DISSERTATIONES GEOGRAPHICAE UNIVERSITATIS TARTUENSIS
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NITROGEN AND PHOSPHORUS TRANSFORMATION IN RIPARIAN BUFFER ZONES OF AGRICULTURAL LANDSCAPES IN ESTONIA

VALDO KUUSEMETS

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VALDO KUUSEMETS

TARTU UNIVERSITY
PRESS

Institute of Geography, Faculty of Biology and Geography, University of Tartu, Estonia.

The Faculty Council of Biology and Geography, University of Tartu, has on the May 14, 1999 accepted this dissertation to be defended for the degree of Doctor of Philosophy (in Geography).

Opponent: Dr Lena B.-M. Vought, Department of Ecology, Limnology, University of Lund, Sweden.

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Tartu Ülikooli Kirjastuse trükikoda
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CONTENTS

ABSTRACT

Kuusemets, V. 1999. *Nitrogen and phosphorus transformation in riparian buffer zones of agricultural landscapes in Estonia.* Dissertationes Geographicae Universitatis Tartuensis No 8, Tartu University Press. Tartu.

The aim of the thesis was to investigate purification efficiency of different type of riparian buffer zones in Estonia consisting of native riparian communities. Buffer zones are multifunctional elements of landscape that improve water quality, protect air and soil, and increase biological and landscape diversity. The shallow groundwater quality changes and plant uptake of nutrients were emphasised.

A study of purification efficiency and nutrient uptake in plants was made in two riparian buffer zones with a complex of wet meadow and grey alder (*Alnus incana*) stand. In the less polluted Porijõgi test site, the 31 m wide buffer zone removed 50% of total nitrogen (total-N) and 78% of total phosphorus (total-P), while the heavily polluted a 51 m wide buffer zone in Viiratsi retained 87% of total-N and 84% of total-P. Load and purification efficiency displayed a significant relationship. The total-N removal in buffer strips was negative when the input load was less than 0.3 mg l^{-1} and the purification efficiency was positive when the input load exceeded 5 mg l^{-1}. The purification efficiency of total-P was positive when the input load exceeded 0.15 mg l^{-1}. Grass vegetation plays an important role in nutrient retention in riparian buffer zones. Biomass production of the plant community dominated by *Filipendula ulmaria* in the Porijõgi transect was up to 3601 g m^{-2}, the nitrogen assimilation was 85.8 g m^{-2}, the phosphorus assimilation 6.36 g m^{-2}. This is much higher than the production and N and P uptake of the grey alder (*Alnus incana*) community, being 1730, 20.5 and 1.5 g m^{-2}, respectively.

The outflow of total-N was 4.9 and outflow of total-P was 4.8 times lower in well-buffered watershed in comparison to similar watershed with less buffering ability.

An original method for dimensioning riparian buffer zones and buffer strips has been worked out. This is based on topography, soil, hydrological and vegetation parameters.

Regular management of buffer zones (harvesting trees and hay) can degrease the load in riparian buffers and thus, help to increase their efficiency.

Buffer zones and buffer strips along ditches, channels and riverbanks are most attractive poly-functional mitigation elements for watershed management. Therefore designing of buffer zones should be one important part for landscape planning.

ORIGINAL PUBLICATIONS

This thesis is based partly on unpublished data and partly on the following papers, which are included as appendices at the end of the thesis:

I Mander, Ü., **Kuusemets,** V. and Ivask, M. 1995. Nutrient dynamics of riparian ecotones: A case study from the Porijõgi River catchment, Estonia. *Landscape and Urban Planning*, 31, 333–348.

II Mander, Ü., Lõhmus, K., **Kuusemets,** V. and Ivask, M. 1997. The potential role of wet meadows and grey alder forests as buffer zones. *Buffer Zones, Their Processes and Potential in Water Protection, Proc. of the Int. Conf. on Buffer Zones.* Haycock, N. E. *et al.* (eds.). Oxford, Quest Environmental, Foundation for Water Research, pp. 35–46.

III **Kuusemets,** V., Mander, Ü., Ivask, M. and Lõhmus, K. Phosphorus retention in riparian buffer zones in agricultural landscapes in Estonia. 1998. In: *Proceedings of OECD workshop Practical and innovative measures for the control of agricultural phosphorus losses to water. 16–19 June 1998*, Belfast, UK, 114–115.

IV **Kuusemets,** V., Mander, Ü., Lõhmus, K. and Ivask. M. 1999. Shallow Groundwater Quality and Plant Uptake of Nitrogen and Phosphorus in Complex Riparian Buffer Zones. *Journal of Environmental Quality* (*submitted*).

V Mander, Ü., **Kuusemets,** V., Lõhmus, K. and Mauring, T. 1997. Efficiency and dimensioning of riparian buffer zones in agricultural catchments. *Ecological Engineering*, 8, 299–324.

VI **Kuusemets,** V. and Mander, Ü. 1999. Eecotechnological measures to control nutrient losses from catchments. *Water Science and Technology.* (*in press*).

Author's contribution

Publication I:	The author is fully responsible for the fieldwork, data collection and analysis and partly in writing the manuscript.
Publication II:	The author is responsible for the fieldwork, data collection and analysis and partly in writing the manuscript.
Publication III:	The author is fully responsible for the fieldwork, data collection and analysis and in writing the manuscript.
Publication IV:	The author is fully responsible for the fieldwork, data collection and analysis and in writing the manuscript.
Publication V:	The author is fully responsible for the fieldwork, data collection and analysis and partly in writing the manuscript.
Publication VI:	The author is partly responsible for the fieldwork, data collection and analysis and fully responsible in writing the manuscript.

1. INTRODUCTION

Rivers and streams of different order together with their riparian zones are essential multifunctional elements of the world-wide ecological network (Bischoff and Jongman, 1993; EECONET Declaration, 1993: Baldock *et al.*, 1994). Buffer zones and buffer strips have found to be important tools for achievement of ecological requirements. As land-water ecotones, riparian areas can effectively control nutrient fluxes from adjacent agriculturally used areas that has been widely described in many regions (Peterjohn and Correll, 1984; Lowrance *et al.*, 1984; Pinay and Decamps, 1988; Cooper, 1990; Risser, 1990; Uusi-Kämppä and Yläranta, 1992; Haycock and Pinay, 1993; Vought *et al.*, 1994). Importance of buffer zones has been demonstrated in areas with high pollution loads, such as feedlots (Doyle *et al.*, 1977; Young *et al.*, 1980; Dillaha *et al.*, 1989) or manure accumulating sites (Mander, 1985).

In some countries the complex structure of buffer zones is officially recommended or legislatively stated. In the USA the recommended complex buffer zone consists of three parts which are perpendicular to the stream bank or lake shore (sequentially from agricultural field to the water body): a grass strip, a young (managed) forest strip and an old (unmanaged) forest strip (Lowrance, 1991). In Estonia, stream and ditch banks covered with grey alder (*Alnus incana* (L.) Moench.) and willow species (*Salix spp.*) have been typical for rural landscapes and ecological network during the last century. However, with the beginning of intensive amelioration and land reclamation in 1960's, the traditional mosaic of rural landscapes has changed significantly. Many grasslands, small wetlands, woodlots, trees rows, stone fences and other ecologically important landscape elements were liquidated in order to enlarge and homogenise agricultural fields. Channels and ditches were usually constructed without forest/bush vegetation. Despite the dominating mentality of Soviet "agroindustrial" development, which did not pay attention to ecological objectives, the first experimental reaches of ditches with forest buffers were re-established in the 1970s to demonstrate the economical benefit of such ecological engineering measures (Mander, 1985). The legislative protection of riparian biotopes began also in the 1970s. Based on Estonian landscape planning, riparian buffer zones and buffer strips are often termed "water protection zones" and "water protection strips", respectively (Mander, 1989). In 1996 the Ministry of Environment constituted the regulation Guidelines of Landscape Ecological and Water Protection Measures for Building and Maintaining of Land Reclamation Systems. The regulation determines principles for designing and establishment of buffer zones and buffer strips based on the principles worked out during present study. Principle of buffer zones and buffer strips is applied also in the Law on Protection of Marine and Freshwater Coasts, Shores and Banks, 1994.

In the early 1970s there were only few papers that reported efficiency of buffer zones then there are now over 400 such papers per year. Despite of increasing rate of publication, our knowledge concerning the water purification processes in buffer zones is far from adequate (Correll, 1997). Because retention and efficiency rates vary greatly under different climatic and physico-geographical conditions, few proposals have been presented with design criteria for buffer zones and their establishment and management (Dillaha and Inamdar, 1997; Mander *et al.*, 1997b). The optimal vegetation and the most effective buffer width is still unclear (Correll, 1997).

Main objectives of this paper are:

1) to investigate purification efficiency of different type of riparian buffer zones in Estonia consisting of native riparian communities;
2) to clarify the influence of complex riparian buffer zones on nitrogen and phosphorus removal;
3) to verify the methods for dimensioning of riparian buffer zones.

2. BUFFER ZONES:
MAIN FUNCTIONS AND PROCESSES

2.1. Main functions of riparian buffers

Riparian biotopes have the following essential functions: (1) filtering of polluted overland and subsurface flow from intensively managed adjacent agricultural fields, (2) protecting banks of water bodies against erosion, (3) filtering polluted air, especially from local sources (*e.g.*, large farm complexes, agrochemically treated fields), (4) avoiding intensive growth of aquatic macrophytes by canopy shading, (5) improving the microclimate in adjacent fields, (6) creating new habitats in land/inland water ecotones, and (7) creating more connectivity in landscapes due to migration corridors and stepping stones (Mander, *et al.*, 1997b).

2.2. Hydrological and soil conditions of buffers

The key problem of nutrient removal in riparian buffers is the intensity of biological and physico-chemical processes that depend highly on the hydrological and soil conditions of the buffer zone. The major factors of hydrological conditions are volume and pathway of the water that is moving through the buffer (Correll, 1997). Main components of the water pathways are: overland flow; shallow ground water; deep ground water and stream water (Hill, 1990; Burt, 1997). All these pathways are mixing in riparian buffer zone area constituting hyporheic zone that plays important role of the purification processes in buffers (Stanford and Ward, 1988; Triska *et al.*, 1989; Hendricks and White, 1991). Domination of different components of pathways will determine different purification processes dominating in buffer. The pathways and the volume of the water movement are at the same time dependent upon the morphology and soil conditions of the valley where the stream is situating. Soil mineralogy is a very important determinant of the chemical composition of external inputs, whereas soil texture to a large extent determines the relative proportions of surface water and groundwater inputs (Correll, 1997)

2.3. Main removal processes taking place in buffer zones

2.3.1. Nitrogen removal

Main nitrogen removal processes are (Kadlec and Knight, 1996): (1) uptake and storage in vegetation; (2) microbial immobilisation and storage in the soil as organic nitrogen; (3) microbial conversion to gaseous form of nitrogen (denitrification); (4) ammonia volatilisation.

Storage in vegetation depends mostly on the age of the riparian community and is low where the net production is also low. Thus, young successional stages of forests and meadows show higher removal rates. Nitrogen storage in the soil has not been quantified in any riparian ecosystem study (Weller *et al.*, 1994). The soil nitrogen pool in riparian zones is large and spatially variable so small errors in bulk density or soil nitrogen analysis lead to large uncertainties in total soil nitrogen.

