Agriculture, Ecosystems and Environment 86 (2001) 1–21

Agriculture Ecosystems & Environment

www.elsevier.com/locate/agee

Review

Landscape fate of nitrate fluxes and emissions in Central Europe A critical review of concepts, data, and models for transport and retention

Daniel Haag*, Martin Kaupenjohann

Institute of Soil Science and Land Evaluation (310), Hohenheim University, Emil-Wolff-Strasse 27, D-70599 Stuttgart, Germany

Received 13 December 1999; received in revised form 11 July 2000; accepted 29 August 2000

Abstract

Agroecosystems are leaky systems emitting nutrients like nitrate, which affect ecosystems on a range of scales. This paper examines the fate of nitrate on the landscape level focussing on how landscape components either facilitate or impede N translocation from the field to the stream (headwater). According to their role in landscape metabolism, two categories of landscape components are distinguished, ecotones/retention compartments and conduits/corridors. Conduits such as macropores, preferential interflow-paths, drainage tiles and streams rapidly relocate nitrate to headwaters. Retention compartments like the capillary fringe/saturated zone and riparian vegetation eliminate N through denitrification. The differential role of compartments is illustrated with quantitative examples from the literature. On the landscape level retention potential for N is spatially variable and quantitatively limited, while its realisation is uncertain. Notwithstanding, the literature indicates that on a watershed basis the bulk of total N input is retained; thus the potential is discussed for the retention of nitrate on different scales, i.e. the field, landscape, regional and global scale. The transitory retention of excess nitrate in soil and subsoil solution, soil organic matter, groundwater and riparian vegetation may delay nitrate discharge to the aquatic system for decades, contributing to the low emission factors on basin scale. The adverse effects arising from denitrification are discussed, presenting data on the emission of nitrous oxide from the entirety of the different landscape compartments. It is concluded that reliance on landscape metabolism and self-purification postpones the problem of global N overload and partially transfers it to the atmosphere. An assessment scheme is presented which in the face of the unpredictability of ecosystem and landscape behaviour is risk oriented (instead of impact oriented). The scheme uses a budget approach, which accounts for the critical role of corridors and considers the scale and scope of N emissions. A conceptual framework for the remediation of N overload is presented which rests on the realisation of cycling principles and zero-emission approaches on all scales of agricultural production and which pleads for regional approaches that transcend sectoral boundaries and take account of overall regional N fluxes. © 2001 Elsevier Science B.V. All rights reserved.

Keywords: Corridor; Central Europe; Landscape; Nitrate; N; Retention; Scale; Scope; Transport

* Corresponding author. Tel.: +49-711-4593636; fax: +49-711-4594071.
E-mail address: haagd@uni-hohenheim.de (D. Haag).

PII: S0167-8809(00)00266-8

1. Introduction

Agricultural systems are ecosystems that are maintained in an immature state due to human intervention (Odum, 1969). Control is largely external (Odum, 1984), manifested by frequent external inputs of nutrients and energy, which are large compared to internal fluxes and cycling. As plants are regularly removed from the system, plant and decomposer activity are decoupled. Compared to natural ecosystems, agroecosystems are leaky systems with greater amounts of nutrients flowing in and out (Hendrix et al., 1992; Magdoff et al., 1997). The emitted substances are dispersed in the environment by transformation and transport processes. Transformation processes break up molecules, augment the number of "small molecules" (Addiscott, 1995) and thus increase entropy. Transport processes distribute substances along gradients of potential energy in the environment of agroecosystems.

Intensive N fertilisation and disrupted N cycles have brought about the emission of considerable amounts of N compounds. In terrestrial ecosystems N is mostly translocated as nitrate, which is subject to mass flow and leaching. Average nitrate leaching from terrestrial ecosystems in Central Europe is 15 $kg\,ha^{-1}\,yr^{-1}$: N leaching is 15.9 $kg\,ha^{-1}\,yr^{-1}$ in Germany (Werner, 1994), 15.0 $kg\,ha^{-1}\,yr^{-1}$ in the watershed of Lake of Constance, the second largest European lake (Prasuhn et al., 1996), and 14.7 $kg\,ha^{-1}\,yr^{-1}$ in the canton Bern in Switzerland (Prasuhn and Braun, 1995).

The scope of N impacts ranges from adverse effects on (ground-)water quality over acidification and eutrophication of aquatic ecosystems to loss of biological diversity, and to impacts on atmosphere and climate, e.g., nitrous oxide as greenhouse gas (Lehn et al., 1995; Vitousek et al., 1997a). Ecosystems on a variety of scales are affected by N emissions. On the local scale, groundwater quality and headwaters are affected. On the regional scale, rivers and lakes receive large N loads, roughly half of it deriving from agriculture; e.g., in the European Union rivers receive 55% (Isermann and Isermann, 1997) and in Germany 44% (Werner, 1994) of total N input from agriculture. Agricultural activities account for 64% of N input into the Lake of Constance and to natural background concentration for only 36% (Prasuhn et al., 1996). Rivers discharging into seas are a major conveyor of N. With respect to N, the North Sea drainages are among the most disturbed regions: Average net anthropogenic N input into watersheds is 3900 $kg\,km^{-2}\,yr^{-1}$, 83% of which derive from fertilisers. The resultant discharge to the sea is 1450 $kg\,N\,km^{-2}\,yr^{-1}$ on average (Howarth et al., 1996). This paper therefore focuses on the fate of agricultural N in Central Europe.

2. Assessing N fluxes in agroecosystems

A variety of approaches has been developed to assess the N fluxes arising from agricultural production and to evaluate potential impacts on the environment.

On the field scale, the risk of N loss is assessed with index models, budget approaches and simulation models. Index models characterise risks only qualitatively. Examples are DRASTIC (Aller et al., 1987) and KUL (Eckert and Breitschuh, 1994; Kerschberger and Eckert, 1994). Index methods such as DRASTIC correlate only weakly with measured nitrate inputs into the groundwater (Canter, 1997), hence they are only suitable for the tentative screening of problem areas. Budget approaches indicate site specific risk of N loss and potential disequilibria (Bach, 1987; PARCOM, 1994; Wendland, 1994). Simulation models for the N cycle represent processes of the N cycle at point and field scale (de Willigen, 1991; de Willigen and Neetson, 1985; Groot et al., 1991). They have been applied to study the effect of certain agricultural measures on emissions on field scale (e.g., Dijkstra and Hack, 1995; Line et al., 1993; Rode et al., 1995). However, the simulation of N dynamics and the assessment of output potentials neither address the path nor the fate of nitrate emissions.

Recently, attempts are made to adapt life cycle assessment procedures to agricultural production systems (Vito, 1998). Life cycle approaches assess the impact of agricultural production systems on the environment in terms of effect potentials; they disregard the spatial dimension and setting.

On a catchment scale, agricultural non-point-source (Ag-NPS) models are employed. They usually are built on field-scale models of losses that are aggregated at the catchment scale. Ag-NPS models in conjunction with GIS applications have been used to investigate the relation between land use (i.e. land cover pattern and land use proximity to stream channels) and N chemistry (Hunsaker and Levine, 1995; Tufford et al., 1998) and to study the impact of best management practices on water quality (Hession et al., 1989; Prato and Shi, 1990; Tim and Jolly, 1994). Models are compared by Novotny (1986), Line et al. (1993), while Loague et al. (1998) draw attention to the uncertainties intrinsic to this approach. Key limitations of the Ag-NPS models are twofold (Merot and Durand, 1997). Firstly, they are distributed models resting on the assumption that parameters for each individual cell are perfectly known and that the catchment response is the aggregation of the functioning of the cells. Secondly, the classical Ag-NPS models such as ANSWERS or AGNPS do not explicitly take account of retention zones like hedges or riparian vegetation, overlooking processes which are essential for the functioning of buffer zones.

The mentioned approaches only crudely address the role of the landscape into which agricultural sites and affected ecosystems are embedded and in which transport and retention of matter take place. Leached nitrate passes a number of compartments and landscape elements prior to discharge to the aquatic system. Having left the root zone, nitrate passes the vadose zone (subsoil) and a capillary fringe, eventually reaching an aquifer. Often distinct aquifer storeys coexist, in particular an unconfined shallow aquifer may be underlain by (semi-)confined, deeper aquifers. Lateral transport of nitrate takes place in interflow, drainage tiles and aquifers. A riparian zone may be crossed prior to discharge into a stream. The hydrological setting and the resultant hydrological routing can be rather complex, steering contact times and time lags between in- and output and retention. Retention of nitrate is either due to plant uptake or to denitrification. While the first represents temporary storage in the system, the latter leads to the elimination of N from the system. The steering factors and conditions of denitrification in laboratory and field have been discussed elsewhere (Ferguson, 1994; Groffman et al., 1987). The different compartments function as "landscape organs" (Rapport et al., 1998) contributing to a specific landscape metabolism. With the metabolism metaphor the idea of the "self-purification" of both terrestrial and aquatic systems is associated. Yet where in the landscape retention actually takes place and whether retention potentials can be sustained in the long run, is not clear.

In the following landscape metabolism and its potential elements are investigated. Based on the concepts of hierarchy theory, sustainability and landscape diversity (Barrett, 1992), a conceptual framework is developed for the distinction of interfaces and corridors. Interfaces or ecotones are landscape organs attenuating matter fluxes and their impact on aquatic media; corridors lead to the rapid translocation of matter, increasing environmental risks. A review is provided of the retention or transport potential of the different compartments along the way from the field to headwaters, which dominate water quality downstream and which consequently should have priority in water protection (Haycock et al., 1993). The retention potential of landscapes is critically discussed and the wide scale and scope of nitrate losses is highlighted. Finally, a risk assessment scheme and concepts for remediation are sketched, taking account of the unpredictability of ecosystem behaviour and of the importance of balanced budgets and closed nutrient cycles. It is concluded that sustainable agricultural management should avoid end-of-the-pipe solutions (relying, e.g., on the retentive potential of riparian vegetation), but employ scalar system approaches, in which natural cycling principles should be the benchmark for best management.

3. Conceptualisation of nitrate transport and retention

Landscapes are heterogeneous "patch-works", in which spatial pattern and processes interact (Turner, 1989) to produce domains in which either retention or transport of matter dominates. The ensuing landscape elements operate as biogeochemical processors of matter, governing matter fluxes and budgets on the landscape level (Frede and Bach, 1995). Ecosystem theory conceives landscape elements as components of a nested, inclusive hierarchy with holons as the basic units (Ahl and Allen, 1996; Allen and Hoekstra, 1992). Transfers and processes inside a holon are more intensive than the connections between different holons, while process rates exhibit steep gradients at the margins of holons (Müller, 1992). Holons are delimited by boundaries which act as differentially permeable membranes facilitating some ecological flows but impeding others (Wiens et al., 1985).

3.1. Retention elements

Boundaries are locations where the rates of ecological transfers tend to change abruptly; they increase landscape resistance (Forman, 1995), and they are important control points for material flux (Naiman et al., 1988). Spatially they are expressed as transitional zones or ecotones (Hansen et al., 1988), particularly at aquatic–terrestrial interfaces (Naiman, 1990). Ecotone width depends on the type of flux under consideration, with physicochemical flows creating the widest ecotones (Gilbert et al., 1990). Retention in transition zones is due to storage in pools with long turnover times, e.g., nutrient stocks in vegetation (Johnston, 1991) or the passive soil carbon pool with turnover times of up to 1000 years (Parton et al., 1988); retention also includes elimination and transfer to the atmosphere (denitrification). Retention is largely determined by retention time and area of contact. Accordingly, water retention time is the most critical factor for N removal in wetlands (Jansson et al., 1994a). From a landscape health perspective interfaces are critical landscape organs (Rapport et al., 1998), regulating the flow of materials across landscapes and acting as sinks in landscape transport (Tim and Jolly, 1994).

