Simulating nitrogen pollution potential in surface and subsurface runoff in Ontario using EPIC model

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[1]*School of Engineering, University of Guelph, Guelph, Ontario N1G 2V7, Canada; and* [2]*Environmental Monitoring and Reporting Branch, Ministry of the Environment, Etobicoke, Ontario M9P 3V6, Canada. *Email: rrudra@uoguelph.ca*

Rudra, R.P., S.I. Ahmed, N.A. McLaughlin and P.K. Goel. 2011. **Simulating nitrogen pollution potential in surface and subsurface runoff in Ontario using EPIC model**. Canadian Biosystems Engineering/Le génie des biosystèmes au Canada. 53: 1.1–1.18. This study focuses on the possible potential of nitrogen contamination of water resources due to land application of manure and fertilizer in agricultural regions of Ontario. The EPIC (Erosion/Productivity Impact Calculator) model was used to partition nitrogen loads in percolating and runoff waters under corn production system by dividing the province into four regions. Water balance and nitrogen loads were estimated for different types of soils, land slope gradient, soil organic matter, rate of nitrogen application, types of tillage, and the presence and absence of subsurface drainage systems. Results indicated that reasonable annual nitrogen leaching predictions could be obtained through accurate soil hydraulic characterization and monthly runoff curve number adjustment. The regression equations reflected the general trends of rate of application and organic matter and were found to be the most important factor in quantifying the nitrate loads in the infiltrating water for all soil types. Increasing slope gradient translated into an increasing portion of nitrate moving with subsurface lateral flow based on the model results. The statistical analyses also showed that the equations can satisfactorily predict nitrate loads. The strength of the equations over the MCLONE4 module is shown by the reduced annual deviation from the observed nitrogen loads. In addition, the current recommended rate seemed to represent a cut-off limit, below which nitrogen loads in percolating water did not substantially decrease. Although, crop yield did not significantly increase with increased nitrogen application rates, the nitrogen loads did. It shows that the current recommended rates for the different regions are probably appropriate to minimize nitrogen losses while maximizing yield. The high nitrogen losses from the Western region may result not only from higher nitrogen application rates, but also from the higher probability of leaching events caused by the longer cropping period. Overall, the approach and the results of the study could be helpful in effective source water protection planning. **Key words:** water quality, computer modeling, management strategies.

Cette étude vise à quantifier les risques de contamination des ressources hydriques résultant de l'épandage de fumiers et de fertilisants dans les régions agricoles de l'Ontario. Le modèle EPIC (Erosion/Productivity Impact Calculator – calculateur de l'impact érosion/productivité) a été utilisé pour différencier les charges d'azote dans les eaux de percolation et de ruissellement dans un système de production de maïs en divisant la province en quatre régions. Le bilan en eau et les charges en azote ont été estimés pour différents types de sols, pentes des champs, contenus en matière organique, taux d'épandage d'azote, types de travail du sol et présence ou absence de système de drainage souterrain. Les résultats ont indiqué que des prédictions raisonnables de lessivage annuel d'azote pouvaient être obtenues par une caractérisation hydrique des sols adéquate et un ajustement du ''curve number'' du ruissellement mensuel. Les équations de régression ont représenté les tendances générales du taux d'épandage et du contenu en matière organique et se sont avérées être le facteur le plus important pour quantifier les charges d'azote dans les eaux d'infiltration pour tous les types de sol. Selon les résultats obtenus du modèle, une augmentation de la pente des champs se traduisait en une augmentation de la proportion d'azote déplacé via l'écoulement latéral souterrain. Les analyses statistiques ont aussi montré que les équations pouvaient prédire de manière satisfaisante les charges en azote. La précision des équations par rapport au module MCLONE4 est démontrée par une déviation annuelle réduite par rapport aux charges d'azote observées. De plus, les taux actuellement recommandés semble représenter un seuil limite sous lequel les charges en azote dans l'eau de percolation ne montraient pas une diminution substantielle. Cependant même si le rendement des cultures n'augmentait pas de façon significative avec une augmentation des taux d'épandages d'azote, les charges d'azote augmentaient. Ceci démontre que les taux courant d'épandage pour les différentes régions sont probablement appropriés pour minimiser les pertes en azote tout en maximisant les rendements. Les pertes en azote élevées dans la région de l'ouest peuvent ne pas découler seulement de taux d'épandage d'azote élevés mais aussi provenir d'une plus grande probabilité d'évènements de ruissellement causés par une période de culture plus longue. En résumé, l'approche et les résultats de cette étude pourraient constituer des éléments utiles pour une planification efficace de la protection des ressources hydriques. **Mots clés:** qualité de l'eau, modélisation informatique, stratégies de gestion.

INTRODUCTION

Nitrogen and phosphorus pollution of water resources from intensive agriculture and livestock operations is a growing concern in the Canadian Great Lakes Basin (Chambers et al. 2001). In addition to the Walkerton tragedy in 2000, there have been numerous cases in Ontario where groundwater and surface water pollution have been linked to excessive manure and litter disposal (Rudolph et al. 1998; Gillham 1991; Frank et al. 1991). Fleming and Fraser (1999), while investigating the changes in land use in several watersheds over the past four decades, reported that the nitrate concentrations in surface waters have been steadily increasing since the 1960s, the highest levels occurring in agricultural watersheds. Frank et al. (1991) surveyed 183 wells in Ontario and observed 15% with nitrate-nitrogen (NO_3-N) exceeding 10 mg L^{-1}. Goss et al. (1998) performed surveys of 1237 farm wells and observed 37% did not meet standard levels for at least one type of contaminant (coliform, nitrate, or herbicides), and 14% of the wells exceeded NO_3-N drinking water standards.

A holistic approach is needed to reduce the growing impact of agriculture activities on water quality. During the past decade, a number of field studies (Frank et al. 1991; Rudolph et al. 1998; Thoma et al. 2005; Jaynes et al. 2008) and modeling studies (Bakhsh et al. 2004; Wang et al. 2006) have been initiated in North America to understand problems related to fertilizer and manure management. Many studies have also been conducted in Ontario to evaluate the environmental impact and alternative management options for the province (Goss et al. 1995; Kachonoski et al. 1998; Wall et al. 1998; Ahmed et al. 2007). However, the considerable amount of time and expense, and other problems, have limited the impact of these field studies. To overcome these limitations, hydrologic and water-quality computer modeling approaches have been found to be a better alternative for researchers and policy makers. In addition, it is also important to test and validate these models for a wide variety of climatic and agricultural conditions (Wang et al. 2006).

Jaynes et al. (2008) evaluated the effects of greater tile depth and denitrification walls on both sides of a tile to reduce nitrate losses from tile drains. They found that denitrification walls reduce flow-weighted nitrate concentrations by greater than 55% without affecting the grain yield. Thoma et al. (2005) investigated the effect of tillage and nutrient application methods on tile flow and surface inlet water quality under continuous corn system and observed few statistically significant differences in water quality among the treatments.

Bakhsh et al. (2004) used the Root Zone Water Quality Model (RZWQM Team 1992) to study nitrate losses in subsurface drains in Iowa, USA, and found the model's efficiency to be in close agreement with the observed data for tile flow (99%) and nitrate concentration in tile drains (80%). The percent difference between observed and simulated corn yields was also less than 10%. Ahmed et al. (2007) investigated the long-term impact of nitrogen management practices, followed for corn production, on the NO_3-N in subsurface drainage water and crop yield using RZWQM in Ontario and reported that the relative long-term effectiveness of nitrogen application fluctuates significantly from year to year in response to weather patterns. The crop rotation from corn–soybean to corn–soybean–soybean results in a greater reduction in NO_3-N loads in the tile outflow on silt loam soil than on sandy loam soil.

The predictive ability of EPIC (Erosion/Productivity Impact Calculator) model (Williams et al. 1990) was evaluated by Wang et al. (2006), who concluded that the EPIC successfully predicted annual, monthly, and daily surface runoff modeling efficiency (EF) greater than 0.5 and R^2 larger than 0.7. The crop prediction efficiency for simulation of crop yield was 0.96. The EF of EPIC in simulating annual sediment, soluble phosphorus, and nitrate losses ranged from 0.59 to 0.87.

The EPIC model was also evaluated by Chung et al. (2002) for simulating NO_3-N losses to streams in north-central USA for three cropping systems. Predicted average tile flows and NO_3-N losses generally improved following calibration of Soil Conservation Service (SCS) Curve Number (CN) (SCS 1972); however, due to the absence of a preferential flow component there was poor agreement between the observed and simulated monthly tile flows and NO_3-N losses.

Field-scale water budget and water quality models have proven to be useful tools for agricultural water management. Although computer modeling has been widely acceptable as an effective tool to evaluate Best Management Practices (BMPs), one of the important factors is the selection of a computer model according to local environmental conditions. MCLONE4 is a computer program to evaluate and rate a planned or operating manure-handling system for Ontario conditions (Centre for Land and Water Stewardship 1999). The name MCLONE is derived from "Manure" and five criteria of the program: Cost, Labour, Odour, Nutrients, and Environment. The purpose of MCLONE4 was to allow livestock producers to evaluate simultaneously the technical, economic, and environmental risk attributes of alternative manure-handling systems.