In riparian wetland ecosystems (both in forests and meadows) denitrification is believed to be the most significant factor for nitrogen removal (Lowrance, 1992; Pinay *et al.*, 1992; Brüsch and Nilsson, 1993; Haycock and Pinay, 1993; Pinay *et al.*, 1993). However, direct evidence is limited by the lack of accurate field methods for measuring denitrification and by the enormous spatial and temporal variability of denitrification (Tiedje *et al.*, 1989; Weller *et al.*, 1994). Table 1 shows the variability of the intensity of the most important processes relevant to nitrogen removal in riparian ecosystems, including those that control the transformation and fluxes of nitrogen within ecosystems (*e.g.*, mineralisation, nitrification, nitrogen fixation, and dissimilative reduction of nitrate to ammonium). Table 1 shows that denitrification varies greatly (<1–1600 kg ha^{-1} yr^{-1}) but vegetation uptake, especially in riparian meadows, also display considerable variation (<10–350 kg ha^{-1} yr^{-1}). Thus, different processes can play a leading role in nitrogen removal.

2.3.2. Phosphorus removal

Main processes that remove phosphorus in buffer zones are (Kadlec and Knight, 1996): (1) sedimentation of particulate phosphorus and chemical precipitation; (2) soil sorption; (3) removal of dissolved inorganic phosphorus by plant uptake; (4) microbial immobilization and storage in the soil as organic phosphorus. Additionally phosphorus can be released by phosphine volatilisation (Devai *et al.*, 1988; Kadlec and Knight, 1996), however this process is not sufficiently measured and described.

Table 1. Intensity of processes relevant to nitrogen removal in riparian buffer ecosystems

Process	Intensity rate (kg N ha^{-1} yr^{-1})	Ecosystem	Source
N mineralisation	24–400	Temperate hardwoods in riparian zone	Melillo, 1981 (cit. after Bowden, 1986a)
	[1]3.7–456	Pine-oak forest, swamp forest	Zak and Grigal, 1991
	[2]46–113	Riparian fen and bog	Verhoeven *et al.*, 1990
	[2]1.1–219	Old-field site (potential buffer zone)	Robertson *et al.*, 1988
Nitrification	38–80	Oak-beech forest	Tietema *et al.*, 1987
	[3]<1–100	Freshwater wetlands	Bowden, 1987
	[1]13–29	Pine-oak forest, swamp forest	Zak and Grigal, 1991
	[2]0.73–183	Old-field site (potential buffer zone)	Robertson *et al.*, 1988
	[3]1580–1930	Tidal freshwater marsh	Bowden, 1986b
Denitrification	0.16–475	Riparian deciduous forests Riparian meadows	Weller *et al.*, 1994; Fustec *et al.*, 1991
	20–1600	Grassland buffer strip	Brüsch and Nilsson, 1993
	[4]114–2880	Riparian forests	Groffmann *et al.*, 1991
	[1]<0.04–2895		Ambus and Christiansen, 1991
		Riparian meadow	Cooper, 1990
	[5]2.6–2960	Riparian successional forest	Klingensmith and Van Cleve, 1993
	[3]1400–23650		
Vegetation uptake	30–220	Riparian flooded meadows	Leonardson *et al.*, 1994
	34–90	Experimental grass and forest buffer strips	Uusi-Kämppä and Yläranta, 1996
	[3]<10–350	Freshwater wetlands	Bowden, 1987
	[3]15–30	Riparian meadow	Prach and Rauch, 1992
N fixation	5–10	Temperate hardwoods	Melillo, 1981 (cit. after Bowden, 1986a)
	30–164	Riparian alder stands	Klingensmith and Van Cleve, 1993
Dissimilative reduction of nitrate to ammonium	[6]0.02–1.2	Riparian fen	Ambus *et al.*, 1992
	[3]5	Tidal freshwater marsh	Bowden, 1986b and 1987
Ammonia volatilisation	30–56	Riparian forests	Kim, 1973 (cit. after Bowden, 1986a)
	3.5–16.5	Riparian meadow	Woodmansee, 1978 (cit. after Bowden, 1986a)
	95–114	Riparian grazed sheep pasture	Denmead *et al.*, 1976 (cit. after Bowden, 1986a)

[1]in mg N m^{-2} d^{-1} [4]in g N ha^{-1} d^{-1}
[2]in µg N cm^{-2} d^{-1} [5]in mg N m^{-2} h^{-1}
[3]in g N m^{-2} yr^{-1} [6]µg N m^{-2} h^{-1}

The microbial pool is usually small and quickly becomes saturated. Therefore, the microbial uptake is often considered a part of the soil adsorption (Richardson, 1985). In absolute terms, both the soil adsorption and the vegetation uptake are on a comparable level, varying from 0.1 to 236 and from 0.2 to 50 kg P ha^{-1} yr^{-1} (Table 2). An interesting mechanism to control phosphorus storage in the soil and sediments is the inactivation by nitrates. Andersen (1982) demonstrated that oxidised nitrogen can buffer the redox potential of the lake sediment up to 16 kg P ha^{-1} yr^{-1}. The same mechanism can possibly play an important role in buffer zones.

There are also studies that show that phosphorus, in contrast, can be released from the wetland soils of riparian zones (Richardson and Marshall, 1986; Vanek, 1991). Table 2, presents the most important processes controlling phosphorus retention in riparian buffers.

Table 2. Intensity of processes relevant to phosphorus removal in riparian buffer ecosystems

Process	Intensity rate (kg P ha^{-1} yr^{-1})	Ecosystem	Source
Adsorption in soil and sediments	1.72–7.3	Floodplain wetland	Yarbro, 1979 (cit. after Mitsch and Gosselink, 1993)
	15–38	Riparian fen	Richardson and Marshall, 1986
	36	Alluvial cypress swamp	Mitsch *et al.*, 1979
	[1]236	Sawgrass tidal marsh	Hsieh, 1988
	[1,4]4.1–28.6	Constructed riparian wetlands	Mitsch *et al.*, 1995
Sedimentation	[1]5.9–130	Constructed riparian wetlands	Fennessy *et al.*, 1994
Incorporation of organic P into peat	0.05–2.4	Temperate wetlands	Richardson, 1985
Vegetation uptake	16.6–50.1	Riparian fen	Richardson and Marshall, 1986
	5–8	Experimental grass and forest buffer strips	Uusi-Kämppä and Yläranta, 1996
	[1]2–4,5	Riparian meadow	Prach and Rauch, 1992
	[1]185	Sawgrass tidal marsh	Hsieh, 1988
	8.7	Alluvial cypress swamp	Mitsch *et al.*, 1979
	[2]2.6–16.6	Constructed riparian wetlands	Mitsch, 1992
Microbial uptake	2.3–14.4	Riparian fen	Richardson and Marshall, 1986
P inactivation by NO_3^-	[5]0.14–16	Eutrophic lake sediments	Andersen, 1982
Release	0.13–0.30	Aspen-birch forest	Timmons *et al.*, 1977
	0.46	Riparian forested bog	Verry and Timmons, 1982
	26–42	Riparian fen	Richardson and Marshall, 1986
	[3]7.3–1044	Riparian forested wetland	Clausen and Johnson, 1990

[1]in g P m^{-2} yr^{-1}
[2]in mg P m^{-2} $week^{-1}$
[3]in kg P ha^{-1} d^{-1}
[4]retention mostly based on adsorption in sediments
[5]calculated after the depth and bulk density of sediments

2.4. Role of plants in buffer zones

The role of vegetation in buffer zones is still not clear (Correll, 1997). There are two main vegetation types of buffer zones studied in different works: grass zones and forested zones. Haycock and Pinay (1993) found that forested buffer zones are more effective than grass buffer zones, while other works show good purification in grass buffer zones (Peterjohn and Correll, 1984; Groffman *et al.*, 1991; Correll, 1997). Grass strips are considered sediment traps by which a large portion of nutrients, especially P is deposited from surface flow (Mitsch *et al.*, 1979; Peterjohn and Correll, 1984; Yoyng *et al.*, 1980; Cooper *et al.*, 1987; Magette *et al.*, 1989). At the same time grass buffer strips can also remove dissolved nutrients (Peterjohn and Correll, 1984; Uusi-Kämppä and Yläranta, 1992; Haycock and Burt, 1993; Vought *et al.*, 1994) which shows that grass strips are an important part of buffer zones. Although the assimilation of nutrients by herbs in buffer zones has been described in only a few works, results indicate a high nutrient assimilation ability. Van Oorschot (1994) measured above-ground N and P uptake in riparian communities up to 7.1 and 1.07 g m^{-2}, respectively. Prach and Rauch (1992) estimate the N and P removal hay from a floodplain to be 15–30 and 2–4.5 g m^{-2}, respectively.

2.5. Purification of polluted overland and subsurface flow

Riparian buffers can effectively retain sediments from overland flow (Mitsch *et al.*, 1979; Cooper *et al.*, 1987; Magette *et al.*, 1989). The filtering process in riparian buffer biotopes has an nonlinear (often exponential) character: on the upper part of buffer strips (*i.e.* on the border between arable land and buffer community) the amounts of adsorbed and transformed substancies are essentially larger than those in the middle and lower parts of the buffer (Doyle *et al.*, 1977; Mander, 1985; Knauer and Mander, 1989; Vought, *et al.*, 1994).

The role of buffers in the removal of soluble nutrients is more complicated to assess. However, several case studies worldwide suggest that different riparian ecosystems can significantly decrease the nitrogen and phosphorus concentration in both overland flow and groundwater (Peterjohn and Correll, 1984; Pinay and Décamps, 1988; Knauer and Mander, 1988; Jordan *et al.*, 1992; Osborne *et al.*, 1993; Vought *et al.*, 1994).

The load-retention relationship in various ecosystems has been discussed in several studies. Fleischer *et al.* (1991) stimulated an intensive discussion on the role of wetlands controlling nutrient fluxes in landscapes. They proposed that the water pollution problems caused by nitrogen loading from non-point pollution sources can be solved by wetlands, because no limits had been found on the retention capacity of nitrogen in wetland ecosystems. Mander *et al.* (1997b) had similar efficiency of buffer ecosystems where buffer strips studies and phosphorus retention was included. However, presentation in the same plot (load vs. retention) of various ecosystems with very different nutrient concentration dynamics only shows the main trends and does not give information about retention dynamics in individual wetlands. Arheimer and Wittgren (1994) showed a significant difference between two individual wetlands with different seasonal hydrological and hydrochemical dynamics. Therefore, they suggested that comparison of nutrient retention from different studies and extrapolation of results from one region to another is consequently only possible if detailed background data is available. A similar scheme of load-retention relationship should work for buffer strips as well. Nevertheless, studies on individual riparian buffers show a very strong positive correlation between nitrogen load and removal (Haycock and Pinay, 1993).

3. STUDY AREA AND METHODS

3.1. Description of study area

3.1.1. Porijõgi watershed

The study of nutrient cycling and its relation to the agriculture started in 1985 at Porijõgi watershed area, South Estonia.

The central and northern parts of the catchment lie within the South-East Estonian moraine plain 5–10 km southern from. The elevation of the moraine plateau is from 30 to 60 m above m.s.l., relief is undulated (slopes achieve normally 5–6%), and the landscape is dissected by primeval valleys (Varep, 1964). The southern part of the drainage basin lies on the northern slope of Otepää Heights that is formed from moraine hills and kames with extreme variety of glacial deposits. The elevation of this region is up to 120 m, the relative heights reach 30–35 meters. About 50% of the moraine plain (mostly with podzoluvisols and planosols on loamy sand and fine sandy loam) was used as arable land. In valleys and other low areas gleysols and peatlands, partly used as perennial grasslands, dominate. In forested areas (nowadays, about 45% of the territory), coniferous and mixed forests are most common. In riparian zones alder (*Alnus ssp.*) forests and willow (*Salix ssp.*) bushes occur.

3.1.2. Landscape transects

For studying nutrient cycling in agricultural landscapes and influence of buffer zones to the water quality 6 transects in 1991 and one in 1994 were established.

The transects are situated on slopes adjacent to streams, following of surface water flow and crossing agricultural fields and different riparian plant communities. Six transects (Porijõgi, Sipeoja I, Sipeoja II, Tatra, Vända I, Vända II) were studied in 1992–1993. The Porijõgi and Viiratsi (established in 1994) transects were main study areas in 1994 and 1995.