3.2. Corridors

Corridors are conduits connecting holons and elements of larger scales (Allen and Hoekstra, 1992). Corridors are expressed structurally as preferential flow-paths on different spatial scales. They usually are part of a hierarchical pattern of flow-paths. For example in funnel flow, water is gradually congregated into preferential flow-paths and its movement can be conceptualized as a network of tributaries merging into rivers (Ju and Kung, 1997). Macropore networks have been found to be continuous laterally (interflow) and vertically (Mosley, 1982). Other examples for the hierarchical pattern of corridors are linear forms of erosion (Helming and Frielinghaus, 1998), and the network of streams and rivers (Petts, 1994). Typical corridors are illustrated schematically in Fig. 1.

In corridors matter translocation is rapid, so that residence time is shortened, retention zones are bypassed and spatial distances are bridged. Substances are "flushed through" corridors and internal processing of matter entailing transformation, cycling and retention is restricted (Fig. 2). Contact and interaction with corridor boundaries is limited. For example in soils there is hardly any lateral interaction between corridor and soil matrix in macropore or funnel flow (Ju and Kung, 1997). In the fluvial system of headwater catchments, the physical and chemical processes are dominated by longitudinal processes as well (Petts, 1994).

While holons, boundaries/interfaces and corridors are conceived theoretically, spatially explicit compartments can be classified as retention, intermediary and conduit compartments (Fig. 2), based on overall partitioning between transport and retention of matter.

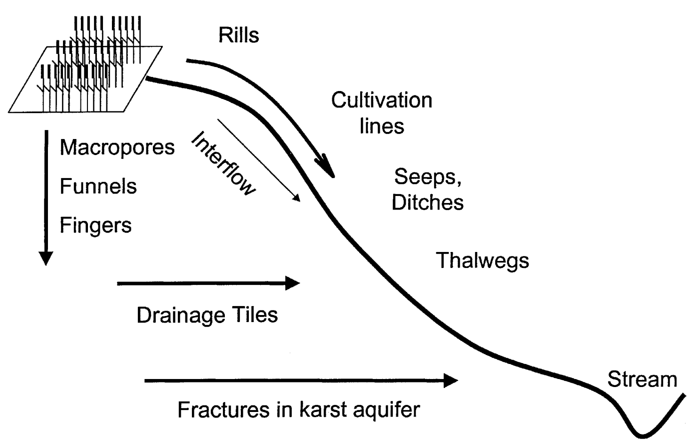

Fig. 1. Corridors in an agricultural landscape. Corridors are doorways of the agricultural system, through which substances bypass on-site and off-site retention zones and are conveyed directly and quickly to the aquatic system. Note the hierarchy of surface corridors, ranging from rills to streams.

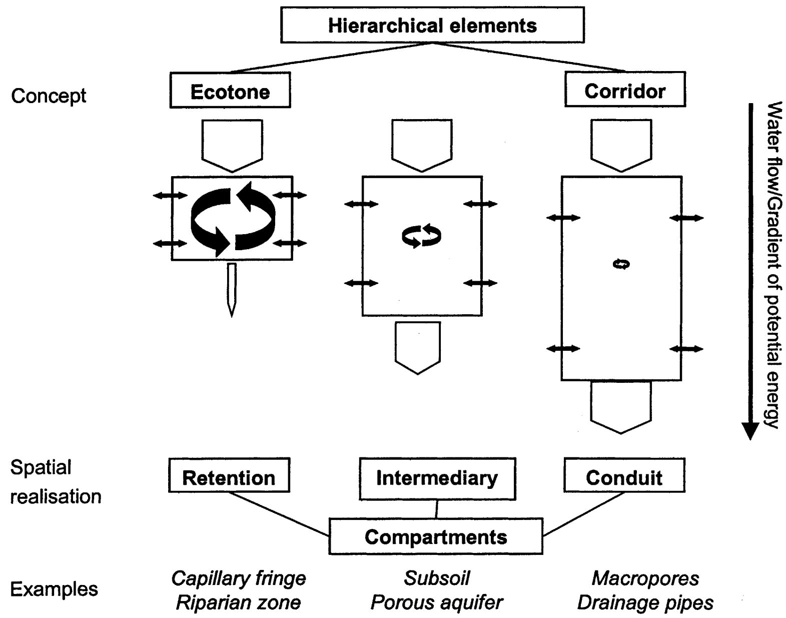

Fig. 2. Classification of landscape elements and compartments. Ecotones and corridors are conceived conceptually. Compartments are explicit sections of space, which are distinguished according to overall matter processing rate. Water flow follows gradients of potential energy. Towards the lateral boundaries of the compartments process rates decline. Internal cycling (indicated by circular arrows) and residence time (indicated by reciprocal of length) varies considerably. In conduits residence times are particularly low. The terms corridor/conduit and ecotone/retention compartment will be used interchangeably in the text.

3.3. *Focus on nitrate leaching to headwaters*

Agricultural contaminants differ with respect to their affinity to determine transport mechanisms. Based upon their soil–solution-partitioning coefficient they can be assigned preferential transport mechanisms (Fig. 3). Nitrate as a highly water-soluble substance is prone to leaching with mass flow. The Lake of Constance study illustrates the dominance of leaching as transport mechanism. Leaching accounted for 79% of NPS, while run-off was a minor source (3%) and erosion was relevant in the Alpine parts of the watershed only (Prasuhn et al., 1996). Under certain conditions, runoff plays a more prominent role, e.g., in some major estuaries, such as Delaware Bay and Chesapeake Bay, NPS runoff from terrestrial ecosystems accounted for half or more of total N inputs (Cronan et al., 1999). Yet as in Central Europe up to 80% of river water stems from groundwater (Hamm, 1991) and owing to the general relevance of leaching this paper focuses on subsurface processes. A characteristic sequence of compartments nitrate traverses on its way from the field to the stream is shown in Fig. 4.

From a water quality perspective, protection of headwaters should have priority (Haycock et al., 1993), as on a catchment scale 60–70% of the water in large rivers enters the system via first- to third-order streams (Vought et al., 1994). According to Kirkby (1978) even 90% of the flow of rivers comes from headwaters, defined as first- and second-order streams. Thus low-order streams contribute the highest percentage to the loading of rivers with nutrients and pesticides (Bach et al., 1997). The approach of this study, therefore stresses the loading of headwaters.

4. Retention in landscape compartments

The different compartments on the way from the field to the headwater are highlighted (Fig. 4) and their role in landscape N metabolism is illustrated with experimental data from a variety of studies in the following section.

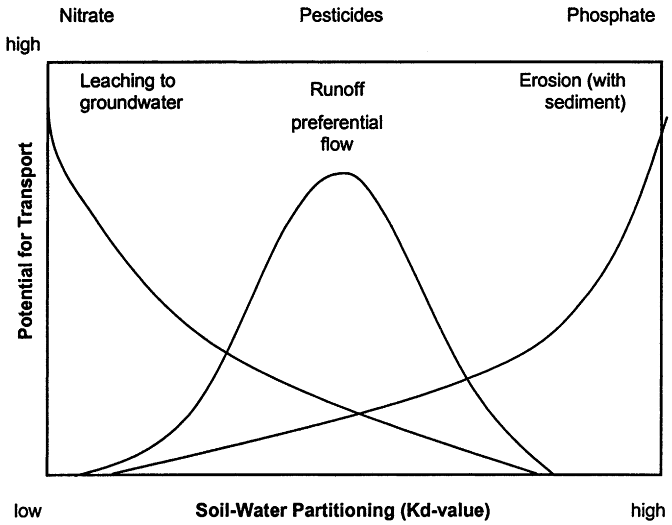

Fig. 3. Affinity of agricultural contaminants to different mechanisms of transport as a function of their soil–water partitioning coefficient. For nitrate, leaching is the dominant transport process, while superficial transport in run-off water and with eroding soil is of minor importance (adapted from Logan, 1993).

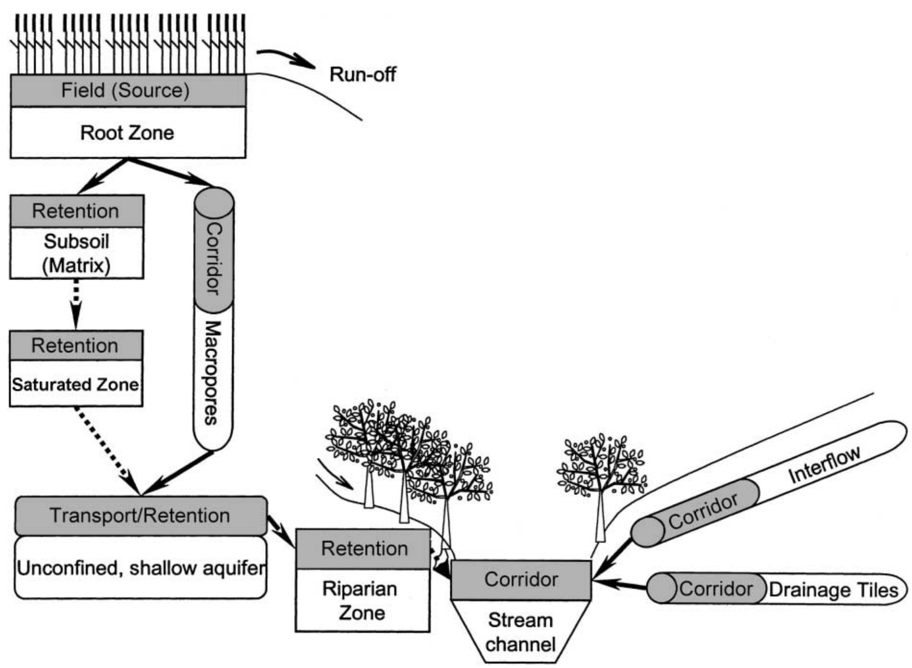

Fig. 4. Schematic of corridors and retention compartments. The sequence of compartments depends upon the specific hydrological setting and is spatio-temporally variable.

4.1. Soil and subsoil

Organic carbon is the key limiting factor for denitrification in subsoils, so that movement of carbon from the soil surface is necessary to support denitrification (Rice and Rogers, 1993). Anaerobic conditions are another precondition. Soil morphology, particularly the existence of stratified layers within the soil profile, impeding water and solute movement may contribute to the creation of conditions favourable for denitrification (Zakosek and Zepp, 1993). Depending upon soil type and agricultural land use denitrification losses ranged from 1 to 223 N kg ha^{-1} yr^{-1} in a number of field experiments (Wendland, 1992).

However, denitrification in subsoil and intermediate vadose zone may be insignificant under certain conditions (Rice and Rogers, 1993; Zakosek and Zepp, 1993): For example unstratified coarse textured soils either lack organic carbon or anaerobic conditions. Fine textured soils may lack organic carbon; e.g., in some loess subsoils denitrification has been shown to be insignificant due to the lack of organic C and thus played no role in the reduction of nitrate transfer into the groundwater (Heyder, 1993). Under normal field conditions subsoil denitrification potential and its rate of recovery tend to be low (Zakosek and Zepp, 1993). Residence time of leachate in soil and underlying substrates varies from days (karst) to decades (fine-textured, thick substrates without fissures), thus N passage to aquifers may be retarded considerably (Hölting et al., 1995).

4.2. Groundwater and aquifers

Groundwater and aquifers diverge with respect to landscape position, chemical characteristics, permeability and vulnerability to agricultural inputs (Hölting et al., 1995). Three aquifer types can be distinguished (Davis and DeWiest, 1991; Hölting, 1980): Unconsolidated, porous aquifers (gravel, sand), consolidated aquifers (cracks in solid rock) and karst aquifers (fractures). Retention takes place in transition zones (Gilbert et al., 1990), while fissures and fractures serve as conduits. Depending upon permeability and biological/chemical characteristics, aquifers as a whole can act as conduits (e.g., karst aquifers with wide fissures) or as retention compartments (e.g., aquifers with low permeability and high denitrification potentials). Groundwater transport usually is slow compared to superficial water flow and can retard discharge of nitrate to streams for years or decades (see below).

