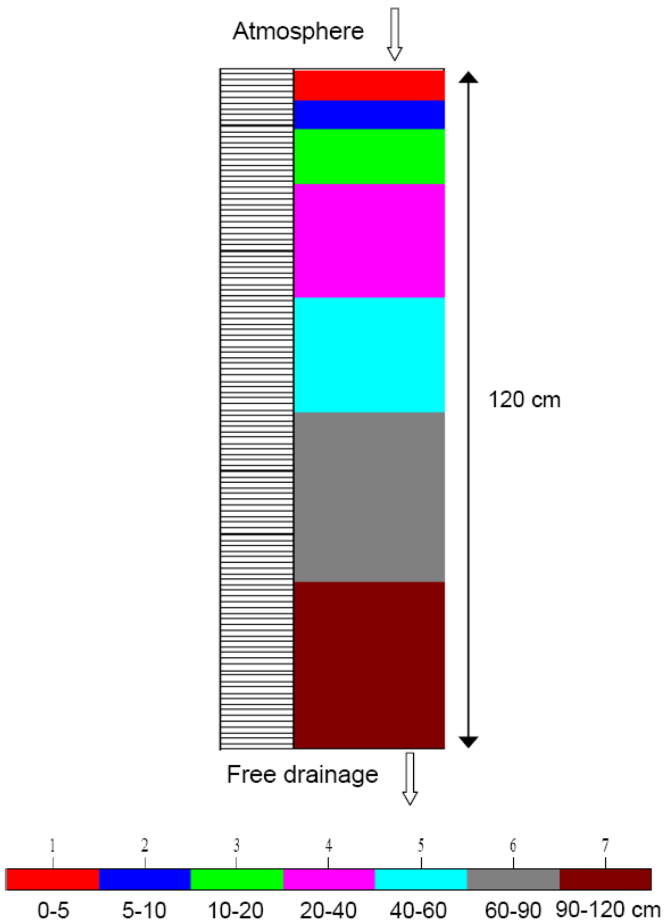


Fig. 4.1. Conceptual model for the one-dimensional flow domain.

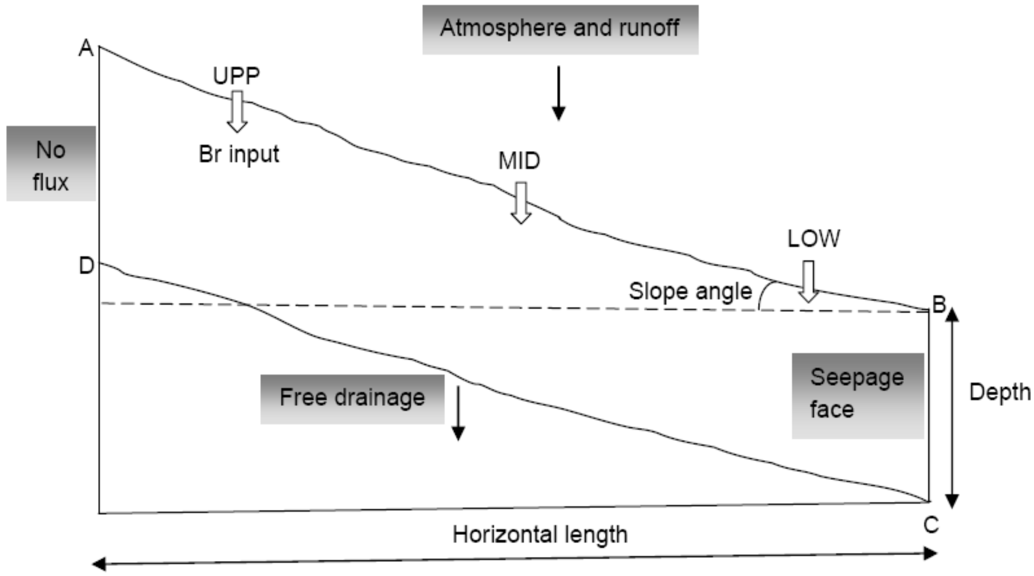


Fig. 4.2. Conceptual model for the two-dimensional flow domain.

vertical dimension (z-direction) was 120 cm. The conceptual 2D domain was designed with the same dimensions (x- and z-directions) as the experimental site.

The domain geometry defined for the 2D simulation was a simple rectangular sloping domain with structured mesh, comprising of triangular finite elements (FE) (Fig. 4.3). The soil materials in the 2D domain were stratified into seven layers (0-5, 5-10, 10-20, 20-40, 40-60, 60-90, and 90-120 cm), arranged parallel to the surface and bottom boundaries of the flow domain (Fig. 4.3a). The sloping domain was compartmentalized into three vertical sub-regions with equal lengths along the horizontal dimension (Fig. 4.3b). The total area for each sub-region in the x-z vertical plane was 20 m^2. Each sub-region represents individual relative landscape position in the downslope direction. This domain definition was designed to capture the mass balance of water and solute at each relative landscape position.

The 2D domain was discretized into grid mesh using the mesh generation feature embedded in HYDRUS-2D/3D program. Since the 2D domain consists of structured finite elements, the horizontal dimension (x-direction) was discretized by 20 cm spacing into uniform grid mesh. However, the vertical dimension (z-direction) was variably discretized into seven segments corresponding to the seven layers of soil materials described above. Finer soil layers were specified close to the surface boundary while coarse layers were at the lower boundary. The FE mesh was discretized into a total of 2,008 nodes and 3,500 computational elements.

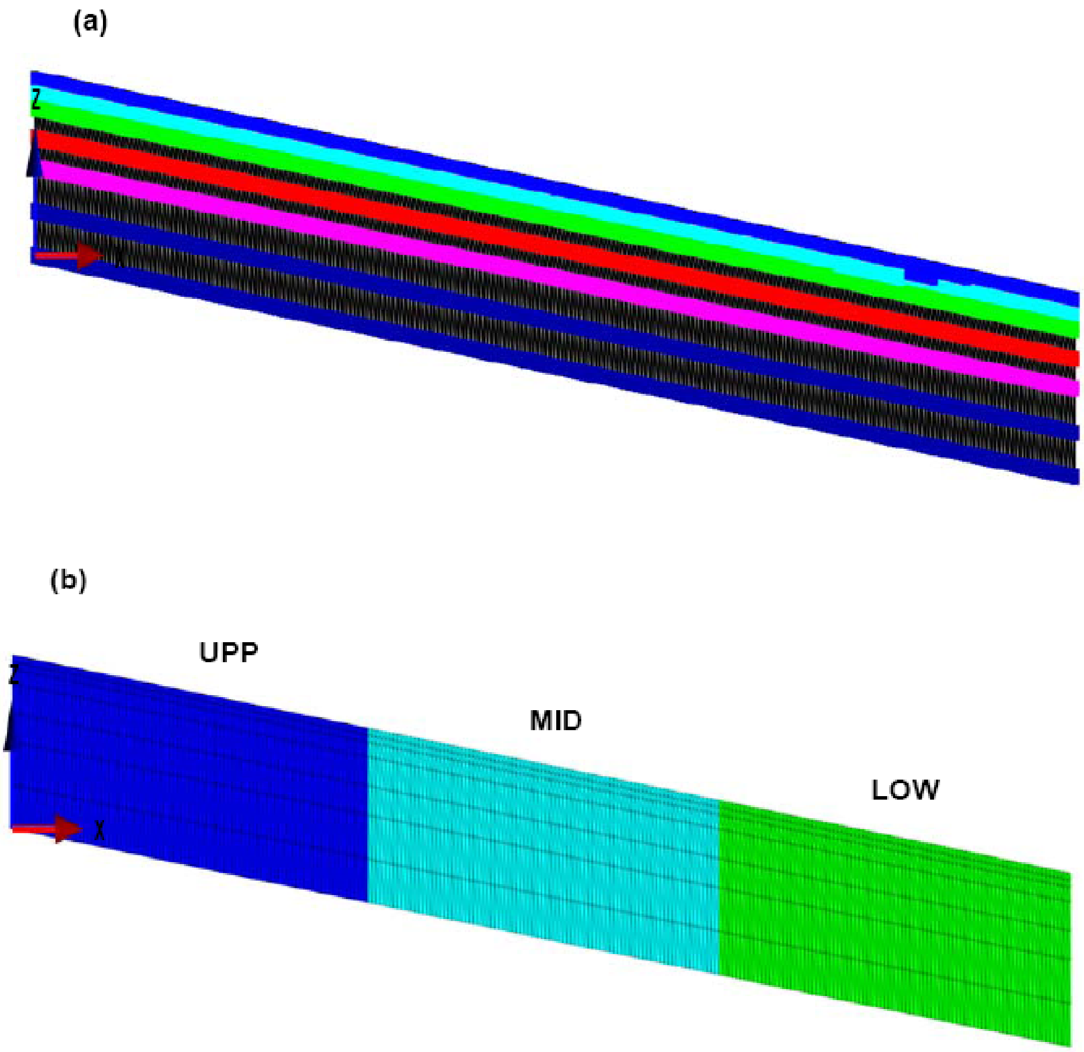


Fig. 4.3. Domain properties for the 2D simulation of flow and transport: (a) soil material distribution, (b) landscape sub-regions.

4.3.4 Water Flow and Solute Transport Parameters

To solve the Richards' equation, knowledge of soil hydraulic properties such as the soil moisture characteristic curve and unsaturated hydraulic conductivity function are required. The analytical model of van Genuchten-Mualem implemented in the HYDRUS code was selected to describe the hydraulic conductivity and retentivity functions. The retentivity function according to van Genuchten model (van Genuchten 1980) is based on the capillary model theory. The statistical pore-size distribution model of Mualem (1976) was applied to the closed-form equation of van Genuchten soil-hydraulic functions to predict the equations for the unsaturated hydraulic conductivity.

The retentivity function in HYDRUS program is defined as:

$$\theta(h) = \begin{cases} \theta_r + \dfrac{\theta_s - \theta_r}{\left[1 + |\alpha h|^n\right]^m} & h < 0 \\ \theta_s & h \geq 0 \end{cases} \qquad [4.7]$$

where

$$m = 1 - \frac{1}{n} \qquad n > 1 \qquad [4.8]$$

where $\theta(h)$ is the soil water content (as a function of the pressure head, h); θ_r and θ_s denote the residual and saturated water content (cm^3 cm^{-3}), respectively; α is a scaling parameter also known as the inverse of the air-entry value; m is an empirical coefficient; and n is a pore-size distribution index. The hydraulic conductivity function is expressed as:

$$K(h)=\begin{cases} K_{sat} S_e^l \left[1-(1-S_e^{1/m})^m\right]^2 & h<0 \\ K_{sat} & h\geq 0 \end{cases} \qquad [4.9]$$

where $K(h)$ is the unsaturated hydraulic conductivity; K_{sat} is the saturated hydraulic conductivity; l is a pore-connectivity parameter, estimated as 0.5 for many soils (Mualem 1976); S_e is the effective soil water content, defined as:

$$S_e = \frac{\theta - \theta_r}{\theta_s - \theta_r} \qquad [4.10]$$

The van Genuchten-Mualem equations described above contain six hydraulic parameters (θ_r, θ_s, α, n, K_{sat}, and l). These parameters were estimated directly from field-measured percent sand, silt and clay, bulk density, and water content at field capacity (Tables 2.1-2.2), using the pedotransfer functions of HYDRUS. The pedotransfer functions are based on neural network predictions of soil hydraulic properties. The Rosetta Lite program (Schaap et al. 2001) incorporated into the HYDRUS code was used to compute the pedotransfer functions in a hierarchical manner from soil textural fractions, bulk density, and water retention points.

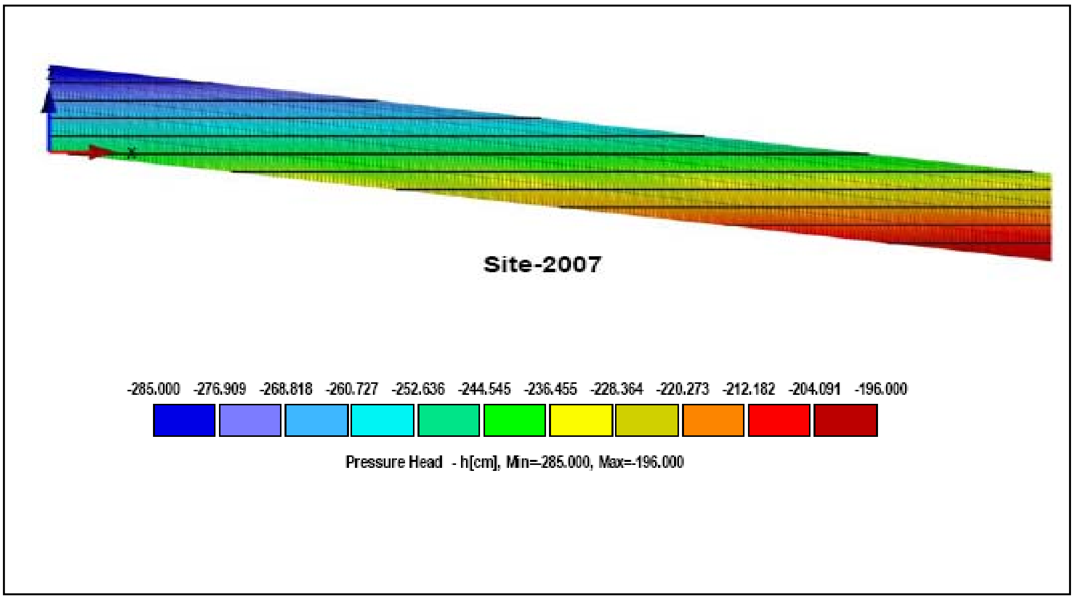
The longitudinal dispersivity, λ_L, for 1D solute transport was set equal to 10 cm for all the layers in the soil profile. The longitudinal dispersivity for the 2D simulation was set at 20 cm with depth, while the transverse dispersivity, λ_T, was one-tenth of λ_L (Abbaspour et al. 2001; Coquet et al. 2005; Gardenas et al. 2005). The longitudinal dispersivity selected for this study was within the range of 5 to 20 cm reported for solute transport at field scale (Warrick 2003; Jury and Horton 2004). The molecular diffusion coefficient of bromide in free water was specified as 1.62 cm^2 d^{-1}. This value was similar to those reported by Coquet et al. (2005) and Kohne and Gerke (2005). The tortuosity factor, based on Moldrup's formulation (Moldrup et al. 1997), was selected for calculating the tortuosity coefficient of bromide in the liquid phase.

4.3.5 Initial and Boundary Conditions

Initial conditions for water flow in HYDRUS can be specified as pressure head or water content. Water contents are known to vary considerably in the soil profile due to spatial heterogeneity (Hillel 1998; Coquet et al. 2005), while the pressure heads are relatively uniform and thereby recommended for computing the initial conditions for water flow in the HYDRUS program. However, there were no field data on pressure heads. Therefore, soil water contents determined during the baseline measurements were used to estimate the initial pressure heads. The water flow process was initialized using the baseline water contents with a model run of 10 days to derive the pressure heads (estimated).

The initial pressure head was specified for each soil layer in the 1D profile, while the average value across the landscape was used for the 2D domain. The initial pressure head for the 2D flow was assumed to increase linearly with depth and downslope (Fig. 4.4). In the solute transport module, the initial bromide concentration was set to zero for both 1D and 2D simulations.

The surface and bottom boundary conditions for the 2D water flow were similar to those used for 1D simulation. The left side of the 2D flow domain corresponds to the summit at the upland region of the landscape, and thereby set as no flux. Seepage face condition was set along the right side boundary at the



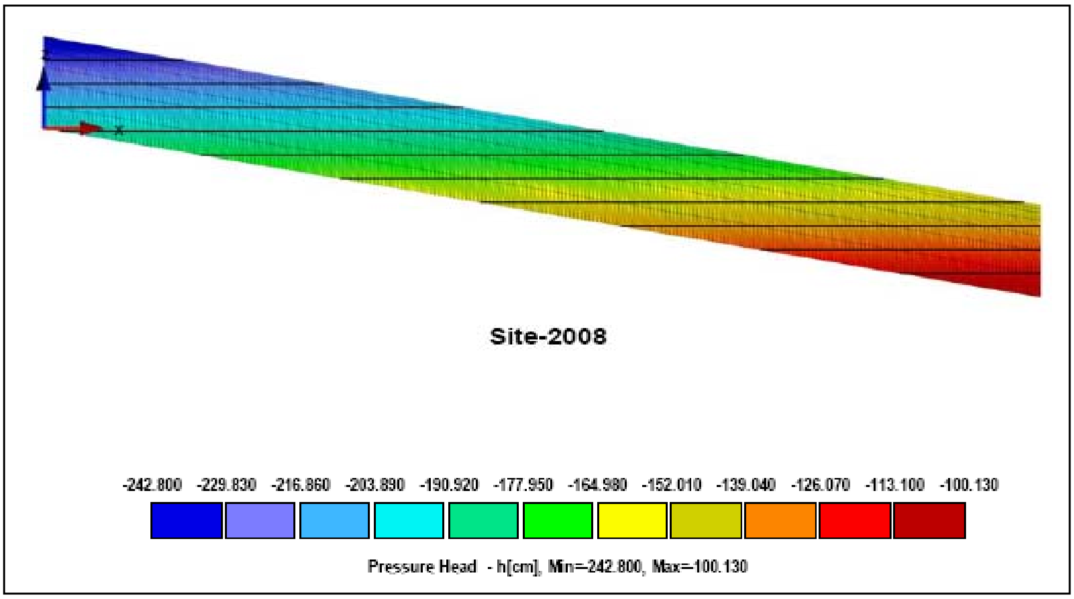


Fig. 4.4. Linear distribution of initial pressure heads across the landscape in Site-2007 (top) and Site-2008 (bottom).

lower slope region. In the seepage face BC, the code assumes that the pressure head is equal to zero along the seepage face boundary. Therefore, flow occurs across the right side boundary once the boundary becomes saturated.

The bromide application was accomplished using surface flux (precipitation) at the rate of 200 kg Br^- ha^{-1} in HYDRUS-1D with the Cauchy (third-type) boundary condition. Zero concentration gradient was imposed at the lower boundary of the soil profile. In HYDRUS-2D computation, the Cauchy type boundary condition was applied to all the domain boundaries using the vector code for atmospheric BC, except at the left side where no flux was specified. The points of bromide application along the surface boundary were represented using the vector code for variable flux BC. To obtain the actual mass balance in the 2D domain, the length of solute application was set at 100 cm in the x-direction (in contrast to the 40 cm length in the three-dimensional field microplot).

4.3.6 Meteorological Variables and Time Information

Local weather data for daily precipitation and air temperature were used to compute time-variable boundary records. Precipitation data from the MAFRI weather station were used to compute rainfall between April and October. However, since snowfall data were not available at the MAFRI weather station (Fig. 2.5), the ENVIR snowfall record was used to gap-fill the precipitation between November and March of the following year. Meteorological parameters such as evaporation and transpiration were estimated using the Hargreaves equation (Jensen et al. 1997) as implemented in the HYDRUS program.

The Feddes model (Feddes 1977) and the S-shape functions of van Genuchten (1985) were used to compute root water uptake. The Feddes model describes root water uptake as a function of soil water pressure head, while the S-shape function is an extended formulation of Feddes by including osmotic stress in the sink term. Plant growth parameters such as crop height, root depth, and leaf area index (LAI) were specified for the crop in each growing season. Phenological data for canola in 2007 were adapted from the Canola Growers Manual (Canola Council of Canada online resources), while those for winter wheat in 2008 were obtained from Fowler (2002) and McCullough and Hunt (1993). The performance of the two root water uptake models was evaluated by comparing 1D simulated soil water contents with the field data in Site-2007.

The model simulation was run for 382 days (1 May 2007 to 16 May 2008) for Site-2007, and 377 days (1 May 2008 to 12 May 2009) for Site-2008, corresponding to the duration of field experiments in both years. Bromide application was implemented on Day 37 (6 June 2007) of the simulation in Site-2007, and on Day 40 (9 June 2008) in Site-2008. The simulation was started with water flow prior to tracer application in order to equilibrate the system for the solute transport. The initial time step, dt, was set at 0.01 d, while the minimum and maximum permitted value of the time increment, dt_{min} and dt_{max}, were computed as 1e−005 and 0.05 d, respectively.

The soil water contents measured during and after the growing season in Site-2007 were simulated using HYDRUS-1D. The output print times for the simulated water flow and solute transport correspond to the following events in both site-years: (i) 24 hours after bromide application; (ii) harvest; (iii) fall soil sampling in October; (iv) 15th of January in winter; (v) spring soil sampling in May.

4.4 Results and Discussion

4.4.1 Estimated Soil Hydraulic Parameters

The pedotransfer functions generated six soil hydraulic parameters namely, θ_r, θ_s, α, n, K_{sat}, and l (Tables 4.1−4.2). The pore-connectivity parameter, l, was estimated to be 0.5 for all the soil materials, thus, was not listed in the table. It should be noted that the identical hydraulic properties estimated for depths 0-5 and 5-10 cm in Site-2007 (Table 4.1) was due to the average soil physical properties within 0-10 cm depth (Table 2.2) used for predicting the parameters. The residual soil water content (θ_r) was generally uniform within the soil profile in Site-2007. Conversely, the saturated water content (θ_s) decreased with depth, as it ranged from 0.37 to 0.57 $cm^3\ cm^{-3}$ across the landscape. The parameters α and n are empirical coefficients affecting the shape of the hydraulic functions. The largest values for the bubbling pressure (α) were within the top 0-10 cm depth.

