Environmental Management (2010) 45:1201–1222
DOI 10.1007/s00267-010-9448-8

Environmental Indicators to Assess the Risk of Diffuse Nitrogen Losses from Agriculture

Uwe Buczko · Rolf O. Kuchenbuch

Received: 9 March 2009 / Accepted: 21 January 2010 / Published online: 20 March 2010

Abstract Diffuse Nitrogen (N) loss from agriculture is a major factor contributing to increased concentrations of nitrate in surface and groundwater, and of N_2O and NH_3 in the atmosphere. Different approaches to assess diffuse N losses from agriculture have been proposed, among other direct measurements of N loads in leachate and groundwater, and physically-based modelling. However, both these approaches have serious drawbacks and are awkward to use at a routine base. N loss indicators (NLIs) are environmental management tools for assessing the risk of diffuse N losses from agricultural fields. They range in complexity from simple proxy variables to elaborate systems of algebraic equations. Here we present an overview of NLIs developed in different parts of the world. NLIs can be categorized into source-based, transport-based, and composite approaches. Several issues demand more attention in future studies. (1) Is incorporation of leaching losses and gaseous losses into one single NLI warranted? (2) Is it sufficient to restrict the focus on the rooted soil zone without considering the vadose zone and aquifer? (3) Calibration and validation of NLIs using field data of N loss seems not sufficient. Comparisons of several different NLIs with each other needs more attention; however, the different scaling of NLIs impedes comparability. (4) Sensitivity of input parameters with regard to the final NLI output needs more attention in future studies. (5) For environmental management purposes, factors addressing management decision by farmers deserve more attention.

Keywords Environmental indicator · Agricultural nutrient management · Risk assessment · Nitrate leaching · Nitrogen loss indicator · Non-point source pollution

Introduction

Diffuse nitrogen (N) losses from agricultural fields are the major cause of increasing nitrate concentrations in ground- and surface waters and have been an environmental concern since several years (e.g., Bach 1987; Strebel and others 1989; Wendland and others 1993; ten Berge 2002; Behrendt and others 2003; Delgado and others 2008). Excessive nitrate concentrations can have toxic effects in drinking water (e.g., Townsend and others 2003, but see also the critical discussion in Powlson and others 2008) and cause eutrophication in surface waters (Vitousek and others 1997; Wolfe and Patz 2002). Gaseous N losses in form of N_2O are an important factor in global warming and the destruction of the stratospheric ozone layer (IPCC 2007), whereas ammonia volatilization contributes to soil acidification and eutrophication (Follett and Delgado 2002). Moreover, fertilizer and manure N that is not used by growing crops but lost to the environment instead represents an economic loss.

For management and environmental planning purposes, it is necessary to assess the risk and magnitude of diffuse N losses from agricultural fields and how they are impacted by management practices, climate and weather, soil properties, etc. (e.g., Meisinger and Delgado 2002; Havlin 2004).

U. Buczko (✉)
University Rostock, Justus-von-Liebig-Weg 6, 18059 Rostock, Germany
e-mail: Uwe.Buczko@uni-rostock.de

R. O. Kuchenbuch
Agricultural Analysis and Research Institute (LUFA) Rostock, Graf-Lippe-Straße 1, 18059 Rostock, Germany

Utilization of experimental methods to determine actual amounts of N losses, such as analysis of leachate water obtained by suction cups (Pamperin 2002; Sieling and Kage 2006), pan lysimeters (Jemison and Fox 1994), monolith lysimeters (Chichester 1977; Bohne and others 1997; Knappe and others 2002), analysis of percolate from tile drains (Hofmann and others 2004; Tiemeyer and others 2008), analysis of groundwater samples (de Ruijter and others 2007), and also the N-min method (Wehrmann and Scharpf 1979), is restricted because routine application of such labor-demanding methods is mostly not viable, measurements can be made only after management decisions have been taken (i.e., too late), and the experimental data are often not amenable to generalization (because of the effects of different years with varying weather patterns, different management practices, fertilizer application rates, etc.).

On the other hand, with more or less complex, physically based N transport models (e.g., Wu and McGechan 1998; Ma and Shaffer 2001; Cannavo and others 2008), it is—at least in principle—possible to quantify N losses for various environmental conditions and management practices (e.g., Gollany and others 2004). However, in general such models require many input data, contain many weakly constrained parameters and are often difficult to operate. All these factors severely restrict a routine use of physically based models for assessment of N loss from agricultural fields.

As an alternative, simplified models have been developed since about two decades for use as indicator (or index) approaches for N loss assessment (in the following termed NLI = Nitrogen Loss Indicator, plural: NLIs) (e.g., Follett and others 1991; Shaffer and Delgado 2002; Schröder and others 2004; Magette and others 2007; Delgado and others 2008; Bockstaller and others 2008). Although broadly related with environmental indicators (e.g., Villa and McLeod 2002; Rees and others 2008), agri-environment indicators (Bockstaller and others 2008; Hajkowicz and others 2009), and groundwater vulnerability indicators (e.g., Mazari-Hiriart and others 2006; Neukum and others 2008), NLIs nevertheless are a distinct group of environmental pollution risk indicators, focussed on assessment of non-point source pollution of nitrogen compounds (mainly nitrate) from agriculture. The various NLI approaches differ with respect to their complexity, incorporated loss processes, data requirements and type of output (e.g., risk classes, quantified amounts of N loss, etc.). To support a decision as to which NLI method could be suited for a given site, management options, and data availability, a comprehensive and global overview of NLIs is needed. Previous studies concentrated on specific aspects of NLI approaches: Shaffer and Delgado (2002) presented a review of N soil leaching indices developed in the USA. Bockstaller and others (2008, 2009) discussed several NLIs (i.e., agri-environmental indicators for N loss in their terminology) developed and used in France. Ten Berge (2002) and de Ruijter and others (2007) reviewed and tested several NLIs used in the Netherlands. Schröder and others (2004) reviewed several basic nutrient loss indicators such as manure input or nutrient balances. Magette and others (2007) and Delgado and others (2008) reviewed tersely several NLIs.

Here we want to provide a comprehensive and critical overview of various NLI approaches from different parts of the world which have been developed during the past decades. We discuss advantages and disadvantages of the NLIs and outline the major problems and open questions which should be dealt with in future studies.

Overview of N Loss Indicators

The purpose of a NLI is to assess the potential N loss (or the risk of N loss) from agricultural fields, based on relatively simple and generally available input data (e.g., Schröder and others 2004; Bockstaller and others 2008; Delgado and others 2008). Compared with phosphorus indices (Sharpley and others 2003; Buczko and Kuchenbuch 2007), NLI approaches vary more widely in their structure and incorporated processses, which may be due primarily to the complexity of the N cycle (Delgado and others 2006). The focus of most NLIs is on N loss by soil leaching into groundwater or tile drains. The time scale of NLIs is usually one year (i.e., the crop cycle), and the spatial resolution the field or farm scale. The point of reference can be (1) the soil zone (“which amounts of N are lost from the soil?”), (2) the surface of the water table (“what nitrate loads possibly enter the groundwater surface?”), (3) the groundwater system (“what nitrate concentrations can be expected in the groundwater?”), or (4) the groundwater outflow into surface waters (“what nitrate loads remain after groundwater passage?”).

Some NLI approaches yield dimensionless scores which are rated into vulnerability classes, whereas in some approaches nitrate loads or nitrate concentrations in percolation water or groundwater are quantified. Compared with relatively complex physically based models of N dynamics, NLI approaches use a larger, integrated timescale (at least 1 year), data requirements are less demanding, and the output is in general qualitative or semiquantitative and evaluated in terms of risk classes (“low”, “medium”, “high”, or similar). Whereas several composite NLIs allow to indicate which factors cause increased N losses and which management options could be chosen to reduce N losses, this is not so straightforward with simpler NLIs. On the other hand, calculation of complex NLIs can require many data and in fact be too demanding for routine use. Therefore, the utilization of either complex or simple NLIs may be preferable, depending on data availability and requirements as regards the output and the conclusions which shall be drawn from the calculations.

Although somewhat different classification schemes for NLI approaches have been presented (e.g., Bockstaller and others 2008), we use here a classification into (1) NLIs based on source terms, (2) NLIs based on transport terms, and (3) composite NLI approaches (Table 1). NLIs based on transport terms are divided into groundwater vulnerability indices (abbreviation "TG") and approaches focussed on the hydrology of the soil zone (abbreviation "TS"). The composite NLIs are classified into approaches based on scores (abbreviation "CS") and quantitative models with (predominantly) physical units, either calculated using a single equation (abbreviation "CE") or by more complex sets of several equations (abbreviation "CC"), which in many cases consider different N loss processes (Table 1).

In the following sections, the NLIs are described according to this classification. Although we endeavoured to describe the different NLIs in comparable detail, somewhat more attention is focussed on more widely used, well-documented and general applicable NLIs (such as DRASTIC or EF), compared with NLIs which seem to be more local and/or in an experimental stage of development (such as NO-NI or OMAFRA-NI). The pertinent factors and properties of all NLIs discussed in this review are compiled in Table 2.

NLI Based on Source Terms

Several relatively simple NLIs have been proposed based entirely or predominantly on N source terms (e.g., ten Berge 2002; Shaffer and Delgado 2002; Schröder and others 2004; de Ruijter and others 2007; Bockstaller and others 2009). In principle, the N amount which is possibly available to diffuse N losses can be assessed by two different types of approaches: either by calculating a N input/output balance, usually on an annual basis, or by measuring directly the mineral N content in the soil profile, usually immediately before the start of the main leaching period, i.e. in autumn for climatic conditions of central Europe or North America. Although both natural and anthropogenic N sources are considered in these approaches, the proportion of anthropogenic N will be more important in most agroecosystems.

N Balance (NBal, S1)

N balances are among the most common NLIs used in the EU (Goodlass and others 2003). They can be calculated for a whole farm ("farm-gate balance"), for the soil surface, or for the soil system (Oenema and others 2003). Soil system balances are the most detailed and would be preferable in most cases for utilization as a NLI.

In its most basic form the annual N balance (NBal, S1, a list of abbreviations is compiled in Table 3 of the Appendix) comprises merely N fertilizer application rates minus N export by harvested products. Calculation of a complete soil system N balance which accounts for all possibly relevant components of input (fertilizer/manure application, atmospheric deposition, mineralization, N from crop residues) and output (extraction by harvested crops, immobilization, ammonia volatilization, denitrification, soil leaching, erosion, surface runoff), is rarely feasible, because the necessary data are mostly not available. This applies especially when immobilisation and mineralization of N are not in equilibrium (i.e., for shifting cultivation, changing nutrient inputs, or changing soil carbon pools). N balances have been evaluated as a NLI in several studies (e.g., van Eerdt and Fong 1998; Jansons and others 2003; van Beek and others 2003; Sieling and Kage 2006; Rankinen and others 2007; Schröder and Neeteson 2008). Some authors use the term "N available to leaching" instead (e.g., Shaffer and Delgado 2002).

However, when N balances are calculated simply as the difference between N application rate and N extraction by harvested crops (e.g., Bach 1987; Sieling and Kage 2006), without accounting for dynamic changes in the N status of the soil (mineralization, immobilization) (Lord and others 2002; Oenema and others 2005), the N balance proves to be a poor predictor of N amounts lost actually to the environment on the scale of a single year, and is only weakly correlated with N losses actually measured (Schröder and others 2004; Sieling and Kage 2006; de Ruijter and others 2007; Rankinen and others 2007). A further drawback is that N balances per se yield no information about the pathways of N loss and which factors could possibly contribute to N losses. Better correlations between N balances and measured N loss are usually obtained when longer-term data are considered (e.g., Haferkorn 2000; Sieling and Kage 2006) or for grassland (Lord and others 2002; Ten Berge and others 2002).

In essence also a N balance approach, the Nitrogen Risk Index for the Lombardy region of northern Italy (Provolo 2005) is based primarily on N applications at the farm scale and the N requirements of the crop.