4.2.1. Denitrification studies

Substantial denitrification has been observed in a variety of aquifers (Hiscock et al., 1991; Korom, 1992; Lowrance and Pionke, 1989; Mariotti, 1994; Rice and Rogers, 1993; Spalding and Parrot, 1994), while in other aquifers little or no denitrification activity was observed (Hiscock et al., 1991; Lowrance, 1992; Lowrance and Pionke, 1989; Mariotti, 1994; Rice and Rogers, 1993). Actual and potential denitrification depend on biological and chemical characteristics and on hydrology (Mariotti, 1994). The key limiting factor of heterotrophic denitrification is organic carbon availability, while populations of denitrifiers exist in both shallow and deep aquifer systems (Hiscock et al., 1991; Mariotti, 1994). Autotrophic denitrification, requiring an inorganic source for oxidation, e.g., pyrite, is uncommon in groundwater (Hiscock et al., 1991).

4.2.2. Shallow unconfined aquifers

Denitrification may be an important mechanism for reducing nitrate within selected landscape positions, especially in near proximity to the water table (Steinheimer et al., 1998), i.e. in the transition zone between unsaturated and saturated zones. Correspondingly, it appears to be of greatest significance in shallow unconfined aquifers (Rice and Rogers, 1993), where denitrification is considered an important mechanism attenuating nitrate concentration (Lowrance and Pionke, 1989; Montgomery et al., 1997). Within the lower Rhine region in Germany nitrate reductions for three shallow ground water catchments were 16, 63 and 70% of the nitrate reaching the aquifer (Obermann, 1982). In a superficial pleistocene aquifer, dissolved carbon leached into groundwater yielded maximum potential denitrification of 65 mg l^{-1} nitrate (Leuchs, 1988).

4.2.3. Hydrological setting

The hydrological setting is crucial for denitrification particularly in shallow aquifers. In Central Europe three typical constellations were found, showing the wide range of denitrification potential and stressing the relevance of organic carbon (Obermann, 1991). Firstly, consolidated aquifers with little soil cover and high permeability in combination to high nitrate inputs entailed correspondingly high nitrate output; discharge of nitrate was only delayed. Secondly, unconsolidated aquifers with low amounts of organic carbon in combination with limited nitrate input led to partial elimination of nitrate. Thirdly, unconsolidated aquifers with high amounts of organic carbon caused almost complete elimination of nitrate.

4.3. Terrestrial–aquatic interfaces and riparian zones

There seems to be general agreement that the land–water interface regulates water quality in agricultural watersheds (Dillaha et al., 1989), making riparian buffers the most important factor controlling entry of non-point source nitrate in surface water (Gilliam et al., 1997). Thus buffer zones are attributed an enormous potential for the control of water-based pollution (Haycock et al., 1997). Riparian zones may improve water quality due to sedimentation, plant uptake, retention in soil and microbial processes (Correll, 1997; Johnston, 1991; Vought et al., 1994). Particularly denitrification, which ultimately exports N from the system, is very common in wetland ecotones (Gilbert et al., 1990).

4.3.1. Field and laboratory studies

Denitrification losses from riparian forests in Georgia and Maryland ranged from 61 to 89% of N inputs, while retention ranged from 39 kg ha^{-1} (32 kg ha^{-1} due to denitrification and 7 kg ha^{-1} due to net retention within the system) to 74 kg ha^{-1} (Johnston, 1991). In riparian zones of the river Garonne in France, denitrification was so intensive that approximately 30 m of groundwater flow under a woodlot were enough to remove the entire nitrate (Pinay et al., 1990). A riparian zone located below and adjacent to a field-sized watershed planted with soybeans eliminated up to 93% of groundwater nitrate (Line et al., 1993). In a large number of studies riparian nitrate removal exceeded 90% (Hill, 1996) and removals of 90% seem to be common. However, at least some wetlands seem to retain little if any N. In a study of five wetlands in Ontario, Devito et al. (1990) reported net retention ranged from −12% to +4%. The overall range of N retention in wetlands is around −30% to +100% (Johnston, 1991), i.e. depending upon wetland, net release of nitrate and complete retention of nitrate are possible.

Denitrification *potentials* have been studied in field and laboratory. Mesocosm experiments yielded denitrification potentials of 29 and 171 kg ha^{-1} yr^{-1} for similar sites (Addy et al., 1999) demonstrating the influence of land use legacy. Under incubated laboratory conditions an average of 76 kg ha^{-1} yr^{-1} was assessed, while soil amended in situ with N reached values of 160 up to 1340 kg ha^{-1} yr^{-1}. However, under unamended in situ conditions, average was only 2 kg ha^{-1} yr^{-1} (Johnston, 1991) demonstrating that *actual* denitrification in riparian zones is easily overestimated.

4.3.2. Hydrological setting

A major factor for the realization of retention potentials and the effectiveness of buffer zones is hydrological setting (Fig. 5) (Addiscott, 1997; Correll, 1997; Gilliam et al., 1997; Haycock et al., 1997). It determines residence time, which is the single most important variable for water quality improvement (Fennessy and Cronk, 1997). For example, in a controlled situation at least 10 days of water retention was needed to remove N (Hillbricht-Ilkowska, 1995). Riparian forests of different hydrological positions thus vary in nutrient retention (Risser, 1990) and buffer zones work well only under determined hydrological conditions (Hill, 1996). Effective removal is restrained to riparian zones with permeable surface soils and sediments that are underlain at a depth of 1–4 m by an impermeable layer that produces shallow subsurface flow of groundwater across the riparian area. Riparian zones connected to large aquifers may be less effective as interaction with vegetation and soils is restricted. To improve the buffer function, water regime is to be managed aiming at increased residence time within the system (Haycock et al., 1993).

4.3.3. Optimum width

There is no consensus regarding width of riparian zones, except that minimum width is 10 m (Haycock et al., 1993), while less than 5–10 m provide little protection of aquatic resources (Castelle et al., 1994).

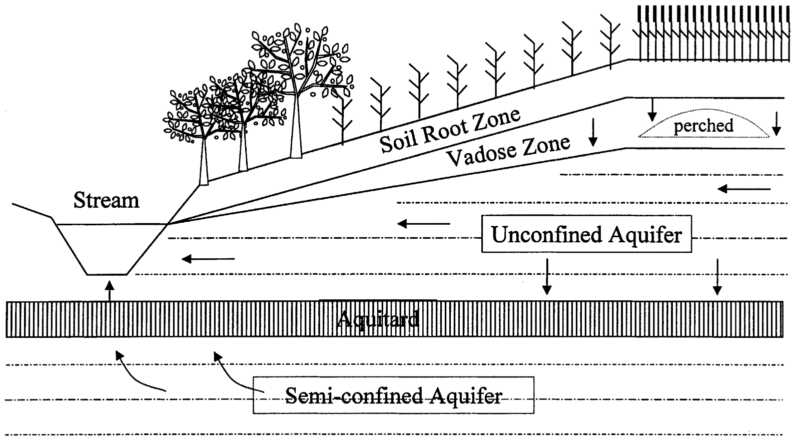

Fig. 5. Schematic of vadose zone, aquifers and flow directions in a typical riparian zone in a humid climate (adapted from Lowrance and Pionke (1989)). The hydrological setting determines whether leached nitrate is subject to riparian retention or bypasses it. Drainage tiles and interflow are not depicted.

Nitrate reductions of 100% seem to be approached by a width between 10 and 20 m (Vought et al., 1994) or 20 and 30 m (Fennessy and Cronk, 1997). Given the complexity of the riparian setting, a useful retort to the question of width is "how wide do you want it?" (Haycock et al., 1997).

4.3.4. Sustainability of retention

Seasonal and long-term sustainability of riparian buffers is controversial as well (Addiscott, 1997). The seasonal sustainability of retention in riparian zones may be maintained in summer by vegetation uptake and during the dormant season by denitrification, as denitrification takes place as soon as the soil temperature exceeds 4°C (Haycock et al., 1993). Other authors, however, stress the seasonal variability of retention, the role of extreme (e.g., storm) events and the decoupling of peak emissions and maximum of retention activity (Addiscott, 1997; Hill, 1996). Long-term sustainability may be affected by declining availability of organic carbon for denitrification and decreasing uptake by old vegetation (Haycock et al., 1993). Moreover there may be an upper limit for the retention of agricultural loads. In wetlands only amounts below 200 kg N ha^{-1} yr^{-1} could be removed satisfactorily (>80%), while the long-term application of higher loads resulted in removal of less than 40% (Hillbricht-Ilkowska, 1995).

4.4. Aquatic–aquatic interfaces: hyporheic zone and sediments

The hyporheic zone is an active ecotone between the surface stream and groundwater. Connections are bidirectional (Bencala, 1993); exchange of water, nutrients, and organic matter occur in response to variations in discharge and porosity (Boulton et al., 1998). Particularly sediments act as sinks for nitrate that discharges to streams and rivers (Gilbert et al., 1990; Pfenning and McMahon, 1996). Laboratory incubation suggests that nitrate is rapidly depleted below the sediment–water interface (Hill, 1997). In the sediments of the river Dorn in Oxfordshire denitrification accounted for 15% of nitrate entering under baseflow conditions (Fennessy and Cronk, 1997). Estimates of the magnitude of N removal during the summer season, when streams are frequently at base flow range from <10% to 76% in a number of studies (Hill, 1997). However, potential denitrification tends to be limited by organic carbon and low temperatures; e.g., potential denitrification measured at 4°C was 77% lower than at 22°C in lab experiments on Australian river sediments, supposedly contributing to high nitrate concentration in the river during winter (Pfenning and McMahon, 1996). In any case, overall in-stream denitrification will be much less than in adjacent riparian wetlands (Fennessy and Cronk, 1997).

5. Transport in corridors

5.1. Preferential flow

Preferential flow takes place in *macropores*, *fingers* and *funnels* (Ju and Kung, 1997; Jury and Flühler, 1992; Stagnitti et al., 1995). Preferential flow has been observed under a variety of conditions, from sandy to clayey soils. Biopores, e.g., well-connected root channels of wheat (*Triticum* spp.), alfalfa (*Medicago sativa* L.) and corn (*Zea mays* L.) may induce preferential flow (Li and Ghodrati, 1994). Preferential flow is not predictable in advance from field analysis (Bouma, 1992; Jury and Flühler, 1992). Rapid movement of nitrate along macropores has been observed (Bouma, 1992). For example, in a heavy clay soil rapid nitrate leaching via preferential flow through mesopores and macropores was observed leading to average nitrate concentrations of 70 mg l^{-1} and maximum concentrations of 136 mg l^{-1} in drain discharge (Bronswijk et al., 1995). While gaps in the N balance often are attributed to denitrification, bypass flow may sometimes be a more important process (Dekker and Bouma, 1984).

5.2. Interflow

Interflow has been observed as an important mechanism for the rapid transport of nitrate towards streams, particularly under stormflow and snowmelt conditions (Göttlicher-Göbel, 1987; Mosley, 1982; Peter, 1987). In forested watersheds average subsurface flow velocities were as high as 0.3 cm s^{-1}, due to flow along macropores and along layers at which permeability changed abruptly. (Mosley, 1982). In small watersheds, nitrate peaked in streams due to interflow after stormflow (Peter, 1987). At the beginning of the winter leaching period, nitrate concentrations in the interflow of a loess site peaked, while denitrification was low (Steininger et al., 1997). Preferential flowpaths may circumvent retention zones, as e.g., has been demonstrated for riparian zones in Britanny (Bidois, 1999).