The estimated saturated hydraulic conductivities (K_{sat}) were greatest for the top 0-10 cm depth at each slope position. At this depth (0-10 cm), the greatest K_{sat} was at the MID slope, followed by the LOW slope, and the smallest was at the UPP slope. However, the magnitude of K_{sat} in the soil profile was greatest at the LOW slope compared to other landscape positions.

Table 4.1. Estimated soil hydraulic parameters in Site-2007 using the pedotransfer functions of HYDRUS.

Landscape	Depth (cm)	θ_r (cm^3 cm^{-3})	θ_s (cm^3 cm^{-3})	α (cm^{-1})	n	K_{sat} (cm d^{-1})
UPP	0-5	0.083	0.555	0.032	1.36	83.3
	5-10	0.083	0.555	0.032	1.36	83.3
	10-20	0.076	0.497	0.013	1.35	39.6
	20-40	0.076	0.433	0.011	1.38	10.9
	40-60	0.095	0.424	0.003	1.64	0.990
	60-90	0.083	0.422	0.008	1.36	3.19
	90-120	0.083	0.421	0.008	1.39	3.12
MID	0-5	0.061	0.549	0.033	1.28	137
	5-10	0.061	0.549	0.033	1.28	137
	10-20	0.095	0.461	0.003	1.64	5.35
	20-40	0.084	0.441	0.005	1.49	5.53
	40-60	0.077	0.439	0.009	1.39	9.91
	60-90	0.078	0.418	0.005	1.49	4.15
	90-120	0.078	0.389	0.004	1.47	0.850
LOW	0-5	0.069	0.569	0.038	1.31	130
	5-10	0.069	0.569	0.038	1.31	130
	10-20	0.080	0.496	0.011	1.36	32.2
	20-40	0.077	0.502	0.009	1.39	45.8
	40-60	0.072	0.433	0.009	1.41	10.8
	60-90	0.083	0.395	0.003	1.57	0.800
	90-120	0.066	0.369	0.007	1.37	1.82

Table 4.2. Estimated soil hydraulic parameters in Site-2008 using the pedotransfer functions of HYDRUS.

Landscape	Depth (cm)	θ_r (cm^3 cm^{-3})	θ_s (cm^3 cm^{-3})	α (cm^{-1})	*n*	K_{sat} (cm d^{-1})
UPP	0-5	0.069	0.566	0.038	1.31	147
	5-10	0.077	0.493	0.013	1.34	36.1
	10-20	0.076	0.441	0.017	1.34	18.6
	20-40	0.073	0.423	0.015	1.34	12.9
	40-60	0.072	0.364	0.004	1.48	0.380
	60-90	0.067	0.372	0.011	1.31	2.40
	90-120	0.059	0.382	0.014	1.37	9.77
MID	0-5	0.081	0.613	0.044	1.69	171
	5-10	0.074	0.470	0.009	1.40	23.9
	10-20	0.077	0.446	0.008	1.42	11.1
	20-40	0.071	0.432	0.015	1.37	16.8
	40-60	0.065	0.367	0.012	1.29	2.64
	60-90	0.065	0.394	0.014	1.36	9.48
	90-120	0.062	0.388	0.012	1.39	9.01
LOW	0-5	0.055	0.572	0.051	1.32	208
	5-10	0.067	0.555	0.038	1.36	138
	10-20	0.073	0.495	0.011	1.37	46.6
	20-40	0.075	0.447	0.013	1.34	15.2
	40-60	0.068	0.371	0.006	1.36	0.870
	60-90	0.068	0.389	0.007	1.42	3.43
	90-120	0.063	0.378	0.006	1.48	4.17

The soil hydraulic parameters estimated for Site-2008 (Table 4.2) were similar to those in Site-2007. The hydraulic conductivity that was estimated for the 0-10 cm depth in Site-2008 was greater than in Site-2007. The K_{sat} within 0-10 cm depth in Site-2008 ranged from 91.5 cm d^{-1} at the UPP slope to 173 cm d^{-1} at the LOW slope; greater than the range of 83.3 to 137 cm d^{-1} estimated in Site-2007. In all three landscape positions in 2008, the smallest K_{sat} values were at the 40-60 cm depth. This depth corresponds to the region with the greatest bulk density in the field (Table 2.3). In both site-years, however, estimated K_{sat} generally decreased with depth. The magnitude of K_{sat} in the soil profile had the following trend: LOW > MID > UPP, while the average K_{sat} at each landscape position was similar for Site-2007 and Site-2008.

The performance of the pedotransfer functions was evaluated by comparing field-measured K_{sat} with estimated K_{sat}. The agreement between field measurements and model simulation was tested using the coefficient of determination, R^2. The saturated hydraulic conductivity was estimated for the same depth intervals as the field measurements. While there were large differences in K_{sat} values between field measurements and model simulation at the upper and middle slope positions in Site-2007, the trend in estimated K_{sat} with depth and among landscape positions was similar to the field measurements in both site-years (Tables 4.3-4.4). The data indicated that the pedotransfer functions predicted the K_{sat} reasonably well, particularly in Site-2008. It should be noted that these hydraulic parameters were predicted directly from measured soil physical properties based on the neural network pedotransfer functions. This is unlike the conventional inverse solution based on parameter optimization.

Table 4.3. Measured and estimated saturated hydraulic conductivity in Site-2007.

Landscape	Depth (cm)	Measured K_{sat} (cm hr^{-1})	Estimated K_{sat} (cm hr^{-1})	R^2	P value
UPP	0-15	0.58	2.54	0.00	0.9996
	15-30	1.99	0.84		
	30-60	1.02	0.16		
	60-90	0.04	0.13		
MID	0-15	2.60	1.98	0.20	0.5666
	15-30	3.07	0.22		
	30-60	1.23	0.31		
	60-90	−	0.17		
LOW	0-15	3. 79	3.20	0.93	0.0358
	15-30	2.19	1.59		
	30-60	1.92	0.88		
	60-90	−	0.03		

It was impossible to obtain realistic measurements at 60-90 cm depths at the MID and LOW landscape positions in Site-2007.

Table 4.4. Measured and estimated saturated hydraulic conductivity in Site-2008.

Landscape	Depth (cm)	Measured K_{sat} (cm hr^{-1})	Estimated K_{sat} (cm hr^{-1})	R^2	P value
UPP	0-15	1.74	2.07	0.97	0.0131
	15-30	1.12	0.65		
	30-60	1.05	0.10		
	60-90	0.99	0.10		
MID	0-15	2.05	2.64	0.87	0.0586
	15-30	1.04	0.56		
	30-60	0.27	0.29		
	60-90	0.20	0.40		
LOW	0-15	3.09	5.66	0.79	0.0318
	15-30	2.16	1.04		
	30-60	1.28	0.14		
	60-90	0.66	0.14		

4.4.2 Estimated Soil Water and Boundary Fluxes in Site-2007

The temporal changes in soil surface water fluxes, potential evapotranspiration, root water uptake, and soil water fluxes across the lower boundary were estimated in the HYDRUS-1D. The estimated soil surface water flux (vTop) depicts net infiltration of precipitation (downward flux) when flux is negative, or net evaporation (upward flux) for a positive flux (Fig. 4.5a). The estimated temporal distribution of surface flux was similar among landscape positions in Site-2007. The estimated upward flux was smaller than the net infiltration.

The maximum evapotranspiration (PET) estimated for Site-2007 (Fig. 4.5b) was 0.62 cm d^{-1} as shown on Day 140 (22 September 2007), corresponding to 36 days after harvest. However, the actual root water uptake (vRoot) indicated a maximum water uptake of 0.35 cm d^{-1} on Day 90 (Fig. 4.5c), 14 days before harvest. The root water uptake in 2007 was less than 0.05 cm d^{-1} within the first 30 days, but increased to 0.1 cm d^{-1} by Day 42 (31 days after seeding). The steep increase in vRoot indicated increased capacity of roots to absorb water due to crop growth and increased atmospheric demand for water due to warmer temperature. Following harvest, on Day 104, the model showed that root water uptake continued for the next 100 days. However, vRoot declined markedly to almost zero by Day 200 and remained below 0.1 cm d^{-1} throughout the remaining part of the simulation (Fig. 4.5).

The estimated water flux across the bottom of the soil profile (vBot) depicts inflow and outflow for a positive and negative flux, respectively. The negative vBot represents the amount of water that is leached below the root zone. The

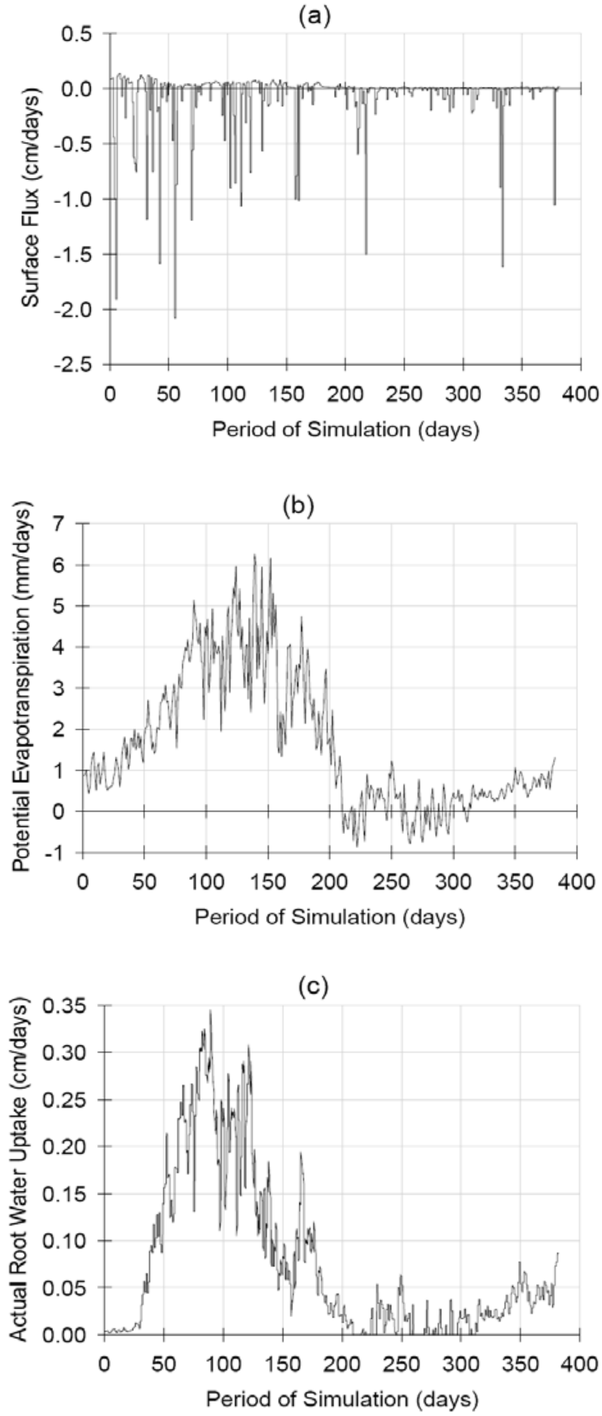


Fig. 4.5. Estimated soil surface water flux and root water uptake in Site-2007: (a) Surface flux of water, where negative flux represents net infiltration and positive flux represents net evaporation; (b) Potential evapotranspiration, which is the maximum amount of water that can be lost by evaporation and transpiration; (c) Actual root water uptake.

bottom flux showed that there was no inflow of water into the soil profile from the water table in Site-2007 (Fig. 4.6). At the UPP slope position, the maximum vBot was −0.1 cm d^{-1} on Day 49. The vRoot and vTop on Day 49 were 0.139 and −0.044 cm d^{-1}, respectively, while precipitation was 0.15 cm d^{-1} (Fig. 4.6a). The results suggest that the large estimated outflow on Day 49 was due to percolation of antecedent soil water. Afterwards, water outflow (leaching) from the soil profile decreased to zero by Day 200. The maximum outflow at the MID slope was −0.11 cm d^{-1} (Fig. 4.6b), greater than that estimated for the UPP slope, while the maximum vBot at the LOW slope (Fig. 4.6c) was smaller than in other landscape positions as −0.08 cm d^{-1}.

The estimated cumulative bottom flux is a measure of the total amount of water moving out of the soil profile at the lower boundary of the flow domain. The cumulative outflow from the soil profile at the end of simulation (Day 382) in Site-2007 was MID (−8.0 cm) > UPP (−7.14 cm) > LOW (−4.89 cm) compared to the cumulative precipitation and root water uptake of 38.7 and 29.8 cm, respectively.

4.4.3 Estimated Soil Water and Boundary Fluxes in Site-2008

The magnitudes of net infiltration estimated for Site-2008 (Fig. 4.7a) were greater than in Site-2007. The temporal distribution of PET in Site-2008 (Fig. 4.7b) was similar to that in Site-2007, while the maximum root water uptake in Site-2008 was 0.5 cm d^{-1} which occurred on Day 111, six days after harvest (Day 105) (Fig. 4.7c). The 2008 simulation confirmed that the model predicts a water uptake period much longer than in reality.

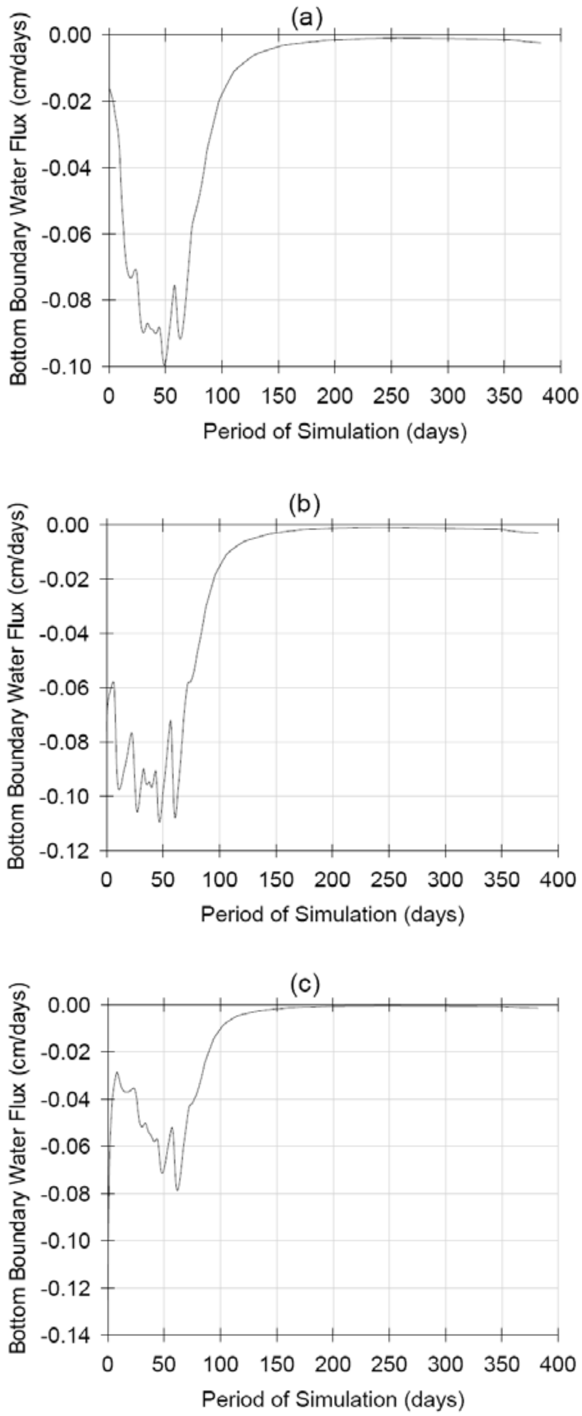


Fig. 4.6. Estimated soil water flux at the bottom boundary in Site-2007 at: (a) Upper slope position; (b) Middle slope position; and (c) Lower slope position. Note that negative flux represents downward movement and positive flux represents upward movement of water from the water table.

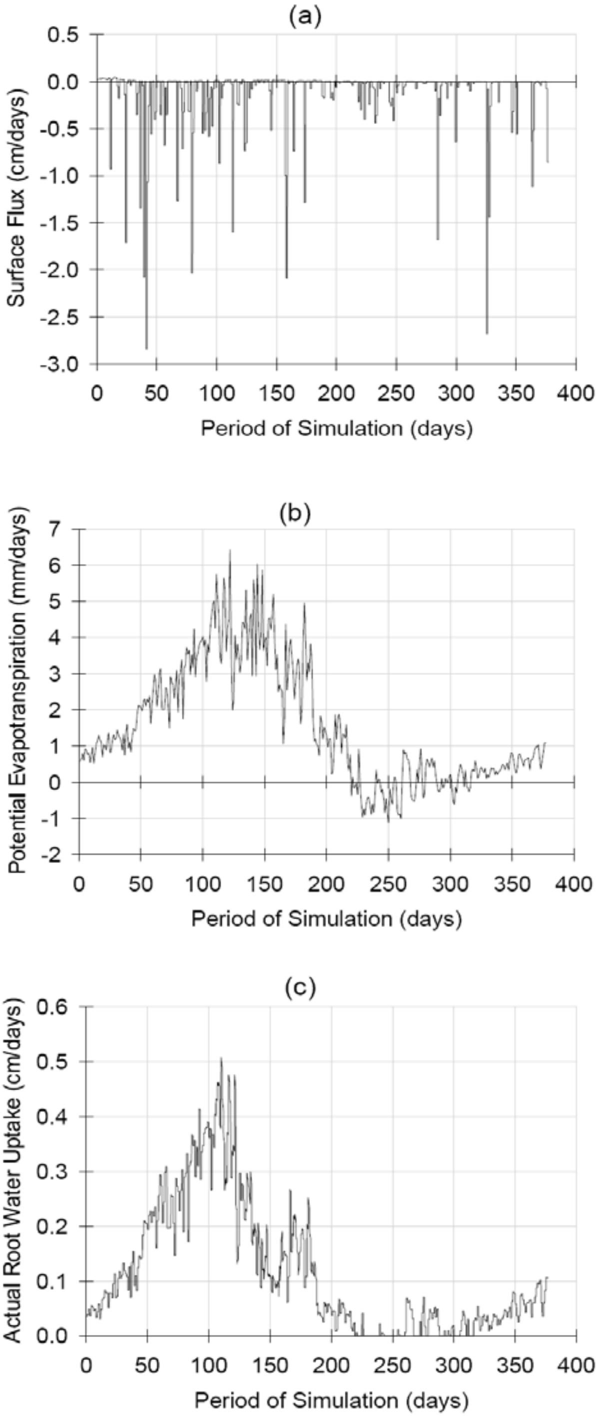


Fig. 4.7. Estimated soil surface water flux and root water uptake in Site-2008: (a) Surface flux of water, where negative flux represents net infiltration and positive flux represents net evaporation; (b) Potential evapotranspiration, which is the maximum amount of water that can be lost by evaporation and transpiration; (c) Actual root water uptake.

The model probably assumed a long period of vegetative growth, unlike the short growing season typical of the Canadian prairies. Nevertheless, the vRoot in Site-2008 (Fig. 4.7) reflected the greater effects of winter wheat on soil water consumption early in the season, compared to canola in Site-2007 (Fig. 4.5).

In spite of the greater water uptake in Site-2008, the estimated bottom fluxes (Fig. 4.8a-c) were greater than in Site-2007. The trend in cumulative outflow among landscape positions in Site-2008 was MID (−10.1 cm) > LOW (−9.42 cm) > UPP (−7.67 cm), while the cumulative precipitation and root water uptake were 50.9 and 42.2 cm, respectively. According to model simulation, it is clear that the greater precipitation in Site-2008 overwhelmed the effect of root water uptake on soil water distribution, thereby resulting in more water moving out of the soil profile compared to Site-2007.

Estimated water fluxes in both site-years indicated that crop water uptake was the major sink for soil moisture depletion as opposed to net leaching. The trend in estimated outflow among landscape positions was not consistent with the field data on bromide leaching. The field study showed that the ranking for bromide leaching was LOW > UPP > MID. The variability in bromide leaching among landscape positions was attributed to differences in soil physical properties and dissipation of water from upslope regions. It should be noted that the estimated cumulative outflow in the 1D model did not consider effects of landscape on soil water dynamics. As such, the cumulative outflow in HYDRUS-1D depends only on differences in soil hydraulic parameters, which may not be consistent with the trend in bromide leaching among landscape positions.

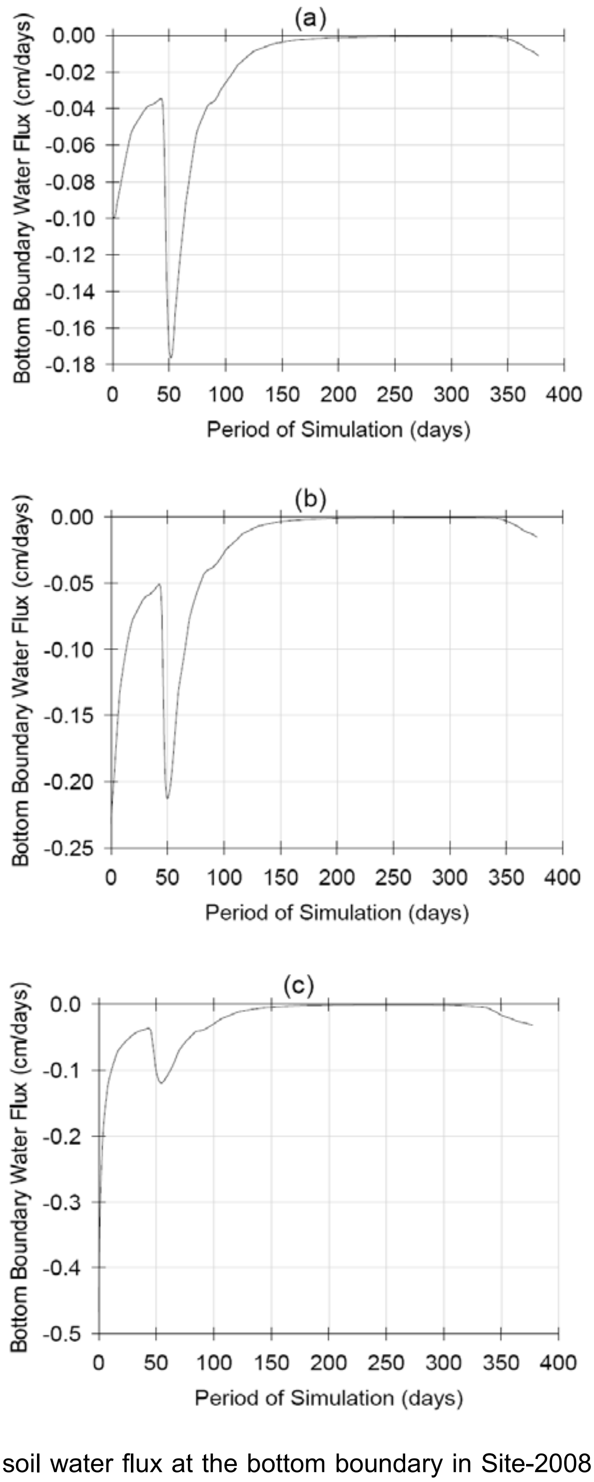


Fig. 4.8. Estimated soil water flux at the bottom boundary in Site-2008 at: (a) Upper slope position; (b) Middle slope position; and (c) Lower slope position. Note that negative flux represents downward movement and positive flux represents upward movement of water from the water table.