Considering the importance of effective nutrient management, this study explores the potential of a field-scale modeling approach for nutrient management under Ontario conditions. The main objective of this study was to apply the EPIC model to Ontario conditions to identify the most important factors in determining the extent of nitrogen contamination in receiving waters, develop a nitrogen prediction tool for application to many scenarios, and establish cutoff criteria to distinguish between degrees of pollution potential. The specific objectives of the study were: (1) to perform multiple scenario simulation of nitrate loads in tile drainage and groundwater and statistically analyze the performance of the EPIC results based on regional characterization; and (2) to develop a method of determining the pollution potential at different sites for implementation within MCLONE4.

Model selection

Currently, numerous models have the capability of simulating nutrients and other pollutants above and below the ground surface. However, selection of a proper hydrological model has always been a challenge for users. As described by Donigian and Rao (1988), choosing the appropriate model for a specific problem depends on the level of analysis required. The selection process requires the acknowledgement of required accuracy, amount of input information available, nature and format of the output and the ease of model application (Thomas et al. 1998). An index approach based on objective and subjective attributes was used for the selection of the model used in this study (Von Euw et al. 1992). Each shortlisted model was evaluated for proper simulation of hydrological effects on nitrogen fate and transport, representation of spatial scale, and the capability to simulate surface runoff, tile flow and ground water input. Further, several factors were defined within each criterion and given relative weights based on their importance and the model was rated based on their assessment.

After a comprehensive evaluation of a variety of models, such as EPIC (Williams et al. 1990), DRAINMOD-N (Skaggs 1982), GLEAMS (Knisel 1993), LEACHM (Wagenet and Hutson 1989), GLEAMS-DRAINMOD (Thooko et al. 1994), PRZM3 (Carsel et al. 1984), and RZWQM using the previously defined criteria, the EPIC model was selected for the present study. A number of studies are available on the evaluation of various components of EPIC. EPIC has the capability to simulate appropriate long-term water budgets (Steiner et al. 1987; Chung et al. 1999; He et al. 2006), transport of nitrogen to the surface and subsurface water systems (Edwards et al. 1994; Chung et al. 1999; King et al. 1996; Wang et al. 2006), nitrogen content in the soil media (Smith et al. 1990; Beckie et al. 1995; Warner et al. 1997; Roloff et al. 1998), and the long-term impact of management practices.

Description of the EPIC model

The EPIC (Erosion/Productivity Impact Calculator) model was originally developed to determine the relationship between erosion and soil productivity and was later modified to simulate processes in the entire plant-soil-atmosphere environment (Beckie et al. 1995). The physical components of EPIC are divided into seven main groups: hydrology, weather, erosion, nutrients, plant growth, soil temperature and tillage (Williams et al. 1990), having a direct effect on nitrogen transport in surface, soil and ground water. The runoff amount is estimated using the SCS Curve Number method, which is based on a retention parameter. Soil erosion, required for the calculations of sediment-bound nitrogen calculations, is computed by modified Universal Soil Loss Equation (Wischmeier and Smith 1978). Mineral nitrogen is added into the system by soil organic matter decomposition, residue decomposition, and fertilizer application. The soil organic pool is simulated in two groups: fresh organic matter and stable organic matter (or humus). EPIC also simulates nitrogen transformations in the soil, plant, and water matrix. Nitrogen can be lost to surface water by runoff and erosion and can percolate to ground water depending upon rate of water flow through soil profile. More details on the model can be found in Williams et al. (1990).

METHODOLOGY

Classification of Ontario into regions

To estimate the possible potential of nitrogen pollution in Ontario, the province was divided into various regions, and multiple scenarios affecting NO_3-N loads in tile drainage and groundwater were applied depending on the regional characterization.

The nitrogen loss algorithm developed was intended to integrate within MCLONE 4. Therefore, it was important that the developed module functions in accordance with MCLONE4's current input. Geographically, the MCLONE4 program divides the province into four regions based on the crop heat units: Southwestern, Eastern, Central and Northern (Fig. 1).

Soil. To properly describe the soil types to obtain realistic simulation results, three soil series representing sandy, loam, and clay textures were chosen for each region. Soils in each region were parameterized using information available in Ontario soil survey reports. Field capacity and wilting point (Table 1) were estimated from general soil characteristics using the method given by Saxton et al. (1986), and the results obtained were compared with similar local studies reported by Selirio et al. (1978). Saturated hydraulic conductivity, shown in Table 2, was estimated within the general bounds defined by Cook et al. (1985). Soil bulk density was estimated using the approach outlined by Pettapiece (1995). Soil organic matter in the A horizon varied between 1 and 5%, generally occurring in most Ontario soils.

The soil types were grouped together into three classes based on similar properties as given in Tables 1–3. Sandy, sandy loam, and loamy sand textures were grouped as sandy soils. Sandy clay loams, sandy loams, loams, silty loams, and silts were classified as loams. Clay loams, silty clay loams, clay and silty clays were grouped as clay soils. These three soil groups were represented in each region by the soil series presented in Table 3. The soil series selected in each region tend to be widely used for agricultural activities.

Topography. The Ontario Drainage Guide (OMAFRA 2000) was used to characterize the topography of these regions. It was found that the particular soil types in each region were generally found on similar slope gradients in all regions. Both sandy and loamy soils exist on slope gradients between 0 and 15%, whereas clay soils exist on fairly flat landscapes (between 0 and 2% slope gradients).

Climate. The annual average temperatures are highest at the extreme south of the province and decrease with increase in latitude. Precipitation is highly variable throughout the province, and the long-term average annual amounts vary between 660 and 1016 mm. In characterizing the different climatic conditions for the four regions, 36 yr of weather data from four sites (including daily precipitation, maximum and minimum temperatures, and wind speed) were used. Sites located in Harrow, Ottawa, Guelph, and Kapuskasing were used to represent weather conditions in the Western, Eastern, Central, and Northern climatic regions, respectively. The estimate of potential heat units for the Western, Eastern, Central, and Northern regions, based on the daily temperature data, were 1550, 1550, 1350, and 1050, respectively. The crop heat unit approach, used in MCLONE4, and differences in soil types, topography, and climatic conditions were followed to divide the province of Ontario into four regions: Southwestern, Eastern, Central and Northern (Fig. 1).

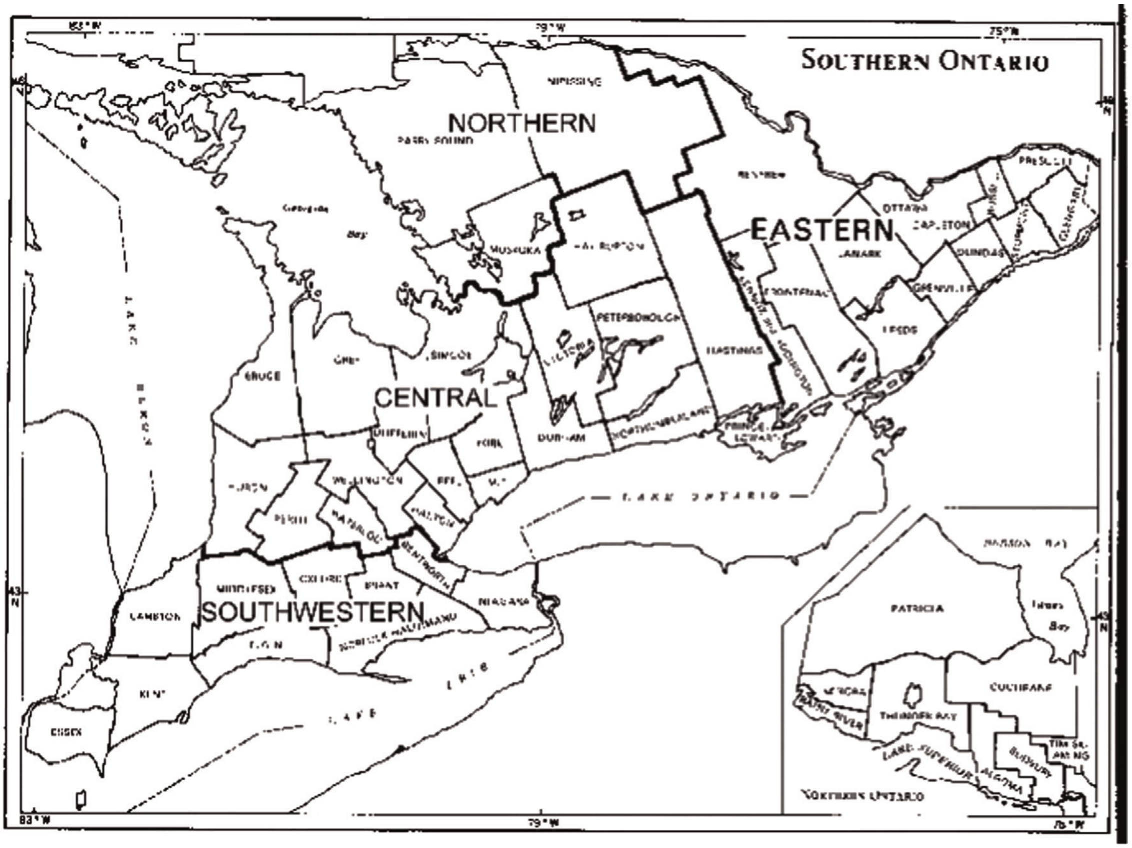

Fig. 1. Regions of Ontario used in the development of scenarios based on physiography and management practices (modified from Webber 1975).