The Porijõgi transect is located on the slope of a primeval valley where agricultural activities stopped in 1992. The transect crosses following plant communities: abandoned field (cultivated last time in 1992) on planosols and podzoluvisols, abandoned cultivated grassland (mowed last time in 1993) on colluvial podzoluvisol (dominated by *Dactylis glomerata* and *Alopecurus pratensis*), an 11 m wide wet grassland on gleysol (two parallel communities, one dominated by *Filipendula ulmaria,* another by *Aegopodium podagraria*), a 20 m wide grey alder stand (*Alnus incana*) on gleysol (Fig. 1A). The Viiratsi transect is situated in the Sakala heights (Varep, 1964) consisting of moraine hills and undulated plains with a variety of glacial deposits. The transect is located on the moraine plain in the vicinity of a pig farm (with about 30 000 pigs during the study). Almost all the slurry from the pig farm is spread on the neighbouring fields and whole area is heavily impacted by pig slurry. The transect crosses following plant communities: field on planosols and podzoluvisols (slurry was spread in autumn 1994), an 11 m wide grassland (*Elytrigia repens-Urtica dioica*) and young grey alder (*Alnus incana*) trees strip on colluvial podzoluviol, 12 m wide wet grassland (*Filipendula ulmaria*) on gleysol, 28 m wide grey alder (*Alnus incana*) forest on podzoluvic gleysol (Fig. 1B).

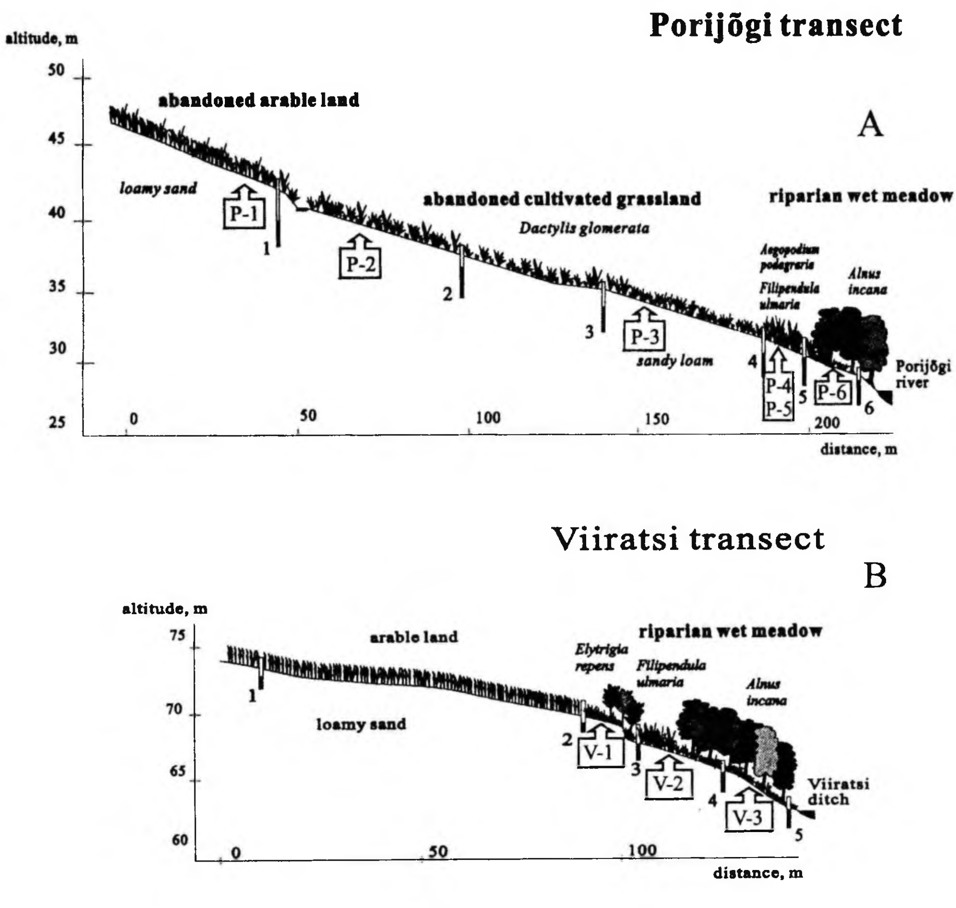

Figure 1. Study transects in complex riparian buffer zones in South Estonia. A — Porijõgi; B — Viiratsi. 3 — shallow groundwater sampling point; P-1 — phytomass sampling plots.

All other transects are situating on the moraine plain on the slope of valleys. The Tatra crosses following plant communities: arable land, grassland (slope of valley), cultivated grassland, sedge fen, spring fen, alder strip. The Sipeoja I and Sipeoja II transects cross arable land and alder forest. The Vända I transect crosses only arable land and Vända II arable land and grassland.

3.2. Materials and methods

3.2.1. Field experiments and laboratory analysis

Stream discharge was measured and water samples were taken for analysing in laboratory once per month from 16 sampling points from Porijõgi River catchment including 5 closing sampling points of subwatersheds since 1987. On 6 transects (Porijõgi, Sipeoja I, Sipeoja II, Tatra, Vända I, Vända II) shallow groundwater samples were collected once a month from piezometers installed on the borders of plant communities during January 1992 to August 1993. On Porijõgi and Viiratsi transects shallow groundwater samples were collected in every two weeks during June 1994 to December 1995. Shallow groundwater discharge was estimated on the basis of both Darcy's law and by gauging with weirs installed in soil water seeping sites. The surface water from Porijõgi River was analysed at the South-Estonian Laboratory of Environment Protection. In 1992–1993 the shallow groundwater was analysed in the laboratory of the Institute of Environment Protection, 1992–1993 in laboratory of Estonian Agricultural University. Filtered water samples were analysed for NH_4-N, NO_2-N, NO_3-N, total-N (in 1994–1995), PO_4-P, total-P and organic matter concentrations (on the base of BOD_5, only for surface water) following standard methods for examination of water and wastewater quality (APHA, 1989).

3.2.2. Phytomass study

The phytomass (*i.e.*, standing crop) samples were collected from all riparian plant communities during the maximum flowering time of the dominant plant species (2^{nd} and 3^{rd} week in July; see Milner and Hughes, 1968). Sampling plots (six in Porijõgi and three in Viiratsi) were installed in typical areas of the community. The above ground biomass was collected from three replicate quadrates (1×1 m) in each community. Below ground root biomass was collected from soil cores taken by auger (diameter 158 mm) from a depth of up to 40–50 cm in three replicates from each location. Roots were washed of soil and from dried roots and aboveground biomass dry weight was measured and N and P content was analysed in the laboratory of Estonian Agricultural University. To estimate the aboveground biomass and productivity of grey alder forests, dimension-analysis techniques (Bormann and Gordon, 1984; Rytter, 1989; Huss-Danell and Ohlsson, 1992) were used. At both test sites (age 14 in Porijõgi transect and 40 years in Viiratsi transect) 17 and 5 model trees per plot, respectively, were felled to collect data on the following tree components: stem (wood and bark), secondary branch growth (wood and bark), primary branch growth, leaves, generative organs (Lõhmus *et al.*, 1996). The relative increments of the wood and barks of an overbark fraction were assumed to be equal. Root systems for 6 and 3 out of the sampled 17 and 5 trees respectively were excavated. The dried weight of all tree components was measured and N and P content in dried biomass were analysed in laboratory of Estonian Agricultural University.

3.2.3. Statistical analysis and calculations

Kruskal-Wallis test was perfomed due to inhomogeneity of variances to analyse N and P concentration changes between measurement points and the test for Binary Sequences for load versus purification efficiency analyses using *Statgraphics Plus 7.1*. The regression analyse of relation between input load and appearance of negative removal of nutrients was performed by *Microsoft Excel 97.*

Removal efficiency E (%) of N and P in buffer communities was estimated as:

$$E = 100\ \% * (Q_{in}C_{in} - Q_{out}C_{out})/\ (Q_{in}C_{in}) \qquad (1)$$

where Q_{in} and Q_{out} = inflow and outflow values ($m^3\ d^{-1}$), respectively; C_{in} and C_{out} = concentration values ($mg\ l^{-1}$), respectively.

The percentage of buffered stream banks was calculated as follows:

$$B = l_b/\Sigma l_t * 100\% \qquad (2)$$

l_b = length of all stream banks in the catchment (m), having buffer zone or strip between field and stream.
Σl_t = total length of stream banks in the catchment (m).

The percentage of unbuffered stream banks was calculated as follows:

$$U = l_u/\Sigma l_t * 100\% \qquad (3)$$

l_u = length of all stream banks in the catchment (m), having agricultural fields up to the bank of stream.

The percentage of stream banks with natural communities was calculated as follows:

$$N = l_n / \Sigma l_t * 100\% \qquad (4)$$

l_n = length of all stream banks in the catchment (m), bordering with natural plant communities.

4. RESULTS

4.1. Shallow groundwater

The first study to determine the efficiency of buffer zones was carried out on 6 landscape transects in 1992–1993 (Mander *et al.*, 1995). The study showed that buffer zones have good purification efficiency of nitrogen and phosphorus in comparison with non-buffered locations. For instance, the Vända transect was crossing fields on the both slopes of the ditch and had no buffering plant communities. There were no remarkable changes in nutrient content between transect sampling points. The average nutrient content on the boundary of ditch canopy in both sides was 9.9 and 16.2 mgN l^{-1} of total inorganic nitrogen (TIN) and 0.11 and 0.09 mgP l^{-1} of total-P. At the same time in locations, where was well developed buffer zone, the nutrient content decreased despite of high input value. In Sipeoja transect the average input of TIN was 25.5 mgN l^{-1} and total-P 0.17 mgP l^{-1}, output values were 0.9 and 0.09 mg l^{-1}, respectively.

The detailed study of nutrient variation in buffer zones was done in Porijõgi and Viiratsi transects in 1994–1995 (Mander *et al.*, 1997a; Kuusemets *et al.*, 1998; Kuusemets *et al.*, 1999).

The shallow groundwater nitrogen load in two transects was different. The total-N content in Viiratsi transect reached up to 138 mg l^{-1} in the cultivated field after slurry application. In Porijõgi transect the highest total-N load was 19.5 mg l^{-1}. The results show that there was a considerable decrease in nitrogen content through the Viiratsi transect buffer zone (see Table 3). The average total-N decreased during the study period (1994 to 1995) from 23 mg l^{-1} in the field (transect point 1, Fig. 1B) to 3.1 mg l^{-1} at the end of the buffer zone (transect point 5). However, this decrease was not significant ($P>0.05$). There was considerable decrease in nitrogen through the first 2 m of the buffer zone where the average total-N decreased from 23 to 14.3 mg l^{-1} in the grass community (transect point 2) and in the following 11 m wide wet grass community to 5.6 mg l^{-1} (in transect point 3). In the alder forest zone the nitrogen content decreased to 3.1 mgN l^{-1}. This change from point 2 is highly significant ($P<0.01$) compared to the values at points 4 and 5. The change was highly significant also when comparing points 3 and 4, and 3 and 5.

The nitrogen load in the Porijõgi transect was lower than in the Viiratsi transect (Table 4). The average total-N content for study period (1994 to 1995) was 1.4 mg l^{-1} in the abandoned field (sampling point 1, Fig. 1A) which increased to 3.0 mg l^{-1}in the cultivated grassland (sampling point 2). Two meters from the edge of the abandoned cultivated grassland the total-N content decreased to 1.9 mg l^{-1}. By the end of the buffer zone (sampling point 6) the average total-N content was 1.5 mg l^{-1}. The only significant decrease (*P<0.05*) was between sampling points 3 to 6.