4.4.4 Simulated and Measured Soil Water Contents in Site-2007

The soil water contents measured during and after the growing season in Site-2007 (Soil MC) were simulated (Feddes and S-shape) using the HYDRUS-1D model (Fig. 4.9-4.11). The model simulations of soil water contents were evaluated using the coefficient of determination, R^2 (Akinremi et al. 2005). Due to large variability in water content in the field data, the R^2 was derived for each replicate of the soil profile while the mean of observations was plotted with the model prediction (Table 4.5).

At the UPP slope position there was a good agreement in soil water contents between model predictions and field data (Soil MC) during the growing season (Fig. 4.9). The simulated soil water distribution was similar for the Feddes root water uptake model and the S-shape function. On 9 November 2007, estimated soil water contents were smaller than observed values, while amounts of soil water estimated by the Feddes model were smaller than those simulated by the S-shape function (Fig. 4.9). The smaller water contents in the model predictions after harvest may be related to the continuation of water uptake beyond harvest (Day 104) in the model (Fig. 4.5).

During the growing season at the MID slope position, measured soil water contents were smaller than the model prediction within the top 60 cm depth in most cases (Fig. 4.10). After harvest, however, simulated water contents were smaller than measured water contents (Fig. 4.10f). On Nov. 9, the trend in soil water distribution between model predictions and field measurements at the MID

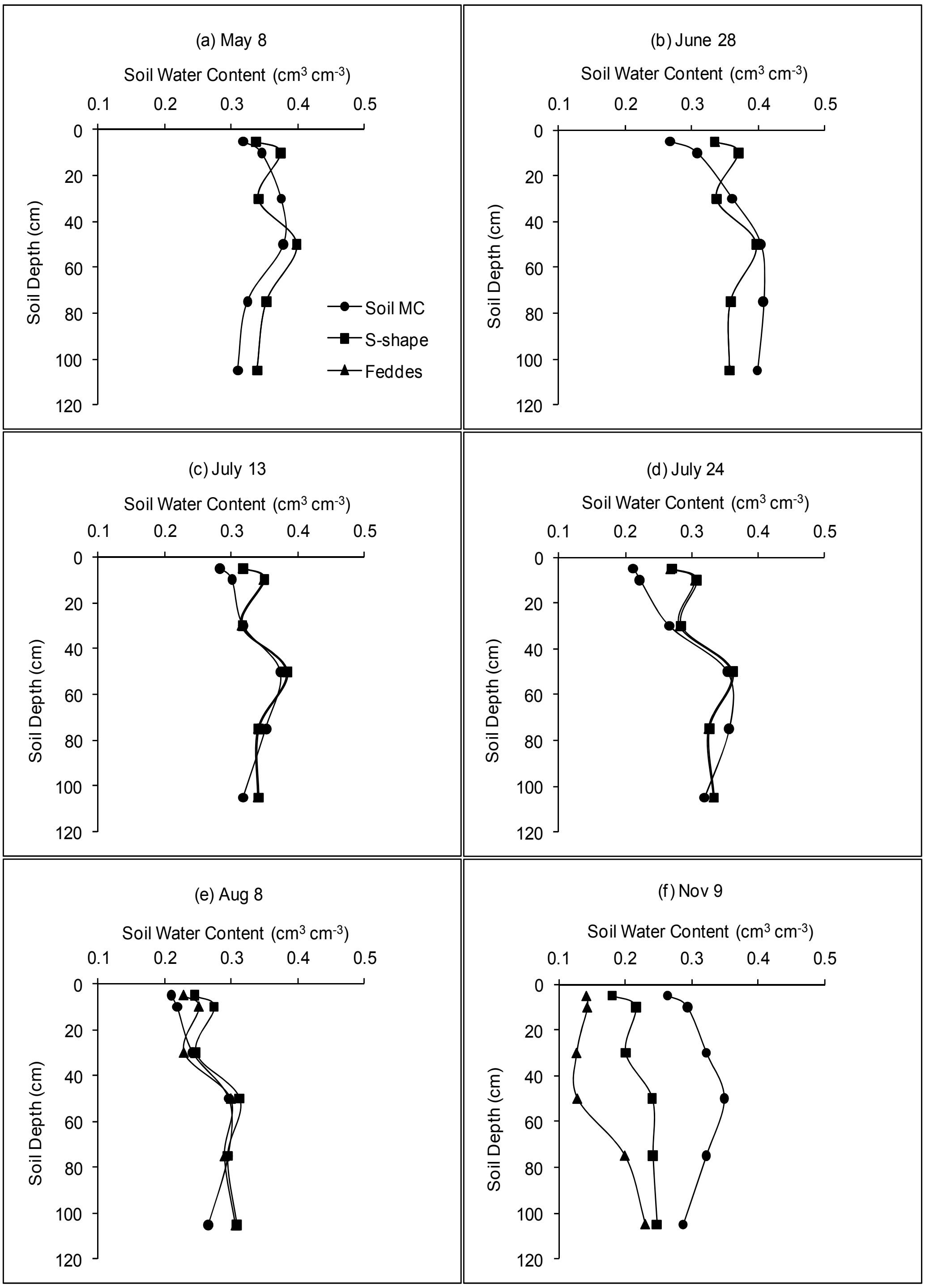


Fig. 4.9. Measured and simulated 1D soil water contents at the upper slope in Site-2007. Soil MC = measured soil water content; S-shape = simulated soil water content using the S-shape function of van Genuchten; Feddes = simulated soil water content using the original Feddes function for root water uptake.

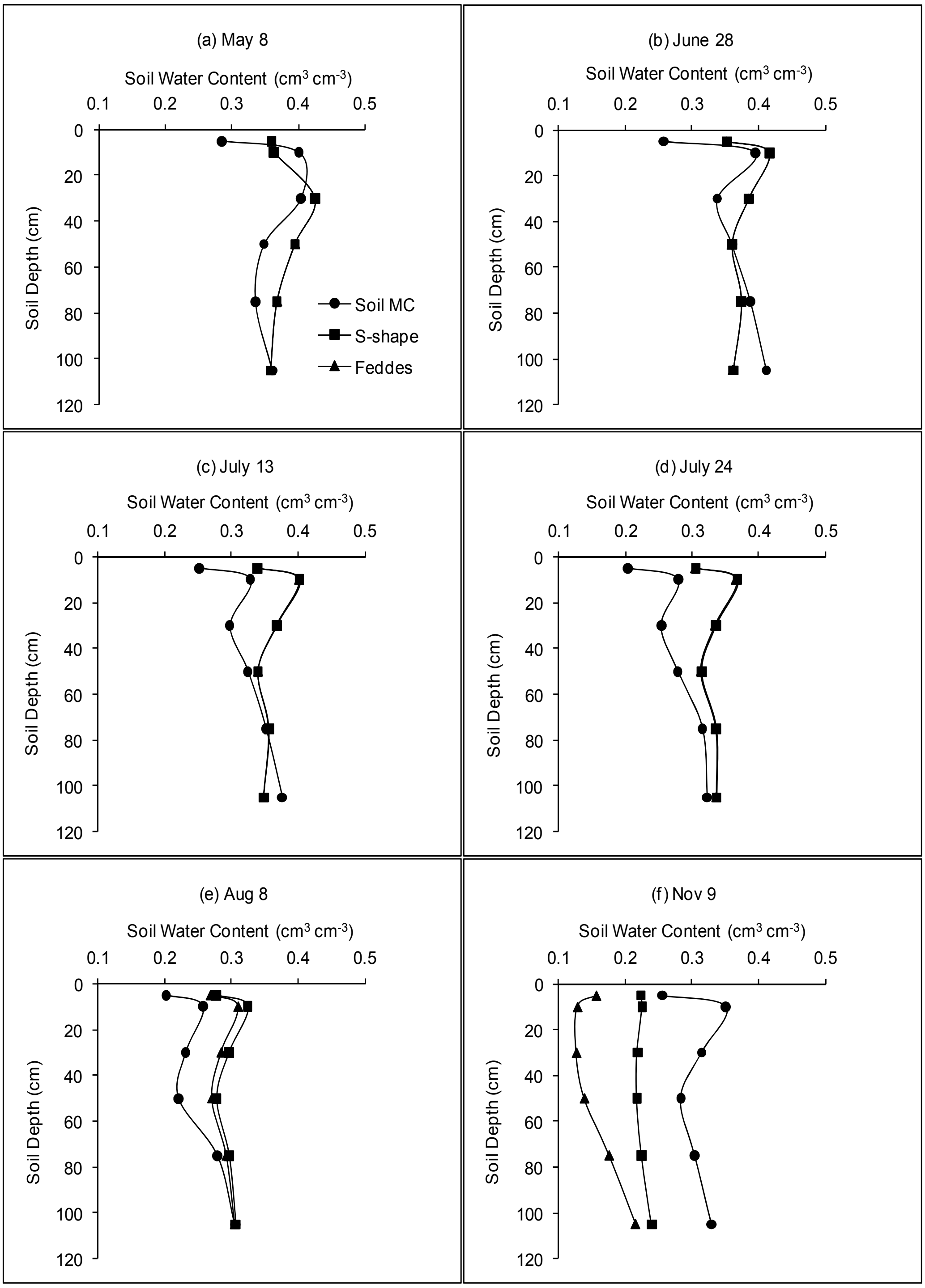


Fig. 4.10. Measured and simulated 1D soil water contents at the middle slope in Site-2007. Soil MC = measured soil water content; S-shape = simulated soil water content using the S-shape function of van Genuchten; Feddes = simulated soil water content using the original Feddes function for root water uptake.

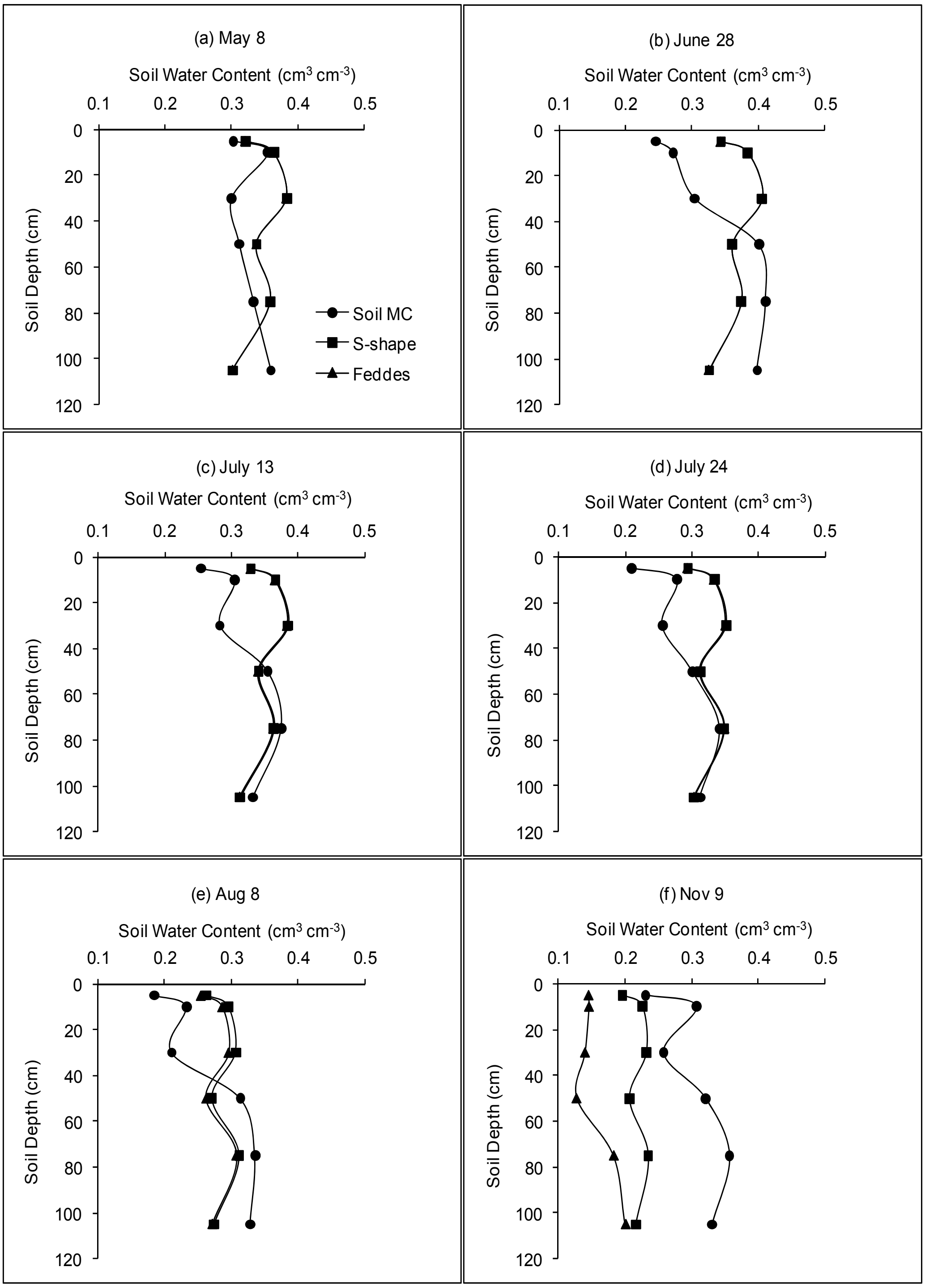


Fig. 4.11. Measured and simulated 1D soil water contents at the lower slope in Site-2007. Soil MC = measured soil water content; S-shape = simulated soil water content using the S-shape function of van Genuchten; Feddes = simulated soil water content using the original Feddes function for root water uptake.

Table 4.5. Evaluation of HYDRUS-1D model predictions of soil water contents, using independent replicates that were measured in Site-2007.

Landscape	Dates	R^2		
		Rep 1	*Rep 2*	*Rep 3*
UPP	May 8	0.41	0.68 *	0.04
	June 28	0.43	0.25	0.22
	July 13	0.84 *	0.21	0.40
	July 24	0.65 *	0.56*	0.69 *
	Aug 8	0.45	0.37	0.54
	Nov 9	0.00	0.41	0.43
MID	May 8	0.27	0.78 *	0.64 *
	June 28	0.56*	0.31	0.02
	July 13	0.09	0.01	0.01
	July 24	0.27	0.31	0.10
	Aug 8	0.53	0.56*	0.37
	Nov 9	0.10	0.48	0.11
LOW	May 8	0.01	0.05	0.13
	June 28	0.04	0.00	0.11
	July 13	0.03	0.02	0.05
	July 24	0.20	0.06	0.12
	Aug 8	0.04	0.03	0.00
	Nov 9	0.19	0.15	0.45

* statistical significance at the 0.05 level, respectively.

slope position was similar to that at the UPP slope. At the LOW slope position, the differences in soil water distribution between model simulation and field measurements (Fig. 4.11) were greater than what was observed at other landscape positions. After harvest, the underestimation of simulated water contents at the LOW slope position was similar to those at the UPP and MID slope positions.

The R^2 values showed that the agreement between model predictions and measured water contents was largely affected by spatial variability in soil water contents among replicates of soil profiles (Table 4.5). Overall, the best agreement between the simulated water distribution and the field measurement was at the UPP slope position, where the R^2 was as large as 0.84 ($P < 0.05$). In contrast, the model performed poorly for the estimated soil water content at the LOW slope position. Akinremi et al. (2005) also identified the influence of variability in soil properties, the amount of rainfall received by the lysimeter, and non-Darcian flow of water due to preferential flow, on uneven distribution of water and chloride across the field. Unlike the variability in field measurements, the model prediction assumed a uniform one-dimensional flow which was analyzed by the single porosity hydraulic model (Simunek et al. 2003; Jarvis 2007).

Since the water contents simulated using the S-shape function had a better agreement with the field data after harvest compared to the Feddes model, the S-shape function of van Genuchten (1985) was selected for simulating root water uptake in the water flow and solute transport processes.

4.4.5 One-Dimensional Water Flow and Solute Transport in Site-2007

In the model simulation, the vertical distribution of soil water at the UPP slope showed a temporal variation in soil water content among output dates in Site-2007 (Fig. 4.12a). On 6 June 2007, 24 hours following solute application in Site-2007 the estimated vertical distribution of soil water indicated a wet soil regime. At this time, the estimated average water content in the soil profile at the UPP slope was 0.36 $cm^3\ cm^{-3}$. This volumetric water content is equivalent to 43.7 cm of water in the soil profile, following 9.1 cm of precipitation between 1 May 2007 and 6 June 2007.

The estimated soil water content declined to 0.25 $cm^3\ cm^{-3}$ by 15 October 2007 (Fig. 4.12a) which corresponds to the fall season soil sampling date in Site-2007. The estimated soil water content slightly increased to 0.27 $cm^3\ cm^{-3}$ on 16 May 2008. This may have been due to soil water recharge by the snowfall after January 2008, in combination with the early spring rainfall. This indicated that the HYDRUS model considers snowfall as rainfall in the water flow process.

HYDRUS-1D estimated that bromide had moved down to 10-20 cm depth by 24 hours after application, while 64% of the bromide applied remained at the soil surface (0-5 cm depth) at the UPP slope position in Site-2007 (Fig. 4.12b). The model also indicated that the solute subsequently moved down the soil profile with a centre of mass at 20-40 cm depth on 12 August 2007 (harvest). The vertical distribution of bromide remained unchanged between 12 August 2007 and 15 October 2007. In spite of the precipitation received and reduction in root water uptake between Day 104 (12 August 2007) and Day 168 (15 October

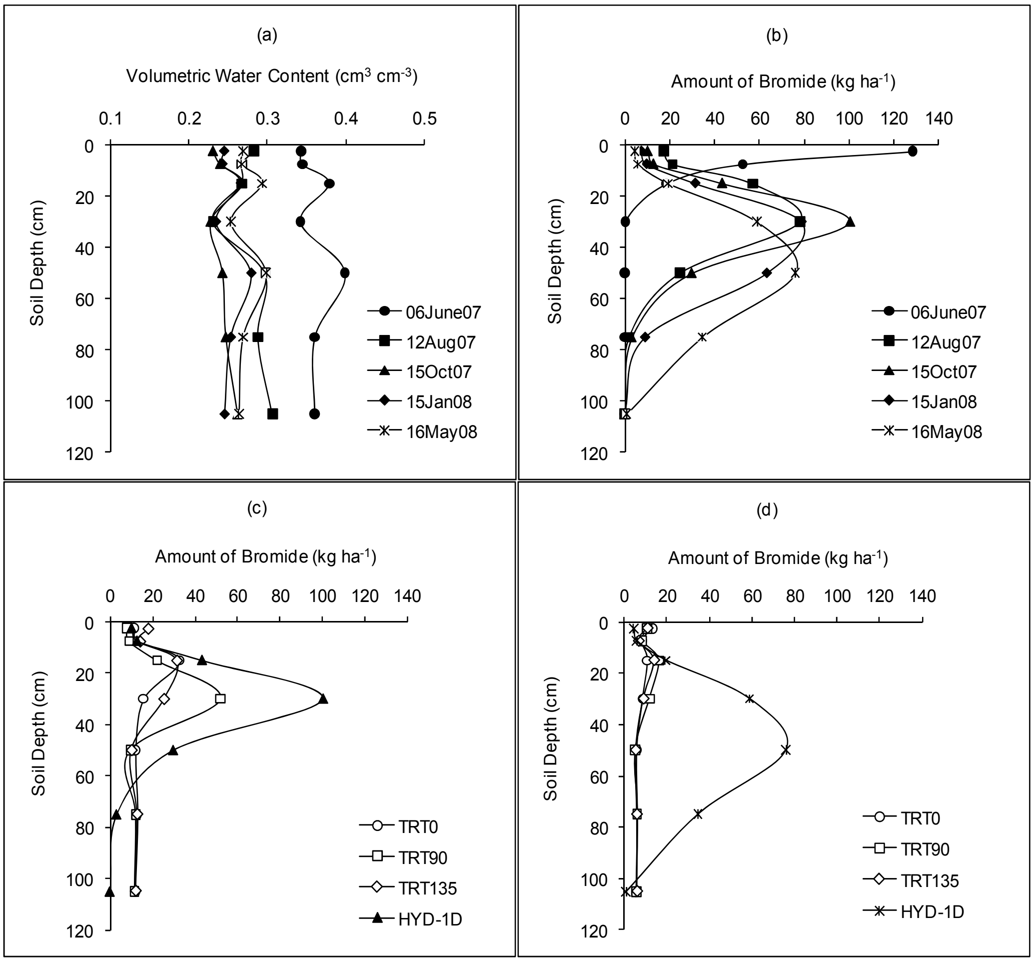


Fig. 4.12. One-dimensional simulation of water flow and solute transport at the upper slope position in Site-2007: (a) HYDRUS-1D estimated volumetric water content at various times of the year; (b) HYDRUS-1D estimated bromide distribution at various times of the year; (c) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 15 Oct. 2007; (d) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 16 May 2008.