"EQUIF" ("EQUIlibre de Fertilisation", S2), is a NLI based on N balances used in France. It accounts for N uptake by crops and fertilizer application, N mineralization of soil organic matter and crop residues, and measured soil mineral N contents in spring (CORPEN 2006; Aveline and others 2009). EQUIF values are calculated as kg N/ha and

Table 1 Overview of NLIs and related approaches to assess the risk of N loss from agricultural fields

Main group	Sub-group	NLI (acronym)	Reference[a]	Loss processes
Source-based (S)		(S1) N balance (NBal)	Oenema and others (2003)	NL
		(S2) "EQUIlibre de Fertilisation" (EQUIF)	Aveline and others (2009)	NL
		(S3) Residual soil mineral nitrogen (RSN)	Schweigert and Zimmermann (2003)	NL
		(S4) N application rate (NFertApp)	Bockstaller and others (2009)	NL
		(S5) N use efficiency (NUE)	Shaffer and Delgado (2002)	NL
		(S6) N concentrations of maize plants at silage maturity (Ncm)	Herrmann and others (2005)	NL
Transport-based (T)	Groundwater vulnerability (TG)	(TG1) DRASTIC	Aller and others (1987)	NL, VZ, GW
		(TG2) Protection function of the vadose zone ("Schutzfunktion der Grundwasserüberdeckung"—SG)	Hölting and others (1995)	NL, VZ
		(TG3) Aquifer Vulnerability Index (AVI)	van Stempvoort and others (1993)	VZ
		(TG4) Multivariate Logistic Regression for nitrate contamination of groundwater (MLR)	Nolan (2001)	NL, VZ, GW
	Based on hydrology of soil zone (TS)	(TS1) Exchange frequency of the soil solution (EF)	Müller (2004)	NL
		(TS2) Drainage index ("Indice de drainage") (P/RU)	CORPEN (2006)	NL
		(TS3) Leaching Index (LI)	Williams and Kissel (1991)	NL
Composite NLI (C)	Based on scores (CS)	(CS1) Colorado vulnerability map (CO-VM) and matrix (CO-VMX)	Ceplecha and others (2004)	NL, VZ
		(CS2) "Environmental Sustainability" (EnSus)	Woods and others (2006)	NL, SR
		(CS3) modified Nitrogen ranking scheme (mNRS)	Magette and others (2007)	NL, SR
		(CS4) Nitrate Leaching Hazard Index for Irrigated Agriculture (NLHI-IRR)	Wu and others (2005)	NL
		(CS5) Nonpoint-Source Agricultural Hazard Index (NPSAH)	Trevisan and others (2000)	NL
		(CS6) N Index of the Ministry of Agriculture, Food and Rural Affairs in Ontario (OMAFRA-NI)	OMAFRA (2003)	NL
		(CS7) Pennsylvania N Index (PA-NI)	Heathwaite and others (2000)	NL
	Model-equation (CE)	(CE1) Indicator risk of water contamination by nitrate-nitrogen (IROWC-N)	De Jong and others (2007)	NL, (DN)
		(CE2) Potential nitrate concentration in leachate (PNCL)	Bach (1987)	NL, (DN)
	Model-complex NLI (CC)	(CC1) Annual Leaching Risk Potential (ALRP)	Pierce and others (1991)	NL, (AV, DN,) VZ
		(CC2) "$I_{\mathrm{N\ losses}}$" indicator (IN)	Pervanchon and others (2005)	NL, AV, DN, NO
		(CC3) Methodology for the evaluation of the risk of nitrate leaching (Méthode d'Evaluation des Risques de Lixiviation des Nitrates)(MERLIN)	Aveline and others (2009)	
		(CC4) N-Index Tier-1 (NIT-1)	Delgado and others (2008)	NL, AV, DN, ER, SR, (VZ)
		(CC5) Norway N index (NO-NI)	Bechmann and others (2009)	NL, DN, ER, SR

[a] Although for some NLIs more than one reference apply, only one ref. is cited here

NL leaching, *VZ* flow through vadose zone, *AV* ammonia volatilization, *DN* denitrification, *ER* erosion, *SR* surface runoff, *GW* groundwater flow, *NO* NO (nitric oxide) emission

Table 2 Overview of N loss indicator approaches

NLI	Climate/ weather	Soil	Management	N-sources	Off-site and belowground factors	Type of equations	Type of output	Sensitivity (%)	Threshold of NConc or NLoss for "high" vulnerability[a]	Application (A)/ validation (V)
NBal (S1)	–	–	indirect (crop, yield)	NFertApp, NCUpt, (NMin, Denitr)	–	Sums input vs. output of N	Amount kg N ha^{-1} (yearly)	Depends on difference between N input and output, often >100%	60 (sand) to 100 (clay) kg N ha^{-1}	A: commonly used in Europe; V: mostly poor correlations with measured N losses
EQUIF (S2)	–	–	Indirect (crop, yield)	NFertApp, NCUpt, NMin (humus, crop residues), RSN	–	Sums input vs. output of N, measured data (RSN)	Amount kg N ha^{-1} $year^{-1}$, rated into 6 classes	Similar as for NBal	40 kg N ha^{-1}	V: with meas. RSN(h)
RSN (S3)	–	–	–	Measured RSN in different soil depths (0–90 cm)	–	Measured data	Amount kg N ha^{-1} at fixed date (before leaching season)	Depends on accuracy of measurements	40 kg N ha^{-1}	A: several countries (Germany, Netherlands, USA); V: NConc(l),(gw)
(S4) N application rate (NFertApp)	–	–	–	NFertApp	–	Data provided by farmer	Amount kg N ha^{-1} $year^{-1}$	Depends on accuracy of data	170 kg N ha^{-1} $year^{-1}$	A: e.g. France, Netherlands
NUE (S5)	–	–	Indirect (crop, yield)	NFertApp, NCUpt	–	NCUpt/NFertApp × 100	%- Values	Similar as for NBal	na	V: NLoss by leaching, irrigated systems
Ncm (S6)	Prec(a) (qualitatively classified as "wet" or "dry")	–	–	Measured N conc. in plants (maize) (Ncm)	–	Exponential regression Ncm vs. NConc(l), critical Ncm for NConc(l) = 11.3 mg NO_3–N l^{-1}	Critical Ncm 10.5 (wet years)/7.82 g (dry years) N kg^{-1} dry matter	>>100 %	NConc >11.3 mg NO_3–N l^{-1}	A,V: simulated NConc(l)
DRASTIC (TG1)	-(Recharge)	Soil texture	–	–[b]	Depth to groundwater, (recharge), aquifer type (porosity), HC of vadose zone and aquifer, topography	Weighted sum of 7 factors (with scores 1–10)	Scores, range 23–230	4–22% (depending on weighting factor)	na	A, V: NConc(gw), different sites worldwide
SG (TG2)	SeepRate (recharge)	aWHC(rz)	–	–	Sediment type and thickness of vadose zone layers	Sum of soil and vadose zone terms (scores)	Score, range about 500–2000, higher scores denote lower vulnerability	20% (sediment, thickness) – 100% (recharge)	na	A, V: NConc(gw) (e.g. Magiera 2002)
AVI (TG3)	–	–	–	–	Thickness (d) and HC of vadose zone layers	Sum of ratio d(i)/HC(i) of vadose zone layers (physical units)	Rh (years), log10-values rated into AVI vulnerability classes	>>100% for layers with low HC; for d and high HC < 100%	na	A, V: NConc(gw) (e.g. Magiera 2002)

Table 2 continued

NLI	Climate/ weather	Soil	Management	N-sources	Off-site and belowground factors	Type of equations	Type of output	Sensitivity (%)	Threshold of NConc or NLoss for "high" vulnerability[a]	Application (A)/ validation (V)
MLR (TG4)	–	% Well-drained soils	% Cropland/pasture	NFertApp	Population density (nat. log), depth to water table, fractures in aquifer	Multivariate logistic regression with six variables	Predicted probability of NConc(gw) >4 mg NO_3 l^{-1}	5% (NFertApp) to 55% (% cropland, depth to water table)	4 mg NO_3 l^{-1}	A: GW vulnerability map USA; V: NConc(gw) >4 mg NO_3 l^{-1}
EF (TS1)	SeepRate (= f(prec(a), ET_{pot}))	WHC(rz)	–	–	–	Transport divided by storage term (physical units)	%-Values (range 0–300)	SeepRate 100%, WHC(rz) depending on magnitude, often >>100%	na	A: Germany, various sites
P/RU (TS2)	Prec(ls)	WHC(rz)	–	–	–	Transport divided by storage term (physical units)	Dimensionless, range 1–12	Prec(ls) 100%, WHC(rz) depending on magnitude, often >>100%	na	A: France, various sites
LI (TS3)	Prec(a), prec(ls)	HSG	–	–	–	Nonlinear regression of percolation vs. prec(a)	Index with length units (e.g., inches)	Prec(a), HSG: >>100%; prec(ls) < 100%	na	A, V: USA, various sites
CO-VM (CS1)	–	Soil drainage class (index)	Land use, irrigation index	(Land use)	Depth to aquifer, presence of aquifer	Sum of scores	Scores, range 0–11	30%, except irrigation index (8%) and presence of aquifer (100%)	NConc >10 mg NO_3–N l^{-1}	V: NConc(gw) >5 mg NO_3–N l^{-1} (n = 576), $r^2 = 0.78$
CO-VMX (CS1)	–	Soil texture	Irrigation efficiency, manure/fertilizer application timing, BMP	NFertApp, NManApp	–	Sum of scores	Scores, range 1–16	25%	RSN >50 kg N ha^{-1}	V: with RSN (n = 4), $r^2 = 0.32$–0.95
EnSus (CS2)	Prec(a), ET	Profile available water (≈ aWHC(rz)), organic and hydric soils, factor for low HC	–	Land use as proxy for potential N sources	–	Source term multiplied by transport term	Scores, range 0.4–50 (rated into 5 classes)	100%; ET >100% for low values	30 kg N ha^{-1} $year^{-1}$ ("N pressure")	A: mapped for New Zealand
mNRS (CS3)	–	Preferential flow paths (including subsurface drains)	"Dirty water" applications, cropping system, farmyard risk	Nutrient application rate and timing	Aquifer vulnerability, subsoil type	Source term multiplied by transport term	Scores, range 4–448	Source factors 20–40%; transport factors 100%	Related to crop N requirements	V: grassland, NConc(gw), $r^2 > 0.35$ if N from grazing livestock considered
NLHI-IRR (CS4)	–	HC, texture	Irrigation system, crop type	–	–	Multiplication of 3 score terms	Scores, range 1–80	100%	na	A: irrigated agriculture California; V: qualitative
NPSAH (CS5)	Precipitation, temperature ("control factor")	–	Agronomic practices, irrigation ("control factors")	NFertApp ("hazard factor")	Slope ("hazard factor")	Multiplication of 5 score terms	Scaled index, range 1–10	100%	na	A: risk map for Cremona province (Italy)

Table 2 continued

NLI	Climate/ weather	Soil	Management	N-sources	Off-site and belowground factors	Type of equations	Type of output	Sensitivity (%)	Threshold of NConc or NLoss for "high" vulnerability[a]	Application (A)/ validation (V)
OMAFRA-NI (CS6)	–	HSG	Crop type, cover crop, manure application timing	NBal, manure applications after harvest	–	Sum of scores for NBal and manure applications after harvest	Scores, range 0–12, rating of scores depends on HSG	50%	135 kg N ha^{-1} $year^{-1}$	A: Ontario, Canada
PA-NI (CS7)	–	Soil texture, HC	Method of fertilizer/manure application	NFertApp, NManApp	–	Source term multiplied with transport term (scores)	Scores, range 0–32	Source factors: 25%; transport factors: 50%	150 kg N ha^{-1} $year^{-1}$	A: 40 ha experimental watershed in Pennsylvania
IROWC-N (CE1)	Prec(a), ET_{pot}	aWHC	–	NBal (NFertApp, NCUpt, NDep, NFix)	–	Ratio of source and transport term (physical units)	NConc(l)	40–100%	NL >20 kg N ha^{-1} and NConc >10 mg NO_3–N l^{-1}	A: mapped for large parts of Canada
PNCL (CE2)	Prec(a), ET_{pot}	–	–	NBal (NFertApp, NCUpt, NDep, NFix)	–	Ratio of source and transport term (physical units)	NConc(l)	NBal 100%; SeepRate variable, >>100% for small values	NConc >50 mg NO_3 l^{-1} (= 11.3 mg NO_3–N l^{-1})	A: mapped for Germany
ALRP (CC1)	Prec(a), prec(ls)	HSG, porosity	–	NBal (NFertApp, NMin, NIrrig, NFix, NCUpt, Denitr, NH_3-volat.)	Travel time to aquifer, position of aquifer, vulnerability of aquifer	4 Components	Scores which are rated into vulnerability classes	25–50%	89.6 kg N ha^{-1} $year^{-1}$	A: USA, various sites
IN (CC2)	SeepRate	Soil water retention, rooting depth	Manure type, application method and timing (for NH3-volat.)	RSN(h), NFertApp	–	Minimum of 4 sub-indicators (scaled between 1 and 10) is defined as IN score	Scores (scaled between 1 and 10) with lower values denoting higher risk	Sub-indicators: 100% when lower than others; separate components for leaching: >>100%	NConc >11.3 mg NO_3–N l^{-1}	V: sub-indicator for NL (grassland sites in France)
MERLIN (CC3)	–	Sensitivity of soils for leaching (similar to HSG)	Cover crops	NFertApp, NCUpt, NMin (humus, crop residues), RSN	–	3 Sub-indicators, combined by relationship table	3 MERLIN classes (low, medium, high risk)	50–100% (3 sub-indicators)	40 kg N ha^{-1} (RSN), NConc >11.3 mg NO_3–N l^{-1}	V: meas. NLoss (n = 125)
NIT-1 (CC4)	Prec(a), prec(ls)	HSG, porosity, bulk density, C_{org}	Rooting depth, crop rotation, fertilizer/manure application method and type, split application, tile drainage, buffer width, irrigation	NBal (NFertApp, NDep, NMin, NIrrig, NFix, NCUpt, Denitr, ammonia volat.)	(Runoff class), travel time to aquifer, position and vulnerability of aquifer, distance to surface water	Sum of 15 score terms (0–8)	Scores, range 0–124	6.2% (for each of the 15 components)	>112 kg N ha^{-1}	V: sub-indicators for leaching, NH_3 volatilization and runoff, sites in USA, Argentina, China

divided into 6 classes. It is one of the three components of the MERLIN NLI (see below).