5.3. Drainage tiles

Drainage tiles inducing artificial interflow are particularly rapid conduits. Artificial drainage speeds the movement of water and contaminants such as nitrate, reducing the opportunity for denitrification to take place (Fennessy and Cronk, 1997). In a number of studies, nitrate concentrations have been observed to range from 2 to 20 mg NO_3 l^{-1} under mineral soils (Hamm, 1991). Average annual nitrate N loss to subsurface drains has been shown to range from 14 to 105 kg per annum, with most of the loss occurring in the winter season (Kladikov et al., 1999). Drainage tiles can contribute significantly to water pollution. For example, around 60% of nitrate-N in surface waters in Illinois entered through drainage tiles (Kohl et al., 1971). Flood events can lead to large export of N in tiles; accordingly, a few days of high-flow events led to most of the annual nitrate loss from a tile-drained field (David et al., 1997). In many areas, subsurface drains discharge into surface ditches or streams (Kladikov et al., 1999). Thus large amounts of N may reach streams through drainage tiles emptying directly into the channel without contact with the riparian soil (Vought et al., 1994).

5.4. Surface flow

Superficial preferential flow minimizes contact with the soil matrix and conveys nitrate rapidly and directly into the aquatic system, overrunning retention compartments such as riparian vegetation (Bach et al., 1994, 1997). Preferential flow-paths are part of a hierarchical network (Fig. 1), consisting of intermittent elements such as rills, cultivation lines and tracks, thalwegs and ephemeral gullies (Helming and Frielinghaus, 1998) and of more permanent streamlets. Drainage lines and streamlets change position and features constantly and despite their importance as conduits removing substances quickly from the field they are overlooked easily. For example a typical drainage line or streamlet in Central Germany had a depth of only 3 cm and an average width of 63 cm, giving rise to an overall streamlet surface of 630 m^2 km^{-2} (Bach et al., 1996). Once substances enter preferential flowpaths, retention is minimized.

5.5. Streams

Streams are "bodies of water moving to a position of lower energy" (Bren, 1993); they are highly dynamic in time and space and are difficult to distinguish from lesser forms like drainage lines or seeps. Uptake and denitrification in streams is limited; the bulk of denitrification probably takes place in aquatic ecotones (sediment) and not in the stream channel itself. In a small Scandinavian reach of 7 km length retention was less than 3% of total N transport in the stream (Jansson et al., 1994b). In a Canadian basin denitrification was less than 6% of the annual export of total N from the basin, while macrophyte uptake accounted for 15% (Hill, 1988). In two rivers in the USA, 7 and 35% of the N load received from external sources was denitrified (Fennessy and Cronk, 1997). Annual mass balances indicate that nitrate-N removal ranges from 1 to 5% in many streams, although values of 20% were also estimated (Hill, 1997).

6. Retention of nitrate on different spatial and temporal scales

In a scalar approach to N fluxes and cycles, four levels can be distinguished (Fig. 6). Firstly, the field and adjacent ecosystems. Secondly, a local level which is restricted to low-order streams and ponds and their watershed. Thirdly, a regional level, which encompasses rivers and lakes like the Rhine, the Danube or the Lake of Constance and their respective basin. Fourthly, a global level, which includes seas like the North, the Baltic and the Black Sea and the atmosphere as a sink for gaseous emissions.

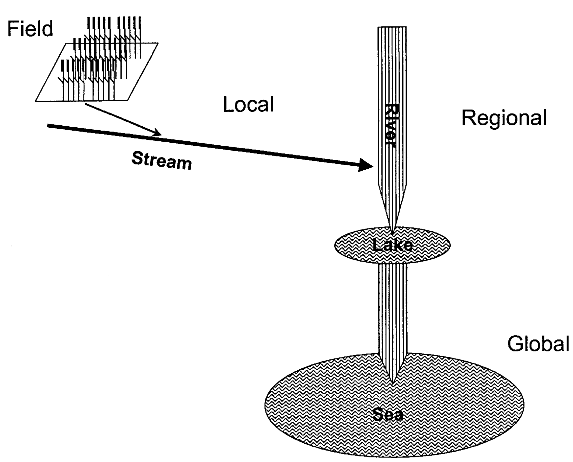

Fig. 6. Scalar approach to water quality, in which four levels are distinguished: The field as the source system including adjacent terrestrial ecosystems, the local level with streams of low-order and occasional ponds, the regional level with rivers and lakes and the global level with seas and the atmosphere.

6.1. Local scale and limitations to retention

On local scale, the capacity of landscape metabolism to retain or eliminate excess N depends upon the pattern and interaction of retention compartments and corridors. Retention and elimination of leached nitrate has been demonstrated for many compartments, but retention is variable, limited and unpredictable as is illustrated for aquifers and for riparian zones.

In groundwater the availability of oxidizable material and residence time limit denitrification. Owing to these constraints in groundwater only has a potential for removing up to $3\,\mathrm{mg\,N\,l^{-1}}$ can be assumed under normal circumstances (Hiscock et al., 1991). Moreover, organic carbon may be depleted at a higher (unsustainable) rate than it is replenished: A number of studies indicates that currently both autotrophic and heterotrophic denitrification potentials are being depleted, with the risk of a nitrate "breakthrough" in the future (Borchers, 1993; Böttcher et al., 1990a,b; Obermann, 1991).

Riparian zones have been attributed a particular significance in water quality protection. However heterogeneity in terms of soils, biogeochemistry and water pathways (Merot and Durand, 1997) complicates the understanding of the mechanisms controlling riparian zone functioning. Accordingly, results concerning actual retention capacities are controversial (Steinmann, 1991) and both high and little or no denitrification have been observed in a number of studies (Groffman and Gold, 1998). Some riparian zones may even release N (Steinheimer et al., 1998). Variability in nitrate removal among sites and within different domains is high (Hill, 1996). Ground water nitrate removal rates may differ even among sites with similar texture, drainage class and morphology (Addy et al., 1999). Caution is required against ascribing specific ground water removal rates to different riparian zones and vegetation. Seasonal and long-term sustainability of the system are also questionable. The restoration of buffer zones with an optimum width of >10 m is difficult to accomplish in densely cultivated agricultural landscapes like in Central Europe. Nevertheless, some authors assume that approximately 50% of the N that is leached is denitrified in riparian forests and groundwater (Groffman and Gold, 1998). Others however claim that "scientists have frequently oversold the ability of wetlands to retain sediments and nutrients" (Johnston, 1991) and that riparian zones can only be a partial solution of a more comprehensive remediation policy (Bidois, 1999). Moreover, the impact of nutrients on wetlands as ecosystems of their own right requires more consideration. In summary, the potential for retention of nitrate on the way from the field to the stream is spatially and temporally restricted and its realization is uncertain.

Corridors connect spatial elements and scales and thus transcend space. Emissions to corridors generally increase environmental risks: Nitrate is rapidly lost from the system of origin circumventing retention potentials and decoupling the N cycle spatially and temporally; eventually emissions and their impact are aggregated on higher scales, where they elude human control. While leading to the rapid translocation of substances, flow in corridors is highly unpredictable.

6.2. Overstrained landscape retention

Anthropogenic N input into terrestrial ecosystems overstrains the capacity of landscapes to retain N. The transfer of N from the atmosphere into the land-based biological N cycle has at least doubled since preindustrial times (Vitousek et al., 1997a), i.e. human activity adds at least as much N to terrestrial ecosystems as do all natural sources combined (Vitousek et al., 1997b). Large parts of this (global) overload are discharged to the aquatic system. Movements of total dissolved N into most of the temperate-zone rivers discharging into the North Atlantic Ocean may have increased by 2–20-fold since preindustrial times, while for rivers in the North Sea region, the N increase may have been 6–20-fold (Howarth et al., 1996). Nitrogen fertilizers eventually end up in estuaries and continental shelves (Kroeze and Seitzinger, 1998).

6.3. Regional scale and retention on basin scale

Although N load to the sea is high, the percentage of total N input into watersheds which is actually discharged is remarkably small: Watersheds in Central and Northern Europe, but also elsewhere discharge only 20% of overall N input to the sea and retain up to 80% (Caraco and Cole, 1999; Howarth et al., 1996). One reason may be denitrification and sedimentation on the regional scale: denitrification in rivers and particularly in riverine ecotones, like wetlands and sediments (Vitousek et al., 1997a) may contribute to N elimination. In-river processes account for losses of around 10–20% of total N inputs (Howarth et al., 1996), while values of 50% can be attained by heavily polluted rivers like the Scheldt (Billen et al., 1985). Retention in lakes and impoundments ranges from 20 to 80% (Howarth et al., 1996). Productive lakes may remove 50% of total N input, with denitrification accounting for one-third, while the rest is trapped in sediments (Jansson et al., 1994a). Nitrogen budgets on basin level indicate that, e.g., in the Rhine basin 85×10^6 kg of N are denitrified (the equivalent of 33% of total input), while in the Elbe 75×10^6 kg (40% of input) are denitrified (Werner, 1994).

6.4. Temporal scales and memory effects

On the local scale, retention may be due to denitrification, but temporary storage in soil (soil organic matter), vegetation and groundwater contribute substantially to the transitory attenuation of nitrate overload. Long residence times in soil and groundwater and the incorporation of N into vegetation and soil organic matter are followed by subsequent, slow release. Apparently, there is a considerable memory effect in ecosystems concerning past nutrient input. In agroecosystems, fertilizer N is incorporated into pools with slow turnover times, increasing N stocks. The major part of leached N derives from the mineralization of organic matter rather than directly from applied fertilizer, as has been shown by a number of studies (Addiscott et al., 1991). For example, in a Rothhamsted experiment nitrate leakage declined to half its initial rate only after 41 years without fertilizer application (Addiscott et al., 1991). Similarly, N released from riparian ecotones tends to originate from within the system, while external nitrate input is absorbed. Nitrogen overload and built-up of organic N have led to the hypertrophication of agricultural soils and landscapes, which may continue to release nitrate for decades, even if nutrient inputs were reduced drastically (Addiscott et al., 1991; Steininger et al., 1997; Vagstad et al., 1997). Due to memory effects, buffer zones may also act as N-source long after the pollution of waterways has been abated (Gilbert et al., 1990). Delay of N translocation in subsurface environments may be considerable; e.g., residence times in aquifers range from less than 1 year (karst) to 103 years (plains of Northern Germany (Wendland, 1992)), though normally maximum residence time in German aquifers is 25–40 years (Bouwer, 1995) with an average of 20 years (Isermann and Isermann, 1997). It can be inferred that "system memory", temporary storage and slow transport can delay the emission of excess N into the aquatic system for decades. In the view of long-term sustainability, the transfer of excess nutrients to transitory storage compartments is no solution. While in conventional agriculture microeconomic time preferences and small-scale system boundaries prevail, sustainable agriculture needs to take account of large-scale and long-term effects (Norton, 1995).

7. Scope of impacts

The environmental impact of nitrate depends on the scalar level under consideration (Isermann, 1993): On a local scale, N emissions may lead to the contamination of groundwater and to the eutrophication and acidification of dystrophic and headwater ecosystems. Headwater streams and their ecotones tend to be particularly sensitive to pollutant inputs (Hamm, 1991). On a regional scale, rivers and lakes are subject to eutrophication, though they often are P limited rather than N limited. In sharp contrast to the majority of temperate-zone lakes, where P is the nutrient that limits primary productivity by algae and other aquatic plants and controls eutrophication, these processes are controlled by N inputs in the majority of temperate-maritime ecosystems (Vitousek et al., 1997a).