2007), the estimated cumulative surface flux was 5.6 cm at this period. The vertical distribution of solute estimated for Day 168 suggests that this amount of infiltrating water was not sufficient to move the centre of mass below 40 cm depth by 15 October 2007. Note that the centre of mass of bromide was measured at 20-40 cm depth in TRT90 in the fall of 2007 (Fig. 4.12c).

By 15 January 2008, the model estimated that bromide had moved further into the soil profile with a concomitant reduction in bromide peak (Fig. 4.12b). In reality, all flow and transport processes were expected to cease during the winter season due to freezing conditions. In contrast, the vertical distribution of bromide that was simulated for January 2008 showed that the model treated snowfall between fall of 2007 and January 2008 as rainfall, thereby moving the solute in the soil profile. According to the model, the centre of mass of bromide had moved down to the middle of the soil profile (40-60 cm depth) by 16 May 2008.

The 1D simulation of bromide at the UPP slope in Site-2007 indicated that bromide did not move past 120 cm depth, suggesting that there was no leaching loss of bromide from the soil profile. This was in spite of 7.1 cm of water moving below this depth as simulated by the model. The lack of bromide leaching in HYDRUS-1D simulation was also reflected in estimates of large amounts of solute remaining in the profiles in fall 2007 and spring 2008, in contrast to the smaller amounts of bromide that were measured in the field under the three N fertility treatments (Fig. 4.12c-d). Nevertheless, amounts of bromide within the top 20 cm depth were similar for the simulated 1D profile and field measurements at the UPP slope position. Of the three fertility treatments at the UPP slope, the distribution of bromide in TRT90 most closely matched the HYDRUS-1D simulation. However, the model estimated a greater amount of bromide at the 20 to 60 cm depth compared to what was measured in TRT90 (Fig. 4.12c).

While the large amount of bromide in the simulated profiles may be partly due to lack of leaching loss, it is also important to note that HYDRUS-1D assumed that bromide transport was strictly by downward movement; thereby ignoring lateral movement. It was shown in the previous chapter on two-dimensional redistribution of bromide that approximately 20% (40 kg ha^{-1}) of the bromide applied was recovered within 25 to 175 cm away from the site of application in the top 20 cm soil layer in the lateral direction in fall 2007. Accounting for lateral bromide movement would reduce some of the differences between the simulated and the measured mass of bromide.

The simulated temporal distributions of soil water and bromide at the MID and LOW slope positions in Site-2007 (Fig. 4.13-4.14) were similar to that at the UPP slope. The 1D model was able to simulate amounts of bromide within the top 20 cm depth in all three N fertility treatments at the UPP and MID slope positions (Fig. 4.12-4.13). However, the 1D model underpredicted bromide losses at the LOW slope position (Fig. 4.14).

4.4.6 One-Dimensional Water Flow and Solute Transport in Site-2008

The simulated temporal distribution of soil water among output dates in Site-2008 (Fig. 4.15-4.17) was similar to that in Site-2007. In Site-2008, the estimated mass of bromide remaining at the 0-5 cm depth by 24 hours after application (10 June 2008) ranged from 32 to 43% relative to the amount applied (Fig. 4.15-4.17). The model indicated that the solute subsequently moved down

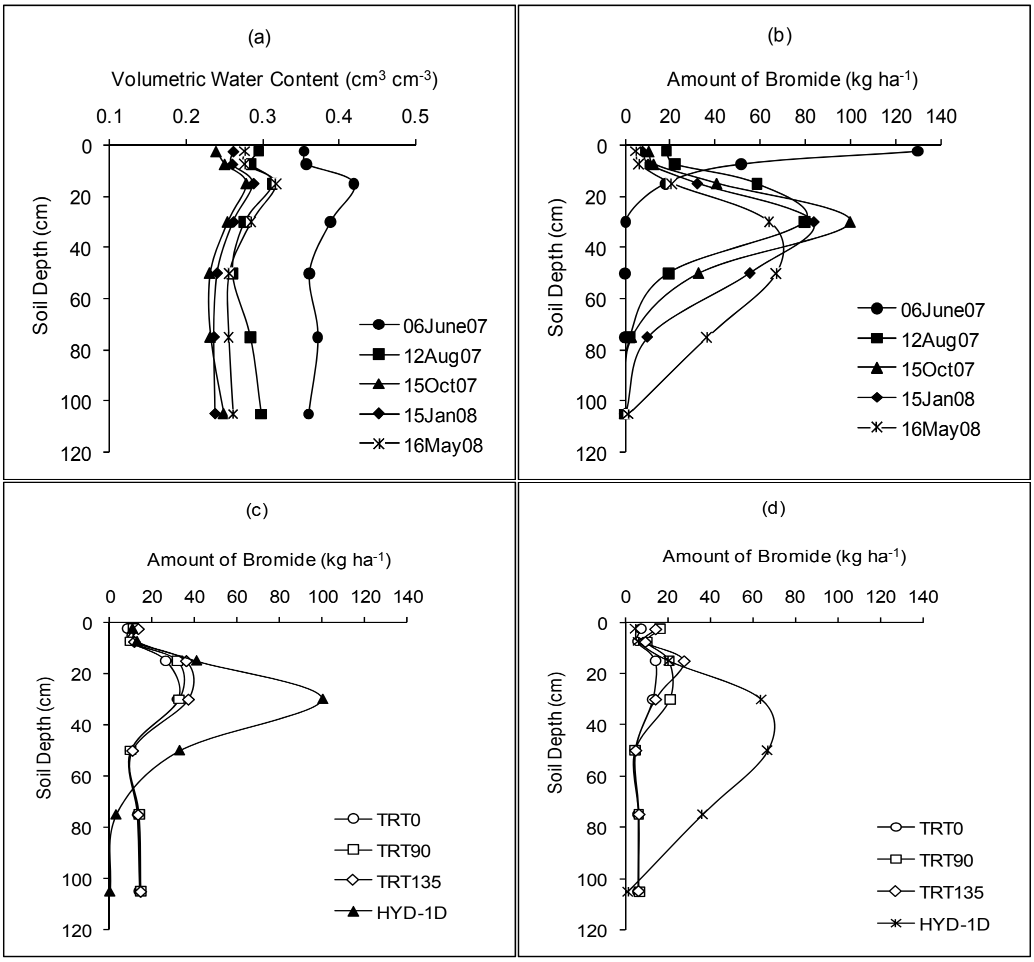


Fig. 4.13. One-dimensional simulation of water flow and solute transport at the middle slope position in Site-2007: (a) HYDRUS-1D estimated volumetric water content at various times of the year; (b) HYDRUS-1D estimated bromide distribution at various times of the year; (c) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 15 Oct. 2007; (d) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 16 May 2008.

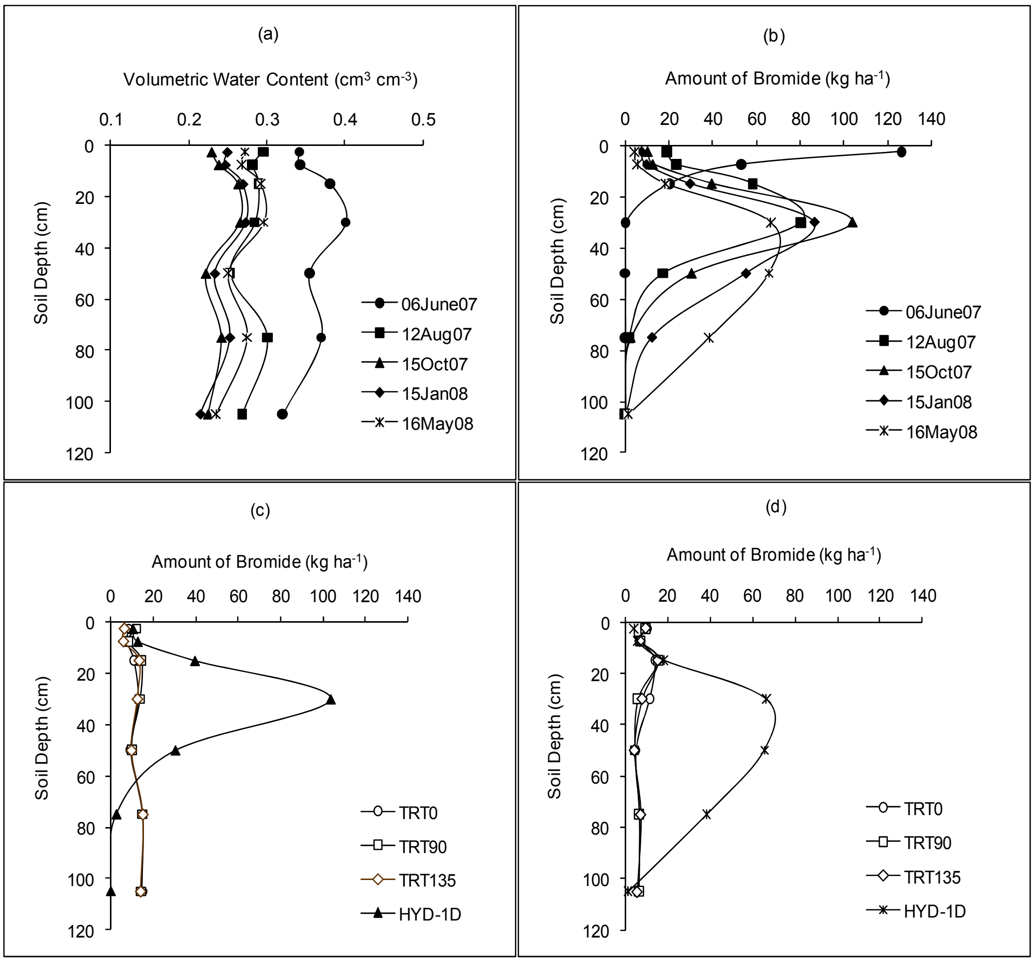


Fig. 4.14. One-dimensional simulation of water flow and solute transport at the lower slope position in Site-2007: (a) HYDRUS-1D estimated volumetric water content at various times of the year; (b) HYDRUS-1D estimated bromide distribution at various times of the year; (c) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 15 Oct. 2007; (d) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 16 May 2008.

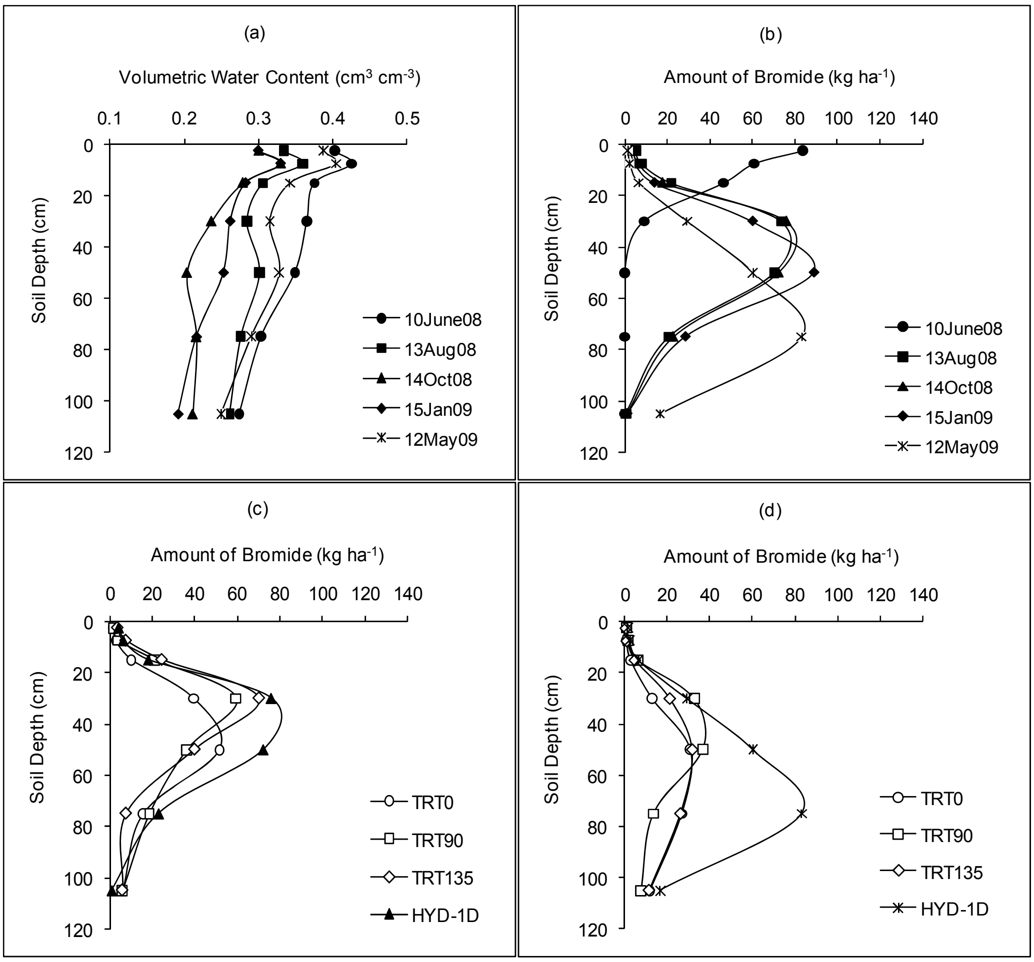


Fig. 4.15. One-dimensional simulation of water flow and solute transport at the upper slope position in Site-2008: (a) HYDRUS-1D estimated volumetric water content at various times of the year; (b) HYDRUS-1D estimated bromide distribution at various times of the year; (c) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 14 Oct. 2008; (d) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 12 May 2009.

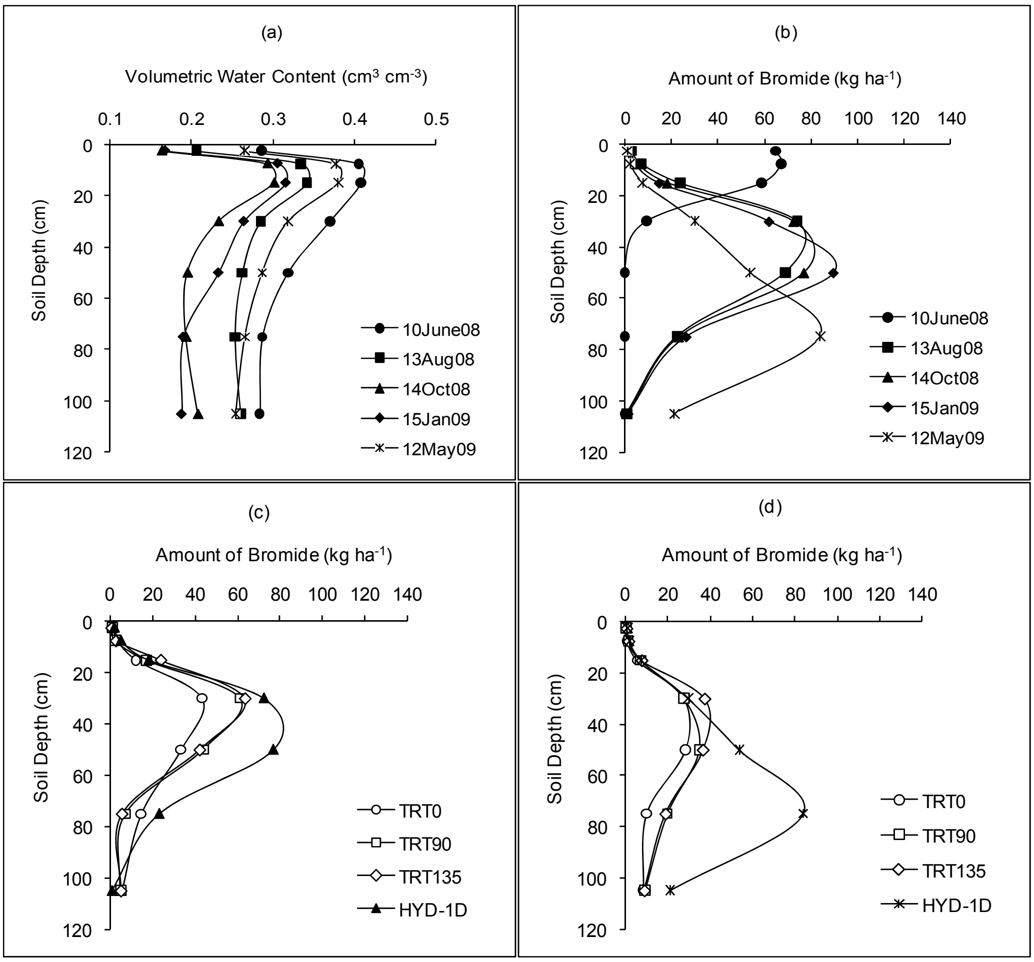


Fig. 4.16. One-dimensional simulation of water flow and solute transport at the middle slope position in Site-2008: (a) HYDRUS-1D estimated volumetric water content at various times of the year; (b) HYDRUS-1D estimated bromide distribution at various times of the year; (c) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 14 Oct. 2008; (d) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 12 May 2009.

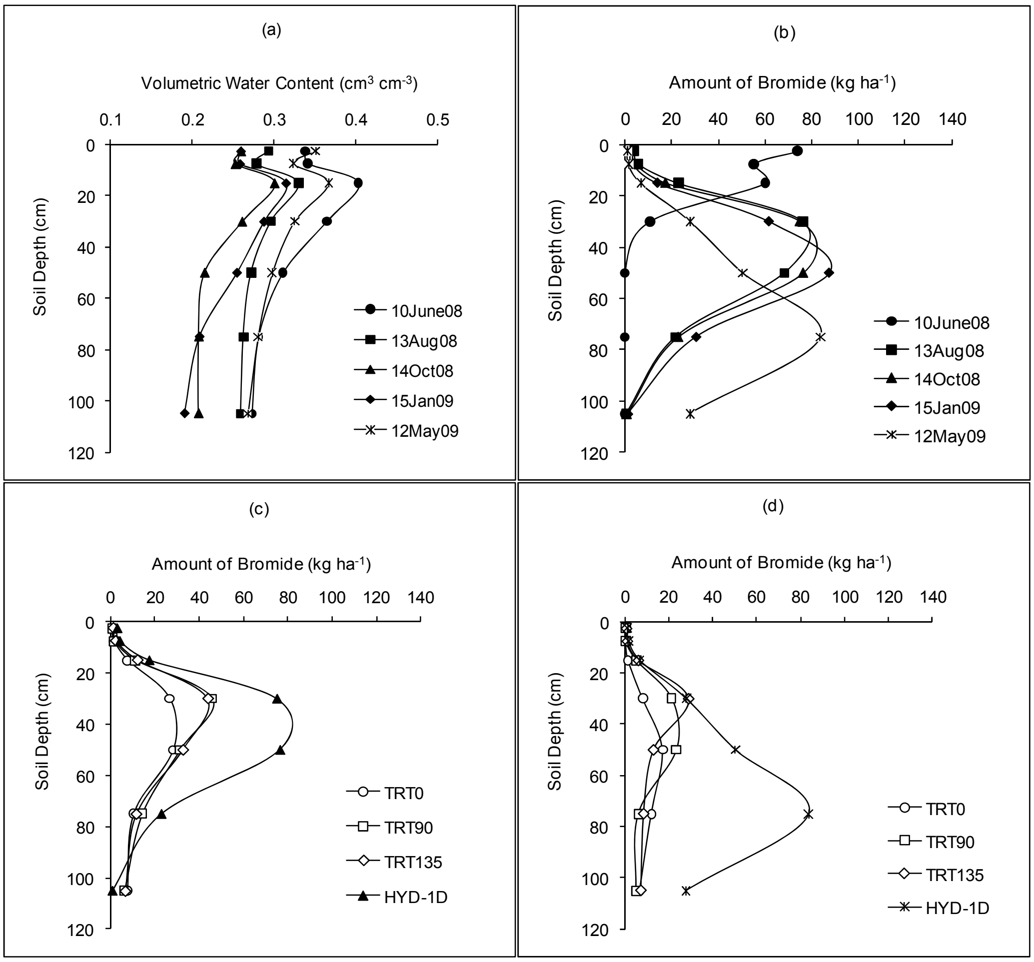


Fig. 4.17. One-dimensional simulation of water flow and solute transport at the lower slope position in Site-2008: (a) HYDRUS-1D estimated volumetric water content at various times of the year; (b) HYDRUS-1D estimated bromide distribution at various times of the year; (c) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 14 Oct. 2008; (d) Comparison between HYDRUS-1D simulated distribution of bromide and field measurements of bromide for different N fertility treatments on 12 May 2009.

to 20-40 cm depth in all slope positions by 24 hours after bromide application in Site-2008.

The simulated centre of mass of bromide was within 20-60 cm depth on all output dates in Site-2008, except on 12 May 2009 (spring 2009 sampling) when the centre of mass had moved to 60-90 cm depth in all landscape positions (Fig. 4.15-4.17). Unlike in Site-2007, the simulated bromide distribution in Site-2008 indicated the potential for solute leaching, as considerable amounts of bromide were estimated to move down to 120 cm depth by spring 2009. The best agreement between the simulated mass of bromide and the measured amounts of bromide was obtained at the UPP and MID slope positions in fall 2008, and also within the top 20 cm depth at the LOW slope in Site-2008.