Residual Soil Mineral Nitrogen (RSN, S3)

The other approach to assess the N amount which is possibly available to diffuse N losses is direct measurement of the mineral N content within the soil profile, usually over the rooted depth (i.e., mostly 60–90 cm, depending on the crop, Wehrmann and Scharpf 1979). This is done preferably in autumn after harvest, if the main leaching period is during winter. This amount is usually called "Residual Soil Nitrogen" (RSN), i.e., the amount of inorganic (mineral) nitrogen remaining in the soil after harvest, before the start of the winter leaching period. It is used as an indicator of possible N leaching into groundwater for instance in Germany (Schweigert and Zimmermann 2003), the Netherlands (Schröder and others 1996; ten Berge 2002; de Ruijter and others 2007), in France (CORPEN 2006), or the USA ("RN index" according to Shaffer and Delgado 2002). RSN according to this definition corresponds to nitrate in the soil measured with the "N-min" method (Wehrmann and Scharpf 1979), which is commonly used to estimate the N fertilizer demand of crops when the soil is sampled before N fertilizer application in spring (crop-specific soil depths, but mostly at 0–60 cm).

If not directly measured, RSN can be estimated from NBal (e.g., ten Berge and others 2002; De Jong and others 2007), or from the N amount in fertilizer applications (ten Berge and others 2002; Pervanchon and others 2005). RSN after harvest, RSN(h), has been correlated with soil N leaching (Chichester 1977; Roelsma 2002) and nitrate concentrations in groundwater (Schröder and others 1996; de Ruijter and others 2007).

Whereas NBal is calculated using data for a whole year, RSN is measured at one point in time. This "snapshot character" of RSN has been criticized variously, because RSN is not fixed and can be subject to changes during the leaching period (e.g., Schröder and others 2004). Moreover, to obtain a representative estimate of RSN at the field scale, a large number of samples may be necessary (i.e., more than 15 samples per ha, Ilsemann and others 2001).

Other Source-Based NLIs

The problems discussed above for N balances apply even more to the use of the N application rate as a NLI (S4), although the N application rate (modified for different crops and yield goals) is used as a NLI for instance in the Netherlands (ten Berge 2002; Schröder and Neeteson 2008), or France (CORPEN 2006).

Nitrogen use efficiency (NUE, S5) is the percentage of applied N that is taken up by the crop. In some soil-crop

Table 2 continued

NLI	Climate/ weather	Soil	Management	N-sources	Off-site and belowground factors	Type of equations	Type of output	Sensitivity (%)	Threshold of NConc or NLoss for "high" vulnerability[a]	Application (A)/ validation (V)
NO-NI (CC5)	(Erosion risk)	Erosion risk[c]	Tillage timing, split application, manure application timing	NBal (NFertApp, NDep, NFix, NCUpt, Incidental NLoss from manure, Denitr), NManApp	Erosion risk	Sum of 3 loss terms: dissolved N (Leaching), particulate N (erosion), incidental N (surface runoff)	N loss in kg N ha^{-1} $year^{-1}$	Variable <100% (each of 3 components), dependent on baseline values	na	A: experimental catchment in southeast Norway

For acronyms of NLIs, see Table 1; for all other abbreviations refer to Table 3 in the Appendix

[a] Thresholds of NConc or loads for "high" vulnerability refer only to the source term; they may be modified by other, transport-related factors

[b] N sources are considered only in modified DRASTIC versions

[c] Erosion risk depends on soil properties, topography, climate

systems, NUE values are highly correlated with NO_3–N leached (for instance, in irrigated systems in Colorado, Shaffer and Delgado 2002).

An indicator for the nitrate concentration in the soil leachate based on N concentrations of maize plants at silage maturity (Ncm, S6), a routinely recorded quality parameter, was suggested by Herrmann and others (2005). This indicator is applicable only for silage maize.

Several other source-based simple NLIs were discussed in Bockstaller and others (2009), for instance number of N fertilizer applications, deviation of the recommended N fertilizer rate, or period of application. Because they seem overly simplistic on the one hand but tightly dependent on local conditions on the other hand, they are not discussed further here.

NLIs Based on Transport Terms

On the other hand, there is a large number of NLIs which take into account only (or primarily) the transport properties of the soil, the vadose zone and/or the aquifer.

Groundwater Vulnerability Indices

In groundwater protection, the concept of "intrinsic vulnerability" (i.e., independent of type of contaminant) of groundwater has been used already for nearly five decades (e.g., LeGrand 1964). The widely used groundwater vulnerability indices are often utilized with respect to vulnerability for diffuse nitrate pollution from agricultural areas.

"DRASTIC" (TG 1) Probably the most widespread groundwater vulnerability index is "DRASTIC" (Aller and others 1987), which therefore is described here in relatively great detail. It is an acronym for the seven factors: *D*epth to groundwater (vulnerability increases with decreasing depth), *R*echarge, *A*quifer type, *S*oil properties (texture), *T*opography (slope angle), *I*mpact of the vadose zone (effectively vadose zone permeability), and (hydraulic) *C*onductivity (of the aquifer). For a specific site, each factor is assigned a rating value between 1 and 10 based on its relative importance for groundwater contamination. The ratings are multiplied with a weighting factor and summed up to yield the final DRASTIC index:

$$\text{DRASTIC index} = D \times w_D + R \times w_R + A \times w_A + S \times w_S + T \times w_T + I \times w_I + C \times w_C \quad (1)$$

Here, capitals denote the rating values for the respective factors and the "w"s with the corresponding subscripts the weighting factors. According to Aller and others (1987), the weighting factors are constants which should not be changed ($w_D = 5$, $w_R = 4$, $w_A = 3$, $w_S = 2$, $w_T = 1$, $w_I = 5$, $w_C = 3$). This yields a possible range of DRASTIC index scores between 23 and 230.

DRASTIC has been extensively used for diffuse nitrate pollution (e.g., Navulur and Engel 1998: USA, Indiana; McLay and others 2001: New Zealand; Rupert 2001: USA; Stigter and others 2006: Portugal; Panagopoulos and others 2006: Greece; Berkhoff 2008: NW Germany).

Correlations between DRASTIC values and nitrate concentrations in groundwater proved very poor when the unmodified original DRASTIC version was used (e.g., Panagopoulos and others 2006). This may be a consequence of the preponderance of factors pertaining to the groundwater and the vadose zone below the rooted soil zone (5 of the 7 factors), and the lack of factors for nitrogen sources and land management in the original DRASTIC formulation. This, among other factors, stimulated the introduction of management and N source factors to the original DRASTIC scheme (for instance, Navulur and Engel 1998; Panagopoulos and others 2006; Stigter and others 2006).

The advantage of the DRASTIC approach is that the required data are in general easily available for extensive regions. Moreover, the method has been tested and applied in numerous studies from all over the world. For purely agricultural applications, DRASTIC may be overly focussed on the groundwater, which may impede the use at the field scale. Measured nitrate concentrations in groundwater are often difficult to assign to specific fields due to lateral groundwater flow and attenuation.

Tile drainage, which has a great impact on nitrate loss from agricultural fields into groundwater (e.g., Vinten and others 1994; Dinnes and others 2002) is not accounted for in DRASTIC. Although according to DRASTIC, groundwater vulnerability increases with decreasing depth to groundwater, lower nitrate concentrations in groundwater have been described for sites with higher groundwater tables, presumably due to increased denitrification (e.g., de Ruijter and others 2007). Moreover, DRASTIC tends to overestimate the vulnerability of porous media aquifers compared to aquifers in fractured media, and several important factors, e.g. organic carbon content and sorption capacity (of the soil, the vadose zone and the aquifer), travel time and dilution are not taken directly into account.

Other Groundwater Vulnerability Indices In Germany, the concept of the "Protection function of the vadose zone" ("Schutzfunktion der Grundwasserüberdeckung"—SG, TG2) (Hölting and others 1995) is used by geological surveys. This approach takes into account the available WHC of the soil, the thickness and hydraulic properties of the vadose zone, and the percolation rate (=SeepRate). The SG has been applied to groundwater contamination risk with nitrate, which is not adsorbed in many soil media and may be treated as a "conservative tracer" (e.g., Magiera 2002) (for tropical soils see discussion in section "General discussion and conclusions").

Another groundwater vulnerability index, which has been used with respect to nitrate contamination, is the Aquifer-Vulnerability-Index (AVI, TG3) (Stempvoort and others 1993). AVI quantifies groundwater vulnerability by means of the hydraulic resistance (Rh) which is calculated from the two parameters thickness d(i) and hydraulic conductivity HC(i) of each vadose zone layer (index i) overlying the aquifer:

$$\mathrm{Rh} = \sum_{i=1}^{n} \frac{\mathrm{d(i)}}{\mathrm{HC(i)}} \tag{2}$$

with n the number of layers above the groundwater table. Log values of Rh (years) are rated into vulnerability categories as <1: extremely high vulnerability, 1–2: high vulnerability, 2–3: moderate vulnerability, 3–4: low vulnerability, >4: extremely low vulnerability. That means, the higher the Rh value, the lower the groundwater vulnerability and the lower the risk of nitrate contamination of groundwater.

In the NLI of Nolan (2001) (MLR, TG4), multivariate logistic regression models based on more than 900 sampled wells were used to predict the probability of exceeding 4 mg $\mathrm{NO_3\ l^{-1}}$ in groundwater in the USA.

Approaches Based on the Hydrology of the Soil Zone

Exchange Frequency (TS1) The "Exchange frequency of the soil solution within the effective root zone" (EF) is often used in Germany (Frede and Dabbert 1999; Müller 2004). Similar NLIs are utilized in other countries, for instance in France (CORPEN 2006, see below). Since the basic principle of these NLIs is similar, the EF is discussed here exemplarily in more detail. It is calculated as the ratio of a transport (annual seepage rate, or groundwater recharge, SeepRate in mm $\mathrm{year^{-1}}$) and a storage term (total or available water holding capacity of the root zone, WHC(rz)):

$$\mathrm{EF} = \mathrm{SeepRate/WHC(rz)} \times 100 \tag{3}$$

Thus, EF has units of % per year. It is assumed that water moves through the soil profile in a homogeneous front (i.e., no preferential flow) and that no surface runoff occurs. Mostly the total water holding capacity (tWHC) is used as a storage term (Frede and Dabbert 1999; Kersebaum and others 2006). However, some authors prefer to use the available water holding capacity (aWHC) as storage term (Hölting and others 1995). When aWHC is used, only water bound in pores larger than diameters corresponding to pF 4.2 (i.e., pores with diameters >0.2 µm, corresponding to plant-available soil water) is accounted for, whereas for tWHC, the total water content at field capacity (i.e., all pores with diameters <50 µm) participates in water flow. Both tWHC and aWHC differ especially in clay rich soils. It is not a priori known which of these is a better predictor of N leaching. Theoretically, one would expect that EF with aWHC as a storage term would better predict N leaching.

A low EF corresponds to a high retention capacity (for water) of the soil in the effective root zone and therefore to a low risk for nitrate leaching. EF values are usually classified into risk classes for N leaching out of the soil zone <70 ("very low"), 70–100 ("low"), 100–150 ("moderate"), 150–250 ("high"), and >250 ("very high") (Müller 2004).

Unless not measured directly, the seepage rate can be estimated for different land use types from precipitation during winter and summer, plant-available soil water, and potential evapotranspiration according to Haude (Frede and Dabbert 1999). WHC(rz) can be assessed based on soil texture, bulk density and soil organic matter content. Effective rooting depth can be estimated from soil texture, bulk density, depth of the soil profile, soil hydrology and occurrence of compacted horizons (Wendland and others 1993; Müller 2004).

The EF approach can be calculated easily and the required parameters are widely available. Due to its simplicity, however, many points of criticism may be raised regarding this approach, among others:

1. No N source and no management factors are considered.
2. The approach implicitly assumes that water moves through the soil profile in a homogeneous front, whereas in many soils, preferential flow phenomena have been observed as a rule rather than as an exception (e.g., Flury and others 1994).
3. Pores with diameters >50 µm are not considered at all in this approach, whereas many studies have shown that such larger pores convey a large part of water and solute fluxes from the soil surface to the subsurface (e.g., Buczko and others 2006).
4. The seasonal distribution of precipitation (and even more, percolation) is not taken into account.