Scenarios used for pollution evaluation

To evaluate the possible pollution potential in various regions of Ontario, scenarios evaluated were divided into two categories. The first category included soil type, land slope gradient, and soil organic matter. Some of them were sensitive parameters for EPIC application. The second category focused on soil, water, and nitrogen management practices (Beckwith et al. 1998; McGechan and Wu 1998l; Chen et al. 2000; Bakhsh et al. 2001). It included rate of nitrogen application for corn, tillage practices, and the presence or absence of subsurface drainage. Model simulations included the combination of all factors in both categories. Table 4 presents the factors levels of variation of the parameters used in this study. The average annual nitrate loads to the surface and ground water were tabulated using 540 model simulations for each region.

Table 1. Common relationships between soil texture and soil water-holding capacities[†]

Class	% sand	% clay	Wilting point (m^3 m^{-3})	Field capacity (m^3 m^{-3})
Sand	90	10	0.09	0.16
Loamy sand	80	10	0.09	0.17
Sandy loam	65	10	0.09	0.20
Sand clay loam	60	30	0.17	0.27
Sandy clay	50	45	0.24	0.34
Loam	40	20	0.13	0.26
Silt loam	25	15	0.11	0.28
Silt	7	10	0.10	0.30
Clay loam	30	35	0.19	0.34
Silty clay loam	10	35	0.19	0.37
Clay	30	50	0.28	0.41
Silty clay	10	45	0.26	0.42

[†]Source: Saxton et al. (1986).

Table 2. Hydraulic conductivity values for different soil textural classes[†]

Textural class	Hydraulic conductivity (m s^{-1})
Gravels, coarse sands	$>4.4 \times 10^{-6}$
Loamy sands and sandy loams	1.4×10^{-6} to 4.4×10^{-6}
Fine sandy loams, loams	0.4×10^{-6} to 1.4×10^{-6}
Loams, silt loams, clay loams	0.14×10^{-6} to 0.4×10^{-6}
Clay loams, clays	4×10^{-8} to 0.4×10^{-8}
Dense, compacted clays	$<4 \times 10^{-8}$

[†]Source: Cook et al. (1985).

Table 3. Soil textural groups and representative soils selected for modeling for each site.

Soil texture	Soil type	Region			
		Western	Eastern	Central	Northern
Sandy, sandy loam, and loamy sand	Coarse sandy	Fox	Kars	Brighton	Wendigo
Sandy clay loams, sandy loams, loams, silty loams, and silts	Medium loam	Guelph	Osgoode	Milliken	Magnetawan
Clay loams, silty clay loams, clay and silty clays	Fine clay	Brookston	Bearbrook	Simcoe	Ste-Rosalie

The nitrogen application rates for corn (Eastern 175 kg ha^{-1}, Western 200 kg ha^{-1}, Central 135 kg ha^{-1}, and Northern 155 kg ha^{-1}) were varied within the recommended application rates as reported in MCLONE4. The amount of nitrogen losses to surface and groundwater from nitrogen (manure and inorganic fertilizer) application were computed as the difference between total nitrogen application and nitrogen used by the crop. In the MCLONE4, the maximum nitrogen application is 30 kg N ha^{-1} above crop requirements. The rate of application represented total fertilizer input minus volatilization losses. This procedure allows for easier implementation within MCLONE4 by separating EPIC's volatilization rate estimates from the results. The current volatilization prediction technology within MCLONE4 is based on local conditions and is considered more accurate than EPIC estimates.

The hydrologic effect of tillage practices (conventional and no-till) was simulated by adjusting the CN values (Table 5). Simulated conventional tillage represented moldboard plowing in the fall, and has an impact on water movement during the winter months. This practice is significantly different from the no-till operations, as it could result in higher subsurface drainage during the non-cropped season, which represents a large portion of the annual drainage amounts (Drury et al. 1996).

Subsurface drainage is a very common water table management practice in Ontario and has a significant impact on the eventual nitrogen sinks. Water table management practices were simulated by including or excluding subsurface drainage. Tile drains were set at a 900 mm depth for sandy and silty soils and at 600 mm depth for clay soil (OMAFRA 2000). The curve number was not adjusted between the drained and non-drained systems as it was assumed that artificial drainage systems would only be implemented if soil moisture conditions were normally above average (CNII) moisture conditions (SCS 1972). The applied CN values are representative of average soil moisture conditions for both tile-drained and naturally drained soils.

Table 4. List of important variables and fertilizer/manure application rates used for EPIC simulation.

Factor	Tested levels
Soil type	Sandy, loam, clay
Rate of nitrogen fertilization	50% manure[†] + 50% inorganic fertilizer
	75% manure + 25% inorganic fertilizer
	100% manure
	100% manure + 15 kg N ha^{-1} manure
	100% manure + 30 kg N ha^{-1} manure
Soil organic matter	1%, 3%, 5%
Slope gradient	0%, 7.5%, 15% (for sand and loam)
	0%, 1%, 2% (for clays)
Tillage	Conventional, no till
Drainage	With tile drain, without tile drain

[†]Percent of crop requirements.

The timing of application of nitrogen, an important factor affecting nitrogen loads in drainage water could not be included in this analysis because the version of EPIC model used in this study does not have the capability to properly change the time of application from year to year in the continuous mode.

Analysis procedure

The EPIC model, calibrated and validated under Ontario conditions by McLaughlin et al. (2006), was applied to simulate various components of water budget and nitrogen totals on three types of soils. Analysis was performed separately on tile-drained and naturally drained soils. Hydrological analysis focused on the long-term average components of the water budget. For tile-drained soils, water which is not intercepted by the crop can either enter the tile drain or bypass and move further toward groundwater. For naturally drained soils (without tile drainage), water which is not intercepted by the plant roots will move downward toward the groundwater. Water can also move laterally within the soil matrix. As a conservative assumption lateral flow was assumed as a part of deep percolation contributing to groundwater.

The nitrogen sinks related to water partitioning for both tile-drained and naturally drained soils have been divided in surface runoff N, tile flow N, other lateral flow N, and deep percolation N. The nitrogen moving with water in the soil profile for tile-drained soils was further divided into two sinks: TOTN and GNDNI. TOTN represents the total amount of nitrogen (tile drain N + lateral flow N + deep percolation N) moving through the soil matrix. GNDNI represents the sum of nitrogen in the lateral flow and nitrogen in deep percolation as shown in Fig. 2. In the naturally drained soils, all the nitrogen moving into and through the soil matrix is lumped together as one sink (GNDNII). GNDNII represents the nitrogen moving with water below the root zone with deep percolation and lateral flow at the edge of the field.

Table 5. Curve number values across the year used for soil types and management practices for EPIC simulation.

Treatment	Jan	Mar	May	PT†	Mid June	July	Sept	HT¥	PW‡	Dec
Sandy – CT§	92	94	80	77	72	67	72	75	73	80
Sandy – NT	92	94	85	71	69	67	69	70	71	85
Loam (B)£ – CT	92	94	89	86	81	76	81	86	84	88
Loam (B) – NT	92	94	91	82	78	74	78	80	82	88
Loam (C)£– CT	92	94	89	89	88	85	88	86	85	89
Loam (C) – NT	92	94	91	88	82	76	80	82	88	90
Clay – CT	94	95	92	92	91	88	91	92	87	90
Clay – NT	94	95	94	88	85	83	85	87	88	92

†Planting date; ¥Harvesting date; ‡Plowing date (for conventional till); November 1 (for no till); §CT: Conventional Till; NT: No Till; £(B/C) Soil Hydrological Group.

RESULTS and DISCUSSION

Water balance

Precipitation, evapotranspiration, infiltration, and surface runoff are the basic components in the water balance of a watershed. Table 6 shows the long-term (30-yr) annual average water balance for three types of soils in all four regions with and without tile drainage. The values shown in Table 6 are the simple mean values of many simulations for various regions of the province. These data also show the effect of tile drainage on partitioning of water into surface and subsurface components. These data clearly indicate that percolation and lateral subsurface flows are closely linked to the presence or absence of tile drainage. In the presence of tile drainage, the other lateral flows are negligible, and percolation is virtually nonexistence in all the regions. However, tile drainage systems have a minor impact on surface runoff. Clay soils had the highest amount of surface runoff (~25%) of total water balance for all four regions. Tile flow was found to be highest for sandy soils and lowest for clay soils. Similar trends were found for other lateral flows and groundwater for these soils for all the regions (Table 6). These results also show that under no-tile conditions, sandy soil contributes more to percolation (27 to 37% of total water budget) than loam and clay soils, and precipitation contributes a higher percentage to percolation in the northern region followed by central, eastern, and western regions. Evapotranspiration (ET) was found to be higher in the West and Central regions, as expected. The lowest ET was from the Northern region. Overall, ET varied from 43–67% of the water budget.