Model simulation of Site-2008 data indicated that the greater precipitation in 2008 enhanced the downward penetration of solute, hence, a better simulation of bromide distribution, compared to Site-2007. In contrast, the field measurements showed greater amounts of bromide in the soil profile in Site-2008, which was an indication of a smaller movement of bromide, compared to Site-2007. The smaller vertical movement of bromide in Site-2008 in the field measurements was attributed to winter wheat enhancing soil water depletion and reducing bromide transport, as opposed to that of canola in Site-2007. According to the model simulation, the effect of precipitation on solute movement overrides that of crop type. Similar to Site-2007, however, there was no evidence of solute leaching in the one-dimensional simulation of bromide transport in Site-2008.

Estimated mass balances for water and solute at various times in Site-2007 and Site-2008 are shown in Tables 4.6-4.7. The accuracy of the numerical

Table 4.6. Mass balance for one-dimensional water flow and solute transport in Site-2007.

Landscape	Event Date[z]	Ave. θ_s[y] ($cm^3\ cm^{-3}$)	WatBalR[x] (%)	Solute[w] ($kg\ ha^{-1}$)	CncBalR[v] (%)
UPP	6June07	0.361	0.014	200	0.00
	12Aug07	0.277	0.040	200	0.00
	15Oct07	0.246	0.028	200	0.00
	15Jan08	0.252	0.025	200	0.00
	16May08	0.274	0.025	200	0.00
MID	6June07	0.373	0.013	200	0.00
	12Aug07	0.288	0.043	200	0.00
	15Oct07	0.248	0.028	200	0.00
	15Jan08	0.255	0.025	200	0.00
	16May08	0.275	0.025	200	0.00
LOW	6June07	0.359	0.015	200	0.00
	12Aug07	0.282	0.048	200	0.00
	15Oct07	0.241	0.031	200	0.00
	15Jan08	0.249	0.027	200	0.00
	16May08	0.270	0.028	200	0.00

[z]Event Date = the date of field event.
[y]Ave. θ_s = average volumetric water content of the entire flow domain.
[x]WatBalR = the relative errors in water mass balance.
[w]Solute = total mass of solute in the entire flow domain.
[v]CncBalR = the relative errors in solute mass balance.

Table 4.7. Mass balance for one-dimensional water flow and solute transport in Site-2008.

Landscape	Event Date[z]	Ave. θ_s[y] (cm^3 cm^{-3})	WatBalR[x] (%)	Solute[w] (kg ha^{-1})	CncBalR[v] (%)
UPP	10June08	0.356	0.094	200	0.000
	13Aug08	0.303	0.087	200	0.001
	14Oct08	0.254	0.071	200	0.000
	15Jan09	0.261	0.054	200	0.000
	12May09	0.331	0.052	200	0.010
MID	10June08	0.337	0.084	200	0.000
	13Aug08	0.278	0.083	200	0.002
	14Oct08	0.227	0.068	200	0.002
	15Jan09	0.238	0.052	200	0.000
	12May09	0.307	0.050	200	0.019
LOW	10June08	0.330	0.082	200	0.000
	13Aug08	0.285	0.083	200	0.000
	14Oct08	0.244	0.067	200	0.000
	15Jan09	0.254	0.051	200	0.001
	12May09	0.316	0.049	200	0.061

[z]Event Date = the date of field event.
[y]Ave. θ_s = average volumetric water content of the entire flow domain.
[x]WatBalR = the relative errors in water mass balance.
[w]Solute = total mass of solute in the entire flow domain.
[v]CncBalR = the relative errors in solute mass balance.

solution in HYDRUS-1D was evaluated internally using the relative errors in the water mass balance (WatBalR %) and solute mass balance (CncBalR %) for the entire soil profile. The maximum acceptable errors for water flow and solute transport in HYDRUS are 1% and 5%, respectively (Simunek et al. 1998, 2005). The errors shown in the mass balance table indicated that the spatial and temporal discretization selected were appropriate for the simulation runs.

The simulated total mass of bromide in the soil profile was identical to the amount of bromide applied (200 kg ha^{-1}), while the mass of bromide measured in all three fertility treatments in both Site-2007 and Site-2008 ranged from 24 to 80% of the mass of bromide applied (Table 4.8). The simulated mass of bromide also remained the same between fall and spring seasons, and was similar among landscape positions. The cumulative precipitation did not reduce the estimated mass of solute in the soil profile over the period of simulation. Also, differences in estimated K_{sat} among landscape positions were expected to result in differences in solute transport. In contrast, there were no temporal and spatial variations in the total mass of solute in the 1D simulation (Table 4.8).

These results showed that HYDRUS-1D model was unable to reproduce the distribution and mass balance of bromide as measured in the field during the two growing seasons. However, the lack of estimated leaching loss of solute may be due to computational or fundamental error internal to the 1D model. As a result of the inability of HYDRUS-1D to reproduce field data, coupled with the fact that solute transport within the landscape is three-dimensional, the next study was conducted to investigate two-dimensional movement of water and solute in the landscape using HYDRUS-2D/3D model.

Table 4.8. One-dimensional simulated and measured solute mass balance in Site-2007 and Site-2008.

Landscape	HYDRUS-1D (simulated)	TRT0	TRT90 (measured)	TRT135
Site-2007				
Fall 2007		kg ha^{-1}		
UPP	200	109	124	123
MID	200	113	122	135
LOW	200	79.9	87.6	77.4
Spring 2008				
UPP	200	58.4	64.0	57.0
MID	200	57.0	87.4	82.1
LOW	200	61.7	56.3	57.7
Site-2008				
Fall 2008				
UPP	200	127	147	159
MID	200	114	139	145
LOW	200	82.7	110	112
Spring 2009				
UPP	200	87.0	99.4	97.8
MID	200	82.9	102	112
LOW	200	47.0	62.6	65.6

4.4.7 Two-Dimensional Water Flow and Solute Transport in Site-2007

The simulated two-dimensional water flow and bromide movement across the landscape in Site-2007 is presented using contour lines in the x-z plane (Fig. 4.18-4.22). Bromide concentration is presented in the liquid phase in mg cm^{-3} while the water content is expressed in cm^3 cm^{-3}. Figure 4.18 shows the contour lines of estimated water and bromide on Day 38 (6 June 2007), which corresponds to 24 hours after bromide application in Site-2007. The model output for soil water showed that the top 10 cm depth was the driest portion of the soil profile, where the soil water content was 0.34 cm^3 cm^{-3} across the landscape (Fig. 4.18). The simulated water distribution indicated that lateral flow was in the downslope direction, reflecting slope effects on soil water flow. The contour lines for the simulated bromide showed that bromide had moved to 10-20 cm depth by 24 hours after a precipitation event of 0.83 cm (Fig. 4.18).

At harvest, on Day 104 (Fig. 4.19), the model indicated that the wettest region of the soil profile was within the 20-60 cm depth. After 11.2 cm of cumulative precipitation from the day of solute application (Day 37), simulated bromide had penetrated all depths in all three landscape positions on Day 104. However, the simulated bromide concentration peak remained within 20 to 40 cm depth on this day (Fig. 4.19). On Day 168, which corresponds to fall sampling date, simulated water was uniformly distributed within the soil profile and across the landscape (Fig. 4.20). At this time, cumulative precipitation from Day 37 was 19.6 cm, which subsequently moved the bromide farther downward in all slope positions (Fig. 4.20).

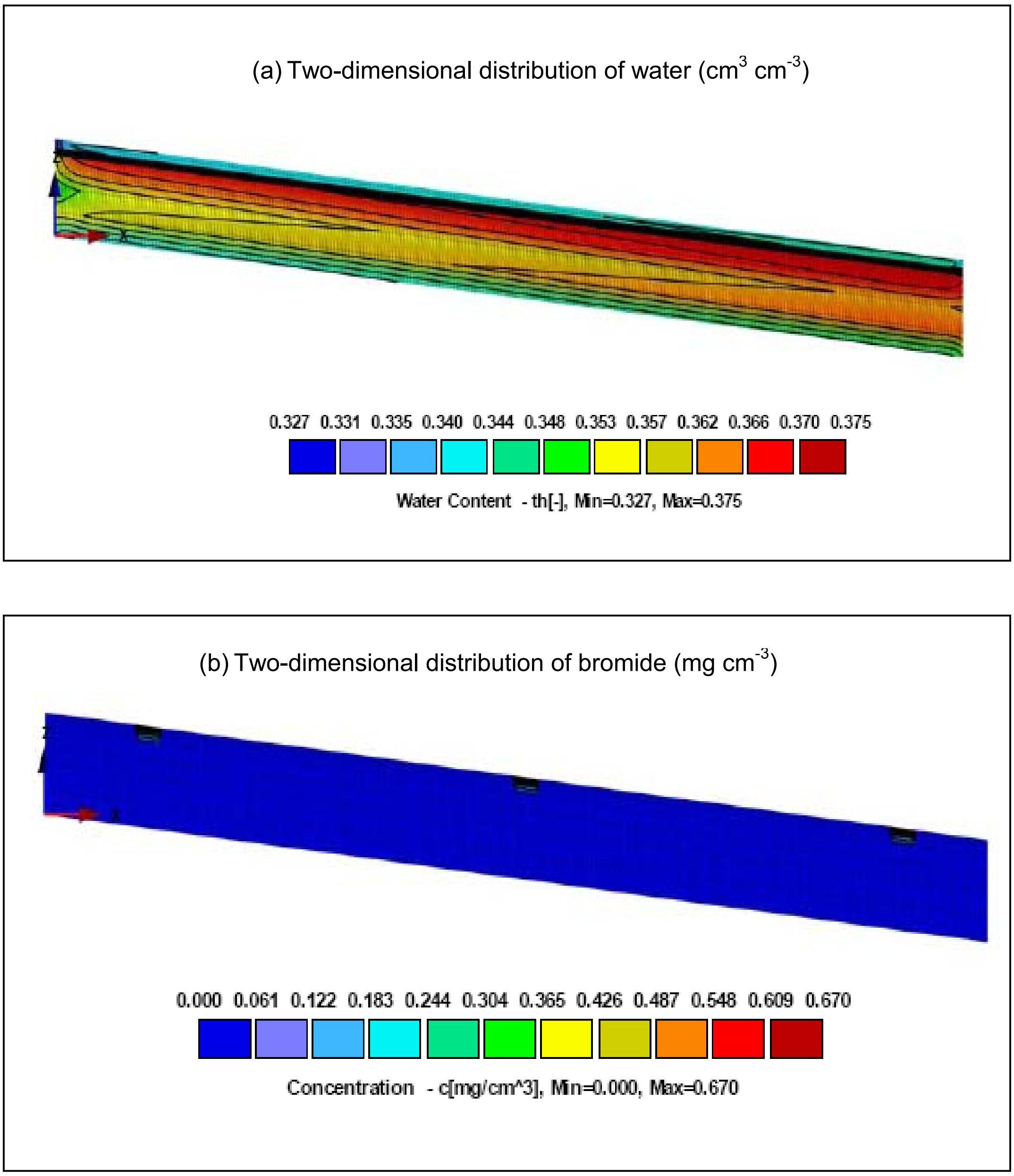
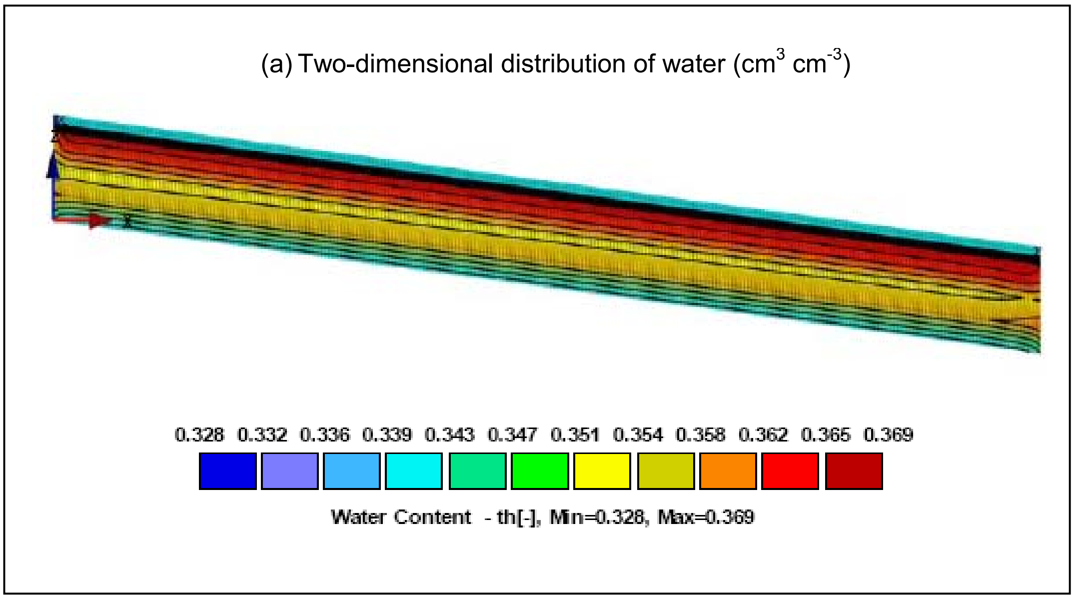


Fig. 4.18. Contours of HYDRUS-2D/3D estimated: (a) Water flow; (b) Solute transport on Day 38 (6 June 2007) in Site-2007.



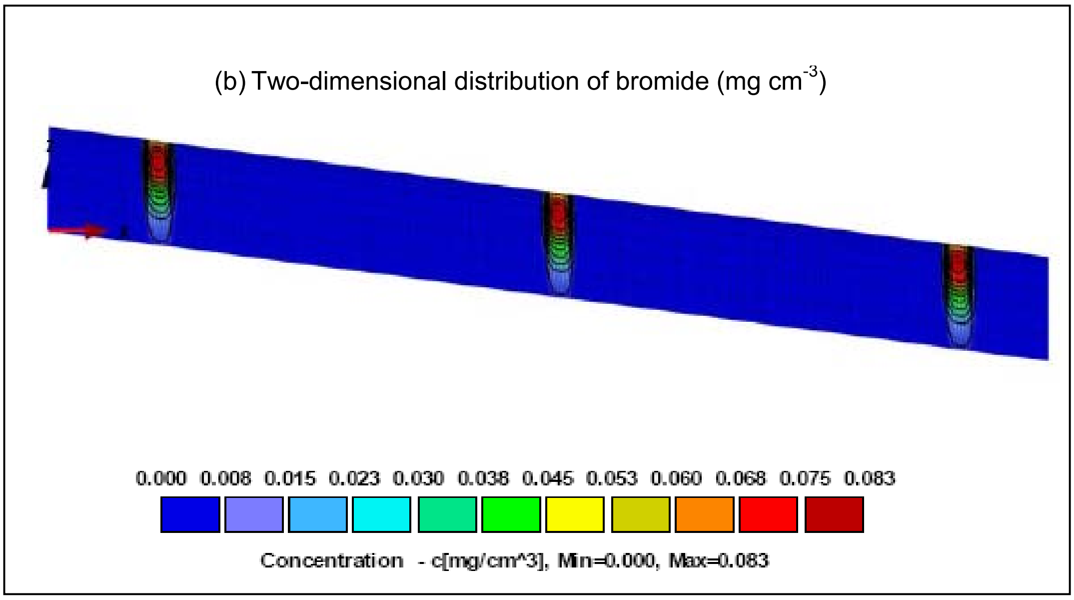


Fig. 4.19. Contours of HYDRUS-2D/3D estimated: (a) Water flow; (b) Solute transport on Day 104 (12 August 2007) in Site-2007.

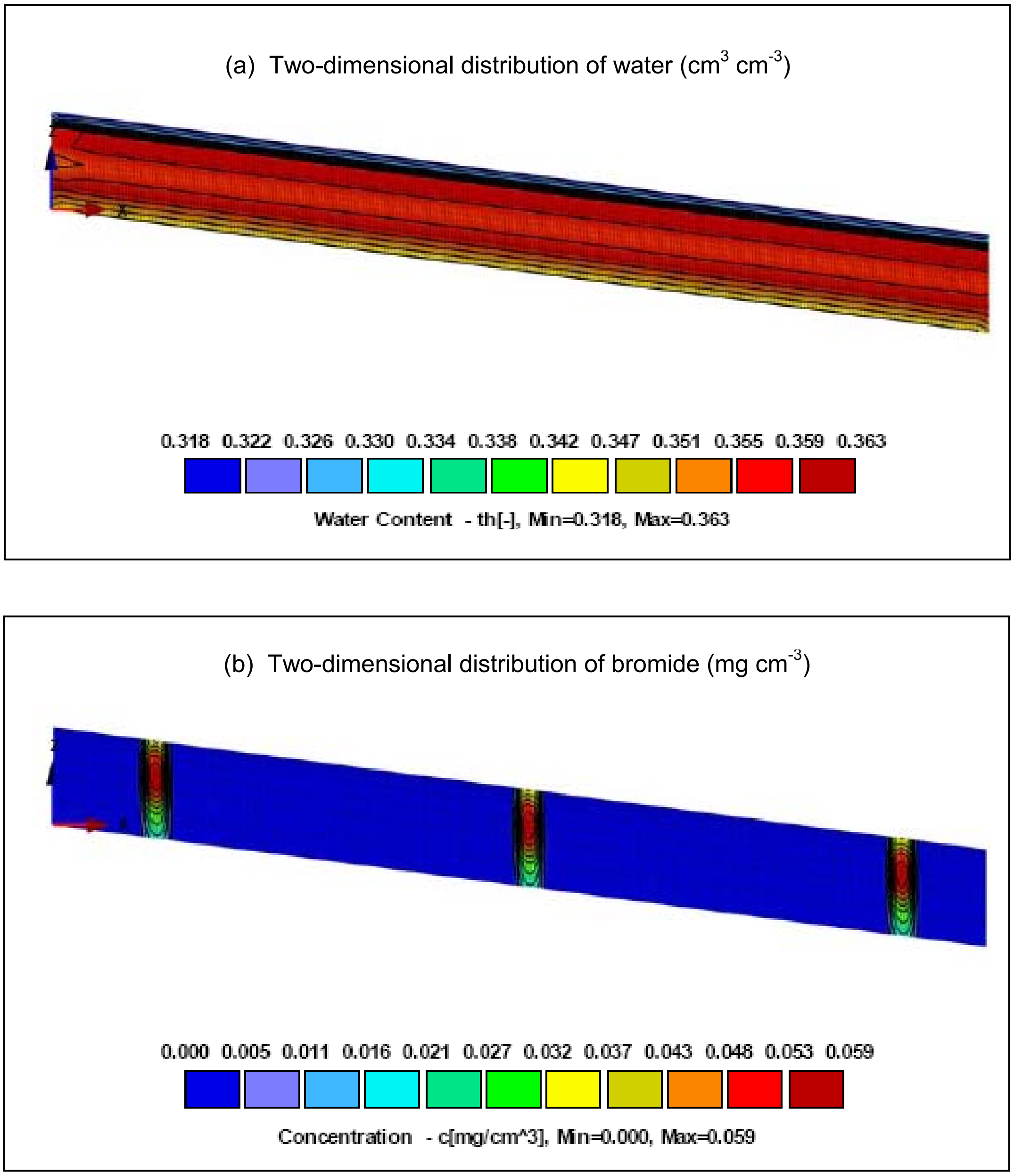


Fig. 4.20. Contours of HYDRUS-2D/3D estimated: (a) Water flow; (b) Solute transport on Day 168 (15 October 2007) in Site-2007.

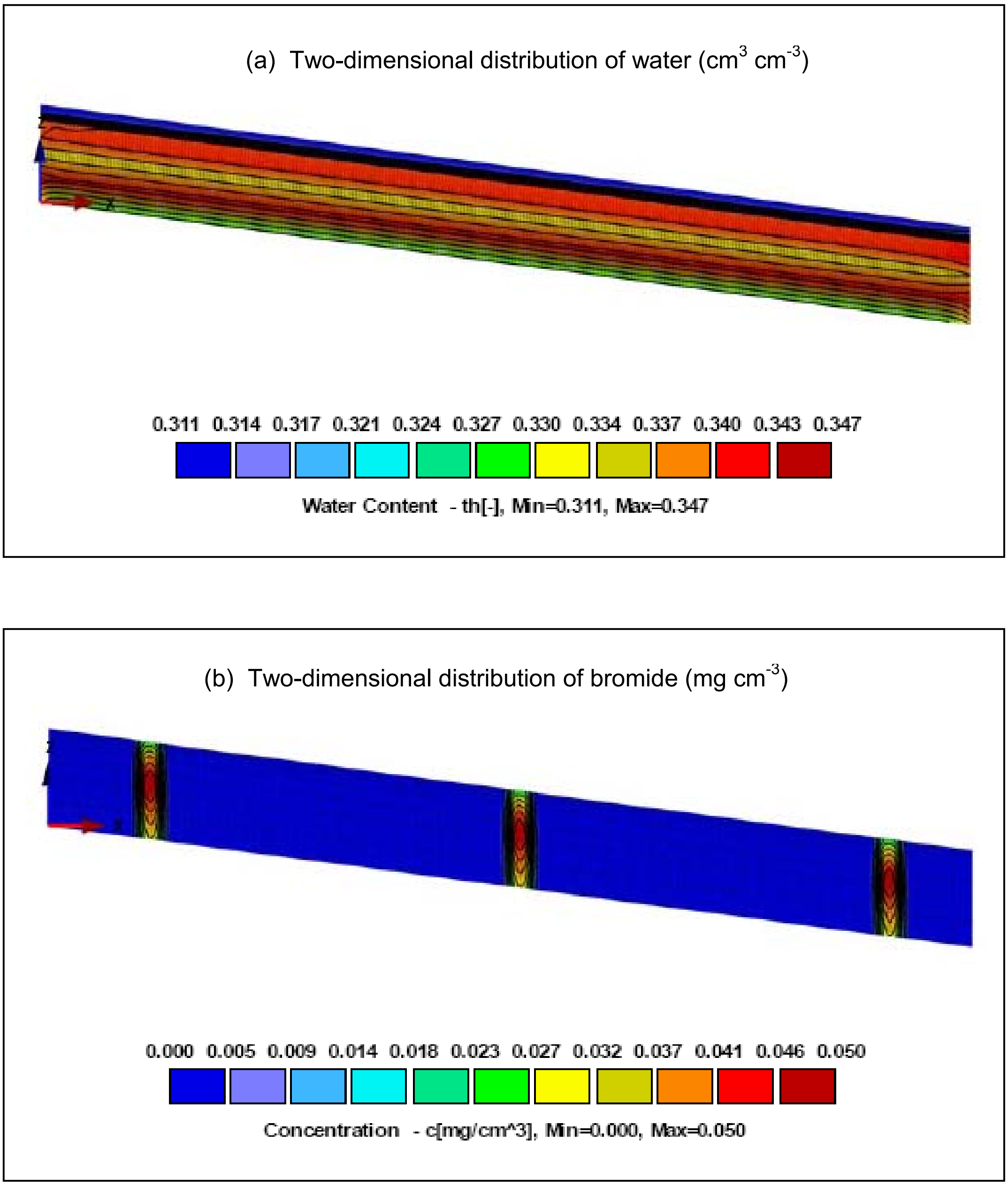


Fig. 4.21. Contours of HYDRUS-2D/3D estimated: (a) Water flow; (b) Solute transport on Day 260 (15 January 2008) in Site-2007.