A source factor for nitrate was introduced in a modification of this approach by Hilmes and others (1998). Kersebaum and others (2006) combined the EF approach with an indicator for groundwater vulnerability (based on aquifer texture, type of vadose zone cover and depth to groundwater table) into a groundwater pollution risk index.

Similar procedures to estimate the exchange frequency of the soil solution are used in various other countries. For instance, in France, a "drainage index" ("Indice de drainage" or "P/RU", TS2) is calculated as the ratio of the precipitation sum during the leaching period (prec(ls)) and aWHC(rz) (CORPEN 2006).

Other Soil Leaching Indices The "Leaching Index" (LI, TS3) proposed by Williams and Kissel (1991) has been extensively used in North America (e.g., Pierce and others 1991; Czymmek and others 2003), but also in Europe (e.g., De Paz and others 2009). Soil hydraulic properties are taken into account by means of the "hydrologic soil group" (HSG), whereas percolation through the soil is estimated using the annual precipitation amount (prec(a)) and its seasonal distribution. The LI is calculated as the product of a Percolation Index (PI) and a Seasonal Index (SI):

$$\mathrm{LI} = \mathrm{PI} \times \mathrm{SI} \tag{4}$$

The PI is calculated as (Williams and Kissel 1991):

$$\mathrm{PI} = \frac{(\mathrm{prec(a)} - 0.4 \times \mathrm{R})^2}{(\mathrm{prec(a)} + 0.6 \times \mathrm{R})} \tag{5}$$

The retention parameter R depends on the HSG. Please note that "percolation" here is identical to "seepage" (SeepRate) used in the description of other NLIs. The SI is determined by the annual precipitation (prec(a)) and the precipitation during the leaching season (prec(ls)) (i.e., fall and winter precipitation for temperate climates): $\mathrm{SI} = (2 * \mathrm{prec(ls)}/\mathrm{prec(a)})^{1/3}$.

The HSG in this form is specific for the USA, and is generally not mapped for regions outside the USA. This impedes the use of the LI for other regions of the world. As in other transport-based NLIs, no source terms for nitrogen are considered, and land use or management factors are not incorporated. Some studies reported that this method therefore does not accurately evaluate the leached NO_3–N amount (Shaffer and Delgado 2002; Van Es and others 2002). The percolation is directly derived from precipitation, without accounting for evapotranspiration, and the PI is very sensitive with respect to the HSG.

The LI forms a part of the ALRP and the NIT-1 (see below).

Poiani and others (1996) combined the LI with nitrate leaching and considered denitrification during groundwater transport of NO_3–N.

Composite NLI Approaches

Score-Based NLIs

Colorado Vulnerability Map (CO-VM) and Matrix (CO-VMX) (CS1) For the US state Colorado, Ceplecha and others (2004) developed an aquifer vulnerability map for nitrate (CO-VM) and a field-scale nitrate vulnerability matrix (CO-VMX) for N loss by subsurface leaching from irrigated fields.

The CO-VM takes into account the presence or absence of a primary aquifer, depth to groundwater, soil drainage class, recharge availability (i.e., irrigation), and land use:

$$\begin{aligned}\text{CO-VM} = {} & \big[\text{Drainage} + (\text{Land Use} + \text{Irrigation Index}) \\ & + \text{Depth to groundwater}\big] \times \text{Presence of primary Aquifer}\end{aligned} \tag{6}$$

Here, "Drainage" denotes the drainage conditions of soils as assessed by soil drainage classes, with values between 1 (poorly drained—"very low vulnerability") and 4 (excessively drained—"high vulnerability"); the indicator "Land Use" is used as a proxy for N sources with values of 0 for open water/ice, 1 for natural vegetation and wetlands, 2 for developed (urban) lands, and 3 for agricultural lands. An "Irrigation Index" value of 1 is added to "Land Use" in case of irrigation. Since natural groundwater recharge is negligible for agriculturally used areas of Colorado, no further indicator for water movement through the vadose zone is incorporated. The indicator for "Depth to groundwater" has values of 1 for >15 m depth, 2 for 6–15 m, and 3 for 0–6 m depth. The indicator for "Presence of primary aquifer" is assigned a value of 1 if the investigated area is located above a primary aquifer. Otherwise it has a value of 0. The possible range of values for CO-VM is 0–11. High index values denote great contamination risk.

The field scale nitrate vulnerability matrix (CO-VMX) is calculated from four factors as:

$$\begin{aligned}\text{CO-VMX} = {} & \mathrm{f}(\text{soil texture}) + \mathrm{f}(\text{irrigation efficiency}) \\ & + \mathrm{f}(\text{nitrogen application rate}) + \mathrm{f}(\text{application timing})\end{aligned} \tag{7}$$

It is assumed that NO_3–N leaching increases with sand content and nitrogen application rate, and with decreasing irrigation efficiency. Rating values between 1 and 4 are assigned to each of these factors. Additionally, one index point is subtracted from the final score if one of several best management practices is applied (e.g., use of slow release fertilizer or nitrification inhibitor, winter cover crop, deep rooted crop).

The statewide CO-VM and the field scale CO-VMX are complementary: CO-VM was intended as a screening procedure with which resources could be focused on "high risk" areas, which should be studied using the field scale CO-VMX.

EnSus (CS2) EnSus ("Environmental Sustainability") (Woods and others 2006) was developed for estimating the risk of N leaching and N loss by surface runoff in New Zealand. N leaching risk is calculated by multiplying a factor for potential soil leaching ("N leaching vulnerability index", i.e., transport factors) and for N sources ("N pressure index" depending on land use). For calculating the N leaching vulnerability index, first a N leaching vulnerability score is calculated:

$$\begin{aligned}&\text{N leaching vulnerability score} = \text{prec(a)}/\text{ET} \times \text{PAW f.}\\&\times \text{slowpermeability f.} \times \text{attenuation factor}\end{aligned} \tag{8}$$

The PAW ("profile available water") factor with values between 1 and 2.4 accounts for increase of N leaching risk if available water content in the soil profile is lower than 200 mm. Profile Available Water is a measure for the soil's capacity to hold water assessed for the soil profile to a depth of 0.9 m and expressed as millimetres of water, i.e., it corresponds largely to WHC(rz). The "Slow permeability factor" has a value of 0.7 for soils with saturated hydraulic conductivity <2.5 mm day^{-1} and 1 for other soils.

The "Attenuation factor" accounts for gaseous N losses by denitrification in poorly drained and/or organic soils and can assume values between 0.1 (very poorly drained organic soils) and 1.

The resulting N leaching vulnerability scores range from 0 to 44. To obtain the N leaching vulnerability index, the scores are classified into 5 categories (with index values in parentheses): 0 to ≤2: Low (1); 2 to ≤3: Mod low (2); 3 to ≤4: Mod (3); 4 to ≤7: Mod high (4); >7: High (5). The N leaching vulnerability index class values 0–5 are used further for multiplication with the N pressure index to obtain the N leaching risk.

The pressure of N inputs to soils is estimated from land use classes, with N pressure index values between 0.4 (native forest) and 10 (pastoral dairy and Horticultural and vegetables). "Normal" arable land has an index value of 8.

The final EnSus N leaching risk is calculated as:

$$\begin{aligned}&\text{EnSus N leaching risk} = \text{N leaching vulnerability index}\\&\quad\times \text{N pressure index}\end{aligned} \tag{9}$$

Modified Nitrogen Ranking Scheme (mNRS) (CS3) The "modified Nitrogen Ranking Scheme" (mNRS) (Magette and others 2007), was developed for nitrogen leaching via soil and groundwater and NLoss by overland flow in grassland agricultural systems in Ireland. It contains the factors: Nutrient application rate (NA) and timing (NT), dirty water applications (DW), cropping system (C), farmyard risk (FY), aquifer vulnerability (AV), subsoil type (SS), hydrological risk (runoff risk) (HR), preferential flow paths (including subsurface drains) (PP). Factors are rated as Low (1), Medium (2), and High (4). If AV is available, then mNRS Site Score = (NA × NT + DW + C + FY) × AV. If AV is not available, mNRS Site Score = (NA × NT + DW + C + FY) × T, where T = SS × PP, or if SS is unavailable T = HR × PP. The mNRS was tested for a grassland dominated dairy farm in western Ireland, using yearly averaged nitrate concentrations in groundwater from nine boreholes. The mNRS scores could be correlated with yearly averaged nitrate concentrations only if N deposited directly by grazing livestock was included in the application (NA) factor (Magette and others 2007).

Nitrate Leaching Hazard Index for Irrigated Agriculture (NLHI-IRR, CS4) The Nitrate Leaching Hazard Index for Irrigated Agriculture in the SW USA (NLHI-IRR) (Wu and others 2005), consists of a "soil hazard index" (SHI, a function of hydraulic permeability and texture, with values between 1 and 5), an "irrigation system hazard index" (ISHI, 1–4), and a "crop hazard index" (CHI, a function of rooting depth, ratio of N in the crop tops to the recommended N application, fraction of the crop top N that is removed from the field with the harvest, magnitude of the peak N uptake rate, whether the crop is harvested at a time when N uptake rate is high; values between 1 and 4).

The NLHI-IRR is calculated with a multiplicative scheme as:

$$\text{NLHI-IRR} = \text{SHI} \times \text{ISHI} \times \text{CHI} \tag{10}$$

The resulting values of NLHI-IRR range between 1 and 80. Values below 20 are considered to be of minor concern, whereas values >20 require more attention (Wu and others 2005).

Nonpoint-Source Agricultural Hazard Index (NPSAH, CS5) A hazard index for agricultural nonpoint-source pollution (not restricted to nitrogen) (NPSAH) was developed for conditions in Northern Italy (Trevisan and others 2000) and is calculated by multiplying hazard factors ("HF") by control factors ("CF"):

$$\begin{aligned}\text{NPSAH} = &\left(\text{HF}_\text{p} + \text{HF}_\text{f} + \text{HF}_\text{te}\right)\\&\times \left(\text{CF}_\text{ap} \times \text{CF}_\text{c} \times \text{CF}_\text{i} \times \text{Cf}_\text{s}\right)\end{aligned} \tag{11}$$

where HF_p is the hazard factor for pesticides, HF_f is the hazard factor for fertilizers (0–5), HF_te is the hazard factor for trace elements; CF_ap is the control factor for agronomic practices, CF_c is the control factor for climate, CF_i is the control factor for irrigation, and CF_s is the control factor for slope.

The hazard factors (HF) represent farming activities that might cause an impact on groundwater, such as use of fertilizers and pesticides, application of livestock and poultry manure, food industry wastewater, and urban sludge. The control factors modify the hazard factor by considering site characteristics (geographical location, slope, agronomic practices, and type of irrigation).

For determining HF_f values, fertilizer application amounts (N + P) are assigned to land use classes. HF_f can assume values between 0 (e.g., forest) and 4 (e.g., permanently irrigated arable land). The control factors have values in the range between 0.9 and 1.1.

The resulting values of NPSAH index are classified into 10 vulnerability classes.

The NPSAH was tested in the province of Cremona, Italy. However, comparisons with field data was not given in Trevisan and others (2000). Whereas this NLI is relatively simple, determining values for the constituent HF and CF values seems to be not straightforward and is focused on the conditions in Northern Italy. To combine the risk for diffuse losses of pesticides, trace elements, P and N into one single index value seems problematical and could obscure the risk when only one of these components is of interest.

OMAFRA-NI (Ontario, Canada) (CS6) The NLI developed by the Ontario Ministry of Agriculture, Food and Rural Affairs (Canada) (OMAFRA 2003) considers N loss by soil leaching and is based on scores for N crop removal balance (NI(CRB)), for manure applications after harvest (NI(NAL)), and the hydrological soil group. In addition, management factors are incorporated (crop type, cover crop, application timing). NI(CRB) and NI(NAL) can have values ranging between 0 (<17 kg N/ha) and 6 (135–202 kg N/ha for NI(CRB) and 90–134 kg N/ha for NI(NAL)).

First, a NI value is obtained from the scores of NI(CRB) and NI(NAL): NI(OMAFRA) = NI(CRB) + NI(NAL). The rating of these NI(OMAFRA) scores depends on the hydrological soil group, i.e. different threshold values for maximum permissible NI(OMAFRA) values are assigned according to the prevailing hydrological soil group. The threshold ranges from 1 for very well drained soils to 9 for poorly drained soils, reflecting the greater risk of N leaching for well drained compared with poorly drained soils.