7.1. Nitrous oxide production

While denitrification may be beneficial for aquatic ecosystems, the production of nitrous oxide due to denitrification leads to problems on a global scale, as nitrous oxide is both a very efficient greenhouse gas (Houghton, 1994) and plays a role in stratospheric ozone depletion (Crutzen, 1970). There is evidence for the emission of nitrous oxide from the entirety of the compartments discussed above (Dowdell et al., 1979; Yoshinari, 1990). Nitrous oxide emissions from soils vary (Freney, 1997). Depending upon fertilizer type 0.07–2.7% may evolve as N_2O (Eichner, 1990). On the average 0.5−1.5% (McElroy and Woofsy, 1985) or 1.25% (Bouwman, 1992) of applied N to agricultural soils may be emitted as N_2O. Subsoil production of nitrous oxide is not known (Rice and Rogers, 1993). In contaminated aquifers, values of 3.4–7.8 kg $N_2O\,ha^{-1}\,yr^{-1}$ have been measured (Ronen et al., 1988). Shallow aquifers are supposed to be more likely sources of N_2O than confined aquifers (Rice and Rogers, 1993). It is inferred that aquifers could account for 5–10% of total global nitrous oxide source (Rice and Rogers, 1993), i.e. 10–20% of biogenic N_2O sources could originate from aquifers. Nitrous oxide production in riparian zone aquifers ranged from 0.026 to 3.7% of N input on Rhode Island (Jacinthe et al., 1998) and 0.65–0.87% of the input in aquifers in Maryland (Weller et al., 1994). Riparian vegetation thus has a high potential to function as hotspot, inducing nitrous oxide production (Groffman and Gold, 1998), although in many cases riparian vegetation may not emit more N_2O than cropland (Gilliam et al., 1997). Rivers and lakes have been observed to emit N_2O as well (Mariotti, 1994; McMahon and Dennehy, 1999). Overall nitrous oxide emissions from rivers, estuaries and continental shelves increase with increasing N loading from 0.3 to 3% or even 6% of denitrification rates; thus approximately 1% of total N input into these systems may be emitted as N_2O (Kroeze and Seitzinger, 1998). Evidently, the contamination of the subsurface environment with nitrate has the potential for increasing the contribution to atmospheric N_2O (Rice and Rogers, 1993). In fact, direct N_2O emissions (2.1 Tg N) may equal indirect emissions (2.1 Tg N) resulting from agricultural N input into the atmosphere and aquatic systems (Mosier et al., 1998). Thus a (nitrate) water quality problem may be traded for an atmospheric problem (Isermann and Isermann, 1997).

In addition, the loss of nitrate from the field has to be considered as the loss of a resource whose production is linked to the consumption of energy (ca. 47 MJ kg^{-1} N fertilizer) and to the emission of atmospherically active substances. On the average 2500 g CO_2, 10 g N_2O and 1 g CH_4 are emitted to produce 1 kg of N fertilizer (Kaltschmitt, 1997).

7.2. Scale and scope as evaluation criteria

For the evaluation of environmental impact, scale and scope have been forwarded as criteria (Gleich, 1998; Scheringer, 1999). Scope may be defined as the ratio of collateral to intended effects, with crop uptake as the main intended effect of N fertilization. Scope increases with the length and complexity of cause–effect chains. The scale of impact ranges from local/reversible to global/irreversible. The local–global dichotomy indicates to what extent impacts can be attributed to local actors (Norton, 1995; Norton and Ulanowicz, 1992); "reversibility" indicates to what extent and with what ease impacts can be subject to control and remediation. Due to decreasing reversibility and attributability, the larger the scale and scope of emissions, the more problematical they are. To disentangle the impact of agricultural emissions hierarchical, scalar approaches may serve as a heuristic tool (Ahl and Allen, 1996; O'Neill et al., 1989; Wagenet, 1998) and as basis of evaluation.

8. Simulation and prediction of nitrate fate?

Simulation models have been forwarded as tools for the prediction, management and evaluation of agricultural emissions, in particular nitrate. For the prediction of biogeochemical processes on compartment or ecosystem level, no valid general models are available (Hauhs et al., 1996; Oreskes et al., 1994). Variability of the degrees of freedom and the self-modifying character of ecosystems (Kampis, 1991; Lange, 1998) invalidate system descriptions along larger time frames. Accordingly the simulation of (micro-)biological processes, e.g., immobilization and denitrification offers particular problems (de Willigen and Neetson, 1985; Marchetti et al., 1997; Stockdale et al., 1997). Moreover, the interaction of scale and physical structure is highly problematic as due to the spatial heterogeneity of ecosystems on all scales, spatial structure is unknowable at any scales of real interest (Beven, 1996). As a consequence transport in conduits (e.g., preferential flow) is unpredictable (Bouma, 1992; Jury and Flühler, 1992; Stagnitti et al., 1995), and upscaling of distributed models is problematic (Blöschl and Sivapalan, 1995). Spatially transferable models have to be calibrated and validated with data from short-term sets, which do not represent the range of natural phenomena (Konikow and Bredehoeft, 1992). Accordingly, short-term extreme events may override average conditions (Petersen et al., 1987), represented by models. Thus an accurate quantitative prediction of N dynamics and nitrate loss from agricultural systems seems impossible (Jury and Flühler, 1992; Richter and Benbi, 1996).

Transition zones present even more severe obstacles to prediction. Variability and heterogeneity in terms of soils, biogeochemistry and water pathways in ecotones are much greater than the additive properties of adjacent resources (Merot and Durand, 1997; Naiman et al., 1988). The non-linearity of retention processes, the intricate physical structure and influence of memory effects (land use legacy) turn riparian zones into singularities (Breckling, 1992), for which a quantitative prediction seems unattainable (Wagenet, 1998). The connection of compartments and ecosystems on the landscape level offers additional problems.

The linkage of fluxes between different compartments is generally not well understood; e.g., the matter transfer between the unsaturated and the saturated zone (Del Re and Trevisan, 1995), and lateral fluxes and the flux of substances between adjacent ecosystems (Grunewald, 1996). Even in detailed, site specific case studies, a mechanistic knowledge of these interactions has not been obtained.

9. A framework for landscape risk assessment

In a framework for sustainable agriculture and in the light of the precautionary principle (O'Riordan and Jordan, 1995; Westra, 1997) system uncertainties as reflected in simulation models for ecosystems need to be acknowledged (Haag and Kaupenjohann, 2000). The concomitant shift from impact-oriented to risk-oriented approaches favours methods which address environmental risks, capacities (Cartwright, 1994) and output potentials and which aim at the identification of problem areas and risky management options. As indicators of (un-)sustainable landscape management budget approaches and simple output potentials are suitable. To indicate the risk of nutrient loss, water and nutrient budgets may be computed. While the compilation of budgets contributes little to the understanding of a system (Stockdale et al., 1997), budgets hint at disequilibria long before measurement or other methods indicate elevated soil concentrations or matter loss with confidence (Baccini and von Steiger, 1993). Output potentials for larger temporal and spatial scales may be more reliable (Stockdale et al., 1997), as larger areas like watersheds tend to behave more determinate than smaller ones (Corre et al., 1996; Groffman et al., 1987; Wagenet, 1998). The budget approach, however, takes no account of the spatial setting into which agricultural sites are embedded. Budgets should thus be part of a larger screening scheme, which could encompass the following categories of risk potentials:

(a) *Site specific risk* which is represented by simple, physical factors and which is linked to the soil, topography and climate (see, e.g., Marks and Alexander, 1992; Gäth and Wohlrab, 1994; Hölting et al., 1995 for Central Europe). For nitrate leaching the frequency of soil water exchange as a function of water surplus and texture class is a useful indicator (Gäth and Wohlrab, 1994).

(b) *Agricultural activity risk* is assessed with budget approaches, indicating long-term risks and providing hints at potential disequilibria (Baccini and von Steiger, 1993; Isermann and Isermann, 1997; Umweltbundesamt, 1997).

(c) *Headwater contamination risk* (local risk): The spatial setting of an agricultural site and of agricultural landscapes are to be accounted for. Corridors, their proximity to agricultural sites and their propensity to matter input deserve particular attention: Transport in conduits tends to increase scale and scope as conduits usually form part of a hierarchical, unidirectional networks. Cartographic approaches may indicate the abundance and proximity of corridors and the abundance of retention compartments within a landscape section. Quantitative measures for landscape pattern (Gustafson, 1998) and GIS applications may facilitate operationalisation.

(d) *Regional and global scale risk* is assessed qualitatively, based on the criteria of scale and scope and quantitatively, based on life cycle assessment (Vito, 1998), which e.g., may indicate overall global warming potential due to N fertilization.

Such a screening approach evaluates risk potentials, while it leaves out of consideration actual matter fluxes. The approach is thus restricted to the identification of key contributor and problem areas; it may be followed by site specific process studies or monitoring of environmental quality.

10. Remediation concepts

System approaches are advocated (Ikerd, 1993) focussing on nutrient cycles (Hendrix et al., 1992; Magdoff et al., 1997), which should be both tight with regard to spatial and temporal scales and close with regard to matter loss, ensuring a maximum of reversibility/controllability. The plot is the valve, where losses ultimately occur, hence optimization of cycles on the plot scale is imperative. As the plot is part of a hierarchy of landuse and production systems, aside with the plot level, the farm and the regional level also call for optimized cycles.

Detachment of (quasi-industrial) dairy and livestock production from the spatial extension of farmland (Steinfeld et al., 1996) imposes major constraints on cycling approaches: While plant production reaches a N efficiency of 57%, overall agricultural N efficiency is only 25%, as 85% of plant production, together with imported feeds, are utilized in animal production (Isermann and Isermann, 1998). With the carrying capacity of agricultural land being overstrained, fields and grassland frequently function as waste-dumps for excess nutrients from livestock (Isermann and Isermann, 1997). As animal production dominates the agricultural N cycle, it becomes a key driver as to N overload.

A shift away from linear concepts, in which wastes (like excretions in animal production or nitrate in plant production) are considered, the norm should lead to integrated systems targeting total throughput, i.e. systems making optimal use of inputs and mimicking natural cycles. Such a concept of “zero emission” has recently been developed for industry (Mshigeni and Pauli, 1996); it could also be useful for industrial agriculture.

The optimization of production systems on farm and larger scales remains within the realm of sectoral approaches. While in Central Europe fertilization accounts for 83% of total net anthropogenic N input (Howarth et al., 1996), but agricultural production is one subsystem in regional N metabolism. Regional approaches which assess matter fluxes among and matter budgets of different sectors (German Council of Environmental Advisors, 1996) are a way of addressing and tackling disequilibria on larger scales. Tools for the assessment of regional metabolism (Baccini and Bader, 1996; Baccini and Brunner, 1991) and quantitative examples, including N fluxes and budgets on a regional level, have been developed recently for Central Europe (Baccini and Bader, 1996; Brunner and Baccini, 1992; Henseler et al., 1992). The identification of key contributors and key fluxes may guide optimization on an integrated, regional level.

11. Conclusions

Different landscape elements exert control on the flux and fate of excess nutrients such as nitrate. The conceptual approach, which distinguishes retention compartments and corridors and which provides for the scalar assessment of risks induced by emissions can be adapted to other agricultural inputs like pesticides. Retention of nitrate on the local scale, ranging from the field to the stream, has been shown to be of limited and/or of uncertain extent in many compartments on the way from the field to the stream. Storage of N in vegetation, soil organic matter and groundwater may delay the emission of excess N for decades, masking past and present N disequilibria and overloads. On the regional level, elimination in rivers and lakes may contribute to the reduction of N discharge to the sea. Notwithstanding, N discharge has experienced a manifold increase in comparison to preindustrial times, leading to the eutrophication of coastal waters. Denitrification and the concomitant production of N_2O together with emissions arising from fertilizer production may shift the issue of N overload from a terrestrial–aquatic to an atmospheric problem.

Current agricultural practices and end-of-the-pipe solutions (e.g., buffer zones) seem rather unsustainable in view of the unpredictability of matter fluxes, of the uncertainties considering retention behaviour of landscape elements, of the often limited, partly non-renewable retention potentials, and of the only temporary storage of N in landscapes. Instead of short-term, small-scale considerations, an integrated system approach should be pursued, which envisages tight and close cycles and the optimization of N fluxes and budgets at site, farm and regional level. On the latter, both the fluxes induced by the agricultural production and the agricultural sector as a whole and the fluxes arising from other human activities need to be assessed and reconciled.