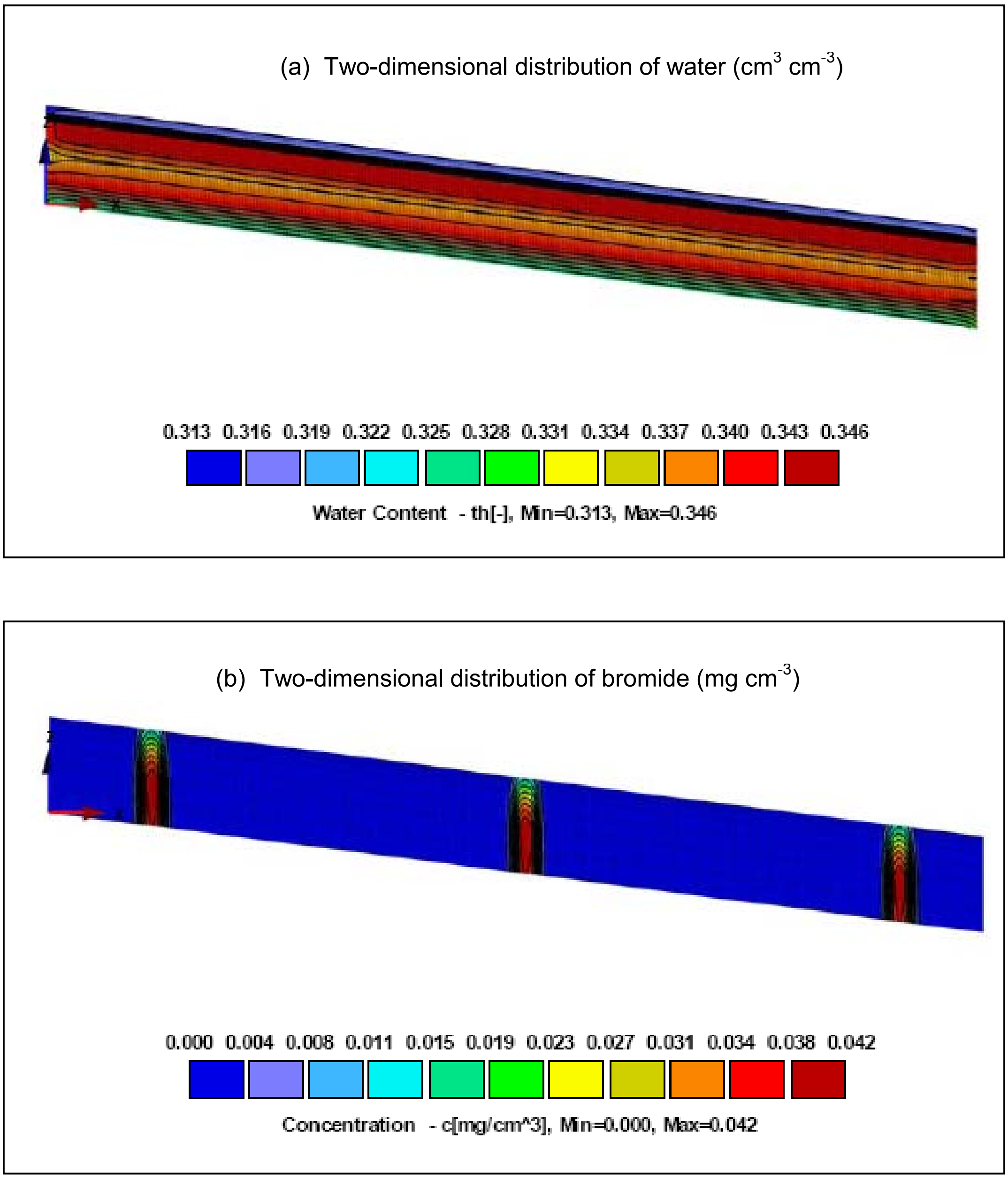


Fig. 4.22. Contours of HYDRUS-2D/3D estimated: (a) Water flow; (b) Solute transport on Day 382 (16 May 2008) in Site-2007.

According to the HYDRUS-2D/3D model, bromide travelled further downward with increases in the cumulative precipitation during the winter period as shown on Day 260 (Fig. 4.21). The final model output in Site-2007 was on Day 382 (Fig. 4.22), which corresponded to the spring sampling in Site-2007. At this time, the cumulative precipitation from the day of bromide application was 30.4 cm. As the landscape became drier with time, lateral flow of water in the soil profile diminished accordingly. On Day 382, the simulated bromide concentration peak was within 60 to 120 cm depth in all slope positions, indicating that considerable amounts of bromide had leached below the 120 cm depth by spring of 2008 (Fig. 4.22).

The 2D model was able to simulate temporal variations in bromide movement in Site-2007 and subsequent reduction in bromide concentration within the soil profile due to leaching losses. These results showed that the HYDRUS-2D/3D model performed better than HYDRUS-1D in simulating the distribution of bromide in the landscape.

4.4.8 Two-Dimensional Water Flow and Solute Transport in Site-2008

The distribution of water and solute within the soil in Site-2008 are presented in Figures 4.23-4.27. On Day 41 (10 June 2008), 24 hours after bromide application, the simulated bromide had moved down to 20-40 cm depth (Fig. 4.23). With total precipitation of 18.8 cm between Day 40 (solute application in Site-2008) and Day 105 (harvest), the simulated bromide concentration peak was within 40 to 90 cm depth on Day 105 (Fig. 4.24).

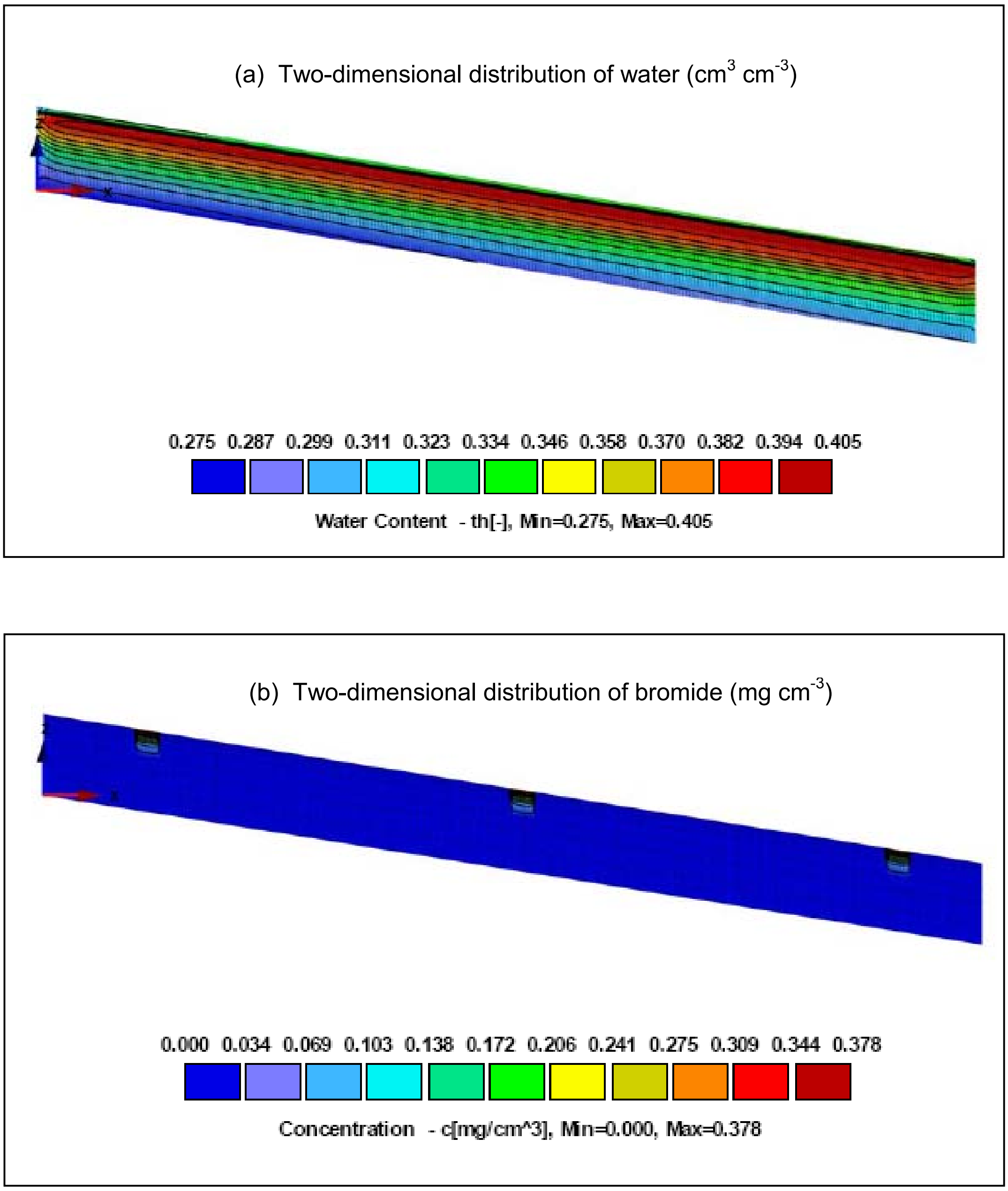
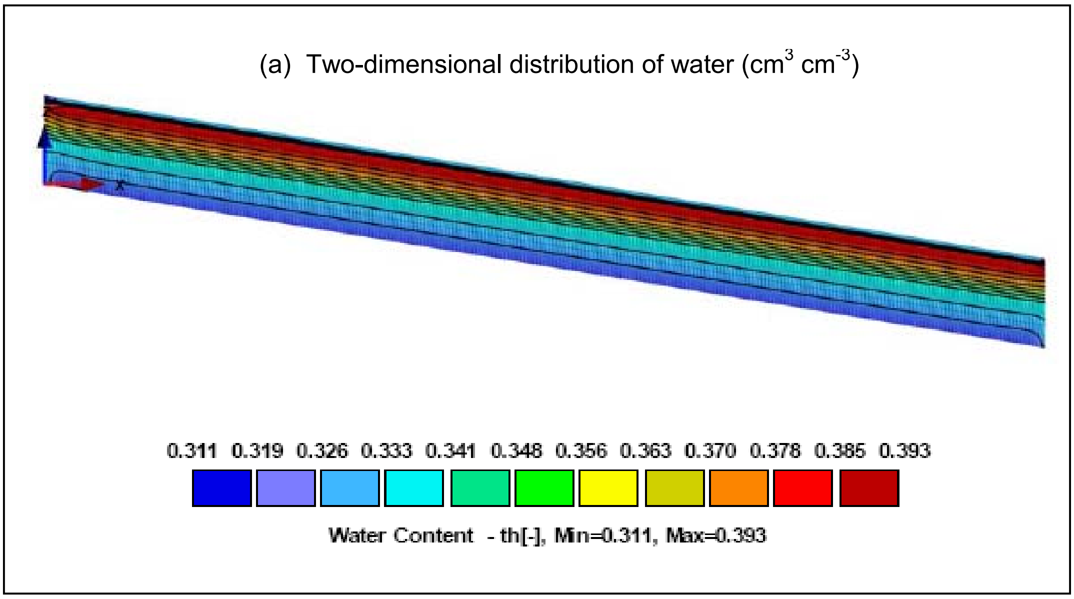


Fig. 4.23. Contours of HYDRUS-2D/3D estimated: (a) Water flow; (b) Solute transport on Day 41 (10 June 2008) in Site-2008.



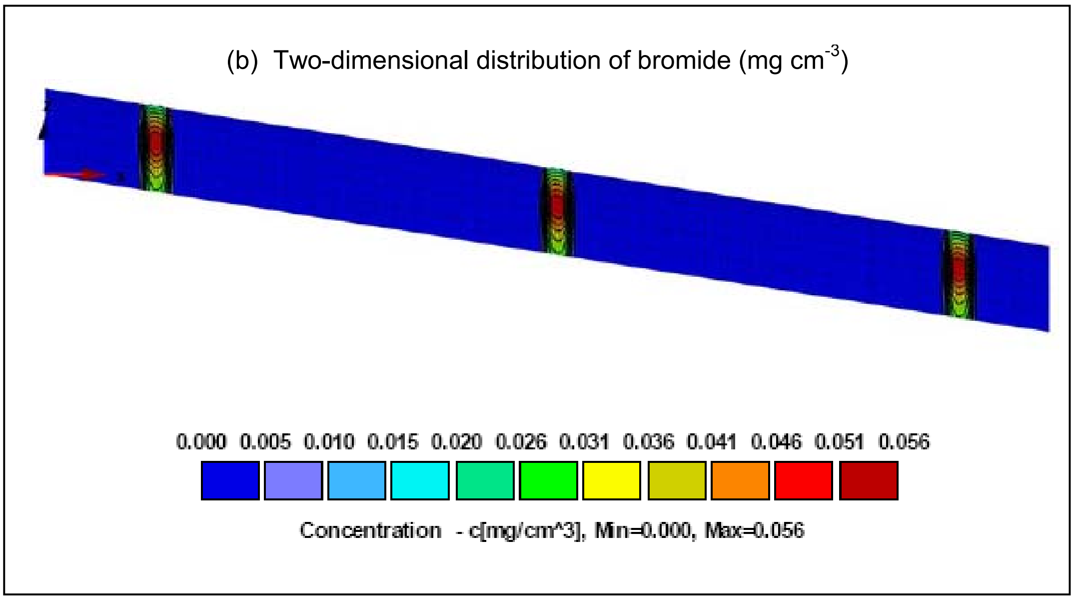


Fig. 4.24. Contours of HYDRUS-2D/3D estimated: (a) Water flow; (b) Solute transport on Day 105 (13 August 2008) in Site-2008.

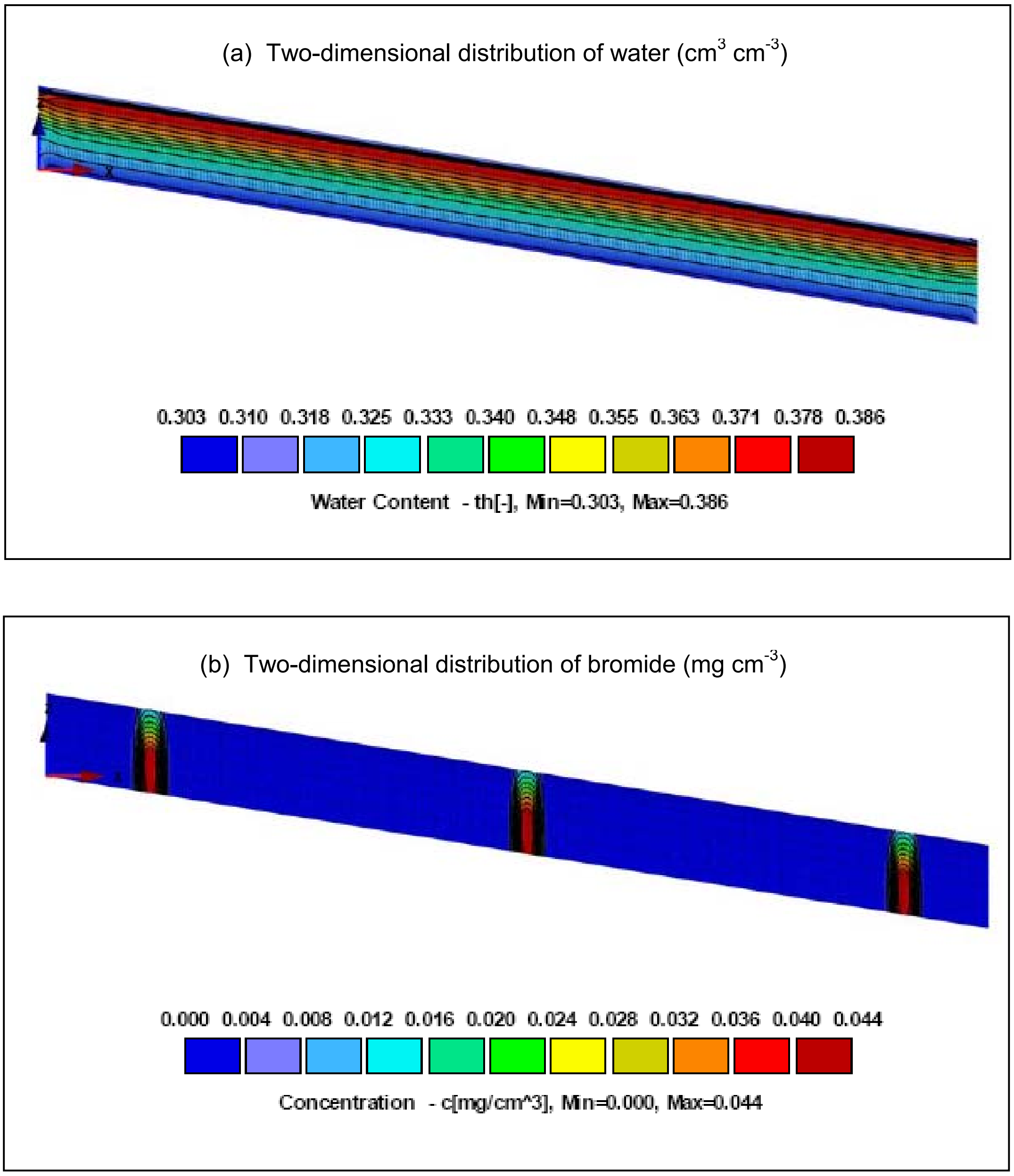
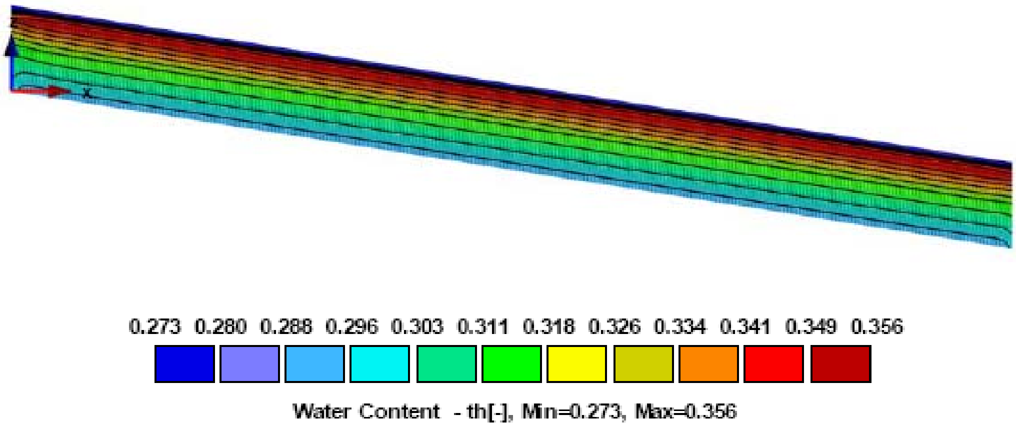


Fig. 4.25. Contours of HYDRUS-2D/3D estimated: (a) Water flow; (b) Solute transport on Day 167 (14 October 2008) in Site-2008.