Pennsylvania N Index (PA-NI) (CS7) The NLI proposed by Heathwaite and others (2000) and McDowell and others (2002) ("Pennsylvania N Index", or "PA-NI") consists of a source term and a transport term which are multiplied to yield the PA-NI score. The source term consists of rating values for: (1) fertilizer application rate; (2) method of fertilizer application; (3) manure application rate; (4) method of manure application. Each rating factor can assume the values 0 (="none"), 1 (="low"), 2 (="medium"), 4 (="high"), and 8 (="very high"). The source term is calculated by summation of the four rating values. The transport term consists of factors for soil texture and hydraulic conductivity. The PA-NI was tested in Pennsylvania, but comparisons with measured nitrate concentrations or losses were not given in Heathwaite and others (2000) or McDowell and others (2002). Climatic parameters or percolation rates are not incorporated into this index. The N source term considers only N in fertilizer and manure applications, i.e., no N extraction by crops.

Model-Type: Simple Equation

IROWC-N (Canada) (CE1) The "indicator risk of water contamination by nitrate–nitrogen (IROWC-N)", developed in Canada (De Jong and others 2007), is based on

1. Residual soil nitrogen (RSN), estimated from the annual N balance;
2. Estimation of NO_3–N leaching by a simplified water balance.

The amount of nitrate leached per year (NL) is calculated as:

$$\mathrm{NL} = \left[\mathrm{RSN} \times \left(\mathrm{prec(a)} - \mathrm{ET_{pot}}\right)\right] / \left[\mathrm{aWHC} + \left(\mathrm{prec(a)} - \mathrm{ET_{pot}}\right)\right] \quad (12)$$

In the IROWC-N, RSN is estimated from the difference between N inputs (fertilizer-N, manure-N, biological fixation, and atmospheric deposition) and N outputs (N removed in crop harvest, N lost from ammonia volatilization and N lost from denitrification), assuming that mineralization and immobilization are balanced; aWHC is defined here as the available water holding capacity up to 100 cm depth. The annual water leaching is estimated as the difference between prec(a) and potential evapotranspiration (ET_{pot}).

The NO_3–N concentration in the leachate (NConc) is calculated as:

$$\mathrm{NConc} = \mathrm{NL} \times 100 / \left(\mathrm{prec(a)} - \mathrm{ET_{pot}}\right) \quad (13)$$

Based on the estimated values for NL and NConc, five IRONC-N classes are distinguished (De Jong and others 2007), with the highest class encompassing NL >20 kg N ha^{-1} and NConc >10 mg NO_3–N l^{-1}. Whereas a map of estimated IROWC-N values for entire Canada has been presented by De Jong and others (2007), these estimates have not been compared with observed nitrogen losses and concentrations.

The IROWC-N index calculates only N loss by leaching through the soil, whereas other pathways of N loss have to be estimated by calculations in separate procedures. N loss by ammonia volatilization and denitrification, which reduces the magnitude of RSN, has to be assessed by separate methods. The assumption that N mineralization and immobilization are balanced may not be valid in many situations, but that applies also to other approaches which include N balances.

Potential Nitrate Concentration in Leachate (PNCL) (CE2) The "Potential nitrate concentration in leachate" (PNCL) in its original form (Bach 1987) is calculated as the ratio of the annual N balance (NBal) (kg N ha^{-1}) and annual seepage rate (SeepRate) (mm):

$$\mathrm{PNCL} = \mathrm{NBal}/\mathrm{SeepRate} \times 4.43 \times 100 \quad (14)$$

The factor "4.43" accounts for re-calculation of N into nitrate, whereas the factor "100" for the transformation into units of mg l^{-1} for PNCL. PNCL is an indicator for the expected mean nitrate concentrations (in mg NO_3 l^{-1}) in the leachate. In the original form (Bach 1987), the underlying assumptions are that no net mineralization/immobilization occurs in the soil (i.e., equilibrium conditions), and all the N-surplus is lost by leaching through the soil profile (i.e., no ammonia volatilization, denitrification losses, no surface runoff). Bach (1987) estimated the PNCL for the Western Federal States of Germany using a raster width of 3 km. In the Atlas of the Nitrate Fluxes in Germany (Wendland and others 1993), PNCL is estimated also for the Eastern Federal States and denitrification (in the vadose zone and in the groundwater) is accounted for.

A rigorous calibration of the PNCL approach using measured data has not been presented in the cited studies. As mentioned in Wendland and others (1998), a comparison of PNCL values (including denitrification) with about 16000 observed nitrate concentrations in the groundwater throughout Germany gave a "good agreement".

A recent, more elaborate form of PNCL accounts for N mineralization and immobilization, and denitrification in the root zone (Frede and Dabbert 1999). The assessment of N mineralization and immobilization, however, is probably beyond routine applications. On the other hand, the quality of PNCL as a NLI depends on the accuracy of the N balance, and the simple N balance used in the original PNCL (Eq. 14) might be too simplistic. A further problem with the PNCL approach is that for an environmental assessment the entity of interest is not nitrate concentration in the leachate but rather in the groundwater. However nitrate concentrations in groundwater are influenced more by nitrate loads transported from the soil zone than nitrate concentrations in the leachate. For a specified period N loads are the product of NConc and seepage rate. For low seepage rates, very high PNCL values are calculated, whereas the actual impact (N load) is low due to the low amount of seepage water.

Model-Type, Complex Approaches

Annual Leaching Risk Potential (ALRP) (CC1) The Annual Leaching Risk Potential (ALRP) index (Pierce and others 1991) was already mentioned in the section "Approaches based on the hydrology of the soil zone", but because it contains an assessment of the N balance and groundwater-related parameters, it is classified here as a "composite NLI". Nitrate leaching into groundwater is assessed as the product of the relative scores ("s") for the NO_3–N amount leached from the root zone during one year (NL), the travel time to the aquifer (TT), the position of the aquifer (PA), and the vulnerability of the aquifer (VA):

$$s(\mathrm{ALRP}) = s(\mathrm{NL}) \times s(\mathrm{TT}) \times s(\mathrm{PA}) \times s(\mathrm{VA}) \quad (15)$$

NL is calculated from "nitrate available to leaching" (NAL), the "leaching index" (LI, see above), and the pore volume of the unit area over the rooting depth (POR, with the same units as LI):

$$\mathrm{NL} = \mathrm{NAL} \times [1 - \exp(-1.2 \times \mathrm{LI}/\mathrm{POR})] \quad (16)$$

NAL is determined from the nitrogen balance, with inputs comprising net N mineralization, N from crop residues, fertilizer applications, symbiotic N fixation, and N in precipitation/irrigation water. Outputs comprise: N uptake by the crops, ammonium volatilization, denitrification, erosion, and surface runoff. The final ALRP score is defined as the log (base 2) of s(ALRP). The ratings for nitrate leaching risk are "very low" (ALRP 0–2); "low" (ALRP 3); "moderate" (ALRP 4); "high" (ALRP 5); "very high" (ALRP 6); "extreme" (ALRP 7); "very extreme" (ALRP 8). The ALRP index is one of the 15 components of the NIT-1 NLI (see below).

The ALRP index combines nitrogen balancing (NAL), climatic data and soil water movement (LI) with underground (off-site) factors. However, the weight on factors pertaining to the vadose zone and the aquifer in this index (3 of the four components, i.e., 3/4) is large compared with other NLIs. Moreover, quantification of those off-site factors is often not straightforward (Shaffer and Delgado 2002). The factors "TT" and "PA" have a very similar meaning (distance to the water table), whereas other properties of the vadose zone are not taken into account. The use of the ALRP is complicated by the fact that values for N loss processes other than leaching must be quantified separately by external calculation procedures. N mineralization (required to calculate NAL) is not customarily measured and may be difficult to determine.

"IN" Indicator (CC2) The "IN" indicator developed in France (Pervanchon and others 2005; Bockstaller and others 2008) takes into account gaseous N losses (NH_3, N_2O, NO) and nitrate leaching.

Volatilization of gaseous NH_3, N_2O and NO is calculated (consecutively in this order) by means of emission factors which are multiplied by the applied amounts of fertilizer and manure. For ammonia volatilization from manure and mineral fertilizers, the emission factors take into account the type of manure/fertilizer, the timing of application, and the mode of incorporation. For N_2O, the emission factor of 0.0125 is modified to account for lower emission when the grassland is cut more than twice (by a factor of 0.7), and higher emission in case of irrigation or for clay soils (factor 1.5), or for organic soils (factor 2). For NO, a constant emission factor of 0.01 is used. Under normal atmospheric conditions, NO is rapidly transformed into the toxic compound N dioxide (NO_2).

Leaching of nitrate is estimated based on the residual mineral soil N at the beginning of the drainage period (i.e., after harvest in autumn) (RSN(h) = "$N_{leachable}$" in Pervanchon and others 2005), and the annual seepage rate (SeepRate). The NO_3–N concentration in leachate (NConc, mg NO_3–N l^{-1}) is calculated as:

$$\mathrm{NConc} = 100 \times ((\mathrm{RSN(h)} \times [\%\mathrm{N}_{\mathrm{leached}}])/\mathrm{SeepRate}) \tag{17}$$

For grassland, RSN(h) is estimated based on fertilization trials in the Netherlands, featuring RSN(h) of zero-fertilization plots and quadratic regression equations, the amount of mineral N inputs (kg N ha^{-1} $year^{-1}$), and the critical N input amount above which soil is overfertilized and RSN(h) increases (ten Berge and others 2002). For arable crops, a soil mineral balance between harvest and the beginning of drainage is calculated, taking into account mineralization of organic matter, crop residues, and crop uptake before winter (Bockstaller and others 2008). Consequently, only data available at the farm and no extra measurements are required.

The parameter $\%N_{leached}$ represents the percentage of N leached below the rooting depth and is calculated using a simplified Burns leaching equation (Burns 1976) as:

$$\%\mathrm{N}_{\mathrm{leached}} = [\mathrm{SeepRate}/(\mathrm{SeepRate} + (\mathrm{SWR}/10))]^{\mathrm{rzd}/2} \tag{18}$$

with rzd the root zone depth (cm), and SWR the volumetric soil water retention (%) (Pervanchon and others 2005).

For each of the four loss pathways, separate sub-indicators I_{NH3}, I_{N2O}, I_{NO} and I_{NO3} are calculated with normalized scores between 0 (high risk) and 10 (no risk), with a reference set at an indicator score of 7. The reference value corresponds to the maximum N level acceptable for the environment. The final "$I_{N\ losses}$" indicator is determined as the minimum of these four sub-indicators: "$I_{N\ losses}$" indicator = Min(I_{NH3}, I_{N2O}, I_{NO}, I_{NO3}).

That means, the sub-indicator indicating the highest risk of N loss is adopted for the total risk of N losses. When estimating the risk of diffuse N losses with the IN NLI, the resulting scores depend strongly on the choice of reference levels. Pervanchon and others (2005) give for the four sub-indicators the following reference values: I_{NH3}: 20 kg N ha^{-1} $year^{-1}$, I_{N2O}: 5.4 kg N ha^{-1} $year^{-1}$, I_{NO}: 1.3 kg N ha^{-1} $year^{-1}$, I_{NO3}: 11.3 mg NO_3–N l^{-1}.

Merlin (CC3) MERLIN (Méthode d'Evaluation des Risques de Lixiviation des Nitrates—methodology for the evaluation of the risk of nitrate leaching) was developed for assessing nitrate leaching in the Poitou–Charentes area (west of France) (Aveline and others 2009). It consists of three sub-indicators:

> EQUIF ("EQUIlibre de Fertilisation"), a N balance which contains crop uptake and fertilizer application, N mineralization of soil organic matter and (if available) measured soil mineral N contents in spring and N mineralization from crop residues. EQUIF values are calculated as kg N ha^{-1} and divided into 6 classes.
>
> IC ("Indicateur de Couverture du sol") an indicator for soil cover in the season after the main crop, but before the beginning of the winter drainage period; it is divided into 3 classes.
>
> SENSIB ("Sensibilité du milieu à l'infiltration") reflecting the influence of soil drainage properties on N leaching, it is divided into 3 classes.

The three sub-indicators are combined by means of a relationship table to obtain three MERLIN classes (1: low risk; 2: intermediate risk; 3: high risk).

The poor fit between MERLIN classes and measured N leaching losses (from 125 lysimeter experimental data with wheat maize oilseed rape in France) was explained mainly by the lack of climate/weather data in the MERLIN calculation by Aveline and others (2009).

"New" N-Index Tier-1 (NIT-1) (CC4) The "New N-Index Tier-1 (NIT-1)" NLI developed by the USDA-ARS (United States Department of Agriculture—Agricultural Research Service) Colorado (Delgado and others 2008) is a comprehensive NLI which accounts for nitrate leaching through the soil, erosion, surface runoff, ammonia volatilization, and denitrification.