Acknowledgements

We are indebted to Gunda Matschonat, Florian Diekmann, Robert Vandré and Jan Siemens for their valuable suggestions on an earlier draft. We thank Claudia Mai-Peter for assistance in the preparation of the manuscript. This work was carried out in the framework of the project “Sustainable production and utilization of energy crops” supported by the German Environmental Foundation (Deutsche Bundesstiftung Umwelt).

References

Addiscott, T.M., 1995. Entropy and sustainability. Eur. J. Soil Sci. 46, 161–168.

Addiscott, T.M., 1997. A critical review of the value of buffer zone environments as a pollution control tool. In: Naycock, N.E., Burt, T.P., Goulding, K.W., Pinay, G. (Eds.), Buffer Zones: Their Processes and Potential in Water Protection. Quest Environment, Hertfordshire, pp. 236–243.

Addiscott, T.M., Whilmore, A.P., Powlson, D., 1991. Farming, Fertilizer and the Nitrate Problem. CAB International, Wallingford, UK.

Addy, K.L., Gold, A.J., Groffman, P.M., Jacinthe, P.A., 1999. Ground water nitrate removal in subsoil of forested and mowed riparian buffer zones. J. Environ. Qual. 28, 962–970.

Ahl, V., Allen, T.F.H., 1996. Hierarchy Theory: A Vision, Vocabulary, and Epistemology. Columbia University Press, New York.

Allen, T.F.H., Hoekstra, T.W., 1992. Toward a Unified Ecology. Columbia University Press, New York.

Aller, L., Bennett, T., Lehr, J.H., 1987. DRASTIC: A standardized system for evaluating ground water pollution potential using hydrogeologic settings. In: EPA (Ed.), EPA 600/2-87/035. EPA, Oklohoma.

Baccini, P., Bader, H., 1996. Regionaler Stoffhaushalt: Erfassung, Bewertung und Steuerung. Spektrum Akad. Verl., Heidelberg.

Baccini, P., Brunner, P.H., 1991. Metabolism of the Anthroposphere. Springer, Berlin.

Baccini, P., von Steiger, B., 1993. Die Stoffbilanzierung landwirtschaftlicher Böden — Eine methode zur Früherkennung von Bodenveränderungen. Z. Pflanzenernähr. Bodenk. 156, 45–54.

Bach, M., 1987. Die potentielle Nitratbelastung des Sickerwassers durch die Landwirtschaft in der BRD. Göttinger Bodenkundliche Berichte 93, 1–186.

Bach, M., Fabis, J., Frede, H.-G., Herzog, I., 1994. Kartierung der potentiellen Filterfunktion von Uferstreifen. 1. Teil: methodik der Kartierung. Z. Kulturtechnik Landentwickl 35, 148–154.

Bach, M., Fischer, P., Frede, H.-G., 1996. Gewässerschutz durch Abstandsauflagen. Nachrichtenbl. Deut. Pflanzenschutzd. 48, 60–62.

Bach, M., Fabis, J., Frede, H.-G., 1997. Filterwirkung von Uferstreifen für Stoffeinträge in Gewässer in unterschiedlichen Landschaftsräumen, Vol. 28. DVWK, Bonn.

Barrett, G.W., 1992. Landscape ecology: designing sustainable agricultural landscapes. J. Sustainable Agric. 2, 83–98.

Bencala, K.E., 1993. A perspective on stream–catchment connections. J. N. Am. Benthol. Soc. 12, 44–47.

Beven, K., 1996. The limits of splitting: hydrology. Sci. Total Environ. 183, 89–97.

Bidois, J., 1999. Amenagement de zones humides ripariennes pour la reconquête de la qualité de l'eau: éxperimentation et modelisation. Dissertation Thesis.

Billen, G., Somville, M., DeBecker, E., Servais, P., 1985. A nitrogen budget of the scheldt hydrographical basin. Neth. J. Sea Res. 19, 223–230.

Blöschl, G., Sivapalan, M., 1995. Scale issues in hydrological modelling — a review. Hydrol. Process. 4, 251–290.

Borchers, U., 1993. Erfassung und Charkterisierung von Substraten in pleoistozänen Aquiferen sowie deren Verwertbarkeit für die mirkobielle Nitratreduktion. Cuvillier, Göttingen.

Böttcher, J., Strebel, O., Duynisveld, W., 1990a. Microbial denitrification in the groundwater of a sandy aquifer. Mitteilgn. Dtsch. Bodenkundl. Gesellsch. 60, 265–270.

Böttcher, J., Strebel, O., Voerkelius, S., Schmidt, H.L., 1990b. Using isotope fractionation of nitrate-nitrogen and nitrate-oxygen for evaluation of microbial denitrification in a sandy aquifer. J. Hydrol. 114, 413–424.

Boulton, A.J., Findlay, S., Marmonier, P., Stanley, E.H., Valett, M., 1998. The functional significance of the hyporheic zone in streams and rivers. Ann. Rev. Ecol. Syst. 29, 59–81.

Bouma, J., 1992. Influence of soil macroporosity on environmental quality. Adv. Agron. 47, 1–37.

Bouwer, W., 1995. Wasser- und Stickstoffumsatz im Boden- und Grundwasserbereich eines Wassereinzugesgebietes in Niedersachsen, Vol. 6. Justus-Liebig-University, Gießen.

Bouwman, A.F., 1992. Soils and the Greenhouse Effect. Wiley, Chichester, UK.

Breckling, B., 1992. Uniqueness of ecosystems versus generalizability and predictability in ecology. Ecol. Model. 63, 13–27.

Bren, L.J., 1993. Riparian zone, stream, and floodplain issues: a review. J. Hydrol. 150, 277–299.

Bronswijk, J.J.B., Hamminga, W., Oostindie, K., 1995. Rapid nutrient leaching to groundwater and surface water in clay soil areas. Eur. J. Agron. 4, 431–439.

Brunner, P.H., Baccini, P., 1992. Regional material management and environmental protection. Waste Mgmt. Res. 10, 420.

Canter, L.W., 1997. Nitrates in Groundwater. CRC Press, Boca Raton, FL.

Caraco, N.F., Cole, J.J., 1999. Human impact on nitrate export: an analysis using major world rivers. Ambio 28, 234–259.

Cartwright, N., 1994. Nature's Capacities and Their Measurement. Oxford University Press, Oxford.

Castelle, A.J., Johnson, A.W., Conolly, C., 1994. Wetland and stream buffer size requirements — a review. J. Environ. Qual. 23, 878–882.

Corre, M.D., Kessel, V.C., Pennock, D.J., 1996. Landscape and seasonal patterns of nitrous oxide emissions in a semiarid region. Soil Sci. Soc. Am. J. 60, 1806–1815.

Correll, D.L., 1997. Buffer zones and water quality protection: general principles. In: Naycock, N.E., Burt, T.P., Goulding, K.W., Pinay, G. (Eds.), Buffer Zones: Their Processes and Potential in Water Protection. Quest Environment, Hertfordshire, pp. 7–20.

Cronan, C.S., Piampiano, J.T., Patterson, H.H., 1999. Influence of land use and hydrology on exports of carbon and nitrogen in a Maine river basin. J. Environ. Qual. 28, 953–961.

Crutzen, P.J., 1970. The influence of nitrogen oxides on the atmospheric ozone content. Quart. J. R. Meteorol. Soc. 96, 320–325.

David, M., Gentry, L.E., Kovacic, D.A., Smith, K.M., 1997. Nitrogen balance in and export from an agricultural watershed. J. Environ. Qual. 26, 1038–1048.

Davis, S.N., DeWiest, R.J.M., 1991. Hydrogeology. Krieger, Malabar, FL.

Dekker, L.W., Bouma, J., 1984. Nitrogen leaching during sprinkler irrigation of a Dutch clay soil. Agric. Water Mgmt. 8, 37–47.

Del Re, A.A.M., Trevisan, M., 1995. Selection criteria of xenobiotic leaching models in soil. Eur. J. Agron. 4, 465–472.

Devito, K.J., Dillon, P.J., Lazerte, B.D., 1990. Phosphorus and nitrogen retention in five Precambrian shield wetlands. Biogeochemistry. 8, 185–204.

de Willigen, P., 1991. Nitrogen turnover in the soil–crop system; comparison of fourteen simulation models. Fertil. Res. 27, 141–149.

de Willigen, P., Neetson, J.J., 1985. Comparison of six simulation models for the nitrogen cycle in the soil. Fertil. Res. 8, 157–171.

Dijkstra, J.P., Hack, M., 1995. Simulation of different management options within integrated arable farming affecting nitrate leaching. In: Schoute, J. (Ed.), Scenario Studies for the Rural Environment. Kluwer Academic Press, Dordrecht, pp. 329–333.

Dillaha, T.A., Sherrard, J.H., Lee, D., 1989. Long-term effectiveness of vegetative filter strips. Water Environ. Technol. (November), 419–421.

Dowdell, R.J., Burford, J.R., Crees, R., 1979. Losses of nitrous oxides dissolved in drainage water from agricultural land. Nature 278, 342–343.

Eckert, H., Breitschuh, G., 1994. Kritische Umweltbelastung Landwirtschaft (KUL) — Eine methode zur analyse und Bewertung der ökologischen situation von Landwirtschaftbetrieben. Arch. Acker- Plf. Boden 38, 149–163.

Eichner, M.J., 1990. Nitrous oxide emissions from fertilized soils: summary of available data. J. Environ. Qual. 19, 272–280.

Fennessy, M.S., Cronk, J.K., 1997. The effectiveness and restoration potential of riparian ecotones for the management of nonpoint source pollution, particularly nitrate. Crit. Rev. Environ. Sci. Technol. 27, 285–317.

Ferguson, S.J., 1994. Denitrification and its control. Antonie van Leeuwenhoek 66, 89–110.

Forman, R.T.T., 1995. Some general principles of landscape and regional ecology. Landscape Ecol. 10, 133–142.

Frede, H.-G., Bach, M., 1995. Landschaftsstoffhaushalt. In: Blume, H.-P. (Ed.), Handbuch der Bodenkunde. Ecomed, Landsberg, pp. 1–34.

Freney, J.R., 1997. Emission of nitrous oxide from soils used for agriculture. Nutr. Cycling Agroecosyst. 49, 1–6.

Gäth, S., Wohlrab, B., 1994. Strategiekonzept der deutschen bodenkundlichen Gesellschaft zur Reduzierung standort- und nutzungsbedingter Belastungen des Grundwassers mit Nitrat. In: DVWK Schriften 106. Wirtschafts- und Verlagsgesellschaft Gas und Wasser, Bonn.

German Council of Environmental Advisors, 1996. Environmental concepts of sustainable use of rural areas. Special Report. Metzler-Poeschel, Stuttgart.

Gilbert, J., Dole-Olivier, M., Marmonier, P., Vervier, P., 1990. Surface-groundwater ecotones. In: Naiman, R.J., Décamps, H. (Eds.), Ecology and Management of Aquatic–terrestrial Ecotones. UNESCO, Paris, pp. 7–21.

Gilliam, J.W., Parsons, J.E., Mikkelsen, R.L., 1997. Nitrogen dynamics and buffer zones. In: Naycock, N.E., Burt, T.P., Goulding, K.W., Pinay, G. (Eds.), Buffer Zones: Their Processes and Potential in Water Protection. Quest Environment, Hertfordshire, pp. 54–61.

Gleich, A.V., 1998. Kriterien zur charakterisierung von Techniken und Stoffen. Z. Umweltchem. Ökotox. 11, 21–32.