The phosphorus values were similar in both transects. The highest average total-P content in Viiratsi transect was 0.43 mg l^{-1} in the field which decreased (not significantly) to 0.17 mg l^{-1} in first buffer zone sampling point (2, Fig. 1B). In grassland-young alder forest strip the average phosphorus content decreased highly significantly (*P<0.001* in comparison to sampling points 1 and 2) down to 0.05 mg l^{-1}. There was a slight but statistically insignificant increase in the last part of alder forest from 0.05 to 0.07 mg l^{-1} (sampling point 5).

In the Porijõgi transect the average P content was highest in the abandoned arable land (0.49 mg l^{-1}, sampling point 1, Fig. 1A) which decreased to 0.27 mg l^{-1}in the abandoned cultivated grassland. The phosphorus content decreased significantly (*P<0.01* in comparison with point 3 and *P<0.05* in comparison with point 2) to 0.06 mg l^{-1}at the beginning of buffer zone (sampling point 4) increased insignificantly in the wet grassland to 0.09 mg l^{-1} and decreased to 0.06 mg l^{-1}in the alder forest.

Table 3. Nutrient variation in shallow groundwater (average ± standard error), soil and the plant biomass in the complex buffer zone in Viiratsi. Sampling points are given in brackets following Fig. 1B

	Field (1)	Grassland (2) (V-1)	Wet Meadow (3) (V-2)	Alder Stand I (4) (V-3)	Alder Stand II (5)
Total-N in shallow groundwater (mg l^{-1})	23 ±7.7	14.3 ±3.6	5.6 ±0.4	3.1 ±0.5	3.1 ±0.7
N assimilation by plants (g N m^{-2})		17.5	21.1	14.0	
Topsoil (0–10 cm) N content (mg g^{-1})		2.16	6.76	9.87	
Soil (10–20 cm) N content (mg g^{-1})		1.66	5.37	5.61	
Total-P in shallow ground-water (mg l^{-1})	0.43 ±0.28	0.17 ±0.04	0.05 ±0.005	0.05 ±0.004	0.07 ±0.01
P assimilation by plants (g P m^{-2})		3.7	4.8	1.1	
Topsoil (0–10 cm) P content (mg g^{-1})		0.49	0.50	0.94	
Soil (10–20 cm) P content (mg g^{-1})		0.37	0.40	0.53	
Plant biomass production (g m^{-2})		1320	1015	1060	

4.2. Nutrient uptake by plants and storage in the soil

The plant biomass production and nutrient uptake was studied in Porijõgi transect (1992–1995), in Tatra transect (1992–1993) and in Viiratsi transect (1994–1995). The highest biomass production and nutrient uptake values were measured in 1992, during last intensive year of agricultural activities in Porijõgi and Tatra transects. In both transects the biomass production was highest in buffer wet meadow: 3601 g m^{-2} in Porijõgi and 4524 g m^{-2} in Tatra. The nitrogen assimilation was 85.8 and 54.3 g m^{-2}, respectively and phosphorus assimilation 6.36 and 5.3 g m^{-2}, respectively.

Table 4. Nutrient variation in shallow groundwater (average ± standard error, soil and the plant biomass in the complex buffer zone in Porijõgi. Sampling points are given in rackets following Fig. 1A

	Field (1) (P-1)	Cultivated grassland I (2) (P-2)	Cultivated grassland II (3) (P-3)	Wet meadow I (4) (P-4)	Wet meadow II (5) (P-5)	Alder stand (6) (P-6)
Total-N in shallow groundwater (mg l^{-1})	1.4 ±0.2	3.0 ±0.8	3.0 ±1.4	1.9 ±0.3	2.3 ±0.4	1.5 ±0.2
N assimilation by plants (g N m^{-2})	11.4	13.2	13.6	21.3†	18.8††	20.5
Topsoil (0–10 cm) N content (mg g^{-1})			2.02		2.82	10.74
Soil (10–20 cm) N content (mg g^{-1})			1.56		2.36	9.54
Total-P in shallow groundwater (mg l^{-1})	0.49 ±0.3	0.29 ±0.01	0.27 ±0.15	0.06 ±0.02	0.09 ±0.03	0.06 ±0.005
P assimilation by plants (g P m^{-2})	2.7	2.1	2.6	3.0†	3.3††	1.5
Topsoil (0–10 cm) P content (mg g^{-1})			0.45		0.63	1.02
Soil (10–20 cm) P content (mg g^{-1})			0.35		0.53	0.9
Plant biomass production (g m^{-2})	1113	1152	1493	1748	1977	1730

† *Aegopodium podagraria,* †† *Filipendula ulmaria.*

In 1994 and 1995 in Viiratsi and Porijõgi transects, the biomass production and nutrient uptake was smaller. Plant nutrient uptake in the Viiratsi transect was higher in the wet meadow *Filipendula ulmaria* association (sampling plot V-2, Fig. 1B), where an average of 21.1 gN m^{-2} yr^{-1} and 4.8 gP m^{-2} yr^{-1} was assimilated in grass (Table 3). This was higher than the nutrient uptake in the alder stand (14.0 and 1.1 g m^{-2} yr^{-1}, respectively). In the Porijõgi transect the uptake of nutrients was also highest in the wet meadow where in the *Filipendula ulmaria* association (sampling plot P-5, see Fig. 1A) assimilated 21.3 and 3.0 g m^{-2} yr^{-1} of N and P, respectively (Table 4). This was higher than annual N and P uptake by the alder forest: 20.5 and 1.5 g m^{-2}, respectively. Nutrient uptake in 1994 reached 33.9 gN m^{-2} in the Porijõgi transect (sampling plot P-5) and 5.5 gP m^{-2} yr^{-1} in the Viiratsi transect (sampling plot V-2).

The nutrient content in soil was highest in the grey alder (*Alnus incana*) community, where the nitrogen content in the Porijõgi topsoil layer (10.7 mg g^{-1}, sampling point 6, see Fig. 1) was comparable with that in Viiratsi (9.9 mg g^{-1}). The phosphorus contents were 1.0 and 5.6 mg g^{-1}, respectively.

4.3. Stream water

The impact of buffer zones to water quality was analysed in Vända, Sipe and Porijõgi upper subcatchments of Porijõgi River catchment (Kuusemets and Mander, 1999). The subcatchments of Vända and Sipe have similar physico-geographical conditions, situating in central part of Porijõgi River catchment. Currently, 68% of Vända and 58% of Sipe subcatchments are used as agricultural land (Table 5). The Porijõgi upper subcatchment has only 6.4% agricultural land, 79% of the catchment is covered by forest. This subcatchment can be used as control area to describe natural outflow rate in the region. In Estonia, the most intensive use of agricultural lands took place at the end of Soviet period, from 1985 to 1990 (Mander and Palang, 1994). For instance, in 1987 the average fertilisation intensity was 150 kg N ha^{-1} and 60 kg P ha^{-1}. Since 1990 the use of fertilisers dropped and constituted in 1994 only 2.3% N and 0.8% P of the level in 1987.

Table 5. Riparian buffer zones, natural communities adjacent to streams, and outflow of N and P (kg ha^{-1} yr^{-1}) from three subcatchments of the Porijõgi River

Subcatchment (stream)	Total area (ha)	Area of agricultural land (ha)	Percentage of buffered stream banks (B)	Percentage of unbuffered stream banks (U)	Percentage of stream banks with natural communities (N)	Outflow of total-N (kg ha^{-1} yr^{-1})	Outflow of total-P (kg ha^{-1} yr^{-1})
Vända	220	150	6	63	31	24.4	0.67
Sipe	900	518	32	51	17	5.0	0.14
Porijõgi upper	1230	79	6	1	93	2.7	0.09

In the Porijõgi catchment, the fertilisation intensity followed the tendency for the whole Estonia. In our study, outflow values for N and P are calculated as average for 1987–1990 (Table 5). The results show very high outflow of nitrogen (24.4 kg ha^{-1} yr^{-1}) and phosphorus (0.67 kg ha^{-1} yr^{-1}) from the Vända subcatchment. It is 9.0 and 7.4 times higher than the outflow from the natural Porijõgi upper subcatchment, respectively. At the same time, from Sipe subcatchment with similar intensity of agricultural use, the outflow of nitrogen and phosphorus were only 1.9 and 1.6 times higher than in the natural subcatchment, correspondingly. Although Sipe stream has 51% percentage of unbuffered stream banks, the outflow of N and P is much lower than that in the Vända subcatchment with 63% unbuffered stream banks. One of the explanations is more complex landscape pattern in Sipe subcatchment that shows better buffering capacity. Here in addition to buffer zone complex structure of landscape plays important role (45% of the 6.6 km long main stream is buffered), the main stream has meandering valley with a well-developed hyporheic zone. The Vända stream is completely straightened, has no meandering parts and only very short fragment of it has buffer zone on both banks that constitutes 7% of the total length (4.8 km) of the stream.

5. DISCUSSION

5.1. Purification efficiency of buffer zones

The study of Porijõgi and Viiratsi transects showed that they have relatively high purification efficiency of nutrients (Mander *et al.*, 1997a; Kuusemets *et al.*, 1998; Kuusemets *et al.*, 1999).

The average removal of total-N and total-P in Viiratsi was 87% and 84%, in Porijõgi transect 50% and 78%, respectively. The water quality improved already within first meters of buffer zone. In Viiratsi transect the purification efficiency of total-N and total-P was 38% and 60% within first 2 m in the buffer grassland, respectively. In Porijõgi transect 2 m from the edge of the cultivated grassland the purification efficiency of total-N and total-P was 37% and 78%, respectively.

To analyse the relation between purification efficiency and input load we calculated the purification efficiency separately for each sampling day for every buffer strip between two sampling points (Kuusemets, *et al.*, 1999). Data were divided by input load concentration into 8 classes with 11 to 36 measurements in each class (Tables 6 and 7). The Tests of Binary Sequences were performed to calculate the level of significance between accurance of positive or negative removal efficiency. The probability of negative removal was calculated by dividing the number of negative removal cases with the total measurement number (%). Comparison of removal efficiency and input load shows that the removal of total-N was negative ($P<0.01$, Table 6) when the input was less than 1.0 mg l^{-1}. For loads between 1.0 to 5.0 mg l^{-1}, removal of total-N showed no significant positive or negative tendency, although positive removal was more common — the probability of negative removal is 30.8 to 40.0% ($P>0.05$). For input loads greater than 5.0 mg l^{-1} the purification efficiency is significantly positive ($P<0.01$), for input loads greater than 42 mg l^{-1} purification efficiency is always positive (Fig. 2A, $P<0.001$). The relation between input load and appearance of negative removal of nitrogen (N_{neg}) is described by logarithmic regression:

$$N_{neg} = 63.0 - 16.84 \mathrm{Ln}(I_{Nmax}) \qquad (5)$$
$$R^2 = 0.78, P<0.001$$

where I_{Nmax} = maximum value of N input class.

For total-P there is no interval for negative removal (Table 7). The input value 0.01 to 0.15 mg l^{-1} yields positive or negative removal. Positive removal is prevalent for input from 0.05 to 0.15 mg P l^{-1} — the probability of negative removal is 18.8 to 38.9% ($P>0.05$). For input greater than 0.15 mg l^{-1}, the purification efficiency of total-P was significantly positive ($P<0.01$), for input loads greater than 2.1 mg l^{-1} purification efficiency is always positive (Fig. 2B, $P<0.01$). The relation between input load and appearance of negative removal of phosphorus (P_{neg}) is described by logarithmic regression:

$$P_{neg} = 7.9-10.20\mathrm{Ln}(I_{Pmax}) \tag{6}$$
$$R^2 = 0.72,\ P<0.01$$

where I_{Pmax} = maximum value of N input class

Table 6. The probability (Tests Binary Sequences) of total-N positive or negative removal versus input load in buffer strips

Input range of total-N in mg l^{-1}	Probability of negative removal (%)	Level of significance	Number of samples (n)
0.02–0.3	100	**	11
0.3–1.0	82.4	*	17
1.0–1.5	30.8	n.s.	13
1.5–2	40.0	n.s.	15
2.0–3.0	38.9	n.s.	18
3.0–5.0	35.3	n.s.	17
5.0–8.0	10.0	**	20
8.0–70	13.3	**	15

* $P< 0.05$, ** $P< 0.01$, n.s. — not significant.