Table 7 presents statistics on average annual water surplus (runoff + subsurface flow + percolation) estimates and evaporation rates simulated by EPIC in comparison with previous studies for these regions. Dickinson and Diiwu (2000) compiled average evapotranspiration amounts for different regions of the province based on recorded precipitation and streamflow data. Their study includes a number of watersheds in each region, within which there is a variety of vegetation, soil types and slope gradients. The estimations provided by Fallow et al. (1999) are based on SHAW (Flerchinger et al. 1996) simulation from one typical soil series in each region using long-term observed climatic data. Evaporation rates were estimated from corn crop parameters over a constant growing season period for the four sites. Surplus water was computed based on runoff and deep percolation computations of the Brooks and Corey water relationship model (Flerchinger 2000). The evaporation estimate provided by Rudra et al. (2000) is based on expected rates from an area with annual crops in southern Ontario. The surplus water estimates were deducted from observed precipitation and streamflow from a number of watersheds in southern Ontario with varying size, topographical features, and soil texture characteristics (Rudra et al. 1998).

Water surplus in the Western region showed a high variability between the different estimation approaches as there was approximately a 200 mm difference between EPIC estimations and those from Fallow et al. (1999). This discrepancy may be attributed to hydraulic differences in different soil textures from EPIC simulations and lack of variability in soil conditions modelled by

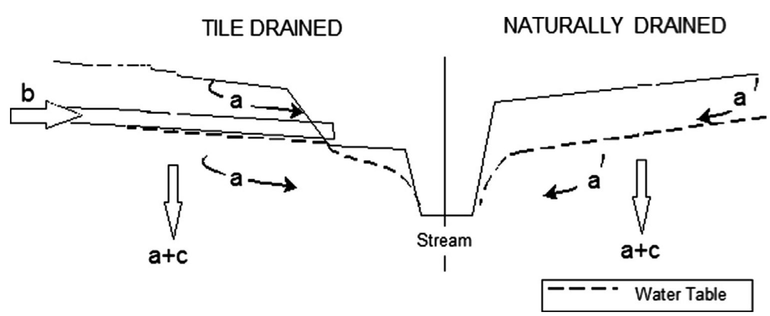

Fig. 2. Nitrogen partition showing lateral flow N (a), tile drain N (b), and deep percolation N (c) for regression analysis for tile-drained and naturally drained soils.

Table 6. The impact of tile drainage on long-term annual average water budget simulated by EPIC for various regions of Ontario.

Region	Soil type	% of precipitation											
		SRO[α]		TF[β]		OLF[γ]		PERCL[Ω]		GND[η]		ET[π]	
		WD[†]	ND[‡]	WD	ND	WD	ND	WD	ND	WD	ND	WD	ND
Western	Sand	12	12	17	0	12	2	1	26	14	29	56	58
	Loam	14	14	11	0	10	2	1	19	11	21	62	63
	Clay	24	24	9	0	1	0	0	8	1	8	65	66
Eastern	Sand	14	13	18	0	12	2	1	28	14	31	53	54
	Loam	19	18	14	0	10	2	0	22	11	25	55	56
	Clay	25	25	16	0	1	0	0	18	2	19	55	55
Central	Sand	15	15	17	0	11	2	0	26	12	29	55	55
	Loam	14	14	10	0	10	2	1	20	11	22	63	63
	Clay	26	26	9	0	1	0	0	10	1	10	63	63
Northern	Sand	15	15	24	0	15	3	2	37	17	40	42	43
	Loam	22	15	16	0	10	2	1	31	12	34	49	49
	Clay	27	27	20	0	1	0	1	23	3	23	48	49

[α]Surface runoff; [†]With drainage system; [‡]No tile drainage system; [β]tile flow; [γ]other lateral flow; [Ω]Percolation; [η]other lateral flow + percolation; [π]Evapotranspiration.

Fallow et al. (1999). There was better agreement between the different methods' surplus water estimates for the other regions. The EPIC findings agreed best with those from Dickinson and Diiwu (2000). This could be a reflection of the variety of soil types and topographical features included in these methods, whereas findings by Fallow et al. (1999) and Rudra et al. (2000) were based on a smaller data set.

Nitrogen balance

Table 8 shows the long-term (30-yr) annual average nitrogen budget for three types of soils in four regions with and without tile drainage. For the Western, Eastern and Central regions, the desired yield of the corn crop was 10 t ha^{-1}, whereas in the Northern region, fertilizer was applied for an expected corn yield of 9 t ha^{-1} (OMAFRA 2001). These data indicate that for Western, Eastern, and Central regions the simulated crop yield was higher than the targeted yield. For the Northern region the simulated crop yield for loam and clay soil was very close to the expected yield; however, the average crop yield was about 10% less than the expected yield for sandy soil. These results also show that over the long-term drainage does not have a significant impact on the crop yield; however, there was significant year-to-year variation.

These data also show that crop uptake is the most dominant component of the nitrogen budget followed by input to the ground water. Plant uptake is a dynamic process within the EPIC model. Its sensitivity to water, nutrient, and aeration stresses provide adjustment to

Table 7. Amount of evapotranspiration and surplus water from various studies for different regions of Ontario.

Source	Western	Eastern	Central	Northern
	Evapotranspiration (mm)			
EPIC	550 ± 36	473 ± 9	504 ± 33	404 ± 26
Dickinson (2000)[‡]	500–550	450	450–500	450
Fallow et al. (1999)[§]	675 ± 68	562 ± 38	497 ± 50	505 ± 37
Rudra et al. (2000)	567£			
	Surplus water (mm)			
EPIC	450 ± 81	321 ± 65	330 ± 33	457 ± 26
Dickinson and Diiwu (2000)	320–420	360–400	300–470	340–530
Fallow et al. (1999)	209	301	364	359
Rudra et al. (2000)	211–329[¥]			

[‡]Personal communication regarding estimation of water balance throughout province based on streamflow and precipitation data.
[§]Partition based on SHAW model results for typical soils in each region.
[£]Calculation for annual crop at one site in Western Region.
[¥]Range of measured runoff for 8 sites in Southern Region.

Table 8. The impact of tile drainage on long-term annual average nitrogen load simulated by EPIC for various regions of Ontario.

		% nitrogen applied													
		SRO[α]		TF[β]		OLF[γ]		PERCL[Ω]		GNDN[η]		Crop uptake		Crop yield (t ha^{-1})	
Region	Soil type	WD[†]	ND[‡]	WD	ND	WD	ND	WD	ND	WD	ND	WD	ND	WD	ND
Western	Sand	6	6	13	0	10	2	1	20	11	23	68	69	10	11
	Loam	11	11	7	0	8	1	0	13	8	15	72	73	11	11
	Clay	4	4	17	0	1	0	0	12	1	13	77	81	11	11
Eastern	Sand	9	9	6	0	7	1	0	12	8	13	75	76	11	11
	Loam	11	11	8	0	5	1	0	10	5	11	74	77	12	12
	Clay	3	4	10	0	0	0	0	7	1	7	84	88	12	12
Central	Sand	0	0	13	0	11	2	0	14	11	16	73	82	9	10
	Loam	12	11	5	0	7	1	0	11	8	12	73	75	11	11
	Clay	5	4	7	0	0	0	0	4	0	4	85	90	11	11
Northern	Sand	8	8	15	0	12	2	0	21	12	24	63	67	7	8
	Loam	26	21	7	0	5	1	0	11	5	13	60	65	8	8
	Clay	3	4	15	0	1	0	0	13	1	13	79	82	8	8

[α]Surface runoff; [†]With drainage system; [‡]No tile drainage system; [β]Tile flow N; [γ]Other lateral flow N; [Ω]Deep percolation N; [η]Other lateral flow N + deep percolation N.

changing environmental conditions. It was expected that nitrogen uptake would be much greater in areas of the province having higher potential heat units and consequently having higher recommended rates. However, the difference in uptake was not equal to the difference in recommended rates of nitrogen application. Actual yield production values for different regions throughout the province also suggest that the increase in yield does not reflect the increase in recommended application rate in terms of nitrogen uptake. These observations may reflect that higher rates of application simply decrease the probability of nitrogen stress deficiency in plants, and result in higher yields. However, the high application rates in these regions may also lead to an increase in environmental pollution.

The simulated N uptake ranged from 61 to 86% of applied nitrogen for tile-drained soils and from 66 to 91% for undrained soils. The maximum N uptake under tile-drained conditions was for clay soils in the Central region and minimum for loam soils in the Northern region. The maximum N uptake in the Central region from clay resulted in about 19% increase in the crop yield. Similar crop yield increase was achieved from clay soil in the Eastern region. Under un-drained conditions, maximum crop yield was obtained from clay and loam soils in the Eastern region where the N uptake was 88 and 77%, respectively. Overall, minimum yield was obtained from all soil types in Northern region, where the N uptake was minimum. There was no consistent pattern in the nitrogen uptake and crop yield with type of soil. The data indicate an increase in yield with nitrogen for both drained and un-drained soils, but there was no definite trend.

The loss of nitrogen in the surface runoff (SRO) was similar for both drained and undrained conditions. The loam soils had maximum nitrogen in the SRO, followed by sand and clay, respectively (Table 8). This was due to sediment bound nitrogen in the surface flow. Loamy soils are more erodible, have high erosion potential and more contribution to sediment and nitrogen to the surface water.