Göttlicher-Göbel, U., 1987. Wasserqualität von Fließgewässern landwirtschaftlich genutzter Einzugegebiete insbesondere bei Hochwasserabflüssen-Justus-Liebig-Universität Gießen, Gießen.

Groffman, P.M., Gold, A.J., 1998. Nitrous oxide production in riparian zones and groundwater. Nutr. Cycling Agroecosyst. 52, 179–186.

Groffman, P.M., Tiedje, J.M., Robertson, G.P., Christensen, S., 1987. Denitrification at different temporal and geographical scales: proximal and distal controls. In: Wilson, J.R. (Ed.), Advances in Nitrogen Cycling in Agricultural Ecosystems. CAB International, Wallingford, pp. 174–192.

Groot, J.J.R., de Willigen, P., Verberne, E.L.J., 1991. Nitrogen turnover in the soil–crop system. Developments in Plant and Soil Sciences, Vol. 44. Kluwer Academic Publishers, Dordrecht.

Grunewald, K., 1996. Großräumige Bodenkontaminationen: Wirkungsgefüge, Erkundungsmethoden und Lösungsansätze. Springer, Berlin.

Gustafson, E.J., 1998. Quantifying landscape spatial pattern: what is the state of the art? Ecosystems 1, 143–156.

Haag, D., Kaupenjohann, M., 2000. Biogeochemical models in the environmental sciences: the dynamical system paradigm and the role of simulation modeling. Hyle — International journal for the philosophy of chemistry (http://www.uni-karlsruhe.de/~ea06/hyle.html). 6, 117–142.

Hamm, A., 1991. Studie über Wirkungen und Qualitätsziele von Nährstoffen in Fließgewässern. Academia Verlag, Sankt Augustin.

Hansen, A.J., di Castri, F., Naiman, R.J., 1988. Ecotones: what and why? In: di Castri, F., Hansen, A.J., Holland, M.M. (Eds.), A New Look at Ecotones. Biol. Inter. Special Issue 17, pp. 9–46.

Hauha, M., Neal, C., Hooper, R., Christophersen, N., 1996. Summary of a workshop on ecosystem modeling: The end of an era? The Science of the Total Environment 183, 1–5.

Haycock, N.E., Pinay, G., Walker, C., 1993. Nitrogen retention in river corridors: European perspective. Ambio 22, 340–346.

Haycock, N.E., Pinay, G., Burt, T.P., Goulding, K.W.T., 1997. Buffer zones: current concerns and future directions. In: Naycock, N.E., Burt, T.P., Goulding, K.W., Pinay, G. (Eds.), Buffer Zones: Their Processes and Potential in Water Protection. Quest Environment, Hertfordshire, pp. 236–243.

Helming, K., Frielinghaus, M., 1998. Skalenaspekte der Bodenerosion. In: Steinhardt, U., Volk, M. (Eds.), Regionalisierung in der Landschaftökologie, Vol. UFZ-Bericht X. UFZ, Leipzig.

Hendrix, P.F., Coleman, D.A., Crossley, D.A., 1992. Using knowledge of soil nutrient cycling to design sustainable agriculture. J. Sustainable Agric. 2, 63–81.

Henseler, G., Scheidegger, R., Brunner, P.H., 1992. Die Bestimmung von Stoffflüssen im Wasserhaushalt einer region. Vom Wasser 78, 91–116.

Hession, W., Huber, L., Mostaghimi, S., Shanholtz, V., McClellan, P., 1989. BMP Effectiveness Evaluation Using AGNPS and a GIS International Winter Meeting. American Society of Agricultural Engineers, Lousiana, pp. 1–18.

Heyder, D., 1993. Nitratveralgerung, Wasserhaushalt und Denitrifikationspotential in mächtigen Lößdecken und einem Tonboden bei unterschiedlicher Bewirtschaftung, Vol. 19. Institut für Bodenkunde Bonn, Bonn.

Hill, A.R., 1988. Factors influencing nitrate depletion in a rural stream. Hydrobiology 160, 111.

Hill, A.R., 1996. Nitrate removal in stream riparian zones. J. Environ. Qual. 25, 743–755.

Hill, A.R., 1997. The potential role of in-stream and hyporheic environments as buffer zones. In: Naycock, N.E., Burt, T.P., Goulding, K.W., Pinay, G. (Eds.), Buffer Zones: Their Processes and Potential in Water Protection. Quest Environment, Hertfordshire, pp. 115–127.

Hillbricht-Ilkowska, A., 1995. Managing ecotones for nutrients and water. Ecol. Inter. 22, 77–93.

Hiscock, K.M., Lloyd, J.W., Lerener, D.N., 1991. Review of natural and artificial denitrification of groundwater. Water Res. 25, 1099–1111.

Hölting, B., 1980. Hydrogeologie. Enke, Stuttgart.

Hölting, B., Haertlé, T., Hohberger, K., Nachtigall, K., Villinger, E., Weinzierl, W., Wrobel, J., 1995. Konzept zur Ermittlung der Schutzfunktion der Grundwasserüberdeckung. Geol. Jb. C 63, 5–24.

Houghton, J.T., 1994. Climate change 1994: radiative forcing of climate change and an evaluation of the IPCC IS92 emission scenarios. Reports of Working Groups I and III of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, p. 339.

Howarth, R.W., Billen, G., Swaney, D., Townsend, A., Jaworski, L., Lajtha, K., Downing, J.A., Elmgren, R., Caraco, N., Jordan, T., Berendse, F., Freney, J., Kudeyarov, V., Murdoch, P., Zhao-Liang, Z., 1996. Regional nitrogen budgets and riverine N and P fluxes for the drainages to the North Atlantic Ocean: natural and human influences. Biogeochemistry 35, 75–139.

Hunsaker, C.T., Levine, D.A., 1995. Hierarchical approaches to the study of water quality in rivers. BioScience 45, 193–203.

Ikerd, J.E., 1993. The need for a systems approach to sustainable agriculture. Agric. Ecosyst. Environ. 46, 147–160.

Isermann, K., 1993. Territorial, continental and global aspects of C, N, P and S emissions from agricultural ecosystems. In: Wollast,

R. (Ed.), Advanced Research Workshop on Interactions of C, N, P and S Biogeochemical Cycle. Springer, Berlin, pp. 79–121.

Isermann, K., Isermann, R., 1997. Globale, territoriale und betriebliche Nährstoffbilanzierung. In: Umweltbundesamt (Ed.), Stoffbilanzierung in der Landwirtschaft, Vol. 20. Umweltbundesamt, Wien, pp. 241–313.

Isermann, K., Isermann, R., 1998. Food production and consumption in Germany: N flows and N emissions. Nutr. Cycling Agroecosyst. 52.

Jacinthe, P.A., Groffman, P.M., Gold, A.J., Mosier, A., 1998. Patchiness in microbial nitrogen transformations in groundwater in a riparian forest. J. Environ. Qual. 27, 156–169.

Jansson, M., Andersson, R., Berggren, H., Leonardson, L., 1994a. Wetlands and lakes as nitrogen traps. Ambio 23, 320–325.

Jansson, M., Leonardson, L., Fejes, J., 1994b. Denitrification and nitrogen retention in a farmland stream in Southern Sweden. Ambio 23, 326–331.

Johnston, C.A., 1991. Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. Crit. Rev. Environ. Control 21, 491–565.

Ju, S.H., Kung, K.S., 1997. Steady-state funnel flow: its characteristics and impact on modeling. Soil Sci. Soc. Am. J. 61, 416–427.

Jury, W.A., Flühler, H., 1992. Transport of chemicals through soil: mechanism, models and field applications. Adv. Agron. 47, 141–201.

Kaltschmitt, M., 1997. Nachwachsende Energieträger: Grundlagen, Verfahren, ökologische Bilanzierung. Vieweg, Braunschweig, Wiesbaden.

Kampis, G., 1991. Self-modifying Systems in Biology and Cognitive Science: A New Framework for Dynamics, Information and Complexity. Pergamon Press, Oxford.

Kerschberger, M., Eckert, H., 1994. Kritische Umweltbelastung Landwirtschaft (KUL) — Analyse und Bewertung der Kategorie Düngung. Arch. Acker- Plf. Boden 38, 361–371.

Kirkby, M.J., 1978. Hillslope hydrology. Willy, Chicester, pp. 389.

Kladikov, E.J., Grochulska, R.F., Turco, G.E., Eigel, J.D., 1999. Pesticide and nitrate transport into subsurface tile drains of different spacings. J. Environ. Qual. 28, 997–1004.

Kohl, D.H., Shearer, G.B., Commoner, B., 1971. Fertilizer nitrogen: contribution to nitrate in surface water in a cornbelt watershed. Science 174, 1331–1334.

Konikow, L.F., Bredehoeft, J.D., 1992. Ground-water models cannot be validated. Adv. Water Res. 15, 75–83.

Korom, S.F., 1992. Natural denitrification in the saturated zone: a review. Water Resour. Res. 28, 1657–1668.

Kroeze, C., Seitzinger, S.P., 1998. Nitrogen inputs to rivers, estuaries and continental shelves and related nitrous oxide emissions in 1990 and 2050: a global model. Nutr. Cycling Agroecosyst. 52, 195–212.

Lange, H., 1998. Are ecosystems dynamical systems? Inter. J. Comput. Anticip. Syst. 3, 169–186.

Lehn, H., Flaig, H., Mohr, H., 1995. Vom Mangel zum Überfluß: Störungen im Stickstoffkreislauf. Gaia 4, 13–25.

Leuchs, W., 1988. Geochemische und mineralogische Auswirkungen beim mikrobiellen Abbau organischer Substanz in einem anoxischen Porengrundwasserleiter. Z. dt. geol. Ges 139, 415–423.

Li, Y., Ghodrati, M., 1994. Preferential transport of nitrate through soil columns containing root channels. Soil Sci. Soc. Am. J. 58, 653–659.

Line, D.E., Arnold, J.A., Osmond, S., Coffey, D., Gale, J.A., Spooner, A., Jennings, G.D., 1993. Nonpoint sources. Water Environ. Res. 64, 558–571.

Loague, K., Corwin, D., Ellsworth, T., 1998. The challenge of predicting nonpoint source pollution. Environ. Sci. Technol. 3, 130–133.

Logan, T.J., 1993. Agricultural best management practices for water pollution control: current issues. Agric., Ecosyst. Environ. 46, 223–231.

Lowrance, R.R., 1992. Groundwater nitrate and denitrification in a coastal plain riparian forest. J. Environ. Qual. 21, 401–405.

Lowrance, R.R., Pionke, H.B., 1989. Transformations and movement of nitrate in aquifer systems. In: Follett, R.F. (Ed.), Nitrogen Management and Ground Water Protection, Vol. 21. Elsevier, Amsterdam, pp. 373–392.

Magdoff, F., Lanyon, L., Liebhardt, B., 1997. Nutrient cycling, transformations, and flows: implications for a more sustainable agriculture. Adv. Agron. 60, 1–73.

Marchetti, R., Donatelli, M., Spallacci, P., 1997. Testing denitrification functions of dynamic crop models. J. Environ. Qual. 26, 394–401.

Mariotti, A., 1994. Dénitrification in situ dans les eaux souterraines, processus naturels ou provoqués: une revue. Hydrogéologie 3, 43–68.

Marks, R., Alexander, J., 1992. Anleitung zur Bewertung des Leistungsvermögens des Landschaftshaushaltes (BA LVL). Zentralausschuss für Dt. Landeskunde, Trier. pp. 222.

McElroy, M.B., Woofsy, S.C., 1985. Nitrous Oxide Sources and Sinks Atmospheric Ozone 1985, Vol. 1. NASA, Washington, p. 81.

McMahon, P.B., Dennehy, K.F., 1999. N_2O emissions from a N-enriched river. Environ. Sci. Technol. 33, 21–25.