Table 7. The probability (Tests Binary Sequences) of total-P positive or negative removal versus input load in buffer strips

Input range of total-P in mg l^{-1}	Probability of negative removal (%)	Level of significance	Number of samples (n)
0.01–0.04	51.4	n.s.	37
0.04–0.05	52.8	n.s.	36
0.05–0.06	23.8	n.s.	13
0.06–0.08	38.9	n.s.	18
0.08–0.10	18.8	*	16
0.10–0.15	26.7	n.s.	15
0.15–0.3	15	**	20
0.3–6	0	**	15

* $P< 0.05$, ** $P< 0.01$, n.s. — not significant.

The results show strong positive correlation between nutrient load and removal. However, buffer zones have upper limits of purification and these regression formulas can not been used in the planning in the case of high input values. This analysis provides limits for buffer strips as water quality purification systems. The input load range (1.0–5.0 and 0.01–0.15 g l^{-1} of N and P, respectively) can be considered to represent natural conditions of buffer strips where water output quality also depends on natural processes taking place in the buffer. This can explain certain increase of total-N and total-P inside both studied buffers. However, the average outflow values from both buffer zones were lower: 3.1 and 0.07 mg l^{-1} of total-N and total-P in Viiratsi, respectively and 1.5 and 0.06 mg l^{-1} in Porijõgi, respectively. Our estimation on denitrification intensity showed that this process does not play a substantial role in nitrogen removal (7.9–20.1 kg N ha^{-1} yr^{-1} in Porijõgi and 8.5–19.3 kg N ha^{-1} yr^{-1} in Viiratsi, Mander *et al.*, 1997a).

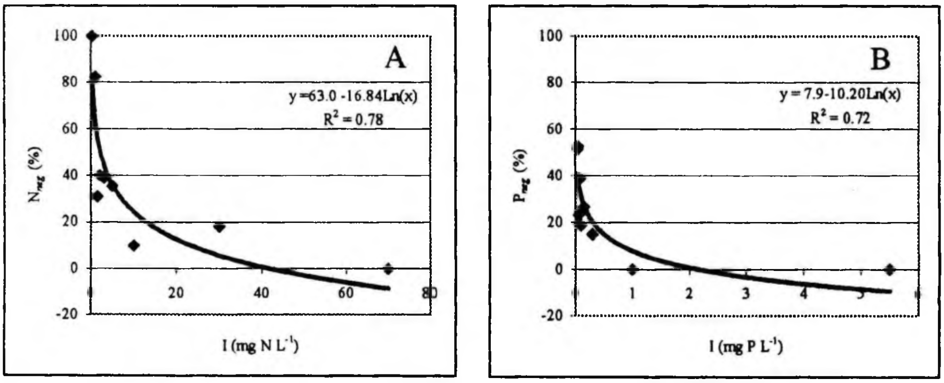

Figure 2. The relation between input concentration (I; mg l^{-1}) and appearance of negative removal of nitrogen (A) and phosphorus (B), (N_{neg} and P_{neg}; %).

5.2. The designing and management of buffer zones

Although many authors for several ecosystems have reported the buffering efficiency of vegetated riparian buffers, there are few examples in the literature about the dimensioning of buffer zones and strips. The first are from the USA, where methods for the determination the parameters of buffer zones adjacent to feedlots and manure land treatment sites (Doyle *et al.*, 1977; Overcash *et al.*, 1981). Phillips (1989) presented calculations to evaluate the buffering efficiency of riparian forests along a coastal plain river. However, calculation methods usable for landscape planning and stream restoration purposes are not much developed. It has been found that the purification of the overland flow within the buffer strips is non-linear; in the upper part of the strips much more organic matter, nitrogen and phosphorus was removed than in the downhill part (Doyle *et al.*, 1977; Mander, 1985; Vought *et al.*, 1994).

Mander (1995) presented dimensioning formulas and nomograph (Fig. 3) for determining width of buffer zones. The formulas are based on hydrological models which appeal on capacity of buffer strips to infiltrate overland flow, and are supplemented with parameters significant for absorption and cation exchange capacity of soils. Basically, this method is comparable to the widely used universal soil loss equation (USLE; see Meyer and Wishmeier, 1969). However, the length-slope factor in this equation, which is a purely empirical relationship, does not account for changes in either surface flow or erosion processes (Moore and Burch, 1986). Therefore, Mander (1995) proposed to use a specific slope length factor. It appeals on overland flow concentration in lower parts of the relief, *e.g.*, in thalwegs.

In addition to the determination of width another open question is vegetation type of buffer zone.

The nutrient uptake in plants indicates, that grass communities play as important role in nutrient retention as grey alder forest. The nitrogen and phosphorus uptake in above-ground biomass was highest in 1992 in wet meadow of Porijõgi transect (84.9 and 5.48 g m^{-2}, respectively). In 1994–1995 the average shoot nitrogen and phosphorus content of herbs in wet meadow of Porijõgi transect was 11.6 and 1.6 g m^{-2}, respectively and in Viiratsi transect 10.6 and 2.3 g m^{-2}, respectively. This gives good opportunity to remove a portion of nutrients by grass mowing and hay cutting while felling of trees can be done with intervals of decades. Cutting should be done during the maximum flowering period of the dominant species when the nutrient content in the shoot biomass is highest (Deinum, 1966). The mowed herbs should be removed after mowing to avoid rapid nutrient loss from hay (see Schaffers, *et al.*, 1998).

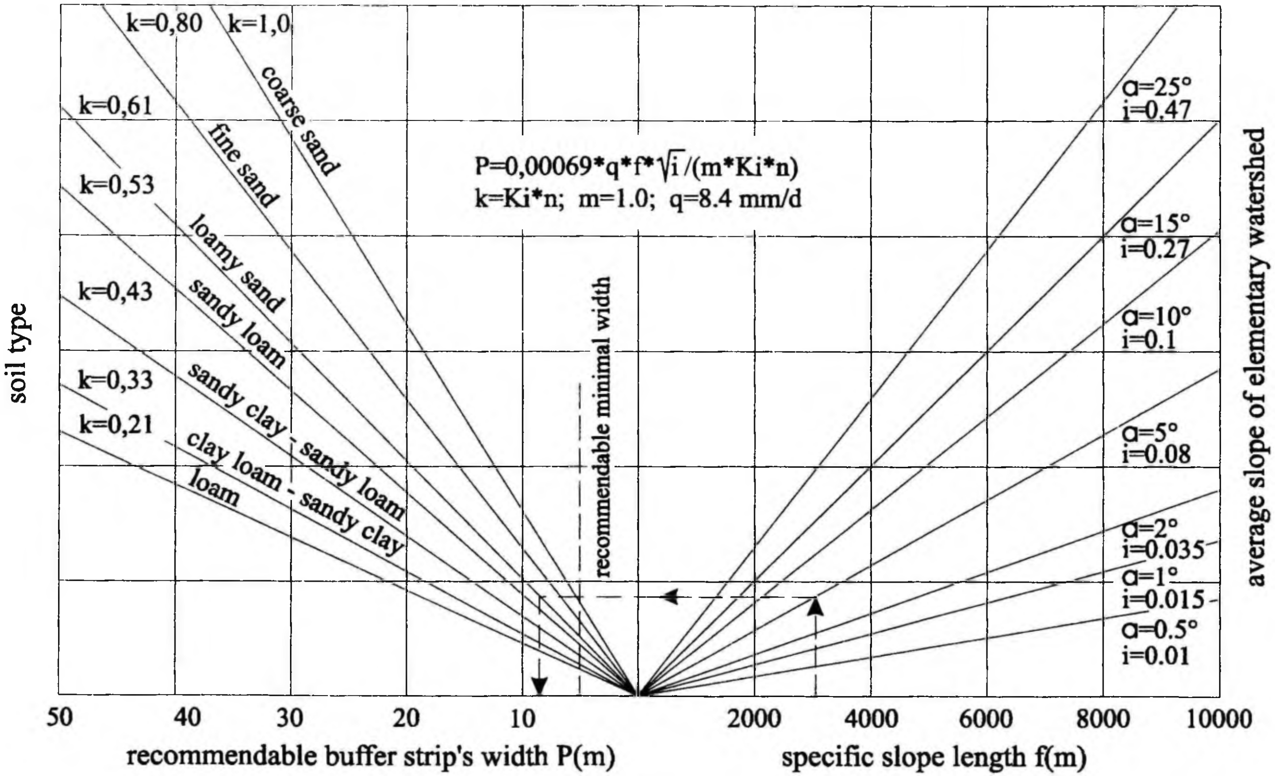

Figure 3. The nomograph for determination of the recommended buffer strtip's width (adopted from Mander and Kuusemets, 1998).

CONCLUSIONS

1. Buffer zones and buffer strips along ditches, channels and riverbanks are most attractive multi-functional mitigation elements for watershed management. Buffer zones and buffer strips have number of positive functions. Some are important to reduce the negative environmental impacts to the water others are beneficial to the landscape and biodiversity. Buffer strips and zones allow stopping pollution at the source where it arises and reducing expenses of measures for improvement water quality in the stream. Designing of buffer zones and strips should be important part of landscape planning.

2. The studied transects show high purification efficiency. The average removal of total-N and total-P in Viiratsi was 87% and 84% and in Porijõgi 50% and 78%, respectively. The water quality had already improved within the first meters of the buffer zones. In Viiratsi transect the purification efficiency of total-N and total-P within the first 2 m of the buffer grassland was 38% and 60%, respectively, whereas the purification efficiency of total-N and total-P of the first 2 m from the edge of the cultivated grassland in the Porijõgi transect was 37% and 78%, respectively. The heavily loaded complex buffer zone consisting of grass and forest strips showed relatively low output concentrations for total-N and total-P that are comparable with the output values from the unloaded transect.

3. Comparison of removal efficiency and input load shows that the removal of nutrients in buffer strips depends on input load level. The removal of total-N was negative (*$P<0.01$*) when the input was less than 1.0 mg l^{-1}. For loads between 1.0 to 5.0 mg l^{-1}, removal of total-N showed no significant positive or negative tendency, although positive removal was more common — the probability of negative removal was 30.8 to 40.0% (*$P>0.05$*). For input loads greater than 5.0 mg l^{-1} the purification efficiency is positive (*$P<0.01$*). For total-P there was no interval for negative removal. The input value 0.01 to 0.15 mg l^{-1} yields positive or negative removal. Positive removal is prevalent for input from 0.05 to 0.15 mg P l^{-1} — the probability of negative removal is 18.8 to 38.9% (*$P>0.05$*). For input greater than 0.15 mg l^{-1}, the purification efficiency of total-P was positive (*$P<0.01$*). This indicates that buffers have certain area of unpredictability for input loads where removal of nutrients is not sufficiently predictable.

4. The results show that complex buffer zones of grass and forest strips are very effective in N and P retention. This kind of complex can be recommended for buffer strip design where grass strips considered as sediment traps but also as important mechanism for dissolved N and P removal. Both

features provide the opportunity to remove part of the nutrients from the system. In addition to efficient nutrient purification potential, forest buffer strips have many other environmentally important functions such as protection against soil erosion, filtering polluted air, canopy shading, and increasing biological and landscape diversity.

5. An original method for dimensioning buffer zones and buffer strips for agricultural catchments has been worked out. It takes into to consideration topography (*i.e.*, specific slope length and slope), soil factors, hydrological parameters and land cover type. This method has shown satisfactory results in planning the riparian buffer zones and buffer strips in Estonia.