Each of 15 site characteristics are rated as "very low" ("column factor", cf = 0), "low" (cf = 2), "medium" (cf = 4), "high" (cf = 6), and "very high" (cf = 8). The 15 site characteristics are: (1) Leaching index (LI); (2) Nitrogen available to leach potential; (3) Estimated nitrate leaching; (4) Nitrogen budget use method; (5) N susceptible to volatilization method; (6) Proximity of nearest field edge to stream or lake; (7) Rooting depths and crop rotation; (8) Aquifer leaching potential risk (ALPR); (9) Tile drainage; (10) NH_3 volatilization; (11) Denitrification; (12) Soil erosion (wind and water); (13) Runoff class; (14) Irrigation erosion; (15) Vegetative buffer.

The Nitrogen available to leach (NAL) is calculated based on an annual N balance Nitrate leaching is assessed using the "nitrate leached" (NL) approach similar as in the ALRP (Eq. 16).

NH_3 volatilization losses are calculated as a function of fertilizer type, application method, and weather conditions; they are highest for surface-applied urea under dry weather conditions and lowest for incorporated NH_4NO_3 under humid weather conditions. Denitrification is calculated as a function of SOM content and drainage conditions of the soil (highest for poorly drained soils with high SOM, lowest for well-drained soils with low SOM).

NIT-1 scores are calculated by summing up the column factors estimated for each of the 15 site characteristics:

$$\text{NIT-1} = \sum_{i=1}^{15} \text{cf}(i) \tag{19}$$

Here, "*i*" denotes the number of the site characteristic and "*cf*(*i*)" the corresponding "column factor". Index scores for separate N loss pathways can be calculated (nitrate leaching component, surface transport component, atmospheric component).

Total NIT-1 scores are rated as 0–24 "None or very low", 24–52 "Low", 52–83 "Medium", 83–107 "High", 107–128 "Very High".

The NIT-1 has been tested using measured NO_3 concentrations and loads in soil leachates from field experiments in the USA (Colorado, Nebraska, New York, Ohio) and China, measured denitrification and NH_3 volatilization from Argentina und NO_3 loads in surface runoff from Alabama (Delgado and others 2008).

Compared with other NLIs, the NIT-1 is relatively complex and requires more input parameters. Whereas most of the NLIs discussed here are intended for special or local conditions, the NIT-1 is intended as an assessment tool which can be applied worldwide. It has been adapted with minor modifications for use in California and Mexico.

A few of the factors in NIT-1 seem redundant: for instance, the ALRP component incorporates the LI and a N balance (Pierce and others 1991), which are utilized in addition as separate site characteristics of the NIT-1. The NIT-1 consists of many components which are connected by an additive scheme. Consequently, the influence of a single factor on the calculated NIT-1 score is restricted (relative sensitivity about 6%, Table 2), although in reality that factor could have an overwhelming influence on N losses (e.g., tile drainage). This problem is alleviated by considering both the total NIT-1 score and the results of separate sub-indicators in the analysis (Delgado and others 2008).

Norway N Index (NO-NI) (CC5) The Norway NLI (NO-NI) (Bechmann and others 2009) is calculated as the sum of three components:

$$\text{NO-NI} = \text{dissolved N} + \text{particulate N} + \text{incidental N} \tag{20}$$

The results of NO-NI are N losses in kg N ha^{-1} $year^{-1}$ which are classified into risk classes. "Dissolved N" (i.e., N transported through soil leaching) is calculated as a N balance (N sources–N removal):

$$\begin{aligned}\text{N sources} &= \text{NDep} + \text{NFertApp} + \text{NFix} + \text{Manure amount} \\ &\times (\text{inorg. N content} \times \text{inorg. N correction} + \text{org. N content} \\ &\times \text{org. N correction}) + \text{Manure applied previous year} \\ &\times \text{org. N content} \times (1 - \text{correction org. N previous year}) \\ &- \text{Incidental N loss from manure previous year} \\ &+ \text{Factor} \times \text{N uptake previous autumn} + \text{Tillage timing N}\end{aligned}$$

$$\begin{aligned}\text{N removal} &= \text{NCUpt} + \text{Correction split application} \\ &+ \text{N uptake autumn} + \text{Amount of straw incorporated} \\ &\times \%\ \text{N in straw} + \text{Denitrification N} \times \text{Drainage factor}\end{aligned}$$

"Particulate N" is the amount of particle bound N lost by erosion and is calculated as the product of erosion risk (based on a modified USLE approach, in kg soil ha^{-1}) and %N in soil.

"Incidental N" is the amount of soluble N lost by surface runoff. It is calculated as the product of the manure amount applied, its organic N content and an "application timing risk factor".

The NO-NI has been tested in the 4.5 km^2 Skutterud catchment in south-eastern Norway (Bechmann and others 2009) but no output calibration with measured N losses has been presented.

For a NLI exhibiting a relatively high degree of complexity as the NO-NI, it is remarkable that it contains no climate and no (direct) soil factors (soil properties are considered only for the estimate of denitrification and erosion risk). Consequently, N leaching through the soil is represented solely by the N balance, which has the same disadvantages as described for the "pure" NBal NLI approaches (see section "N balance (NBal, S1)").

Discussion and Conclusions

Many environmental indicator approaches to assess the risk of diffuse nitrogen losses from agricultural fields have been developed during the past decades. These are variously termed "N indicators" or "N indices"; here, we used the generic term "N loss indicators" (NLI). NLI approaches vary with regard to their complexity, the considered factors and processes, the required input data, and the general focus (e.g., loss from root zone, vulnerability of groundwater, influence of irrigation). For the sake of clarity, NLIs were classified here into source-based, transport-based and composite approaches. Only the composite NLIs, which are classified her into score-based and model-type approaches, contain both source factors, management factors, and transport factors.

Desirable properties of environmental/ecological indicators in general have been discussed extensively in the literature. Based on Kelly and Harwell (1990), Cairns and others (1993), Dale and Beyeler (2001), and Rees and others (2008), an ideal NLI should be:

1. Easy to measure/calculate (i.e., measurement of the constituent factors in composite NLI, "ease of use");
2. Easy to understand and communicate ("comprehensibility") (because NLI are intended also for use by non-specialists);
3. Scientifically sound;
4. Anticipatory (i.e., NLI should be calculated before relevant decisions are taken);
5. Sensitive;
6. Integrative (i.e., relevant factors for N loss should be combined);
7. Responsive to management factors (because these are the factors that can be influenced);
8. Robust with respect to confounding influences of factors not considered in the NLI.

Similarly, Bockstaller and others (2008) discuss 6 criteria for NLIs: simplicity, legibility, pedagogy, sensitivity, flexibility, usefulness.

A comprehensive evaluation of these criteria would require, among other things, feedback from several users for each NLI. Due to the large number of NLIs discussed here, a full evaluation of the NLIs with respect to these criteria was not feasible and therefore should be the subject of future work.

Although the definition of the targeted user group (e.g., scientists, farmers, water managers, extension services) and the purpose of a NLI (i.e., ex ante or ex post analysis) are important (e.g., Bockstaller and others 2008), they are often not explicitly stated in the original descriptions of NLIs. Most NLIs are implicitly intended for anticipatory use (i.e., in order to predict the effect of management practices etc. on Nloss), however using NLIs ex post may also be of interest in order to evaluate which factor has been responsible for the observed N loss patterns and help to learn from this for future activities. Whereas the results of NLI calculations may be of concern for various types of users, the calculations themselves, especially of the more complex NLIs, will often be performed by specially trained extension services and scientists.

Essentially all of the NLIs discussed here were developed for more or less temperate climatic conditions, although several of them may be applied also in other climates (for instance the NIT-1 NLI). We are not aware of any NLIs which were explicitly developed for use under tropical climatic conditions. Such NLIs should consider the N loss conditions which differ in some respects from those in temperate climates (e.g., Wong and others 1990; Sierra and others 2003): the proportion of ammonium of total soil N is higher compared with soils in temperate climates because at high soil temperature, the rate of ammonification is higher than the rate of nitrification and nitrification may be inhibited in acid tropical soils. Moreover, tropical oxisols may have a high anion exchange capacity which contributes to nitrate retention and delays nitrate leaching. On the other hand, in the wet tropics, water and concomitant N leaching rates may be much higher than in temperate climates.

Especially in dry climatic conditions of the developing world, wastewater is commonly used as irrigation water (e.g., Jimenez 2005), although such practices are reported also from more "developed" countries, for instance New Zealand (Barton and others 2005). Such wastewaters can contain high N concentrations; for instance, Barton and others (2005) describe N loading resulting from wastewater irrigation of >400 kg N ha^{-1} $year^{-1}$. Whereas factors accounting explicitly for N in wastewater are included in the mNRS (CS3) and NPSAH (CS5) NLIs, and as nitrate concentration of irrigation water in the source term of NIT-1 (CC4), factors accounting for application of liquid animal waste (slurry) are included in several NLIs discussed here.

The parameter determining the harmfulness of nitrate in the environment is mostly the nitrate concentration in ground- or surface water. However, the risk caused by N losses from the soil zone is more aptly expressed by N loads, i.e., the total amount of N that enters the ground- or surface water. This is reflected by the fact that in most NLIs, the source term is based on loads (in kg N ha^{-1}) rather than on concentrations (in mg N l^{-1}). Connected with this, although threshold nitrate concentrations are justified when considering ground- and surface waters, they are of limited value when considering leachate water, because the N load from the soil is determined by both nitrate concentration and seepage rate.

Most of the NLIs are restricted to N loss by leaching through the soil profile. This may be justified, because leaching is in most cases the dominant pathway for N losses and excessive nitrate concentrations in groundwater or surface water are the most tangible and localized negative effect of diffuse N losses, whereas the effect of gaseous N losses seems to be more on a global scale (stratospheric ozone depletion, global warming, etc.) and therefore less tangible. The question as to what extent and in what form leaching and atmospheric N losses can be merged into a single NLI (as for instance in the IN or NIT-1 NLIs) needs more attention in future studies, since the impact of those pathways on the environment is obviously different. For instance, the bulk of the denitrification product N_2 produces no detrimental environmental effects (in contrast to nitrous oxyde, N_2O) and therefore the assessment of the N_2O/N_2 ratio produced during denitrification (which is mainly a function of nitrate concentration, O_2 partial pressure, temperature, availability of C_{org} and pH) would be possibly more meaningful (although certainly adding complexity) in a NLI than denitrification losses as a whole.

Another point, however, is that many of the NLIs developed from the "agricultural point of view", restrict their focus to the soil zone and estimate the N losses that leave the rooted soil zone . It has been shown in many investigations that the fate of diffuse N losses, and therefore their eventual impact on the environment, are very much influenced by the properties of the unsaturated (vadose) zone beneath the root zone (thickness, hydraulic conductivity, texture, organic matter content) and the aquifer (e.g., Wendland and others 1993; de Ruijter and others 2007). Therefore, for an assessment of the environmental impact of diffuse N losses and for water management purposes, the amount of N that leaves the rooted zone alone is probably not sufficient. On the other hand, common groundwater vulnerability indices are more focused on the physical properties of the vadose zone and the aquifer and neglect N sources and the soil. Steps to combine these two points of view were taken for instance by Pierce and others (1991) (ALRP), Wendland and others (1993) (EF, PNCL), Kersebaum and others (2006) (EF combined with groundwater vulnerability assessment) and Delgado and others (2008) (NIT-1).

It seems that many NLIs have not been tested (validated) extensively against field data of measured N losses or N concentrations in groundwater. For several approaches, no comparisons with field data were conducted (or at least published). Even for widely used NLIs (e.g., DRASTIC, EF), calibration and validation against field data seems not sufficient. Comparisons of several different NLIs among each other and with field data is even more scarce (see introduction). A problem when comparing different NLIs is the scaling: typically, each NLI has a different scale, which hampers the comparison of NLIs among each other.

Another aspect which deserves more attention is the sensitivity of the indicators, i.e., what changes in the NLI values are induced by variation of one input parameter. For NLIs in which several factors are summed up for calculation of the indicator value, the relative sensitivity of the separate parameters is low (Table 2, cf. Bockstaller and others 2008): for instance, the relative sensitivity of each of the 15 site characteristics of the NIT-1 is only about 6%, i.e., if only one site characteristic varies, the final NIT-1 value varies by only 6%. Moreover, in complex NLIs such as NIT-1, the different components which are aggregated to yield the final NLI score often have largely divergent meaning (for instance "Tile drainage" vs. "Irrigation erosion" in the NIT-1). To alleviate this problem, both the results for the separate components and for the composite final NLI are evaluated (Bockstaller and others 2008; Delgado and others 2008). In DRASTIC, due to the lower number of site factors (7) and the varying weighting factors, the relative sensitivity ranges between 4 and 22%. On the other hand, in NLI approaches in which the indicator values are obtained by multiplication of the components, the sensitivity is much higher: for instance, in the mNRS of Ireland the relative sensitivity of each of the transport factors is as large as 100%, and for other NLIs, the sensitivity may well exceed 100% (Table 2). These differences in the sensitivity of the various NLIs are important because they imply a valuation about the relative influence of the separate factors on N loss risk. Therefore, the differences in the sensitivities of the various NLI approaches have an influence on their capacity as N loss indicators. Clearly, this problem demands further attention, for instance by extensive comparative field and NLI studies.