About 6 to 18% of the N was lost through the tile-drained water (Table 8). Clay and sandy soils showed a trend of higher N loss through tile drains than did loam soils. For other lateral flow N loss, tile flow system dominated when compared with no drain system. However, sandy soils lost the greatest amount of N as compared with clay and loam soils for both the systems. The N loss through percolation was much higher from undrained soils than from drained soils. Again, sandy soils lost the greatest amount of N when analyzed for drained and undrained systems.

The nitrogen lost in the groundwater showed a similar pattern for drained and undrained soils (Table 8). However, losses of N with tile drains were lower than the losses with no tile systems. Overall, the maximum contribution was from sandy soils followed by loam and clay, respectively. The range of nitrogen loss to the ground water was 1 to 12% for tile-drained soils and 4 to 24% for undrained soils. This is because most of tile flow occurs during the spring and very little occurS during the late spring, summer and early fall. More surplus water available in loam and clay period results in higher nitrogen uptake.

Based on comparisons with other studies, the annual water budget (Table 7) and nitrogen estimates (Table 8) for different regions of the province provided by EPIC are reasonable. Nitrogen uptake estimates were considered appropriate in association with the predicted yields of different regions and the high variability in observed uptake rates (OMAFRA 2001).

These modeling results provide general guidelines for various regions and major textural soil classes. However, when considering these results it is important to keep in mind the strengths and weaknesses of the EPIC model. The model considers all the nitrogen dynamic processes by first order equations. EPIC does not have the capability to simulate individual events; therefore, it can not provide estimates on a short-term basis, such as on an event basis. Nevertheless, this approach provides realistic annual average estimates without complete knowledge of the factors affecting nitrogen loads in surface and groundwater during the most critical periods. These critical periods are very important, as field measurements have shown that much of the annual nitrogen losses occur during a few large events, and mostly during the spring period. There is also some uncertainty concerning the plant uptake process, which is a very important component of the nitrogen budget. These results provide better guidelines than using the annual nitrogen budget approach. The modeling approach considers temporal variability of processes affecting the water budget and nitrogen dynamics, and provides realistic actual temporal trends of nitrogen loads in surface runoff, tile outflow, and ground water.

Factors affecting nitrogen loads

Figures 3–5 show the effect of slope gradient, rate of N application, soil organic matter content, and tillage practices on nitrate loads in water moving through the soil profile TOTN for tile-drained loamy, sandy, and clay soils. The results presented in these figures are the mean of the simulated values for various variables considered for the analysis.

Although many of the factors had a significant effect on the nitrate loads in these different sinks (based on ANOVA tests), some factors tended to be more important and showed definite trends. These data show that slope gradient has little to no effect on the magnitude of nitrate loads in water percolating through the soil. However, slope gradient can effectively increase the amount of sediment-bound nitrogen loads in surface runoff, hence reducing the amount of nitrogenous material at the soil surface, as shown in Table 8.

The rate of application, as defined in Table 4, tends to have the greatest impact on nitrate loads in water percolating through the soil for all type of soils (Fig. 3b, 4b, and 5b). The actual rate of application includes the amount of available nitrogen applied in manure and inorganic fertilizer. For all soil types, there is a trend of small increase in loss until the rate of application exceeds recommended rates. The recommended application rate represents a point of change in slope gradient on the curve, after which nitrate leaching losses increase substantially.

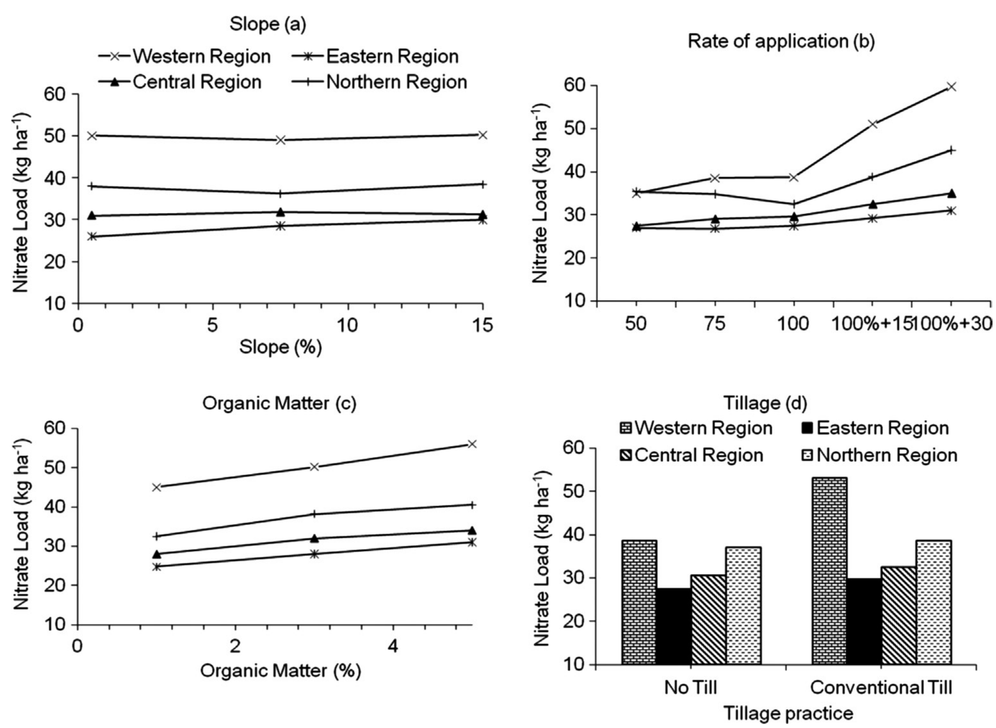

Fig. 3. Effect of slope gradient, rate of application, soil organic matter and tillage practices on nitrate loads in water moving through soil profile (TOTN or lateral flow N + tile drain + deep percolation N) for tile-drained sandy soils.

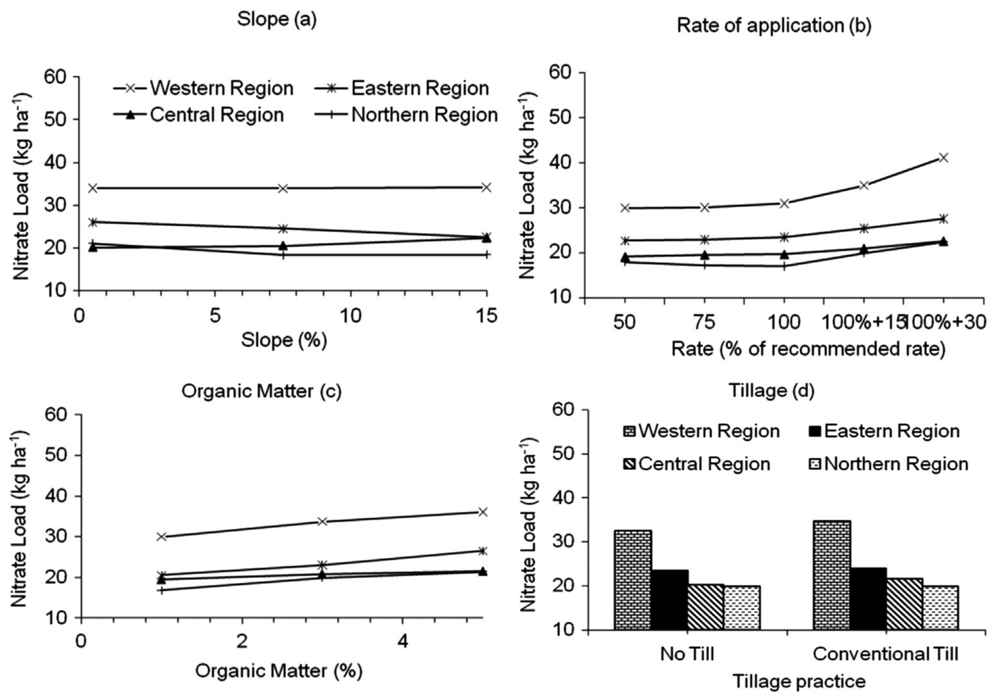

Fig. 4. Effect of slope gradient, rate of application, soil organic matter and tillage practices on nitrate loads in water moving through soil profile (TOTN or lateral flow N + tile drain + deep percolation N) for tile-drained loamy soils.