Merot, P., Durand, P., 1997. Modelling the interaction between buffer zones and the catchment. In: Naycock, N.E., Burt, T.P., Goulding, K.W., Pinay, G. (Eds.), Buffer Zones: Their Processes and Potential in Water Protection. Quest Environment, Hertfordshire, pp. 208–217.

Montgomery, E., Coyne, M.S., Thomas, G.W., 1997. Denitrification can cause variable NO_3 concentration in shallow groundwater. Soil Sci. 162, 148–156.

Mosier, A., Kroeze, C., Nevison, C., Oenema, O., Seitzinger, S., van Cleemput, O., 1998. Closing the global N_2O budget: nitrous oxide emissions through the agricultural nitrogen cycle.. Nutr. Cycling Agroecosyst. 52, 225–248.

Mosley, M.P., 1982. Subsurface flow velocities through selected forest soil, South Island, New Zealand. J. Hydrol. 55, 65–92.

Mshigeni, K., Pauli, G., 1996. In: Proceedings of the Second Annual UNU World Congress on Zero Emissions. United Nations University, Tokyo.

Müller, F., 1992. Hierarchical approaches to ecosystem theory. Ecol. Model. 63, 215–242.

Naiman, R.J., 1990. Ecology and Management of Aquatic–terrestrial Ecotones, Vol. 4. UNESCO, Paris.

Naiman, R.J., Décamps, H., Pastor, J., Johnston, C.A., 1988. The potential importance of boundaries to fluvial ecosystems. J. N. Am. Benthol. Soc. 7, 289–306.

Norton, B.G., 1995. Ecological integrity and social values: at what scale? Ecosyst. Hlth. 1, 228–241.

Norton, B.G., Ulanowicz, R.E., 1992. Scale and biodiversity policy: a hierarchical approach. Ambio 21, 244–249.

Novotny, V., 1986. A review of hydrological and water quality models used for simulation of agricultural pollution. Developments Environmental Modelling 10, 9–36.

Obermann, P., 1982 Contamination of groundwater in the lower Rhine region due to agricultural activities. In: Proceedings of the International Symposium on the Impact of Agricultural Activities on Groundwater, Prague, pp. 285–297.

Obermann, P., 1991. Wasserwirtschaftliche Bedeutung steigender Nitratgehalte im Grundwasser. In: Schindler, R. (Ed.), DENIPLANT, Vol. 5. Forschungszentrum Jülich GmbH, Jülich, pp. 79–90.

Odum, E.P., 1969. The strategy of ecosystem development. Science 164, 262–270.

Odum, E.P., 1984. Properties of agroecosystems. In: Lowrance, R. (Ed.), Agricultural Ecosystems: Unifying Concepts. Wiley, New York, pp. 5–11.

O'Neill, R.V., Johnson, A.R., King, A.W., 1989. A hierarchical framework for the analysis of scale. Landscape Ecol. 3, 193–205.

O'Riordan, T., Jordan, A., 1995. The precautionary principle in contemporary environmental politics. Environ. Values 4, 191–212.

Oreskes, N., Shrader-Frechette, K., Belitz, K., 1994. Verification validation, and confirmation of numerical models in the earth sciences. Science 263, 641–646.

PARCOM, 1994. Guidelines for calculating mineral balances. Document NUTAG 4/7/1-E.

Parton, W., Stewart, J., Cole, C., 1988. Dynamics of C, N, P and S in grassland soils: a model. Biogeochemistry 5, 109–131.

Peter, M., 1987. Zur Bedeutung des Zwischenabflusses aus landwirtschaftlich genutzten Einzugsgebieten beim Stoffeintrag in Gewässer unter besonderer Berücksichtigung des Nitrats. Mitteilgn. Dtsch. Bodenkundl. Gesellsch. 55 (II), 931–936.

Petersen, R.C., Madsen, B., Wilzbach, M., Magadaza, C.H., Paarlberg, A., Kullberg, A., Cummins, K., 1987. Stream management: emerging global similarities. Ambio 16, 166–179.

Petts, G.E., 1994. Rivers: dynamic components of catchment ecosystems. In: Calow, P. (Ed.), The Rivers Handbook: Hydrological and Ecological Principles, Vol. 2. Blackwell Scientific Publications, Oxford, pp. 3–22.

Pfenning, K.S., McMahon, P.B., 1996. Effect of nitrate, organic carbon, and temperature on potential denitrification rates in nitrate-rich riverbed sediments. J. Hydrol. 187, 283–295.

Pinay, G., Décamps, H., Chauvet, E., Fustec, E., 1990. Functions of ecotones in fluvial systems. In: Naiman, R.J., Décamps, H. (Eds.), Ecology and Management of Aquatic–terrestrial Ecotones. UNESCO, Paris, pp. 7–21.

Prasuhn, V., Braun, M., 1995. Regional differenzierte Abschätzung diffuser Phosphor- und Stickstoffeinträge in die Gewässer des Kantons Bern (Schweiz). Z. Kulturtechnik Landentwickl 36, 309–314.

Prasuhn, V., Spiess, E., Braun, M., 1996. Methoden zur Abschätzung der Phosphor- und Stickstoffeinträge aus diffusen Quellen in den Bodensee.

Prato, T., Shi, H., 1990. A comparison of erosion and water pollution control strategies for an agricultural watershed. Water Res. 26, 199–205.

Rapport, D.J., Gaudet, C., Karr, J.R., Baron, J.S., Bohlen, C., Jackson, W., Jones, B., Naiman, R.J., Norton, B., Pollock, M.M., 1998. Evaluating landscape health: integrating societal goals and biophysical process. J. Environ. Mgmt. 53, 1–15.

Rice, C.W., Rogers, K., 1993. Denitrification in subsurface environments: potential source for atmospheric nitrous oxide. In: ASA (Ed.), Agricultural Ecosystem Effects on Trace Gases and Global Climate Change, Vol. 55. ASA, Madison, pp. 121–132.

Richter, J., Benbi, D.K., 1996. Modeling of nitrogen transformations and translocations. Plant Soil 181, 109–121.

Risser, P.G., 1990. The ecological importance of land–water ecotones. In: Naiman, R.J., Décamps, H. (Eds.), Ecology and Management of Aquatic–terrestrial Ecotones. UNESCO, Paris, pp. 7–21.

Rode, M., Grunwald, S., Frede, H.-G., 1995. Methodik zur GIS-gestützten Berechnung von Nährstoffeinträgen in Fließgewässer durch Oberflächenabfluß mit dem Modell AGNPS. Z. Kulturtechnik Landentwickl 36, 63–68.

Ronen, D., Magaritz, M., Alman, E., 1988. Contaminated aquifers are a forgotten component of the global N_2O budget. Nature 335, 57–59.

Scheringer, M., 1999. Persistenz und Reichweite von Umweltchemikalien. Wiley–VCH, Weinheim.

Spalding, R.F., Parrot, J.D., 1994. Shallow groundwater denitrification. Sci. Total Environ. 141, 17–25.

Stagnitti, F., Parlange, J.-Y., Steenhuis, T.S., Boll, J., Pivetz, B., Barry, D.A., 1995. Transport of moisture and solutes in the unsaturated zone by preferential flow. In: Singh, V.P. (Ed.), Environmental Hydrology. Kluwer Academic Press, Dordrecht, pp. 193–224.

Steinfeld, H., de Haan, C., Blackburn, H., 1996. Livestock–environment Interactions: Issues and Options. FAO, Rome.

Steinheimer, T.R., Scoggin, K.D., Kramer, L.A., 1998. Agricultural chemical movement through a field-size watershed in Iowa: subsurface hydrology and distribution of nitrate in groundwater. Environ. Sci. Technol. 32, 1039–1047.

Steininger, M., Abdank, H., Meissner, R., 1997. Untersuchungen zum diffusen Nitrataustrag aus einem landwirtschaftlich genutzten Kleineinzugsgebieten des Unterharzes. Z. Kulturtechnik Landentwickl 38, 82–86.

Steinmann, F., 1991. Die Bedeutung von Gewässerrandstreifen als Kompensationszonen im Grenzbereich zwischen landwirtschaftlichen Nutzflächen und Gewässern für die Immobilisierung der löslichen Fraktionen von Stickstoff und Phosphor aus der gesättigten phase. Dissertation-Kiel.

Stockdale, E.A., Gaunt, J.L., Vos, J., 1997. Soil–plant nitrogen dynamics: what concepts are required? Eur. J. Agron. 7, 145–159.

Tim, U.S., Jolly, R., 1994. Evaluating agricultural non-point-source pollution using integrated geographic information systems and hydrologic/water quality model. J. Environ. Qual. 23, 25–35.

Tufford, D.L., McKellar, H.M., Hussey, J.R., 1998. In-stream nonpoint source nutrient prediction with land-use proximity and seasonality. J. Environ. Qual. 27, 100–111.

Turner, M.G., 1989. Landscape ecology. Ann. Rev. Ecol. Sys. 20, 171–197.

Umweltbundesamt, 1997 Stoffbilanzierung in der Landwirtschaft. In: Umweltbundesamt (Ed.), Tagungsberichte, Vol. 20. Umweltbundesamt, Wien.

Vagstad, N., Eggestad, H.O., Hoyas, T.R., 1997. Mineral nitrogen in agricultural soils and nitrogen losses: relation to soil properties, weather conditions and farm practice. Ambio 26, 266–272.

Vito, M., 1998. Proceedings of the International Conference on Life Cycle Assessment in Agriculture, Agro-Industry and Forestry, December 3–4, 1998, Brussels, Belgium.

Vitousek, P.M., Aber, J., Howarth, R.W., Likens, E.G., Matson, P., Schindler, D.W., Schlesinger, W.H., Tilman, D., 1997a. Human alteration of the global nitrogen cycle: causes and consequences. Ecol. Appl. 7, 737–750

Vitousek, P.M., Mooney, H.A., Lubchenco, J., Melillo, J.M., 1997b. Human domination of earth's ecosystems. Science 277, 494–499.

Vought, L.B.-M., Dahl, J., Pedersen, C.L., Lacoursiere, J.O., 1994. Nutrient retention in riparian ecotones. Ambio 23, 342–348.

Wagenet, R.J., 1998. Scale issues in agroecological research chains. Nutr. Cycling Agroecosyst. 50, 23–34.

Weller, D.E., Correl, D.L., Jordan, T.E., 1994. Denitrification in riparian forests receiving agricultural discharges. In: Mitsch, W.J. (Ed.), Global Wetlands: Old World and New. Elsevier, Amsterdam, pp. 117–131.

Wendland, F., 1992. Die Nitratbelastung in den Grundwasserlandschaften der alten Bundesländer (BRD), Vol. 8. Forschungszentrum Jülich, Jülich.

Wendland, F., 1994. Modelling the nitrate flow in ground-water provinces of the old federal states of the Federal Republic of Germany. Ecol. Model. 757 (6), 385–397.

Werner, W., 1994. Stickstoff- und Phosphateintrag in Fließgewässer Deutschlands unter besonderer Berücksichtigung des Eintragsgeschehens im Lockergesteinsbereich der ehemaligen DDR Agrarspectrum, Vol. 22. DLG-Verlag, Frankfurt, p. 243.

Westra, L., 1997. Post-normal science, the precautionary principle and the ethics of integrity. Foundations Sci. 2, 237–262.

Wiens, J.A., Crawford, C.S., Gosz, J.R., 1985. Boundary dynamics: a conceptual framework for studying landscape ecosystems. Oikos 45, 412–427.

Yoshinari, T., 1990. Emissions of N_2O from various environments. In: Revsbech, N.P., Sorensen, J. (Eds.), Denitrification in Soil and Sediment. Plenum Press, New York.

Zakosek, H., Zepp, H., 1993. Nitratbewegung im Boden und Untergrund. In: Zakosek, H., Lenz, F. (Eds.), Nitrat in Boden und Pflanze. Ulmer, Stuttgart, pp. 21–35.