6. The nutrient uptake in plants indicates, that plant removal can be important opportunity to remove a portion of nutrients from the buffer zones. The easiest possibility is grass mowing and hay cutting while felling of trees can be done with intervals of decades. The average N and P content in herbal shoots in the wet meadows was 11.6 and 1.6 g m^{-2}, respectively in the Porijõgi transect and 10.6 and 2.3 g m^{-2}, respectively in the Viiratsi transect. Cutting should be done during the maximum flowering period of the dominant species when the nutrient content in the shoot biomass is highest. The mowed herbs should be removed after mowing to avoid rapid nutrient loss from hay.

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REFERENCES

Ambus, P. and Christiansen, S. 1991. Denitrification and other nitrogen cycling processes in a riparian meadow. Project abstracts of the Danish NPo Research Programme. C-Abstracts. Number C13a. Ministry of the Environment, National Agency of Environmental Protection, Denmark, pp. 187–202.

Ambus, P., Mosier, A., and Christensen, S. 1992. Nitrogen turnover rates in a riparian fen determined by ^{15}N dilution. *Biol. Fertil. Soils*, 14, 230–236.

Andersen, J. M. 1982. Effect of nitrate concentration in lake water on phosphate release from the sediment. *Water Res.*, 16, 1119–1126.

APHA, 1981. Standard Methods for the Examination of Water and Waste Water, 15th edition, American Public Health Organization, Washington.

Arheimer, B. and Wittgren, H.-B. 1994. Modeling the effects of wetlands on regional nitrogen transport. *Ambio*, 23, 378–386.

Baldock, D., Beaufoy, G., Bennett, G. and Clark, J. 1993. Nature Conservation and New Directions in the Common Agricultural Policy. Report for the Ministry of Agriculture, Nature Management and Fisheries, the Netherlands. Institute for European Environmental Policy, HPC, Arnhem, pp. 224.

Bishoff, N. T. and Jongman, R. H. G. 1993. Development of Rural Areas in Europe: The Claim for Nature. Netherlands Scientific Council for Government Policy. Sdu uitgeverij, Plantijnstraat, The Hague, pp. 206.

Bormann, B. T. and Gordon, J. C. 1984. Stand density effects in young red alder plantations: Productivity, photosynthate partioning and nitrogen fixation. *Ecology*, 65, 394–402.

Bowden, W. B. 1986a. Gaseous nitrogen emmissions from undisturbed terrestrial ecosystems: An assessment of their impacts on local and global nitrogen budgets. *Biogeochemistry*, 2, 249–279.

Bowden, W. B. 1986b. Nitrification, nitrate reduction, and nitrogen immobilization in a tidal freshwater marsh sediment. *Ecology*, 67, 88–99.

Bowden, W. B. 1987. The biogeochemistry of nitrogen in freshwater wetlands. *Biogeochemistry*, 4, 313–348.

Burt, T. P. 1997. The hydrological role of floodplain withion the drainage basin system. In: N. E. Haycock *et al.* (ed.), *Buffer Zones: Their Processes and Potential in Water Protection.* Proceedings of the International Conference on Buffer Zones, September 1996. Quest Environmental, Foundation for Water Research, Oxford, pp. 21–32.

Brüsch, W. and Nilsson, B. 1993. Nitrate transformation and water movement in a wetland area. *Hydrobiologia*, 251, 103–111.

Clausen, J. C. and Johnson, G. D. 1990. Lake level influences on sediment and nutrient retention in a lakeside wetland. *J. Environ. Qual.*, 19, 83–88.

Cooper, B. 1990. Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. *Hydrobiologia*, 202, 13–26.

Cooper, J. R., Gilliam, J. W., Daniels, R. B., and Robarge, W. P. 1987. Riparian areas as filters for agricultural sediment. *Soil Sci. Soc. Am. J.*, 51, 416–420.

Correll, D. L. 1997. Buffer zones and water quality protection: general principles. In: N. E. Haycock *et al.* (ed.) *Buffer Zones: Their Processes and Potential in Water Protection.* Proceedings of the International Conference on Buffer Zones, September 1996. Quest Environmental, Foundation for Water Research, Oxford, pp. 7–20.

Deinum, B. 1966. Climate, nitrogen and grass. Mededelingen Landbouwegeschool, Wageningen, 66–11, pp. 1–91.

Devai, I., Felföldy, L., Wittner, I. and Plosz, S. 1988. Detection of phosphine: New aspects of the phosphorus cycle in the hydrosphere. *Nature*, 333, 343–345.

Dillaha, T. A. III and Inamdar, S. P. 1997. Buffer Zones as Sediment Traps or Sources. In: N. E. Haycock *et. al.* (ed.), *Buffer Zones: Their Processes and Potential in Water Protection.* Proceedings of the International Conference on Buffer Zones, September 1996. Quest Environmental, Foundation for Water Research, Oxford, pp. 33–44.

Dillaha, T. A., Reneau, R. B., Mostaghimi, S. and Lee, D. 1989. Vegetative filter strips for agricultural nonpoint source pollution control. *Trans. ASAE*, St. Joseph, Mich., 32, 513–519.

Doyle, R. C., Stanton, G. C. and Wolf, D. C. 1977. Effectiveness of forest and grass buffer strip in improving the water quality of manure polluted runoff. *ASAE Paper* No 77–2501, St. Joseph, MI, pp. 32.

The EECONET Declaration. 1993. Declaration of the Conference "Conserving Europe's Natural Heritage Towards a European Ecological Network", Maastricht, 9–12 November 1993. Ministry of Agriculture, Nature Management and Fisheries.

Fennessy, M. S., Brueske, C. C., Mitsch, W. J. 1994. Sediment deposition patterns in restored freshwater wetlands using sediment traps. *Ecol. Eng.*, 3, 409–428.

Fleisher, S., Stibe, L. and Leonardson, L. 1991. Restoration of wetlands as a means of reducing nitrogen transport to coastal waters. *Ambio*, 20, 271–272.

Fustec, E., Mariotti, A., Grillo, X. and Sajus, J. 1991. Nitrate removal by denitrification in alluvial ground water: Role of a former channel. *J. Hydrol.*, 123, 337–354.

Groffmann, P. M., Axelrod, E. A., Lemunyon, J. L. and Sullivan, W. M. 1991. Denitrification in grass and forest vegetated filter strips. *J. Environ. Qual.*, 20, 671–674.

Haycock, N. E. and Burt, T. P. 1993. The sensitivity of rivers to nitrate leaching. The effectiveness of near-stream land as a nutrient retention zone. In: Thomas, D. S. G. and Allison, R. J. (eds.). Landscape Sensitivity, Wiley, London, pp. 261–272.

Haycock, N. and Pinay, G. 1993. Nitrate retention in grass and poplar vegetated riparian buffer strips during the winter. *J. Environ. Qual.*, 22, 273–278.

Hendricks, S. P. and White, D. 1991. Physicochemical Patterns within a Hyporheic Cone of a Northern Michigan River, with Comments on Surface Water Patterns. *Can. J. Fish. Aquat. Sci.*, Vol. 48, 1645–1654.

Hill, A. R. 1990. Ground water flow paths in relation to nitrogen chemistry in the near-stream zone. *Hydrobiologia*, 206, 39–52.

Hsieh, Y. P. 1988. Dynamics of carbon, nitrogen, and phosphorus cycling in a sawgrass tidal marsh with special reference to the aboveground primary production. *J. Environ. Qual.*, 17, 676–681.

Huss-Danell, K. and Ohlsson, H. 1992. Distribution of biomass and nitrogen among plant parts and soil nitrogen in a young *Alnus incana* stand. *Can. J. Bot.*, 70, 1545–1549.

Jordan, T. E., Correll, D. L. and Weller, D. E. 1992. Nutrient interception by riparian forest receiving inputs from adjacent cropland. *J. Environ. Qual.*, 22, 467–473.
Kadlec, R. H. and Knight, R. L. 1996. Treatment Wetlands. Lewis Publishers, New York, pp. 893.
Klingensmith, K. M. and Van Cleve, K. 1993. Denitrification and nitrogen fixation in floodplain successional soils along the Tanana River interior, Alaska. *Can. J. For. Res.*, 23, 956–963.
Knauer, N. and Mander, Ü. 1989. Untersuchungen über die Filterwirkung verschiedener Saumbiotope an Gewässern in Schleswig-Holstein. 1. Mitteilung: Filterung von Stickstoff und Phosphor. Z. f. Kulturtechnik und Landentwicklung, 30, 365–376.
Kuusemets, V. and Mander, Ü. 1999. Ecotechnological measures to control nutrient losses from catchments. *Water Science and Technology*. Accepted.
Kuusemets, V., Mander, Ü., Ivask, M. and Lõhmus, K. 1998. Phosphorus retention in riparian buffer zones in agricultural landscapes in Estonia. In: R. H. Foy and R. Dils (eds.) *Practical and Innovative Measures for the Control of Agricultural Phosphorus Losses to Water*. OECD Sponsored Workshop at Greenmount College of Agriculture and Horticulture, Northern Ireland, 16–19 June 1998,. Workshop paper abstracts and posters, pp. 114–115.
Kuusemets, V., Mander, Ü., Lõhmus, K. and Ivask. M. 1999. Shallow Groundwater Quality and Plant Uptake of Nitrogen and Phosphorus in Complex Riparian Buffer Zones. *Journal of Environmental Quality*. Submitted.
Leonardson, L., Bengtsson, L., Davidsson, T., Persson, T. and Emanuelsson, U. 1994. Nitrogen retention in artificially flooded meadows. *Ambio*, 23, 332–341.
Lowrance, R. 1991. Effects of buffer systems on the movement of to N and P from agriculture to streams. In: *NPo-forskning fra Miljøstyrelsen*. Conference Contributions. International Conference on N, P and Organic matter. Helsingør, Denmark.
Lowrance, R. 1992. Groundwater nitrate and denitrification in a coastal plain riparian forest. *J. Environ. Qual.*, 21, 401–405.
Lowrance, R. R., Todd, R., Fail, J., Hendrickson, O., Leonard, R. and Asmussen, L. 1984. Riparian forest as nutrient filters in agricultural watersheds. *BioScience*, 34, 374–377.
Lõhmus, K., Mander, Ü., Tullus, H. and Keedus, K. 1996. Productivity, buffering capacity and recourced of grey alder forests in Estonia. In: Perttu, K. and Koppel, A. (eds). *Short Rotation Willow Coppice for Renewable and Improved Environment*. Swedish University of Agricultural Sciences, Uppsala, pp 95–105.
Magette, W. L., Brinsfield, R. B., Palmer, R. E. and Wood, J. D. 1989. Nutrient and sediment removal by vegetated filter strips. *Trans. ASAE*, St. Joseph, Mich., 32, 663–627.
Mander, Ü. 1985. The renovation effect of polluted surface flow in vegetated buffer strips. *Acta et commentationes Universitatis Tartuensis* 701, 77–89. (In Russian, with summary in English).
Mander, Ü. 1989. Kompensationsstreifen entlang der Ufer und Gewässerschutz. Landesamt für Wasserhaushalt und Küsten Schleswig-Holstein. Bericht D10, pp. 61.
Mander, Ü. 1995. Riparian buffer zones and buffer strips on stream banks: Dimensioning and efficiency assessment from catchments in Estonia. In: M. Eiseltová and J. Biggs (eds.) *Restoration of Stream Ecosystems*. IWRB Publication No 37, Slimbridge, Gloucester, UK, pp. 45–64.

Mander, Ü. and Kuusemets, V. 1998. Non-point pollution control by riparian buffer strips: Dimensioning and efficiency assessment. In: *Proceedings of the 3rd International Conference on Diffuse Pollution, Edinburgh, Scotland, 31 August – 4 September 1998*, pp. 136–143.