(a) Two-dimensional distribution of water ($cm^3\ cm^{-3}$)



(b) Two-dimensional distribution of bromide ($mg\ cm^{-3}$)

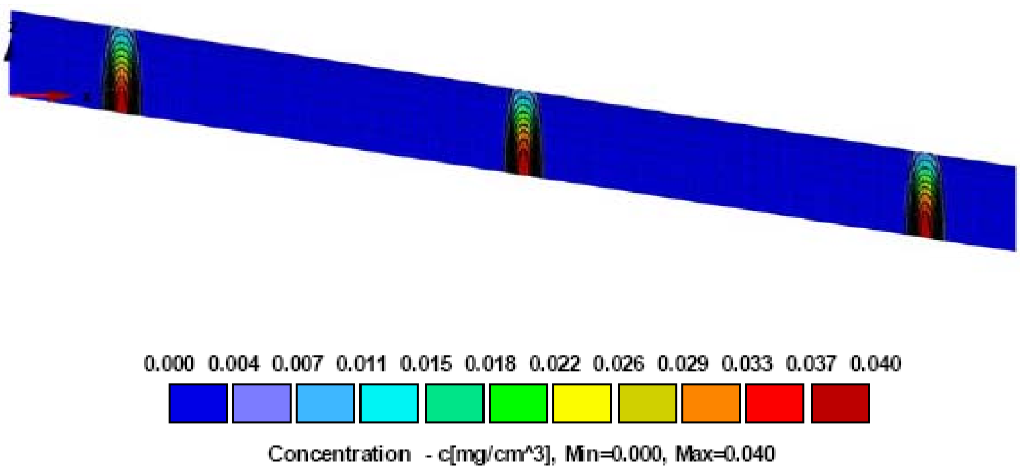
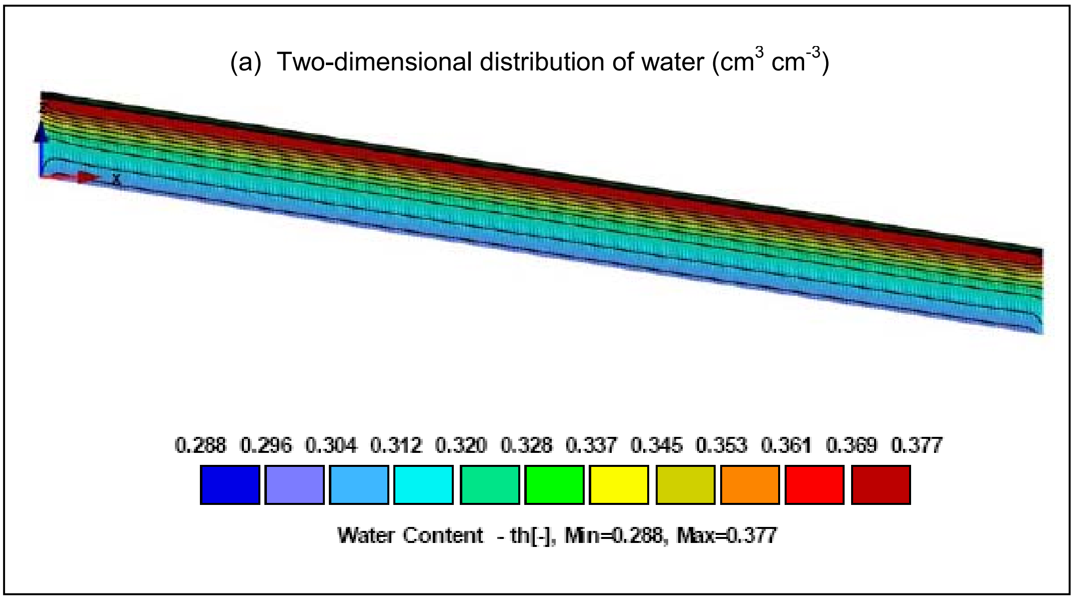


Fig. 4.26. Contours of HYDRUS-2D/3D estimated: (a) Water flow; (b) Solute transport on Day 260 (15 January 2009) in Site-2008.



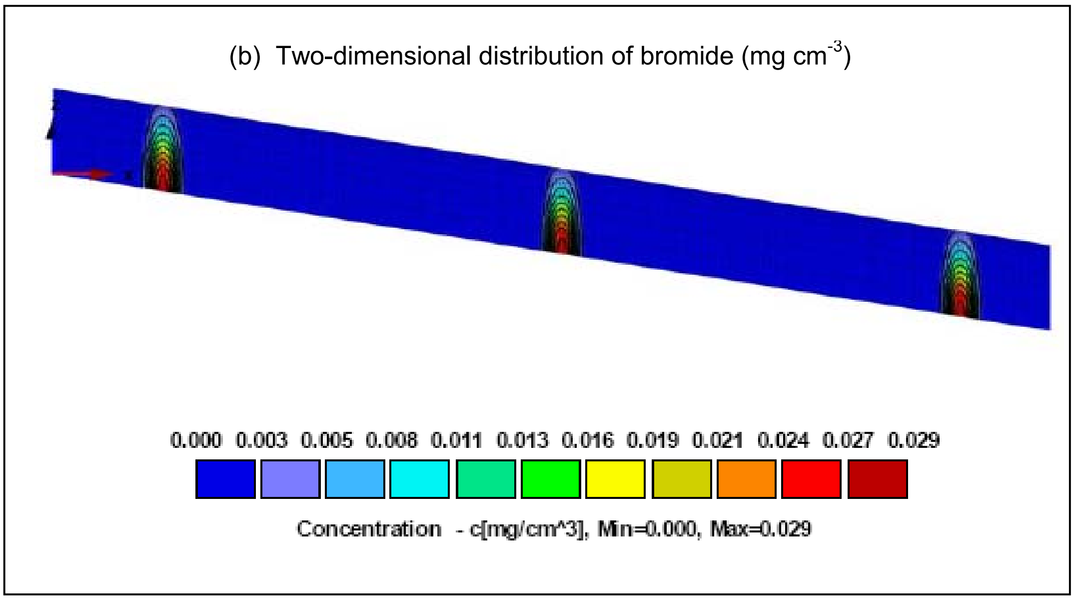


Fig. 4.27. Contours of HYDRUS-2D/3D estimated: (a) Water flow; (b) Solute transport on Day 377 (12 May 2009) in Site-2008.

The depth of bromide penetration increased with time as the bulk of bromide mass was between 60 cm and below the 120 cm depth on Day 167, which corresponds to the fall sampling in Site-2008 (Fig. 4.25). By 15th of January 2009 (Day 260), the simulated bromide had moved farther down below the 120 cm depth (Fig. 4.26). At the end of simulation on Day 377 (spring sampling), the total precipitation from Day 40 was 45.9 cm. According to the model, bromide had leached below the 120 cm depth to a large extent by Day 377 in Site-2008 (Fig. 4.27).

4.4.9 Mass Balance of Two-Dimensional Transport of Bromide in Site-2007 and Site-2008

The mass balance of water and the solute remaining in each sub-region across the landscape in Site-2007 is illustrated in Table 4.9. The 2D model estimated that the mass of bromide declined with time, unlike in the 1D model where the amount of bromide in the soil profile was time invariant. However, there was no significant effect of landscape on solute leaching as the total mass of bromide was the same among landscape positions (Table 4.9).

The model's simulation of bromide transport and subsequent leaching loss in Site-2008 was clearly illustrated in the solute mass balance (Table 4.10). The model estimated that 72% and 35% of the bromide applied remained in the soil profile by Day 167 (fall 2008) and Day 377 (spring 2009), respectively. Estimated amounts of bromide in the soil profile in Site-2008 were generally smaller than in Site-2007. The greater loss of solute in Site-2008 compared to Site-2007 was attributed to differences in precipitation and slope gradient between the two

Table 4.9. Mass balance for two-dimensional water flow and solute transport in Site-2007.

Landscape	Event Date[z]	Ave. θ_s[y] ($cm^3\ cm^{-3}$)	WatBalR[x] (%)	Solute[w] ($kg\ ha^{-1}$)	CncBalR[v] (%)
UPP	6June07	0.356	0.783	184	0.931
	12Aug07	0.354	0.675	190	2.86
	15Oct07	0.353	0.728	184	3.42
	15Jan08	0.336	0.683	169	3.07
	16May08	0.337	0.671	149	2.58
MID	6June07	0.359	0.783	184	0.931
	12Aug07	0.355	0.675	190	2.86
	15Oct07	0.354	0.728	184	3.42
	15Jan08	0.336	0.683	169	3.07
	16May08	0.337	0.671	148	2.58
LOW	6June07	0.361	0.783	184	0.931
	12Aug07	0.355	0.675	190	2.86
	15Oct07	0.354	0.728	183	3.42
	15Jan08	0.336	0.683	169	3.07
	16May08	0.337	0.671	148	2.58

[z]Event Date = the date of field event.
[y]Ave. θ_s = average volumetric water content of the entire flow domain.
[x]WatBalR = the relative errors in water mass balance.
[w]Solute = total mass of solute in the entire flow domain.
[v]CncBalR = the relative errors in solute mass balance.

Table 4.10. Mass balance for two-dimensional water flow and solute transport in Site-2008.

Landscape	Event Date[z]	Ave. θ_s[y] (cm^3 cm^{-3})	WatBalR[x] (%)	Solute[w] (kg ha^{-1})	CncBalR[v] (%)	2007Solute[u] (kg ha^{-1})
UPP	10June08	0.337	0.499	200	0.340	185
	13Aug08	0.349	0.296	180	0.797	187
	14Oct08	0.341	0.310	145	1.25	176
	15Jan09	0.318	0.305	108	1.09	158
	12May09	0.333	0.357	69.3	0.699	137
MID	10June08	0.343	0.499	200	0.340	185
	13Aug08	0.349	0.296	179	0.797	187
	14Oct08	0.341	0.310	144	1.25	176
	15Jan09	0.318	0.305	107	1.09	158
	12May09	0.333	0.357	68.9	0.699	137
LOW	10June08	0.347	0.499	200	0.340	186
	13Aug08	0.349	0.296	179	0.797	187
	14Oct08	0.341	0.310	144	1.25	176
	15Jan09	0.318	0.305	107	1.09	158
	12May09	0.333	0.357	68.6	0.699	136

[z]Event Date = the date of field event.
[y]Ave. θ_s = average volumetric water content of the entire flow domain.
[x]WatBalR = the relative errors in water mass balance.
[w]Solute = total mass of solute in the entire flow domain.
[v]CncBalR = the relative errors in solute mass balance.
[u]2007Solute = total mass of solute estimated for Site-2008 based on Site-2008 receiving precipitation similar to Site-2007.

years. The total precipitation received in 2008-2009 (Site-2008) was greater than that in 2007-2008 (Site-2007) by 12.3 cm. Also, the slope gradient in Site-2008 plot was 3.2°, while that in Site-2007 was 1.7°.

To estimate the effect of slope gradient on bromide leaching, the mass balance of bromide in Site-2008 landscape was calculated using the precipitation data for Site-2007. The results showed that even if precipitation in Site-2008 had been similar to that in Site-2007, amounts of solute remaining in the Site-2008 landscape (Table 4.10) would have been smaller than in Site-2007 (Table 4.9), particularly between fall and spring season. These results confirmed that the greater precipitation and larger slope steepness in Site-2008 both enhanced bromide transport in the model simulation in Site-2008 compared to Site-2007.

The simulation results showed that bromide distribution in both site-years responded more to the vertical movement than to lateral spread (Russo et al. 2005). These results were similar to the measurements obtained from the field which showed that 50-60% of added bromide was recovered vertically within 120 cm depth compared to 10-20% recovered within 200 cm away laterally from the zone of application. Therefore, the 2D simulation and the field study both suggest that the vertical transport of solute in the hummocky landscape was more important than the lateral movement. These findings further demonstrate the potential risk of nitrate leaching and the subsequent contamination of groundwater, particularly when precipitation is above normal in the sub-humid region of the Canadian prairie.

In Site-2007, the model overestimated the mass of bromide remaining in the soil, compared to the amounts measured in the fertility treatments. The simulated total mass of bromide in the fall and spring seasons in Site-2007 ranged from 74 to 92% of the bromide applied, compared to 34-72% in Site-2008 (Table 4.11). However, the total mass of bromide measured in the vertical and lateral components of the landscape was 35-85% of the bromide applied for Site-2007 and 30-90% for Site-2008. In the fall of 2007, there were better agreements between the model simulation and the mass of bromide measured in TRT90 and TRT135, compared to TRT0. The greater loss of bromide in the unfertilized treatment (TRT0) possibly resulted in greater discrepancies in mass of bromide between the model simulation and TRT0. By the spring of 2008, the simulated mass of bromide had declined by 20% of that in fall 2007 while the bromide loss measured in the fertility treatments was 44%.

The results also showed that HYDRUS-2D/3D did not take into consideration the “Campbell hypothesis” (Campbell et al. 1984; 1993), which states that nitrogen fertilization reduces nitrate leaching due to improved crop water uptake. This hypothesis was independently verified using bromide tracer in the field study. Therefore, the effect of N fertility on crop growth, water and nutrient uptake, and the resultant effect on solute leaching should be incorporated into numerical models such as the HYDRUS program.

Unlike in fall 2007, TRT0 had the best agreement with the model simulation in fall 2008 compared to TRT90 and TRT135 (Table 4.11). However, the mass of bromide measured at the LOW slope position in TRT90 and TRT135 was similar to the model simulation. In the spring of 2009, the simulated mass of bromide had declined by approximately 50% of that in fall 2008, compared to 35% in the fertility treatments. Overall, there was a better agreement between the

Table 4.11. Two-dimensional simulated and measured solute mass balance in Site-2007 and Site-2008.

Landscape	HYDRUS-2D/3D (simulated)	TRT0[z]	TRT90[y] (measured)	TRT135[x]
Site-2007				
Fall 2007		kg ha^{-1}		
UPP	184	149	170	167
MID	184	145	152	166
LOW	183	119	137	121
Spring 2008				
UPP	149	69.0	80.0	70.3
MID	148	76.9	103	99.7
LOW	148	71.8	85.9	77.5
Site-2008				
Fall 2008				
UPP	145	143	173	180
MID	144	126	152	158
LOW	144	106	144	140
Spring 2009				
UPP	69.3	96.9	112	109
MID	68.9	93.0	114	123
LOW	68.6	58.9	78.4	79.4

[z,y,x]Data represent both vertical and lateral bromide that was measured in the soil.
[x]The lateral bromide in TRT135 was the average of TRT0 and TRT90.

simulated mass of bromide and those in the fertility treatments in spring 2009, compared to the model simulation in spring 2008.

In the field, the absence of crop growth between fall and spring season resulted in more water in the soil profile, thereby enhancing leaching and reducing the amount of bromide measured in the fertility treatments. In contrast, the model simulation showed that there was root water uptake after harvest in both site-years. However, the model predicted overwinter losses of bromide that was greater than those measured. Therefore, the greater reduction in the simulated mass of bromide by spring 2009 compared to spring 2008 was due to the greater precipitation in Site-2008 and the model's estimate of a larger amount of infiltration between fall 2008 and spring 2009. The precipitation between fall 2008 and spring 2009 in Site-2008 was 17.1 cm, while the precipitation between fall 2007 and spring 2008 in Site-2007 was 10.5 cm. Intuitively, the wetter the soil profile the greater the extent of solute leaching.

In general, the discrepancies in bromide recovery between the model simulation and field data by the spring season may be attributed to differences in the form and timing of precipitation events. Due to freezing condition, it is expected that bromide transport in the field during the snowfall would cease temporarily until spring snowmelt. In contrast, the HYDRUS model considered snowfall precipitation as rainfall. Allowing snow to be treated as rain in the model simulation resulted in continuous movement of bromide, hence, a greater loss of bromide by the spring season compared to the field data. However, the mass of bromide recovered by the spring season in the model simulation was smaller than in the field study in spring 2009 only.

4.4.10 Factors affecting Water Flow and Solute Transport in the Landscape

Russo et al. (2005) identified five factors controlling water flow and solute movement on a hillslope namely, the saturated hydraulic conductivity, the parameters governing the retentivity function, the slope of the terrain, initial and boundary conditions, and water uptake by plant roots.

In the present study, the average soil hydraulic properties of all three landscape positions were used to compute the water flow parameters for each of the seven soil layers in the 2D domain (Fig. 4.3a). The soil hydraulic properties were computed into seven layers in order to reproduce spatial heterogeneity in the soil profile. An alternative scheme to prescribe the water flow parameters in the 2D domain would be to specify the average soil hydraulic properties in the soil profile for each landscape sub-region (Fig. 4.3b). Since the mean of estimated K_{sat} in the soil profile was similar for the two site-years (Tables 4.1-4.2), this suggests that the differences in the estimated solute movement between Site-2007 and Site-2008 were not due to variability in soil hydraulic properties.

In the field study, an important difference between the two site-years was the crop type. The plot in Site-2007 had canola, a spring crop, while the crop in Site-2008 was winter wheat. The early season soil water consumption by winter wheat probably reduced the impact of precipitation on bromide movement in Site-2008. However, the effect of a fall-planted crop such as winter wheat on bromide transport was not properly taken into consideration in the HYDRUS-1D and -2D models. Unlike the field situation, the model probably allowed the large precipitation in Site-2008 to override effects of winter wheat’s early season root water uptake on water flow and bromide transport. The results also suggest that the crop growth parameters specified for winter wheat in the model requires further modifications to better reflect the differences in water uptake between a winter and a spring crop.

The HYDRUS-2D/3D model showed evidence of lateral flow of water in the downslope direction, particularly in the early part of 2007 (Fig. 4.18). However, the trend in lateral flow disappeared as the soil profile became drier with time (Table 4.9). Overall, the potential for vertical transport of bromide in the landscape was greater than the lateral spread, as bromide loss occurred by downward movement in both site-years.

According to the model computation of atmospheric boundary condition, surface ponding is a prerequisite for overland flow while subsurface lateral flow is enhanced by large amounts of water in the soil profile. Based on the model simulation, it appears that the amount of precipitation received at this site was not sufficient to result in saturated soil profile that can enhance lateral flow in the downslope direction. This may also explain the lack of significant variability in bromide mass balance among landscape positions. This is unlike in the field study where the greatest leaching loss of bromide and nitrate was at the lower slope, compared to other landscape positions.

The greatest solute loss measured at the lower slope position in the field study was attributed to the greatest depth of A horizon and K_{sat} compared to other landscape positions. To verify the impacts of K_{sat} on solute transport in the 2D model, the soil material distribution in the flow domain was compartmentalized into three uniform vertical regions (Fig. 4.3b) as opposed to the seven layers of soil hydraulic properties (Fig. 4.3a). The average soil hydraulic parameter in the soil profile was specified at each landscape position. However, the simulated outputs for water flow and bromide transport were similar (data not shown) to those observed when the seven layers of soil hydraulic parameters were used across the landscape. These results suggest that the 2D model was less sensitive to variability in K_{sat} among landscape positions, while bromide transport responded more to precipitation during the model simulation.

4.4.11 Differences in Bromide Transport between HYDRUS-1D and -2D/3D

Although model computations, variable boundary conditions, and flow and transport parameters were similar for HYDRUS-1D and HYDRUS-2D/3D, there were discrepancies in bromide transport between the two models. While there was no solute leaching in HYDRUS-1D, HYDRUS-2D/3D reproduced the measured temporal reduction in bromide concentration in the soil profile.

In spite of estimated cumulative outflow of 10 cm of water in the HYDRUS-1D model, the solute did not move out of the soil profile in the model simulation. The cumulative bottom flux is simply the total amount of water leached from the soil profile during the simulation period. However, HYDRUS-1D showed that water outflow from the soil profile occurred at certain periods which did not coincide with the solute movement. According to the estimated water flux in Site-2007 (Fig. 4.6) and Site-2008 (Fig. 4.8) using HYDRUS-1D, the maximum bottom flux occurred within the first 60 days while the water outflow declined to zero by Day 150. Note that bromide was surface applied with the infiltrating water on Day 37 and Day 40 in 2007 and 2008, respectively. As such, the termination of bottom flux by Day 150 indicated that bromide movement would be restricted within the soil profile, hence, no leaching. This infers that the impact of episodic bottom flux on solute movement is more important than that of the cumulative bottom flux.

Unlike in HYDRUS-1D, there was free drainage of water in the HYDRUS-2D/3D model throughout the period of simulation (Fig. 4.28-4.29). Note that the unit for the estimated free drainage (cm^2 d^{-1}) in HYDRUS-2D/3D indicated a two-dimensional flow domain, as opposed to cm d^{-1} for the estimated bottom flux in HYDRUS-1D. The differences in water leaching between the two models may be attributed to the greater capacity of the 2D model to handle transient flow and heterogeneity in the flow domain due to the presence of triangular FE mesh, compared to the nodal grid mesh in the 1D model.

It is also important to note that the scale and dimension of flow domain in HYDRUS-2D/3D were closer to the three-dimensional field scenario, as opposed to the simple one-dimensional profile in HYDRUS-1D. This may enable the 2D domain to reproduce the system-dependent boundary conditions at the soil-air interface, such as temporal changes in soil water flux due to net infiltration or evaporation, better than the 1D model. However, if there is a large precipitation over the entire period of simulation, the discrepancies in soil water flux between HYDRUS-1D and HYDRUS-2D/3D may be eliminated. The subsequent net infiltration, barring root water uptake, would ultimately promote outflow from the soil profile.