For environmental management and agricultural planning purposes, an anticipatory NLI should contain factors for management options. These are included explicitly only in the composite NLIs (groups CS and CC), whereas the simpler NLIs (groups S, T and CE) are lacking such management factors (although they are considered indirectly in the source-based NLIs).

As can be seen and also indicated by the large number of different approaches, there is no ideal NLI for each purpose, and the question which is the "best" NLI is elusive. The discussion of the various NLI approaches revealed that each one has advantages and disadvantages. The necessary data to "feed" more complex NLIs are probably not available in many cases, and simple NLIs may perform satisfactorily in several cases.

Acknowledgments We want to thank three anonymous reviewers, whose detailed and constructive comments helped improve the manuscript.

Appendix

See Table 3.

Table 3 List of abbreviations

Symbol	Unit	Explanation
aWHC	mm	Available water holding capacity
EF	%	Exchange frequency of the soil solution
EF(aWHC)	%	Exchange frequency, calculated based on available WHC
EF(tWHC)	%	Exchange frequency, calculated based on total WHC

Table 3 continued

Symbol	Unit	Explanation
ET	mm year^{-1}	Evapotranspiration
ET_{pot}	mm year^{-1}	Potential evapotranspiration
HC	m s^{-1}	Hydraulic conductivity
HSG	–	Hydrologic soil group
NAL	kg N ha^{-1} year^{-1}	Nitrogen available to leach
NBal	kg N ha^{-1} year^{-1}	N balance
NConc(l)	mg NO_3–N l^{-1}	Measured (or calculated) NO_3–N concentration in leachate (percolation water)
NConc(gw)	mg NO_3–N l^{-1}	Measured (or calculated) NO_3–N concentration in groundwater
NCUpt	kg N ha^{-1} year^{-1}	N uptake by crops
NDep	kg N ha^{-1} year^{-1}	Atmospheric N deposition
NFertApp	kg N ha^{-1} year^{-1}	N application as mineral fertilizer
NFix	kg N ha^{-1} year^{-1}	N fixation by leguminous plants
NIrrig	mg NO_3–N l^{-1}	Measured NO_3–N concentration in irrigation water
NL	kg N ha^{-1} year^{-1}	Nitrogen leached from the root zone
NLI		Nitrogen Loss Indicator (Index)
NLoss	kg N ha^{-1} year^{-1}	Measured N leaching loss
NManApp	kg N ha^{-1} year^{-1}	N application as manure
NMin	kg N ha^{-1} year^{-1}	N mineralization of organic matter
PI	inches (or other length unit)	Percolation index
PNCL	mg NO_3 l^{-1}	Potential nitrate concentration in leachate
prec(a)	mm year^{-1}	Annual precipitation
prec(ls)	mm year^{-1}	Precipitation sum during leaching season
Rh	year	Hydraulic resistance
RSN	kg N ha^{-1} year^{-1}	Residual soil mineral nitrogen (=content of mineral N as measured with the "N-min" method)
RSN(h)	kg N ha^{-1} year^{-1}	Residual soil mineral nitrogen at harvest (autumn)
SeepRate	mm year^{-1}	Annual seepage rate (drainage) out of the rooted zone
tWHC	mm	Total water holding capacity
WHC(rz)	mm	Water holding capacity of the root zone

References

Aller L, Bennett T, Lehr JH, Petty RJ, Hackett G (1987) DRASTIC: a standardized system for evaluating groundwater pollution potential using hydrogeologic settings. EPA-600/2-87-035, Environmental Protection Agency, Washington, DC

Aveline A, Rousseau ML, Guichard L, Laurent M, Bockstaller C (2009) Evaluating an environmental indicator: case study of MERLIN, an assessment method of the risk of nitrate leaching. Agricultural Systems 100:22–30

Bach M (1987) Die potentielle Nitratbelastung des Sickerwassers durch die Landwirtschaft in der Bundesrepublik Deutschland. Göttinger Bodenkundliche Berichte 93:1–186

Barton L, Schipper LA, Barkle GF, McLeod M, Speir TW, Taylor MD, McGill AC, van Schaik AP, Fitzgerald NB, Pandey SP (2005) Land application of domestic effluent onto four soil types: plant uptake and nutrient leaching. Journal of Environmental Quality 34:635–643

Bechmann M, Stålnacke P, Kværnø S, Eggestad HO, Øygarden L (2009) Integrated tool for risk assessment in agricultural management of soil erosion and losses of phosphorus and nitrogen. Science of the Total Environment 407:749–759

Behrendt H, Bach M, Opitz D, Pagenkopf W-G, Scholz G, Wendland F (2003) Nutrient emissions into river basins of Germany on the basis of a harmonized procedure. UBA-Texte 82, Umweltbundesamt, Berlin

Berkhoff K (2008) Spatially explicit groundwater vulnerability assessment to support the implementation of the Water Framework Directive—a practical approach with stakeholders. Hydrology and Earth System Science 12:111–122

Bockstaller C, Guichard L, Makowski D, Aveline A, Girardin P, Plantureux S (2008) Agri-environmental indicators to assess cropping and farming systems. A review. Agronomy for Sustainable Development 28:139–149

Bockstaller C, Guichard L, Keichinger O, Girardin P, Galan MB, Gaillard G (2009) Comparison of methods to assess the sustainability of agricultural systems. A review. Agronomy for Sustainable Development 29:223–235

Bohne K, Gall H, Zachow B (1997) Simulation von Stickstoff-Austrägen aus Lysimetern. Zeitschrift für Pflanzenernährung und Bodenkunde 160:5–13

Buczko U, Kuchenbuch RO (2007) Phosphorus indices as risk assessment tools in the USA and Europe—a review. Journal of Plant Nutrition and Soil Science 170:445–460

Buczko U, Bens O, Hüttl RF (2006) Tillage effects on hydraulic properties and macroporosity in silty and sandy soils. Soil Science Society of America Journal 70:1998–2007

Burns IG (1976) Equations to predict the leaching of nitrate uniformly incorporated to a known depth or uniformly distributed throughout a soil profile. Journal of Agricultural Sciences 86:305–313

Cairns J, McCormick PV, Niederlehner BR (1993) A proposed framework for developing indicators of ecosystem health. Hydrobiologia 236:1–44

Cannavo R, Recous S, Parnaudeau V, Reaug R (2008) Modeling N dynamics to assess environmental impacts of cropped soils. Advances in Agronomy 97:131–174

Ceplecha ZL, Waskom RM, Bauder TA, Sharkoff JL, Khosla R (2004) Vulnerability assessments of Colorado ground water to nitrate contamination. Water, Air, and Soil pollution 159(1): 373–394

Chichester FW (1977) Effects of increased fertilizer rates on nitrogen content of runoff and percolate from monolith lysimiters. Journal of Environmental Quality 6:211–217

CORPEN (2006) Des indicateurs AZOTE pour gérer des actions de maîtrise des pollutions à l'échelle de la parcelle, de l'exploitation et du territoire, Ministère de l'Écologie et du Développement Durable. http://www.ecologie.gouv.fr/IMG/pdf/maquette_azote29_09.pdf, Paris, p 113

Czymmek KJ, Ketterings QM, van Es HM, DeGloria SD (2003) The New York nitrate leaching index. CSS Extension Publication E03-2, 34 pp

Dale VH, Beyeler SC (2001) Challenges in the development and use of ecological indicators. Ecological Indicators 1(1):3–10

De Jong R, Yang JY, Drury CF, Huffman EC, Kirkwood V, Yang XM (2007) The indicator of risk of water contamination by nitrate-nitrogen. Canadian Journal of Soil Science 87:179–188

De Paz JM, Delgado JA, Ramos C, Shaffer MJ, Barbarick KK (2009) Use of a new GIS nitrogen index assessment tool for evaluation of nitrate leaching across a Mediterranean region. Journal of Hydrology 365:183–194

de Ruijter FJ, Boumans LJM, Smit AL, van den Berg M (2007) Nitrate in upper groundwater on farms under tillage as affected by fertilizer use, soil type and groundwater table. Nutrient Cycling in Agroecosystems 77:155–167

Delgado JA, Shaffer M, Hu C, Lavado RS, Cueto-Wong J, Joosse P, Li X, Rimski-Korsakov H, Follett R, Colon W, Sotomayor D (2006) A decade of change in nutrient management: a new nitrogen index. Journal of Soil and Water Conservation 61:66A–75A

Delgado JA, Shaffer M, Hu C, Lavado R, Cueto-Wong J, Joosse P, Sotomayor D, Colon W, Follett R, DelGrosso S, Li X, Rimski-Korsakov H (2008) An index approach to assess nitrogen losses to the environment. Ecological Engineering 32:108–120

Dinnes DL, Karlen DL, James DB, Kaspar TC, Hatfield JL, Colvin TS, Cambardella CA (2002) Nitrogen management strategies to reduce nitrate leaching in tile-drained midwestern soils. Agronomy Journal 94:153–171

Flury M, Flühler H, Jury WA, Leuenberger J (1994) Susceptibility of soils to preferential flow of water: a field study. Water Resources Research 30:1945–1954

Follett RF, Delgado JA (2002) Nitrogen fate and transport in agricultural systems. Journal of Soil and Water Conservation 57(6):402–408

Follett RF, Keeney DR, Cruse RM (eds) (1991) Managing nitrogen for groundwater quality and farm profitability. Soil Science Society of America, Madison, 357 pp

Frede H-G, Dabbert S (eds) (1999) Handbuch zum Gewässerschutz in der Landwirtschaft. ecomed, Landsberg, 451 pp

Gollany HT, Molina J-AE, Clapp CE, Allmaras RR, Layese MF, Baker JM, Cheng HH (2004) Nitrogen leaching and denitrification in continuous corn as related to residue management and nitrogen fertilization. Environmental Management 33(Supplement1):S289–S298

Goodlass G, Halberg N, Verschuur G (2003) Input output accounting systems in the European community—an appraisal of their usefulness in raising awareness of environmental problems. European Journal of Agronomy 20:17–24

Haferkorn U (2000) Größen des Wasserhaushaltes verschiedener Böden unter landwirtschaftlicher Nutzung im klimatischen Grenzraum des Mitteldeutschen Trockengebietes - Ergebnisse der Lysimeterstation Brandis. PhD dissertation, University Göttingen, 157 pp

Hajkowicz S, Collins K, Cattaneo A (2009) Review of agri-environment indexes and stewardship payments. Environmental Management 43:221–236

Havlin J (2004) Impact of management systems on fertilizer nitrogen use efficiency. In: Mosier AR, Syers JK, Freney JR (eds) Agriculture and the nitrogen cycle. International Council of Scientific Unions/Scientific Committee on Problems of the Environment, Scope, 65, pp 167–178

Heathwaite LH, Sharpley A, Gburek W (2000) A conceptual approach for intergrating phosphorus and nitrogen management at watershed scales. Journal of Environmental Quality 29:158–166

Herrmann A, Kersebaum KC, Taube F (2005) Nitrogen fluxes in silage maize production: relationship between nitrogen content at silage maturity and nitrate concentration in soil leachate. Nutrient Cycling in Agroecosystems 73(1):59–74

Hilmes G, Böckler H, Ilsemann J, Müller U, van der Ploeg RR (1998) Abschätzung und Darstellung des Nitratauswaschungsrisikos aus landwirtschaftlich genutzten Böden im Winter am Beispiel von Niedersachsen. Wasser & Boden 50(10):57–61

Hofmann BS, Brouder SM, Turco RF (2004) Tile spacing impacts on *Zea mays* L. yield and drainage water nitrate load. Ecological Engineering 23:251–267

Hölting B, Th Haertlé, Hohberger K-H, Nachtigall K, Villinger E, Weinzierl W, Wrobel J-P (1995) Konzept zur Ermittlung der Schutzfunktion der Grundwasserüberdeckung. Geologisches Jahrbuch C 63:5–24

Ilsemann J, Goeb S, Bachmann J (2001) How many soil samples are necessary to obtain a reliable estimate of mean nitrate concentrations in an agricultural field? Journal of Plant Nutrition and Soil Science 164:585–590

Intergovernmental Panel on Climate Change (IPCC) (2007) Climate change 2007—the physical science basis. Contribution of working group I to the fourth assessment report of the IPCC

Jansons V, Buismanis P, Dzalbe I, Kirsteina D (2003) Catchment and drainage field nitrogen balances and nitrogen loss in three agriculturally influenced Latvian watersheds. European Journal of Agronomy 20:173–179