Mander, Ü., Kuusemets, V. and Ivask, M. 1995. Nutrient dynamics of riparian ecotones: A case study from the Porijõgi River catchment, Estonia. *Landscape and Urban Planning*, 31, 333–348.

Mander, Ü., Kuusemets, V., Lõhmus, K. and Mauring, T. 1997b. Efficiency and dimensioning of riparian buffer zones in agricultural catchments. *Ecological Engineering*, 8, 299–324.

Mander, Ü., Lõhmus, K., Kuusemets, V. and Ivask, M. 1997a. The potential role of wet meadows and grey alder forests as buffer zones. In: N. E. Haycock *et al.* (ed.), *Buffer Zones: Their Processes and Potential in Water Protection*. Proceedings of the International Conference on Buffer Zones, September 1996. Quest Environmental, Foundation for Water Research, Oxford, pp. 35–46.

Mander, Ü. and Palang, H. 1994. Changes of landscape structure in Estonia during the Soviet period. *GeoJournal*, 33, 45–54.

Meyer, L. D. and Wishmeier, W. H. 1969. Mathematical simulation of the process of soil erosion by water. *Trans. ASAE*, 12, 754–758.

Milner, C. and R. E. Hughes. 1968. Methods for the Measurement of the Primary Production of Grassland. IBP Handbook No. 6. Blackwell Scientific Publications, Oxford, pp. 70.

Mitsch, W. J. 1992. Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution. *Ecol. Eng.*, 27–47.

Mitsch, W. J., Cronk, J. K., Wu, X., Nairn, R. W. and Hey, D. L. 1995. Phosphorus retention in constructed freshwater riparian marshes. *Ecol. Applic.*, 5, 1294–1298.

Mitsch, J. W., Dorge, C. L. and Wiemhoff, J. R. 1979. Ecosystem Dynamics and a Phosphorus Budget of an Alluvial Cypress Swamp in Southern Illinois. *Ecology*, 60 (6), 1116–1124.

Mitsch, W. J. and Gosselink, J. G. 1993. Wetlands. 2nd Edition. Van Nostrand Reinhold, New York, pp. 722.

Moore, I. D. and Burch, G. J. 1986. Physical basis of the length-slope factor in the universal soil loss equation. *Soil Sci. Soc. Am. J.*, 50, 1294–1298.

Osborne, L. L. and Kovacic, D. A. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology*, 29, 243–258.

Overcash, M. R., Bingham, S. C., and Westerman, P. W. 1981. Predicting runoff pollutant reduction in buffer zones adjacent to land treatment sites. *Trans. ASAE*, St. Joseph, Mich., 24, 430–435.

Peterjohn, W. T. and Correll, D. L. 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology*, 65, 1466–1475.

Phillips, J. D. 1989. Nonpoint source pollution control effectiveness of riparian forests along a coastal plain river. *J. Hydrol.*, 110, 221–237

Pinay, G. and Decamps, H. 1988. The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: a conceptual model. *Regulated Rivers: Research and Management* 2, 507–516.

Pinay, G., Fabre, A., Vervier, P. and Gazelle, F. 1992. Control of C, N, P distribution in soils of riparian forests. *Landscape Ecology*, 6, 121–132.

Pinay, G., Roques, L. and Fabre, A. 1993. Spatial and temporal patterns of denitrification in a riparian forest. *Journal of Applied Ecology*, 30, 581–591.
Prach, K. and Rauch, O. 1992. On filter effects of ecotones. *Ekológia* (CSFR), 11, 293–298.
Richardson, C. J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. *Science*, 228, 1424–1427.
Richardson, C. J. and Marshall, P. E. 1986. Processes controlling movement, storage, and export of phosphorus in a fen peatland. *Ecological Monographs*, 56, 279–302.
Risser, P. G. 1990. The ecological importance of land-water ecotones. In: R. J. Naiman and H. Décamps (eds.). *The Ecology and Management of Aquatic-Terrestrial Ecotones*. Man and the Biosphere Series, UNESCO, The Parthenon Publ. Group, Paris, pp. 7–21.
Robertson, G. P., Huston, M. A., Evans, F. C. and Tiedje, J. M. 1988. Spatial variability in a successional plant community: Patterns of nitrogen availability. *Ecology*, 69, 1517–1524.
Rytter, L. 1989. Distribution of roots and root nodules and biomass allocation in young intensively managed grey alder stands on a peat bog. *Plant and Soil*, 128, 363–365.
Schaffers, A. P., Vesseur, M. C. and Sykora, K. V. 1998. Effects of delayed hay removal on the nutrient balance of roadside plant communities. *J. of Appl. Ecol.*, 349–364.
Stanford, J. A. and Ward, J. V. 1988. The hyporheic habitat of river ecosystems. *Nature*, 335, 64–66.
Tiedje, J. M., Simkins, S. and Groffmann, P. M. 1989. Perspectives on measurement of denitrifiction in the field included recommended protocolsfor acetylene methods. *Plant and Soil*, 115, 261–284.
Tietema, A., Duysings, J. J. H. M., Verstraten, J. M. and Westerveld, J. W. 1987. Estimation of actual nitrification rates in an acid forest soil. In: Hutchinson, T. C. and Meena, K. M. (eds.). *Effects of Atmospheric Pollutants on Forests, Wetlands and Agricultural Ecosystems*. NATO ASI Series, Proc. of ARW, Toronto (Canada), 1985. Springer-Verlag, pp. 190–197.
Timmons, D. R., Verry, E. S., Burwell, R. E and Holt, R. F. 1977. Nutrient transport in surface runoff and interflow from an aspen-birch forest. *J. Environ. Qual.*, 6, 188–192.
Triska, F. J, Kennedy, V. C., Avanzino, R. J., Zellweger, G. W. and Bencala, K. E. 1989. Retention and transport of nutrients in a third-order stream in Northwestern California: Hyporheic processes. *Ecology*, 10 (6), 1893–1905.
Uusi-Kämppä, J. and Yläranta, T. 1992. Reduction of sediment, phosphorus and nitrogen transport on vegetated buffer strips. *Agric. Sci. Finl.* 1, 569–574.
Uusi-Kämppä, J. and Yläranta, T. 1996. Effect of buffer strips on controlling soil erosion and nutrient losses in southern Finland. In: Mulamoottil, G. *et al.* (ed.) *Wetlands: Environmental Gradients, Boundaries, and Buffers*. CRC Press Inc., Lewis Publishers, Boca Raton, pp. 221–235.
Vanek, V. 1991. Riparian zone as source of phosphorus for a groundwater-dominated lake. *Water Res.*, 25, 409–418.
Van Oorschot, M. M. P. 1994. Plant production, nutrient uptake and mineralization in river marginal wetlands: the impact of nutrient additions due to former land-use. p. 133–150. In: Mitsch, W. J. (ed.). *Global Wetlands: Old World and New*. Elsevier Science B.V., Amsterdam, pp. 133–150.

Varep, E. 1964. The landscape regions of Estonia. Publications on Geography IV. *Acta et Commerstationes Universitatis Tartuensis*, 156, 3–28.
Verhoeven, J. T. A., Maltby, E. and Schmitz, M. B. 1990. Nitrogen and phosphorus mineralization in fens and bogs. *Journal of Ecology*, 78, 713–726.
Verry, E. S. and Timmons, D. R. 1982. Waterborne nutrient flow through an upland-peatland watershed in Minnesota. *Ecology*, 63, 1456–1467.
Vought, L. B.-M., Dahl, J., Pedersen, C. L. and Lacoursière, J. O. 1994. Nutrient retention in riparian ecotones. *Ambio*, 23, 342–348.
Weller, D. E., Correll, D. L. and Jordan, T. E. 1994. Denitrification in riparian forests receiving agricultural discharges. In: Mitsch, J. W. (ed.). *Global Wetlands: Old World and New*, Elsevier Science B. V., pp. 117–131.
Young, R. A., Huntrods, T. and Anderson, W. 1980. Effectiveness of vegetated buffer strips in controlling pollution from feedlot runoff. *J. Environ. Qual.*, 9, 483–487.
Zak, D. R. and Grigal, D. F. 1991. Nitrogen mineralization, nitrification and denitrification in upland and wetland ecosystems. *Oecologia*, 88, 189–196.

SUMMARY IN ESTONIAN

LÄMMASTIKU JA FOSFORI SIDUMINE PÕLLUMAJANDUSMAASTIKE KALDAÄÄRSETES PUHVERKOOSLUSTES

Veekoguäärsed kaldavööndid ja -ribad on olulised maastiku polüfunktsionaalsed komponendid, millel on olulisi keskkonnakaitselisi funktsioone, nagu vee kvaliteedi parandamine, mulla kaitsmine erosiooni eest, õhu puhastamine, vooluveekogu sängi varjutamine ja makrofüütide vohamise tõkestamine, soodsa mikrokliima loomine, elupaikade rajamine loomadele ja lindudele, ühenduskoridoride moodustamine looduslike tuumalade vahel, bioloogilise ja maastikulise mitmekesisuse suurendamine. Töös käsitletakse pinnasevee kvaliteedi muutust puhverkooslustes ja selgitatakse taimede osa toitainete eemaldamises.

Töö peamisteks eesmärkideks olid (1) lämmastiku ja fosfori sidumise selgitamine kompleksse struktuuriga (erinevatest kooslustest) erisuguse sisendkoormusega kaldaäärsetes puhverökosüsteemides, (2) kaldaäärsete puhvrite laiuse määramise aluste ning veekaitse seisukohalt oluliste majandamismeetmete väljatöötamine.

Peamiselt uuriti kahte maastikuprofiili, mis läbisid põllumaa, kultuurrohumaa, soostunud niidu ja hall-lepiku kooslusi. Porijõe transekt asub Kagu-Eesti lavamaal moreenkünka nõlval — ürgoru veerul, kus põllumajandustegevus lõppes 1992. a. Viiratsi transekt asub Sakala kõrgustikul lainjasse moreentasandikku lõikunud sälkoru veerul seakombinaadi läheduses, kus pideva sealäga laotamise tõttu on väga kõrged toitainete koormused pinnaseveele. Uurimise tulemused näitasid, et 31 m laiune rohumaa-lepiku puhverriba Porijõel eemaldas 50% sisenevast üldlämmastikust ja 78% üldfosforist. Viiratsi 51 m laiune puhver eemaldas 87% sisenevast lämmastikust ja 84% fosforist. Kuigi Viiratsi transektil ulatus lämmastiku sisend 138 mg N l^{-1} ja oli keskmiselt 23 mg l^{-1}, oli puhvrist väljavoolava pinnasevee keskmine lämmastiku sisaldus 3,1 mg N l^{-1}. Porijõel olid vastavad näitajad 3,0 ja 1,5 mg N l^{-1}. Fosfori sisaldused olid mõlemal transektil sarnased. Viiratsis oli fosfori keskmiseks sisendiks 0,43 mg l^{-1}, väljundiks 0,07 mgl^{-1}. Porijõe transektil oli keskmine fosfori sisaldus puhvrisse sisenevas pinnasevees 0,49 ja puhvrist väljuvas vees 0,06 mg l^{-1}. Rohttaimede biomass oli kuni 3601 g m^{-2} hariliku angervaksa (*Filipendula ulmaria*) dominantkoosluses, lämmastiku sidumine taimede poolt samas koosluses oli kuni 85,8 g m^{-2} ja fosfori sidumine kuni 6,36 g m^{-2}.

Hästi väljakujunenud puhvritega Sipe oja valgala väljakannete võrdlus puhverdamata Vända kraavi valgalaga (mõlemad Porijõe alavalgalad) näitas, et kõige intensiivsema põllumajanduse perioodil 1980. aastate lõpus oli Vända valgalalt lämmastiku väljakanne 4,9 korda ja fosfori väljakanne 4,8 korda suurem, võrreldes Sipe oja valgalaga.