The most dramatic losses occurred in the Western and Northern regions (Fig. 3b, 4b, 5b). The Western region has the highest recommended application rates due to the longest growing season. Soils in the Northern region allow high annual drainage because of lower evaporation rates, and have lower plant uptake rates, which results in greater amounts of nitrogen available for leaching. The rate of application had an interesting effect on nitrogen loads in percolating water and crop productivity. Although, crop yield did not increase significantly with increased application rates above recommended rates, the nitrogen loads did. In addition, the current recommended rate seemed to

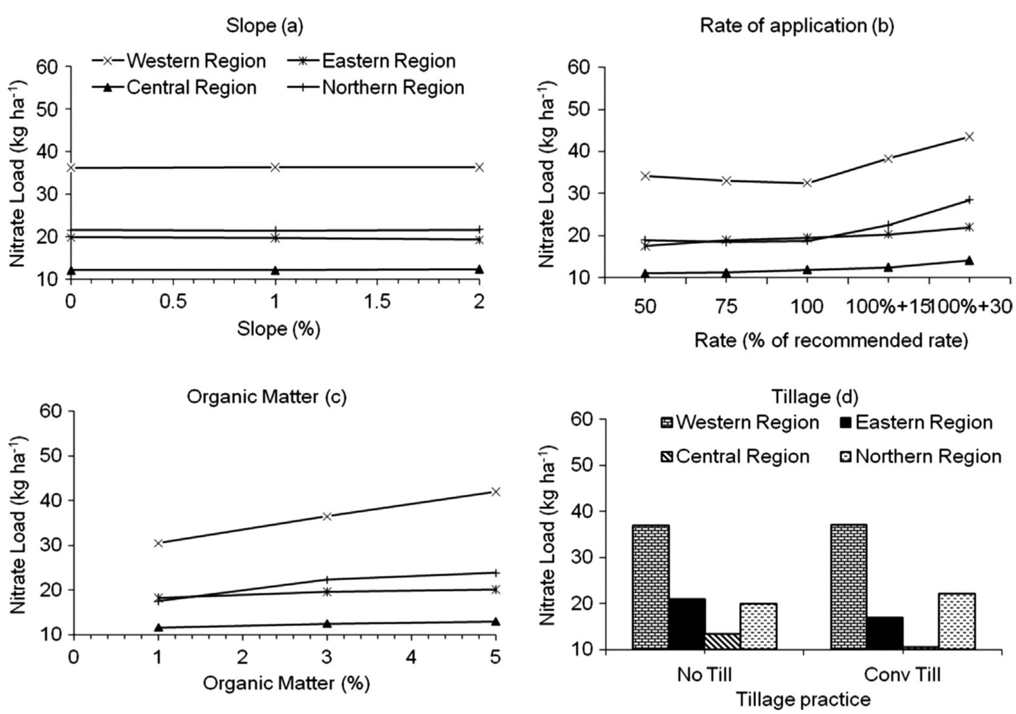

Fig. 5. Effects of slope gradient, rate of application, soil organic matter and tillage practices on nitrate loads in water moving through soil profile (TOTN or lateral flow N + tile drain + deep percolation N) for tile-drained clay soils.

represent a cutoff limit, below which nitrogen loads in percolating water did not substantially decrease.

From this discussion, current recommended rates for the different regions are probably appropriate to minimize nitrogen losses while maximizing yields. The high nitrogen losses from the Western region may result not only from higher nitrogen application rates, but also from the higher probability of leaching events caused by the longer cropping period. Also, the Western region's soils may be more susceptible to intermittent freeze-thaw periods in the winter season, which can lead to substantial loads to groundwater in the non-cropping period.

The soil organic matter content also had substantial effects on nitrate loads in percolating water through the soil profile (Fig. 3c, 4c, and 5c). For the Western and Northern regions, leaching responses were most affected by soil organic matter. In the Western region, a longer growing season promotes higher annual mineralization rates with higher organic matter. In the Northern region, high application rates with respect to crop uptake may also lead to higher mineralization rates with higher soil organic matter content. Higher mineralization rates promote higher nitrogen availability for leaching at the occurrence of high drainage events.

Change in tillage practices generally had small effects on nitrate loads in percolating water in all regions (Fig. 3d, 4d, and 5d). Converting to conventional tillage from no-till practices translates into an increase in annual nitrate loads for sands and loams, and a decrease in nitrate loads for clays. Drainage from sands and loams may increase during the winter season if tillage is implemented in the fall due to the ridge-till effect. Based on the simulation results, nitrate loads were highest for these soil types during the un-cropped season. In clay soils, a large amount of precipitation and snowmelt events during the growing season may result in more runoff than drainage. Also, the high content of fines in clay soil makes the organic particles more vulnerable to be lost by erosion processes. Subsequently, less nitrogen is available to be leached during the growing season.

Figure 6 summarizes the effect of tile drainage on the amount of nitrogen moving into and through the soil matrix for different soil types (TOTN for tile-drained conditions and GNDNII for non tile-drained). Although, the main effects are not presented for non tile-drained soils, the trends were similar. The tile drainage does have substantial effects on the nitrate loads in percolating water due to enhanced drainage. The enhanced drainage of water from the soil matrix by way of tile drainage allows for more rapid movement of water downward. Nitrate contained within the soil can be displaced with the water and becomes less available for plant uptake and immobilization processes. In addition, in soils with higher bulk densities, such as clays and loams, there is a greater possibility of saturated conditions when the soil is not drained, allowing higher denitrification rates and lower leaching loads. This process is most apparent in clay soils.

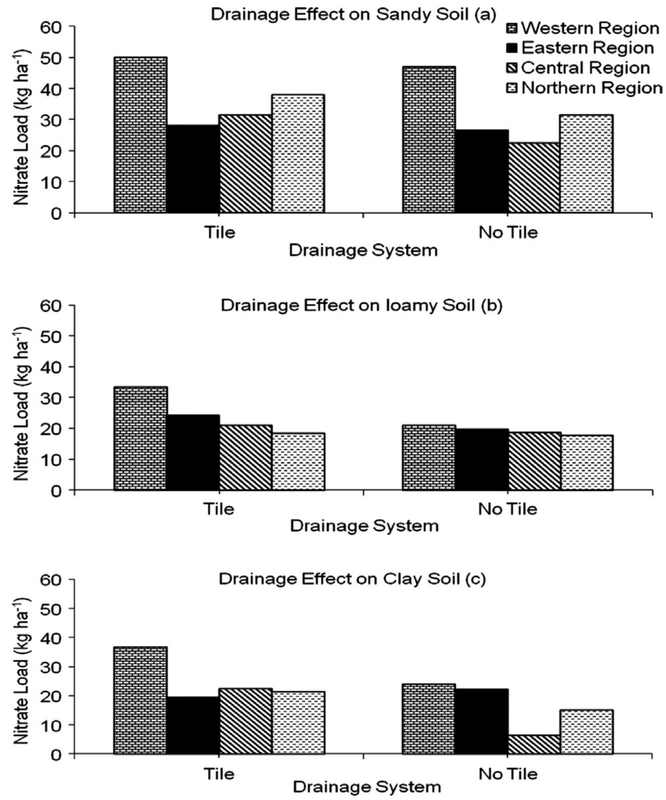

Fig. 6. Effects of artificial drainage on nitrate loads in water moving through soil profile for different soil types.

Multiple regression analysis

The objective of the statistical analysis was to develop predictive equations based on the EPIC simulation results. Based on selection criteria for implementation within MCLONE4, it was important to choose a method that could properly weigh different factors in different scenarios and compute the magnitude of nitrogen loss. Multiple linear regression was adopted as it has been proven as an appropriate tool in developing equations to approximate output from simulation models (Lee and Wang 2003). This regression method considers k independent variables (x_1, x_2, ..., x_k) from which a dependent response is obtained (Walpole and Myers 1993):

$$\hat{e} = b_0 + b_1 x_1 + \ldots + b_k x_k \quad (1)$$

Where $\hat{e}$ is estimated response, b is regression coefficients, and x_i is an independent variable (Eq. 1). The independent variables can represent main effects, interaction terms or multiple degrees of the main effects depending on the nature of the response of each factor.

To select the most appropriate regression model, all possible models were ranked in order of their R^2 value within the SAS statistical package (SAS Institute Inc. 2000). Terms within these models include all the main effects. The effects were only squared (or cubed) if the output did show a quadratic response. For example, the increase in the rate of application did not always correspond to a linear increase in nitrate loads. Interaction terms were included to evaluate whether two terms have compounding effects. The interaction terms included in the analysis were:

OM*RATE, OM*SLOPE, SLOPE*TILLAGE, and OM*TILLAGE.

Where OM is organic matter in soil, RATE is manure/fertilizer application rate, SLOPE is the average slope of the area, and TILLAGE is tillage practices. The OM*RATE term represents an increase in available nitrogen as the nitrogen application may stimulate the organic matter and lead to higher mineralization rates. The OM*TILLAGE factor is representative of the increase in mineralization rates due to incorporation of new material into the soil through conventional tillage. The SLOPE*TILLAGE and OM*SLOPE effects represent the increase in loss of organic nitrogen attached to eroded sediment after tilling a soil on a steep slope gradient. The choice of the most appropriate model was based on an adequate R^2 value (greater than 0.85), model simplicity, and whether the signs of the coefficients properly represented the system and did not violate the model assumptions.

The assumptions made in formulating a regression model are that the residuals (observation – prediction) show random fluctuations around a value of zero. Homogeneous variance is an important assumption made in the regression analysis (Walpole and Myers 1993). To test this hypothesis, residual plots of the models were also analyzed. If the variance was not constant, either transformation of the dependent variable was performed or a different model was selected. The regression equations are expressed in terms of rate (ratio), slope gradient (%), tillage (0 = no till, 1 = conventional tillage), and organic matter (% in the A horizon). The rate ratio represents the nitrogen applied/regional recommended rate for the corn crop.