When the daily precipitation was increased by two-fold in HYDRUS-1D, the mass of bromide remaining on the last day of simulation was 40% of the amount added. However, this was not a realistic amount of precipitation for the

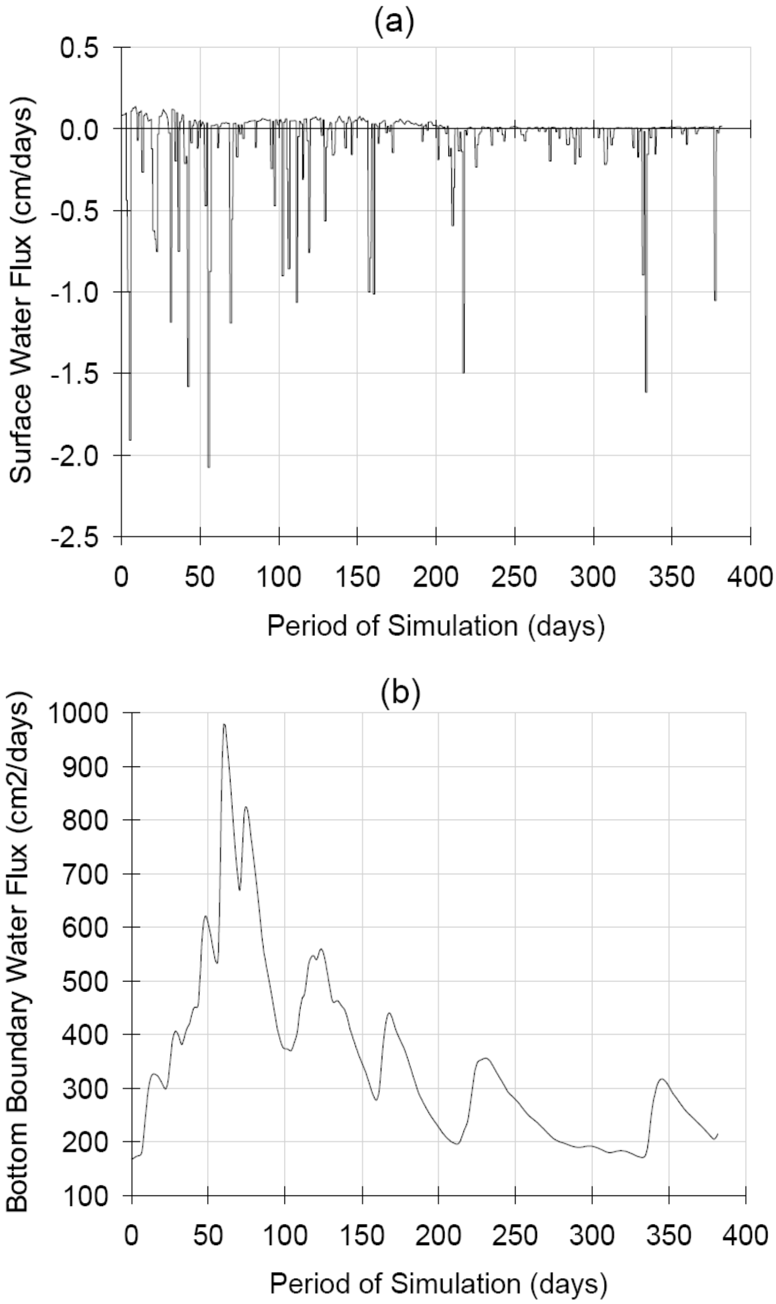


Fig. 4.28. Estimated two-dimensional soil water fluxes in Site-2007 using HYDRUS- 2D/3D: (a) Surface flux of water at the atmospheric boundary; (b) Free drainage flux of water at the bottom boundary.

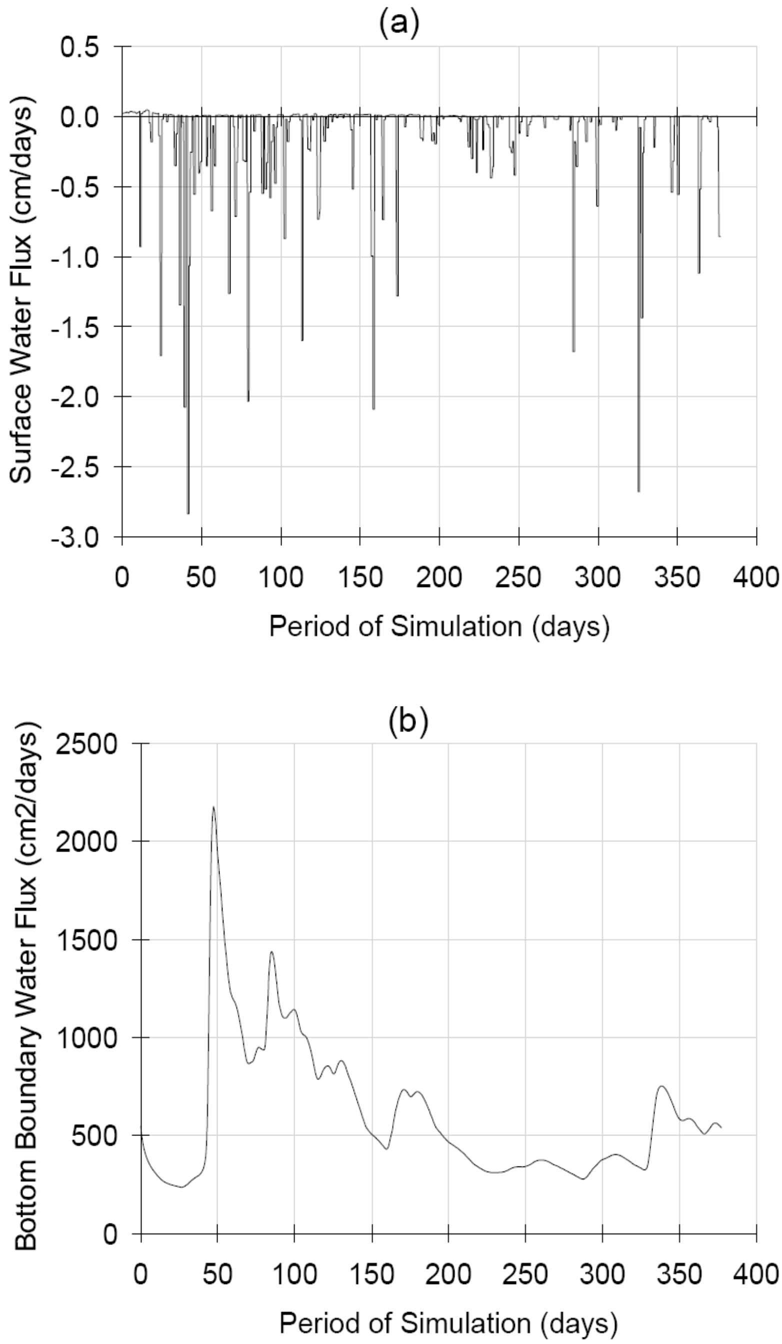


Fig. 4.29. Estimated two-dimensional soil water fluxes in Site-2008 using HYDRUS- 2D/3D: (a) Surface flux of water at the atmospheric boundary; (b) Free drainage flux of water at the bottom boundary.

study area. The cumulative precipitation in Site-2007 was 38.7 cm, and that in Site-2008 was 50.9 cm. Based on long-term records, the total annual precipitation in the study area is 45.9 cm, in which rainfall accounts for 34 cm. Therefore, the estimated cumulative bottom flux may not be a true indicator of solute leaching as compared to the episodic water outflow from the soil profile. The present study showed that HYDRUS-1D was unable to reproduce the movement of bromide that was measured in the field while HYDRUS-2D/3D was able to reproduce the field data on bromide to a reasonable extent.

4.5 Summary and Conclusions

HYDRUS-1D and HYDRUS-2D/3D programs were used to simulate one- and two-dimensional movement of water and bromide in hummocky landscape. The soil hydraulic parameters were estimated from field-measured soil physical properties using the pedotransfer functions in the HYDRUS model. The trend in estimated K_{sat} with depth and among landscape positions was similar to field measurements. Bromide did not move past 120 cm depth in the HYDRUS-1D simulation, indicating that no leaching was predicted in all three landscape positions. This was attributed to early termination of soil water outflow during the simulation period. Overall, HYDRUS-1D was unable to reproduce the measured bromide profiles using the approach of this study.

HYDRUS-2D/3D model reproduced the field study measurements better than HYDRUS-1D. The 2D model showed overwinter reduction in the mass of solute, similar to the decline in measured bromide between fall and spring season. The mass balance of bromide in HYDRUS-2D/3D agrees with some of the fertility treatment measurements. It also showed that the differences in solute transport between Site-2007 and Site-2008 were primarily due to precipitation and slope steepness. The 2D simulation also confirmed that the vertical transport of bromide is the main pathway of solute loss in the landscape compared to the lateral movement, thereby reflecting a high risk of nitrate leaching with above normal precipitation.

A few limitations were observed in HYDRUS-2D/3D. The 2D model did not reproduce the landscape effect and differences in K_{sat} that resulted in variability in bromide transport among slope positions. The 2D model was less sensitive to differences in K_{sat} among landscape positions, with the simulated bromide transport responding primarily to variability in precipitation. The HYDRUS model could not reflect the difference between effects of a winter crop and a spring crop on soil water utilization and bromide transport. Another critical weakness of the HYDRUS model relative to the study area was the model's inability to account for snow accumulation and snowmelt.

Also, it was not possible to test the "Campbell hypothesis" in the model simulation due to lack of consideration for effect of N fertility on crop water uptake and solute transport. Therefore, it is recommended that effects of nutrient on water uptake and its resultant effect on solute leaching should be incorporated into numerical models. Nevertheless, the findings suggest that we can obtain a better understanding and control of nitrate leaching in the sub-humid region of the Canadian prairie by using model information to complement field data.

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OVERALL SYNTHESIS

The downward movement of nitrate-nitrogen below the root zone can result in degradation of groundwater quality due to elevated concentrations of NO_3-N in the water. The lateral movement of nitrate in agricultural landscapes can also contribute to contamination of adjacent surface water bodies. In addition to the associated environmental hazards, leaching losses of nitrate fertilizer from agricultural systems reduce crop utilization of added N, and consequently a reduction in crop yield and an increase in fertilization costs.

Due to effects of topographic attributes on soil water flow, there is a marked variation in solute redistribution within the landscape, and ultimately, the magnitude of nitrate leaching. Knowing which portion of the landscape is most susceptible to nitrate leaching will permit landscape knowledge in general and precision agriculture in particular, to be used as management tools to minimize nitrate leaching. However, it is difficult to accurately predict the fate and movement of nitrate in the field due to various complex processes affecting N dynamics.

To enhance our understanding of nitrate movement in agricultural landscapes, tracer techniques were employed to estimate nitrate distribution and recovery in the soil-plant system. In addition to the tracer studies, numerical modelling of water and solute movement in the landscape was conducted. The objectives of this study were to examine the effect of nitrogen fertilization on two-dimensional redistribution of bromide in a hummocky landscape, and to identify the landscape position with the greatest potential for nitrate leaching in fall and spring seasons using a dual application of bromide and labelled ^{15}N tracers. The field experiment on bromide transport was also simulated using HYDRUS models.

The major findings from the three studies are summarized as follows:

i. Nitrogen fertilization reduced the downward movement of bromide in a cropped soil. The study was therefore able to confirm the "Campbell hypothesis" on nitrate leaching;

ii. Nitrogen fertilization also enhanced the lateral movement of bromide in the downslope direction as a result of its reduced vertical movement;

iii. The lower slope position experienced the greatest losses of solute, thereby exhibiting the greatest potential for nitrate leaching in hummocky landscape particularly in the absence of crop uptake, outside the growing season;

iv. The differences in bromide distribution between the two site-years were attributed mainly to differences in crop type (canola versus winter wheat) and the subsequent water use, indicating that factors other than precipitation affect solute movement in agricultural landscapes;

v. The plant uptake of nitrate-nitrogen was greater than that of bromide, as ^{15}N uptake by canola and winter wheat were 35 and 63% of total

N applied, respectively, compared to 2% of the bromide applied in the plant tissue;

vi. The study showed the tendency for nitrate to leach at the same magnitude as bromide in the absence of crop uptake. This was reflected in the identical proportional change in mass of bromide and ^{15}N in the soil profile between fall and spring season;

vii. With crop uptake, the estimated vertical depth to which 50% of the solute had leached (Q50) indicated that bromide overestimated the downward transport of the added nitrate as the Q50 of ^{15}N was only one-half of that of bromide in both site-years;

viii. The simulation study confirmed that 1D models are inadequate to describe solute transport in the landscape, as HYDRUS-2D/3D model reproduced the field data better than HYDRUS-1D;

ix. The model simulation and field experiment both showed that vertical downward movement is the main pathway of solute loss in the landscape;

x. The 2D model showed that differences in precipitation and slope steepness between the two site-years have significant implications on the temporal variability of solute transport under the Canadian prairie conditions;

xi. The 2D model, however, could not reflect effects of factors due to landscape position, differences in cropping pattern, snow accumulation, and N fertility on bromide transport as observed in the field.

The first study (Chapter 2) was conducted to verify whether addition of N fertilizer will reduce nitrate loss. Campbell and his co-authors have shown that N fertilization at recommended rates reduced nitrate leaching, compared to no fertilization. However, it was not clear whether N fertilization can reduce the leaching of mobile nutrients other than nitrate. The two-dimensional redistribution of bromide was also investigated in the first study to verify which dimension of solute transport is predominant in the landscape, vertical or lateral movement, and to identify which part of the landscape would experience the greatest transport of solute.

The study (Chapter 2) resulted in an independent verification of the process for the “Campbell hypothesis” through the use of bromide tracer. The findings confirmed that the greater solute leaching without adequate N fertilization was predominantly due to more intense hydrological activities resulting from available moisture in the soil profile. As reported by Campbell and others, less water uptake in the absence of sufficient N addition resulted in more water in the soil that moved bromide deeper into the soil profile. Assuming that other factors controlling soil water flow and solute movement remain constant, the study suggests that N fertilization at recommended rates can reduce the leaching losses of hydrologically-dependent nutrients such as nitrate, dissolved phosphorus or suspended particulate P fraction, sulphate, and other soluble chemical components.

Although the study confirmed that the vertical movement of bromide was reduced by N fertilization, it had an unexpected effect of increasing the lateral movement due to the accumulation of bromide in the soil profile (Chapter 2). However, the amount of bromide measured in the soil profile at all slope positions was greater than that in the corresponding lateral component of the landscape. The result indicated that the vertical transport of solute in the landscape was more substantial than the lateral movement. The study also showed that the lower slope position experienced the greatest vertical and lateral movement of bromide, thereby exhibiting the greatest potential for nitrate loss in hummocky landscapes, particularly in the absence of crop uptake.

Despite several attempts to understand nitrate leaching in prairie soils of western Canada, the use of dual tracers has not been adequately explored in field experimentation of nitrate transport in hummocky landscape. The second study (Chapter 3) has made a significant contribution to knowledge by using bromide and labelled ^{15}N tracers to quantify NO_3-N leaching, and to identify the topographical region that is most susceptible to nitrate leaching in a hummocky landscape. In the first experiment in Chapter 2, the lower slope position was the region with the greatest loss of bromide in the landscape. However, past studies have shown that bromide transport often overestimate and exaggerate nitrate leaching potential. To determine the magnitude of overestimation, the vertical transports of bromide, ^{15}N and nitrate were compared in the presence and in the absence of plant uptake of water and nutrient (Chapter 3).

The results in Chapter 3 showed that the lower slope position consistently experienced the greatest loss of bromide, ^{15}N, and NO_3-N in the landscape, thereby exhibiting the greatest potential for nitrate leaching in the fall and spring seasons. In contrast, the smallest transport of all the three solutes was at the middle slope position. Using the dual tracer technique, the study confirmed that bromide loss can overestimate nitrate leaching during the growing season. This was indicated in the crop uptake of ^{15}N by canola and winter wheat as 35 and 63% of total N applied, respectively, compared to the negligible bromide uptake.

In addition to the plant data, the Q50 parameter indicated that bromide overestimated the downward transport of the added nitrate as Q50 of ^{15}N was only one-half of that of bromide in both site-years (Chapter 3). Although the missing mass of bromide is known to exaggerate nitrate leaching, this study showed the tendency for nitrate to leach in the same magnitude as bromide in the absence of crop uptake. This was reflected in the identical proportional change in the amount of bromide and ^{15}N in the soil profile between fall and spring season.

Since solute losses were most substantial at the lower slope position and between fall and spring season, these findings suggest that precision farming techniques such as N fertilization based on soil test recommendation and site-specific hydrologic condition, as opposed to uniform rate of N application, should be considered within the landscape. Fall application of N fertilizer should also be discouraged, as the potential for nitrate leaching was equivalent to that of bromide between fall and spring season when there was no crop uptake.

If we can characterize the variability within the landscape, it may be possible to implement management practices that take advantage of this spatial variability for optimization of input resources such as fertilizer, pesticides and herbicides, and maximization of soil productivity. Integrating the knowledge of the most vulnerable landscape position into the existing precision farming program will enable producers to develop best management practices that can minimize groundwater pollution at the landscape scale.

The field data on bromide transport in Chapters 2 and 3 were used to configure the numerical modelling of one- (1D) and two-dimensional (2D) movement of solute in the third study (Chapter 4). While Akinremi et al. (2005) have evaluated nitrate leaching in the prairie soils of western Canada using LEACHMN model, there is little information on numerical modelling of solute transport in hummocky landscapes. Also, it has been shown that one-dimensional models are inadequate to predict water and solute movement in a field with sloping terrain, due to partitioning of flow into vertical and lateral components of the landscape. HYDRUS-1D and HYDRUS-2D/3D programs were used for the numerical simulation of water and bromide movement in the landscape. The HYDRUS-2D/3D program is a Windows-based model with the capacity to analyze two-dimensional water flow and solute transport. However, the model has not been well tested for solute transport in agricultural landscapes.

Following the simulation of vertical transport of bromide using HYDRUS-1D model, the time invariant amounts of bromide and the subsequent lack of solute leaching from the soil profile confirmed that one-dimensional models are inadequate to predict solute movement in the landscape (Chapter 4). This result was attributed to early termination of soil water outflow during the simulation period. The lack of consideration for lateral flow in the 1D model may also contribute to the inability of HYDRUS-1D to reproduce the field data. HYDRUS-2D/3D model reproduced the field study better than HYDRUS-1D by showing a temporal reduction in mass of bromide, and some considerable agreement in bromide mass balance compared to measured values for the fertility treatments. The 2D simulation and field experiment both showed that vertical downward movement is the main pathway of solute loss in the landscape. These results demonstrate the potential risk for nitrate leaching in hummocky landscape, particularly if large amounts of nitrate in the soil coincide with above normal precipitation event.

There were some limitations with HYDRUS-2D/3D simulation of bromide transport. Firstly, the model could not reflect the variability in bromide transport among landscape positions. Secondly, the effect of differences in crop type on solute transport between the two site-years was not captured in the model simulation. Thirdly, the model's inability to account for snow accumulation and snowmelt was a major drawback in reproducing bromide transport relative to the field measurements. Also, it was not possible to test the "Campbell hypothesis" in the model simulation due to lack of consideration for effect of N fertility on crop water uptake and solute transport.

In the field study, the differences in bromide distribution between Site-2007 and Site-2008 were attributed to different patterns of crop water utilization between the two site-years. Conversely, the model simulation showed that the greater precipitation and slope steepness in Site-2008 resulted in a greater solute leaching compared to Site-2007. In spite of the greater precipitation in 2008-2009 growing season, the field data showed that the vertical and lateral movement of bromide in Site-2008 was smaller than in Site-2007, which received a smaller precipitation in 2007-2008 growing season. The reduced bromide movement in Site-2008 was attributed to early season soil water depletion and greater water consumption by winter wheat, compared to canola in Site-2007. This was unlike the result in the model simulation, due to the model's inability to reproduce the effect of differences in crop type on solute transport between the two site-years.

Amongst the factors mentioned above, the crop type is the only variable that can be used as a management tool to reduce nitrate leaching. A practical recommendation is to adopt crop management practices comprising of N fertilization and crop type that can most effectively utilize water in the soil profile. The findings from the field measurements suggest that including winter wheat in the crop rotation program will reduce excess soil moisture and nitrate leaching, particularly in the sub-humid region of the Canadian prairie.

One of the strategies for mitigating nitrate leaching in agricultural soils is to improve the synchronization between N availability from soil N mineralization and N inputs from fertilizer, and plant N uptake. The agronomic approach to achieving this strategy is concerted efforts to improve the predictability of N requirements for optimum crop growth and yield production. Adequate N fertilization based on site-specific management program and soil N test is also an important strategy for reducing nitrate leaching in agricultural landscapes. As confirmed in the study, low rates of N fertilization may be potentially as detrimental to groundwater quality as excessive rates of N. Application of N fertilizer based on site-specific hydrologic conditions will help minimize the adverse implications of large solute leaching observed at the lower landscape position.

Since NO_3-N is highly soluble and mobile in the soil, the extent of soil water movement in the vadose zone largely determines the magnitude of nitrate leaching. The best way of reducing nitrate leaching in agricultural soils is to select a package of farming practices that minimize water flow through the soil and nitrate accumulation within the soil. This implies that both nitrate fluxes and soil water flow need to be synchronized with crop utilization. A combination of model simulation and field measurements is an effective tool for achieving this synchronization strategy. The management practices suggested above can be recommended to farmers as strategies for promoting viable crop production and environmental sustainability in agricultural landscapes.

Further studies may be conducted in the future to investigate nitrate distribution in hummocky landscape using a labelled ammonium (NH_4^+-^{15}N) tracer source. This is important since the major form of N added to the soil in Manitoba is ammonia-based N fertilizers and manures. In addition to crop N uptake, application of NH_4^+-^{15}N tracer will help to demonstrate the roles of ammonium adsorption to soil components on nitrate leaching. Effects of winter wheat and soil water depletion on bromide transport should be verified by seeding winter wheat and canola on the same plot in the same growing season. This will help to clarify whether the differences in bromide redistribution between the two site-years were predominantly due to differences in crop type, slope aspect, spatial variability between the two sites or differences in precipitation between the two growing seasons.

Although simulation models serve as a useful tool for predicting soil water flow and solute transport, there are still some difficulties with developing accurate numerical and conceptual computations, especially for large-scale transient, multi-dimensional field applications. Also, many of the coefficients used in simulation models are empirical-based. Therefore, it is imperative that these models be tested and validated against realistic field conditions before they are applied to field situations.

The findings from this research study suggest that we can obtain a better understanding and control of nitrate leaching in the sub-humid region of the Canadian prairie by using model information to complement field data.