Jemison JM, Fox RH (1994) Nitrate leaching from nitrogenfertilized and manured corn measured with zero-tension pan lysimeters. Journal of Environmental Quality 23:337–343

Jimenez B (2005) Treatment technology and standards for agricultural wastewater reuse: a case study in Mexico. Irrigation and Drainage 54(Suppl.1):S23–S33

Kelly JR, Harwell MA (1990) Indicators of ecosystem recovery. Environmental Management 14:527–545

Kersebaum KC, Matzdorf B, Kiesel J, Piorr A, Steidl J (2006) Model-based evaluation of agri-environmental measures in the Federal State of Brandenburg (Germany) concerning N pollution of groundwater and surface water. Journal of Plant Nutrition and Soil Science 169:352–359

Knappe S, Haferkorn U, Meissner R (2002) Influence of different agricultural management systems on nitrogen leaching: results of lysimeter studies. Journal of Plant Nutrition and Soil Science 165:73–77

LeGrand HE (1964) System for evaluating the contamination potential of some waste sites. Journal of American Water Works Association 56:959–974

Lord EI, Anthony SG, Goodlass G (2002) Agricultural nitrogen balance and water quality in the UK. Soil Use and Management 18:363–369

Ma L, Shaffer MJ (2001) Review of carbon and nitrogen processes in nine U.S. soil nitrogen dynamics models, Chapter 4. In: Shaffer MJ, Ma L, Hansen S (eds) Modeling carbon and nitrogen dynamics for soil management. CRC Press, Boca Raton

Magette WL, Hallissey R, Hughes K, Cosgrove E (2007) Eutrophication from agricultural sources: field- and catchment-scale risk assessment. Environmental RTDI Programme 2000–2006, final report, 157 pp

Magiera P (2002) GIS-gestützte Bewertung der Verschmutzungsempfindlichkeit des Grundwassers. Geologisches Jahrbuch Sonderhefte 3:1–165

Mazari-Hiriart M, Cruz-Bello G, Bojorquez-Tapia LA, Juarez-Marusich L, Alcantar-Lopez G, Marin LE, Soto-Galera E

(2006) Groundwater vulnerability assessment for organic compounds: fuzzy multicriteria approach for Mexico City. Environmental Management 37(3):410–421

McDowell RW, Sharpley AN, Kleinman PJ (2002) Integrating phosphorus and nitrogen decision management at watershed scales. Journal of the American Water Resources Association 38:479–491

McLay CDA, Dragden R, Sparling G, Selvarajah N (2001) Predicting groundwater nitrate concentrations in a region of mixed agricultural land use: a comparison of three approaches. Environmental Pollution 115:191–204

Meisinger JJ, Delgado JA (2002) Principles for managing nitrogen leaching. Journal of Soil and Water Conservation 57(6):485–498

Müller U (2004) Auswertungsmethoden im Bodenschutz–Dokumentation zur Methodenbank des Niedersächsischen Bodeninformationssystems (NIBIS), 7th edn. Schweizerbart, Stuttgart

Navulur KCS, Engel BA (1998) Groundwater vulnerability assessment to non-point source nitrate pollution on a regional scale using GIS. Transactions of the American Society of Agricultural Engineers 41(6):1671–1678

Neukum C, Hötzl H, Himmelsbach T (2008) Validation of vulnerability mapping methods by field investigations and numerical modelling. Hydrogeology Journal 16(4):641–658

Nolan BT (2001) Relating nitrogen sources and aquifer susceptibility to nitrate in shallow ground waters of the United States. Ground Water 39(2):290–299

Oenema O, Kros H, de Vries W (2003) Approaches and uncertainties in nutrient budgets: implications for nutrient management and environmental policies. European Journal of Agronomy 20:3–16

Oenema O, van Liere L, Schoumans O (2005) Effects of lowering nitrogen and phosphorus surpluses in agriculture on the quality of groundwater and surface water in the Netherlands. Journal of Hydrology 304:289–301

OMAFRA (Ontario Ministry of Agriculture, Food and Rural Affairs) (2003) Nutrient management workbook, section 15: nitrogen index. http://www.omafra.gov.on.ca/english/nm/ar/workbook/workbk3o.htm. Accessed July 31 2008

Pamperin L (2002) Nitratverlagerung in Abhängigkeit von der Bodennutzung, den Standorteigenschaften und der Grundwasserneubildung eines stauwasserbeeinflussten Grundmoränenstandortes. Horizonte – Herrenhäuser Forschungsbeiträge zur Bodenkunde 7

Panagopoulos GP, Antonakos AK, Lambrakis NJ (2006) Optimization of the DRASTIC method for groundwater vulnerability assessment via the use of simple statistical methods and GIS. Hydrogeology Journal 14:894–911

Pervanchon F, Bockstaller C, Bernard PY, Peigné J, Amiaud B, Vertès F, Fiorelli JL, Plantureux S (2005) A novel indicator of environmental risks due to nitrogen management on grasslands. Agricultur Ecosystems and Environment 105:1–16

Pierce FJ, Shaffer MJ, Halvorson AD (1991) Screening procedure for estimating potentially leachable nitrate-nitrogen below the root zone. In: Follett RF, Keeney DR, Cruse RM (eds) Managing nitrogen for groundwater quality and farm profitability. Soil Science Society of America, Madison, WI, pp 259–283

Poiani KA, Bedford BL, Merrill MD (1996) A GIS-based index for relating landscape characteristics to potential nitrogen leaching to wetlands. Landscape Ecology 11(4):237–255

Powlson DS, Addisott TM, Benjamin N, Cassman KG, de Kok TM, van Grinsven H, L'hirondel JL, Avery AA, van Kessel C (2008) When does nitrate become a risk for humans? Journal of Environmental Quality 37:291–295

Provolo G (2005) Manure management practices in Lombardy (Italy). Bioresource Technology 96:145–152

Rankinen K, Salo T, Granlund K (2007) Simulated nitrogen leaching, nitrogen mass field balances and their correlation on four farms in south-western Finland during the period 2000–2005. Agricultural Food Science 16:387–406

Rees HL, Hyland JL, Hylland K, Clarke CSLM, Roff JC, Ware S (2008) Environmental indicators: utility in meeting regulatory needs. An overview. ICES Journal of Marine Science 65:1381–1386

Roelsma J (2002) Nitrate leaching versus residual soil mineral nitrogen. In: ten Berge HFM (ed) A review of potential indicators for nitrate loss from cropping and farming systems in the Netherlands. Plant Research International, Wageningen, pp 105–118

Rupert MG (2001) Calibration of the DRASTIC ground water vulnerability mapping method. Ground Water 39:625–630

Schröder JJ, Neeteson JJ (2008) Nutrient management regulations in the Netherlands. Geoderma 144:418–425

Schröder JJ, van Asperen P, van Dogen GJM, Wijnands FG (1996) Nutrient surpluses on integrated arable farms. European Journal of Agronomy 5:181–191

Schröder JJ, Scholefield D, Cabral F, Hofman G (2004) The effects of nutrient losses from agriculture on ground and surface water quality: the position of science in developing indicators for regulation. Environmental Science and Policy 7:15–23

Schweigert P, Zimmermann P (2003) The N-min content of the soil as an agricultural environmental indicator of water pollution with nitrate. Berichte über Landwirtschaft 81(2):192–207

Shaffer MJ, Delgado JA (2002) Essentials of a national nitrate leaching index assessment tool. Journal of Soil and Water Conservation 57:327–335

Sharpley AN, Weld JL, Beegle DB, Kleinman PJA, Gburek WJ, Moore PA, Mullins G (2003) Development of phosphorus indices for nutrient management planning strategies in the United States. Journal of Soil and Water Conservation 58:137–152

Sieling K, Kage H (2006) N balance as an indicator of N leaching in an oilseed rape–winter wheat–winter barley rotation. Agriculture, Ecosystems & Environment 115:261–269

Sierra J, Brisson N, Ripoche D, Noël C (2003) Application of the STICS crop model to predict nitrogen availability and nitrate transport in a tropical acid soil cropped with maize. Plant and Soil 256:333–345

Stigter TY, Ribeiro L, Dill AMMC (2006) Evaluation of an intrinsic and a specific vulnerability assessment method in comparison with groundwater salinisation and nitrate contamination levels in two agricultural regions in the south of Portugal. Hydrogeology Journal 14:79–99

Strebel O, Duynisveld WHM, Böttcher J (1989) Nitrate pollution of groundwater in Western Europe. Agriculture, Ecosystems & Environment 26:189–214

ten Berge HFM (2002) A review of potential indicators for nitrate loss from cropping and farming systems in the Netherlands. Plant Research International BV, Wageningen, 168 pp

ten Berge HFM, Burgers SLGE, Schröder JJ, Hofstad EJ (2002) 'Partial balance'—regression models for $N_{min,h}$. In: ten Berge HFM (ed) A review of potential indicators for nitrate loss from cropping and farming systems in the Netherlands. Plant Research International, Wageningen, pp 25–60

Tiemeyer B, Lennartz B, Kahle P (2008) Analysing nitrate losses from an artificially drained lowland catchment (North-Eastern Germany) with a mixing model. Agriculture, Ecosystems & Environment 123(1–3):125–136

Townsend AR, Howarth RW, Bazzaz FA, Booth MS, Cleveland CC, Collinge SK, Dobson AP, Epstein PR, Holland EA, Keeney DR, Mallin MA, Rogers CA, Wayne P, Wolfe AH (2003) Human health effects of a changing global nitrogen cycle. Frontiers in Ecology and the Environment 1:240–246

Trevisan M, Padovani L, Capri E (2000) Nonpoint-source agricultural hazard index: a case study of the province of Cremona, Italy. Environmental Management 26:577–584

Van Beek CL, Brouwer L, Oenema O (2003) The use of farmgate balances and soil surface balances as estimator for nitrogen leaching to surface water. Nutrient Cycling in Agroecosystems 67:233–244

van Eerdt MM, Fong PKN (1998) The monitoring of nitrogen surpluses from agriculture. Environmental Pollution 102:227–233

Van Es HM, Czymmek KJ, Ketterings QM (2002) Management effects on N leaching and guidelines for an N leaching index in New York. Journal of Soil and Water Conservation 57(6):499–504

van Stempvoort D, Ewert L, Wassenaar L (1993) Aquifer Vulnerability Index: a GIS-compatible method for groundwater vulnerability mapping. Canadian Water Resources Journal 18(1):25–37

Villa F, McLeod H (2002) Environmental vulnerability indicators for environmental planning and decision-making: guidelines and applications. Environmental Management 29:335–346

Vinten AJA, Vivian BJ, Wright F, Howard RS (1994) A comparative study of nitrate leaching from soils of differing textures under similar climatic and cropping conditions. Journal of Hydrology 159:197–213

Vitousek PM, Aber JD, Howarth RW, Likens GE, Matson PA, Schindler DW, Schlesinger WH, Tilman DG (1997) Human alteration of the global nitrogen cycle: sources and consequences. Ecological Applications 7:737–750

Wehrmann J, Scharpf NC (1979) Der Mineralstickstoffgehalt des Bodens als Maßstab fur den Stickstoffdüngerbedarf (Nmin-Methode). Plant and Soil 52:109–126

Wendland F, Albert H, Bach M, Schmidt R (1993) Atlas zum Nitratstrom in der Bundesrepublik Deutschland. Springer, Heidelberg, 96 pp

Wendland F, Bach M, Kunkel R (1998) The influence of nitrate reduction strategies on the temporal development of the nitrate pollution of soil and groundwater throughout Germany—a regionally differentiated case study. Nutrient Cycling in Agroecosystems 50:167–179

Williams JR, Kissel DE (1991) Water percolation: an indicator of nitrogen-leaching potential. In: Follet RF, Keeney DR, Cruse RM (eds) Managing nitrogen for groundwater quality and farm profitability. Soil Science Society of America, Inc, Madison, pp 59–83

Wolfe AH, Patz JA (2002) Reactive nitrogen and human health: acute and long-term implications. Ambio 31:120–125

Wong MTF, Hughes R, Rowell DL (1990) Retarded leaching of nitrate in acid soils from the tropics: measurement of the effective anion exchange capacity. Journal of Soil Science 41:655–663

Woods R, Bidwell V, Clothier B, Green S, Elliott S, Shankar U, Harris S, Hewitt A, Gibb R, Parfitt R, Wheeler D (2006) The CLUES project: predicting the effects of land-use on water quality—stage II. NIWA client report: HAM2006–096, July 2006 NIWA Project: MAF05502. National Institute of Water & Atmospheric Research, Christchurch

Wu L, McGechan MB (1998) A review of carbon and nitrogen processes in four soil nitrogen dynamics models. Journal of Agricultural Engineering Research 69(4):279–305

Wu L, Letey J, French C, Wood Y, Bikie D (2005) Nitrate leaching hazard index developed for irrigated agriculture. Journal of Soil and Water Conservation 60:90A–95A