The equations reflected the general trends observed earlier in the model simulated results such as rate of application and organic matter were found to be the most important in quantifying the nitrate loads in the infiltrating water for all soil types. Increasing slope gradient translated into an increasing portion of nitrate moving with subsurface lateral flow based on model results. The further analyses (not shown) also show that the equations can accurately predict EPIC's output [high R^2 value (0.88 to 0.99) and observed vs. simulated scatter plots]. These results demonstrated that the nitrogen leaching algorithm can be quite accurately predicted considering soil type, rate of application, soil organic matter, slope gradient, and tillage practice.

Evaluation of predictive algorithms

To evaluate the performance of the developed equations, their results were compared with the observed nitrogen loads, the predictions of MCLONE4, and the EPIC results for the two sites in Ontario, used earlier in the validation phase. Figures 7 and 8 show the regression results against observed loads, EPIC's predictions, and the nitrogen partitioning model in MCLONE4. In the MCLONE4 model, leaching rates were estimated as 2/3 of the annual nitrogen surplus (Goss et al. 1999). For the Ottawa site

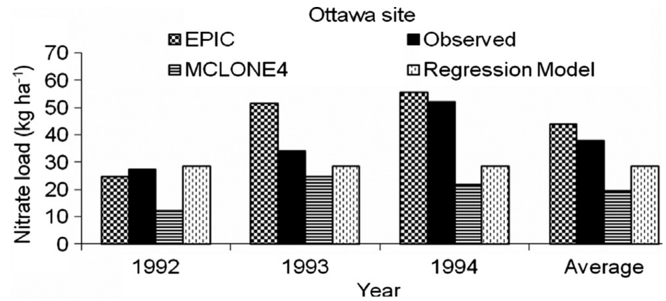

Fig. 7. Comparison of annual nitrate load in tile drained water predicted by various approaches at the Ottawa site.

(Fig. 7), there was a visible increase in observed nitrate loads every year during the study period (1992–1994). This trend was not reflected by the MCLONE4 algorithm or in the regression model because both approaches consider the nitrogen input on an annual basis and are not able to predict the accumulation of nitrogen in the soil from year to year. The EPIC-simulated results showed good agreement with the observed nitrate load data for the years 1992 and 1994. Also, the average values showed good agreement between EPIC and observed results compared with the MCLONE4 and regression model results for the Ottawa site.

At the Woodslee site, the nitrate loads were predicted well in the first year (1992) by both the EPIC and regression model, whereas the MCLONE4 algorithm overppredicted the nitrate load (Fig. 8). For the other years and the averaged values, all the approaches overpredicted the nitrate load. Alfalfa grown the previous year was considered within both approaches (MCLONE4 and regression model) by estimating the mineralization rates externally from the estimates provided by OMAFRA (1994). For the following years, both models overestimated nitrate loads in outflow. In terms of N transformations, it is possible that assumed volatilization rates of urea in these methods (at 9% of applied mineral N) or the assumed crop uptake in both the MCLONE4 and regression models were not appropriate. Also, there was a slight overestimation of nitrate loads in the EPIC simulation results.

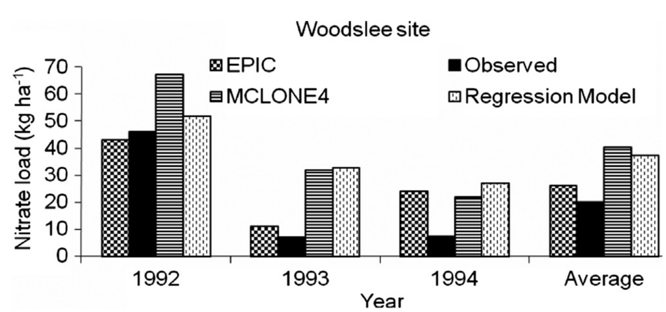

Fig. 8. Comparison of the annual nitrate load in tile drained water for various tillage regimes at the Woodslee site.

For the Ottawa site, the observed average annual nitrate loads for conventional and no-till practices were compared with those of the three prediction methods (Fig. 9). The observed nitrate loads did show slightly less nitrate load for the no-till condition, which was also reflected by the proposed model results. Only EPIC-simulated results showed overestimation of nitrate loads for the no-till system. The trend in nitrate loads was similar for all three methods for conventional tillage system. It is important to mention that the MCLONE4 module does not consider the effects of tillage. Consequently, the observed nitrate loads for both tillage practices compares better with the proposed model and EPIC than with the MCLONE4 module.

As previously discussed, a significant portion of drainage during the spring freeze-thaw period may be caused by preferential flow, which had low nitrate concentrations. Nitrate concentration predicted by EPIC, which considers piston flow, would necessarily be higher than observed under these conditions (Fig. 9). Average annual nitrate loads predicted in the proposed regression model more closely matched the observed loads than those predicted by the MCLONE4 algorithm, although neither approach considers the dynamic processes affecting the hydrology of the study area.

The comparisons of the observed vs. the developed equations, MCLONE4, and EPIC results are presented in Tables 9 and 10. Table 9 shows that the EPIC results had the highest coefficient of determination (R^2) compared to the other two approaches for the Ottawa site. However, for the Woodslee site, the R^2 values were similar for all three methods when compared with the observed ones. The RMSE values were similar for all three sites. Table 10 shows the effect of tillage practices for the Ottawa site predicted by the all three methods. It shows under prediction of nitrate loads by the EPIC model and over prediction by the MCLONE4 and regression models. These results show that tillage effect was not properly simulated by these approaches for the Ottawa site.

Overall, Figs. 7 to 9 show that the predicted nitrate loads, by both the MCLONE4 nitrogen leaching algorithm and the proposed regression model are generally estimated within the range of yearly observed field results. Also, for these sites, year-to-year variability can be quite important as maximum variability ranged from 20 to 70 kg N ha^{-1} yr^{-1} on the two sites. Results from the two predictive approaches both underestimated and overestimated the field observations and were generally comparable in magnitude to one another. The strength of the proposed regression equations over the current MCLONE4 module is shown by the reduced annual deviation from the observed nitrogen loads. Also, annual average observed nitrate loads tend to approach the proposed regression model output more closely for the Ottawa site.

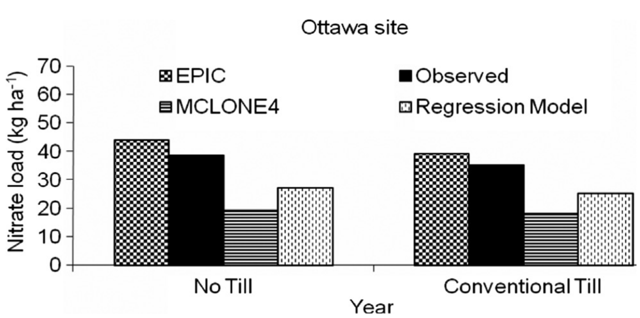

Fig. 9. Comparison of annual nitrate load in tile drained water for no-tillage and conventional tillage predicted by various approaches at the Ottawa site.

Table 9. Statistical evaluation of the nitrate loads obtained by various approaches for the study.

	Statistical analysis			
	Ottawa site		Woodslee site	
Model	R^2	RMSE	R^2	RMSE
EPIC	0.63	17.91	0.84	17.16
MCLONE4	0.31	35.32	0.95	35.76
Regression model	0.49	24.58	0.94	32.89

Long-term pollution level assessment

The long-term nitrogen leaching results from the EPIC model were used to determine the level of pollution of different scenarios in different locations in Ontario. In terms of water partitioning, the groundwater contributions consist of the deep percolating water and subsurface lateral flow. Water surplus or streamflow results are based on overland flow (runoff) and base flow. It was assumed that the base flow rate is governed by groundwater recharge (deep percolation + subsurface lateral flow) without accumulation in the groundwater system. In the case of the tile drainage system, the tile drainage volume was assumed to discharge at the soil surface and added to streamflow.

The average annual amounts of water in different streams were obtained from the simulated water balance in the different regions of the province. Thirty-year average values of surface and groundwater flows were compiled from all 180 developed scenarios for three types of soils in each region as described in Table 6. The amount of water was averaged over a range of slope gradients and tillage practices, which is typical of a watershed area contributing to a water source. Table 11 presents the average amount of water surplus (runoff + subsurface flow + percolation), and the percentage of water contributing into the groundwater system based on the results of EPIC simulations.

Table 10. Evaluation of the effect of tillage practices using various approaches for the Ottawa site.

	Management practices	
Model	No tillage	Conventional tillage
EPIC	−14%	−11%
MCLONE4	51%	49%
Regression Model	29%	29%

Table 11. Average annual water surplus[α] (mm) for various regions in Ontario.

Soil type	Regions			
	Western	Eastern	Central	Northern
		Tile-drained		
Sand	388 (32%)[†]	404 (31%)	375 (27%)	494 (30%)
Loam	336 (31%)	387 (25%)	307 (32%)	437 (24%)
Clay	312 (3%)	383 (6%)	307 (3%)	439 (7%)
		Not tile drain		
Sand	372 (70%)	395 (69%)	370 (66%)	485 (72%)
Loam	324 (60%)	379 (57%)	307 (61%)	432 (69%)
Clay	299 (26%)	382 (43%)	308 (29%)	438 (47%)

[α]Water surplus (overland flow + groundwater recharge).
[†]Value represents surplus water (mm), and the percentages in parentheses represent the partition of surplus water into the groundwater.

To determine nitrate concentrations potentially reaching the groundwater system, the nitrate loads within lateral flow (at the end of the field) and percolation water are divided by the water volumes as shown in the following Eq. 2.

$$N_{gconc} = (N_{lat} + N_{perc})/(WS \times \%GW)*100 \quad (2)$$

where N_{gconc} is the nitrate concentration in water potentially reaching the ground water (mg N L^{-1}) = the groundwater N index, N_{lat} is the nitrate load in water flowing laterally out of field (other than tile drainage) (kg N ha^{-1}), N_{perc} is the nitrate load in water flowing vertically below root zone (percolation) (kg N ha^{-1}), $WS$is the water surplus (mm), $\%GW$ is the percentage of water surplus reaching the groundwater (from lateral flow and percolation), and 100 is the conversion factor.

In determining the surficial nitrogen concentration, the total nitrogen load displaced by water is divided by total water surplus as shown in the following Eq. (3).

$$N_{sconc} = (N_{perc} + N_{lat} + N_{drain} + N_{sed} + N_{run})/ WS \times 100 \quad (3)$$

where N_{sconc} is the nitrogen concentration in surface water (mg N L^{-1}) = the surface water N index, N_{lat}, N_{drain}, N_{perc} are the nitrate loads in lateral, tile drainage and percolation water, respectively (kg N ha^{-1}), N_{sed}, N_{run} are the nitrogen loads in sediment-bound and dissolved forms in overland runoff (kg N ha^{-1}), WS is the wWater surplus (mm). and 100is the conversion factor.

This method assumes that all nitrogen entering the groundwater system will eventually enter a stream or lake through the base flow, and is not transformed while it is present in the groundwater system.

Finally, in order to determine the pollution potential of the site, the highest concentration from the two computations is considered.

$$N_{\lim} = Max(N_{gconc}, N_{sconc}) \quad (4)$$

where N_{lim} is the limiting concentration indicative of the site's pollution potential (mg N L^{-1}). Based on this procedure, the previously described scenarios were classified into three degrees of pollution potential: (a) low: 0–10 mg N L^{-1}, (b) medium: 10–20 mg N L^{-1}, and (c) high: >20 mg N L^{-1}. The 10 mg N L^{-1} limit is based on current drinking water standards. This procedure considers that all water contributions originate from the field where manure is being applied. In many cases, water contributing to an aquifer or surface water also originates from non-cultivated surrounding areas. If one considers that half of the water contribution originates from surrounding areas, which has a very low nitrogen concentration, the water from the manured field may be twice as concentrated before the nitrogen concentration within the sink reaches the critical limit of 10 mg L^{-1}. Thus, the 20 mg L^{-1} mark was established as the medium pollution potential indicator.

From the 540 previously simulated scenarios in each region, the most critical pollution potential was computed and classified according to the N index (Table 12). These EPIC-simulated results show general trends of nitrogen loads in receiving waters and should be considered within the context of EPIC's applicability. For example, trends shown in Table 12 generally show that pollution potential from clay soils is lower than that from soils with different textures. However, as previously discussed at the Woodslee site, preferential flow can be an important component of the water balance when soil cracking occurs (Diiwu et al. 2000). If soil cracking occurs, the pollution potential of clay soils would be higher than is shown in Table 12.

In the case of clay soils, N_{gconc} was never considered for tile-drained soils because of low simulated percolation volume below the tile drains (<30 mm). In the case of clay soils, which were not tile-drained, the annual volume of water percolating below the root zone was higher (>100 mm), and the most critical nitrogen concentration was found to be most often in the groundwater. In addition, the N concentrations were often higher in the groundwater for the non tile-drained sites than in the surface water of the tile-drained sites of clay soils.

For all regions, the degree of pollution potential tends to be similar for both sandy and loamy soils. Although the nitrogen load in percolating water is smaller in loamy soils, its flow volume is also smaller, resulting in concentrations comparable with those in the sandy soils. In addition, for these two soil types, drained soils tend to have higher pollution potential simply due to the fact that higher leaching loads are likely to result from the rapid drainage in tile-drained conditions.

With regard to the sink of most limiting nitrogen concentration, the trend shows that groundwater nitrogen concentrations were most often higher than surface waters for sandy soils (62–100%). Surface water and groundwater both contributed comparably to the limiting nitrogen concentrations in loamy soil conditions (2–100% of instances were limited by groundwater concentrations). Also, instances of high pollution potential caused by high concentrations in surface waters can be more easily

Table 12. Simulated scenarios in each pollution potential category for various soil texture in different regions of Ontario.

Soil Type	Tile-drained			Not tile drained		
	High[§] (≥20)	Medium (10–20)	Low (≤10)	High (≥20)	Medium (10–20)	Low (≤10)
			Western			
Sand	63%	37%	0%	21%	79%	0%
Loam	48%	52%	0%	12%	88%	0%
Clay	0%	100%	0%	80%	20%	0%
			Eastern			
Sand	22%	73%	4%	0%	83%	17%
Loam	22%	78%	0%	0%	54%	46%
Clay	0%	0%	100%	0%	0%	100%
			Central			
Sand	33%	35%	31%	0%	36%	64%
Loam	0%	100%	0%	0%	98%	2%
Clay	0%	0%	100%	0%	0%	100%
			Northern			
Sand	4%	88%	8%	0%	32%	68%
Loam	0%	100%	0%	0%	92%	8%
Clay	0%	0%	100%	0%	14%	86%

[§]These values represent concentrations (mg N L^{-1}).

remedied through treatment or filtering systems at the end of the field or at the tile outlet. This discussion shows that, according to EPIC simulation results, sandy soils will most heavily contribute to groundwater contamination, from which prevention of nitrogen contamination may be most difficult.

Application to nitrogen management

Since the developed regression equations are able to identify those conditions that are more vulnerable to nitrogen losses in water, one can easily identify scenarios which require more attention to prevent nitrogen contamination. They are also effective at identifying sites that may be unsuitable for intensive cropping. In addition, by differentiating groundwater from surface water, it is possible to target management strategies appropriately.

To properly target nitrate management strategies, nitrogen should only be applied at the optimum rate when crops can quickly take up the nitrogen and prevent excessive downward water movement. Also, damping effects caused by the slow release of nitrogen from organic materials can be very useful in minimizing the nitrogen losses. However, for N management strategies, it is important to identify events that are most likely to cause heavy drainage and/or runoff. During these events, it is important to minimize the amount of available nitrogen.

CONCLUSIONS and RECOMMENDATIONS

The simulation results in this study represent a realistic average annual water and nitrogen budget, and a number of conclusions were made:

- Tile drainage can have a significant impact on nitrogen load in surface water, as the tile drains promote downward water movement and nitrate through the soil profile. Most nitrogen movement normally occurs with tile drainage water, mostly in late fall and early spring as a result of less soil cover, high nitrogen availability due to residual N, wetter soil conditions, and events of large magnitude caused by high precipitation and/or snow-melt.
- The rates of nitrogen application and soil organic matter content are important factors affecting the nitrate loads in percolating water.
- Soil type has a significant impact on nitrogen loads in percolating water. Sandy soils have the highest pollution potential, followed by loam and with the lowest for clay.
- Due to climatic conditions, certain regions were more vulnerable to leaching losses. Pollution potential is likely to be highest in the Western region due to higher fertilizer application rates and longer freezing and thawing period.

The equations generated from the regression analysis can provide useful long-term predictions based on a few easily determined variables. In addition, this method is less sensitive to changes in mineralization rates or expected yields than the annual nitrogen budget approach.

Limitations of the chosen approach

- The regression models represent long-term average results; however, they do not represent annual variability caused by fluctuations in temperature, precipitation, soil moisture content, plant uptake patterns, etc.

- The regression algorithms do not consider shallow soil conditions (less than 1.2 m) or other soil conditions that can lead to higher drainage rates (i.e., karst geology). Since the regression models assume corn crop production, the effect of accumulating organic matter on nitrogen leaching has been neglected, as it might occur in other cropping regimes (i.e., legumes). To get more accurate results, the initial soil nitrogen amounts must be included as part of the rate of nitrogen application. However, in order to remain within the limits of the rate parameter of the regression model, the rate of N application must not exceed 30 kg N ha^{-1} above the provincial nitrogen recommended rates by OMAFRA (1994) after volatilization losses are considered.
- The equations developed do not consider the fate of nitrogen beyond the edge of the field or below the root zone; thus, it is included in the "potential" losses. In light of the limitations to the developed regression models, and EPIC model's limitations in describing the effects of nitrogen loads in subsurface drainage water, the model's results have limited accuracy.

As for future directions, the knowledge acquired by such model applications can help in developing an index which could include excess nitrogen from the annual budget computation. Also, hydrological factors such as soil type, soil moisture at application, probability of a significant drainage or runoff event shortly after application, could also be included and weighted according to their relative importance as indicated by model simulations.

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