Samas võib madalate sisendväärtuste puhul täheldada puhverkooslustesse lämmastiku ja fosfori väljakandumist puhvrist. Lämmastiku puhul osutusid alla 1,0 mg N l^{-1} sisendi korral puhastusefektiivsused statistiliselt usaldusväärselt negatiivseks, üle 5,0 mg N l^{-1} korral aga positiivseks. Fosfori puhul ainult negatiivsete puhastusefektiivsustega piirkonda ei esinenud, üle 0,15 mg P l^{-1} korral olid puhastusefektiivsused positiivsed. Nende äärmuste vahele jäävat vahemikku saab käsitleda määramatu piirkonnana, kus võib esineda nii negatiivseid kui ka positiivseid puhastusefektiivsusi.

Koostati originaalne metoodika veekoguäärsete puhvrite laiuse määramiseks sõltuvalt reljeefist, mullaparameetritest, veerežiimist ja taimkattest. Mitmes valgalas tehtud analüüs kinnitab metoodika kasutatavust maastikuplaneerimisel.

Parima puhastustulemuse saavutamiseks tuleks kasutada rohumaa ja puistu kombineeritud puhveralasid. Toitainete eemaldamiseks puhvrist on otstarbekas rohttaimi dominantliigi maksimaalse õitsemise ajal niita, kui toitainete sisaldus taimede maapealses osas on kõige suurem. Niidetud rohi tuleks puhvrist eemaldada, sest toitained hakkavad pärast niitmist heinast välja leostuma. Puhvertsoonide ja ribade kavandamine tuleb ette näha kohalikes üldplaneeringutes, määrates nende laiused ja soovitatava taimekoosluste struktuuri.

PUBLICATIONS

I

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LANDSCAPE
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Nutrient dynamics of riparian ecotones: a case study from the Porijõgi River catchment, Estonia

Ülo Mander[a,*], Valdo Kuusemets[b,1], Mari Ivask[b,1]

[a]Institute of Geography, University of Tartu, 46 Vanemuise Str., EE-2400 Tartu, Estonia
[b]Department of Environment Protection, Estonian Agriculture University, 5 Kreutzwaldi Str., EE-2400 Tartu, Estonia

Abstract

The transformation of nitrogen and phosphorus was studied in various riparian ecotones of the moraine plain and moraine–hilly landscapes in the Porijõgi River catchment area, southern Estonia. Soil water and ground water samples were collected once per month between 1992 and 1993 from piezometers installed between plant communities located along topo-edaphic gradients from moraine uplands to stream valleys at six sites. The production rate, as well as the N and P content in plant biomass from the 1 m^2 plots located near the piezometers was analyzed.

Results show that various riparian ecotones had a significant influence on soil water quality. In the arable land, the mean annual concentration of total inorganic nitrogen ($NH_4+NO_2+NO_3$) in piezometer water was 3–40 mg N l^{-1}, but in the grey alder forest (*Alnus incana*) total inorganic nitrogen never exceeded 1 mg N l^{-1}. The most significant reduction was in the average concentration of NO_3-N content which was reduced from 25 mg l^{-1} under the arable land to 0.5 mg l^{-1} within the alder forest. The average total phosphorus (orthophosphate phosphorus+organic phosphorus) concentration also decreased under the alder stands, being 0.2–1.5 mg P l^{-1} in the arable land and less than 0.2 mg P l^{-1} in the alder forest (less than 0.1 mg P l^{-1} in loamy soils and 0.2 mg P l^{-1} in sandy soils). Wetland herb communities (*Carex elata* association and *Filipendula ulmaria–Cirsium oleraceum–Aegopodium podagraria* community) also had a significant influence on soil water quality. Plant biomass (sum of above- and below-ground biomass) of riparian and wetland communities accumulates up to 70 g N m^{-2} and up to 6 g P m^{-2} during the growth season. Harvesting of riparian herbaceous communities may remove 20–30% of nutrient input. In the cultivated grasslands on sandy colluvial soils with a deep humus layer and sedge fens the content of NH_4-N and total-P in groundwater toplayer significantly increased, rising to 3 mg and 0.5 mg l^{-1}, respectively. Also, it has been assumed that vertical penetration of deep groundwater (contaminated with nitrate and phosphate), may increase the nutrient load to surface water bodies, despite the physical presence of a wide buffer zone (e.g. sedge fen) along the river corridor. From the observations of this work, alder forests and/or willow bushes as buffer strips on and adjacent to the stream banks are recommended to control diffuse water quality.

Keywords: Buffer strips/zones; Grey alder forest; Landscape profiles; Nitrogen and phosphorus removal; Plant uptake; Riparian ecotones

* Corresponding author: Tel. 372-7-430679; Fax. 372-7-435440; e-mail: ylo@geogr.ut.ee.
[1] Tel. 372-7-421597.

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1. Introduction

The flow of material and energy within landscapes can be represented in the form of gradient fields where the transition from peaks and hollows coincide with ecotones (Jagomägi et al., 1988; Correll, 1991). The significance of the ecotone is that all material transfers between ecosystems are regulated at the boundaries. The higher the rate of human management of landscapes, the steeper the gradients and the greater importance of ecotones. The managed systems such as arable lands usually export significant quantities of soil, nutrients, pesticides, heavy metals, and other inorganic ions. Normally, various wetlands and riparian communities regulate the volume and quality of these fluxes through a combination of biological and physical retention and transform other functions (Jordan et al., 1986; Risser, 1990). In recent years, numerous researchers have investigated and quantified the buffering capacity of riparian wetlands, forests and meadows (Nichols, 1983; Yates and Sheridan, 1983; Howard-Williams, 1985; Pinay and Décamps, 1988; Hoffmann, 1991; Ambus and Christiansen, 1991; Correll and Weller, 1992; Brüsch and Nilsson, 1993; Mitsch and Gosselink, 1993). In 1990 an international UNESCO/MAB project 'The Role of Land/Inland Water Ecotones in Landscape Management and Restoration', dealing essentially with the buffering effect of riparian ecosystems, was initiated (Décamps et al., 1990).

Various vegetated buffer strips have been used to both decrease non-point pollution load of water bodies and to filter wastewater from different point pollution activities. The best known example is the overland flow wastewater treatment through grassland slopes (Smith and Schroeder, 1985; Payer and Weil, 1987; Magette et al., 1989; Schwer and Clausen, 1989; Zirschky et al., 1989). In the USA, forest or grassland buffer strips have been reported to control polluted overland flows from feedlots and wastewater land treatment sites (Doyle et al., 1975; Vanderholm et al., 1979; Bingham et al., 1980; Young et al., 1980; Overcash et al., 1981; Pinkowski et al., 1985; Dillaha et al., 1988). The preventive role of buffer strips downslope from fields of intensive manure spreading, especially on frozen soil has also been established (Thompson et al., 1978; Mander, 1985).

The nutrient dynamics of riparian buffers under natural conditions is more complicated, because the variety of riparian biotopes and because long-term studies are needed to obtain a representative view of the nutrient accumulation dynamics. However, different studies demonstrate the importance of riparian buffer strips in nutrient removal (Reddy et al., 1982; Peterjohn and Correll, 1984; Lowrance et al., 1984; Jacobs and Gilliam, 1985; Knauer and Mander, 1989; Brüsch and Nilsson, 1991; Groffmann et al., 1991; Vought et al., 1991; Lowrance, 1992; Mitsch, 1992; Haycock and Pinay, 1993; Uusi-Kämppä and Yläranta, 1993).

In Estonia, investigations on nutrient transformations in riparian buffer strips have been carried out since the end of the 1970s (Mander, 1985). On this base a law protecting water zones and strips, as well as the principle of regulating land-use in riparian areas, were established in 1983. This law states that all water courses and water bodies with a catchment area exceeding 10 km^2 must have a buffer zone 50–1300 m wide. The width depends on the land-use intensity of adjacent territories (i.e. the potential pollution load), and on the use and importance of the water resource. For agricultural areas, the preferable land-use alternative is a perennial grassland with combination of a forest or bush buffer strip directly on riverbanks or lakeshores. The width of those riparian buffer strips is 5–50 m and depends on soil and relief conditions of the adjacent landscape (Mander, 1989). Currently, new water protection laws are being worked out that will include the principle of water-protection zones and strips.

Riparian ecotone investigations continue in the Porijõgi River drainage basin (276 km^2) in southern Estonia. This catchment was chosen for a long-term study of nutrient cycling in agricultural landscapes and especially buffer zones. Continuous monitoring of surface water and groundwater quality within the catchment started in 1987 (Mander et al., 1989). In autumn 1991, six transects were established along topo-edaphic gradient through various riparian ecotones.

2. Study area

The Porijõgi River drainage basin is representative of the whole of southern Estonia. It is located on the border of two landscape regions: the plain of southeast Estonia and the Otepää Heights (Varep, 1964). The central and northern parts of the Porijõgi River catchment area lie within the southeast Estonian moraine plain in the vicinity of Tartu (58°23'N; 26°44'E), the second largest town in Estonia. The elevation of the plateau is from 30 to 60 m above mean sea level. Relief is undulated (slopes are normally 5–6%), and the landscape is dissected by primeval valleys, 3–5 km wide and up to 40 m deep, formed by streams during the Pleistocene and remodeled by glaciers of the last, Weichselian glaciation (Varep, 1964). The southern part of the drainage basin lies on the northern slope of Otepää Heights that is formed from moraine hills and kames with extreme variety of glacial deposits. The elevation of this region is up to 120 m, the relative heights reach 30–35 m.

The bedrock of the whole catchment area is formed by red Devonian sandstone (compact sandstone with clay and aleurolite layers, which is covered by loamy sandy-till of the Weichselian glaciation or glaciofluvial and glaciolacustrine sands and gravels. The Devonian sandstone lies in the depth from 2 m (in lower course) to 60 m (on the hills of Otepää Heights). The depth to the water table varies depending on relief and geomorphologic conditions (0.5–20 m). The upland soils are mostly podzoluvisols, planosols, and podzols on loamy sands and fine sandy loams with the surface soil organic matter content and pH value in the cultivated fields being 1.6–1.9% and 5.6–6.5, respectively. On steeper slopes the soils have been eroded. About 50% of this area is used as arable land. In valleys and other low areas gleysols and peatlands, partly used as perennial grasslands, dominate. The fertilization rate of intensively managed fields was approximately 150 kg N, 70 kg P, and 100 kg K ha^{-1} $year^{-1}$. Application rates have decreased by as much as a factor of 3 over the last 2 years (Mander et al., 1994). In forested areas (about 25% of the territory), coniferous and mixed forests are most common. In riparian zones, alder forests and willow bushes occur. Bogs covered with pine or birch forests are the most common wetland communities on the catchment borders.

The average precipitation during the research period is 780 mm $year^{-1}$ (1992–1993). During the study period, winters have been relatively mild (mean air temperature for the coldest month was approximately −5°C (February 1993)). Snowcover lasted only a few days between November and April 1992/1993.

3. Materials and methods

3.1. Field studies

The Porijõgi River drainage basin was divided into eight sub-catchments, each with different land-use structure. Within this region, six transects were established in autumn 1991 along a topo-edaphic gradient to study the nutrient dynamics in groundwater flow from the moraine plateau to stream valleys (Fig. 1). The six transects were located in four sub-catchments and represent the most common ecotones present in the Porijõgi catchment (Figs. 2–5). Results of four of the six transects are presented in this paper.

In each landscape profile four to six piezometers (plastic pipes, diameter 100 mm, length 1.0–5.5 m, lower part perforated and covered with glass-fiber material) were established on the boundaries between the plant communities. Water samples were collected and the groundwater height was measured once a month from piezometers. The stagnant water was pumped from the piezometer before water samples were taken. Water samples were transported into the laboratory and analysed for NH_4-N, NO_2-N,

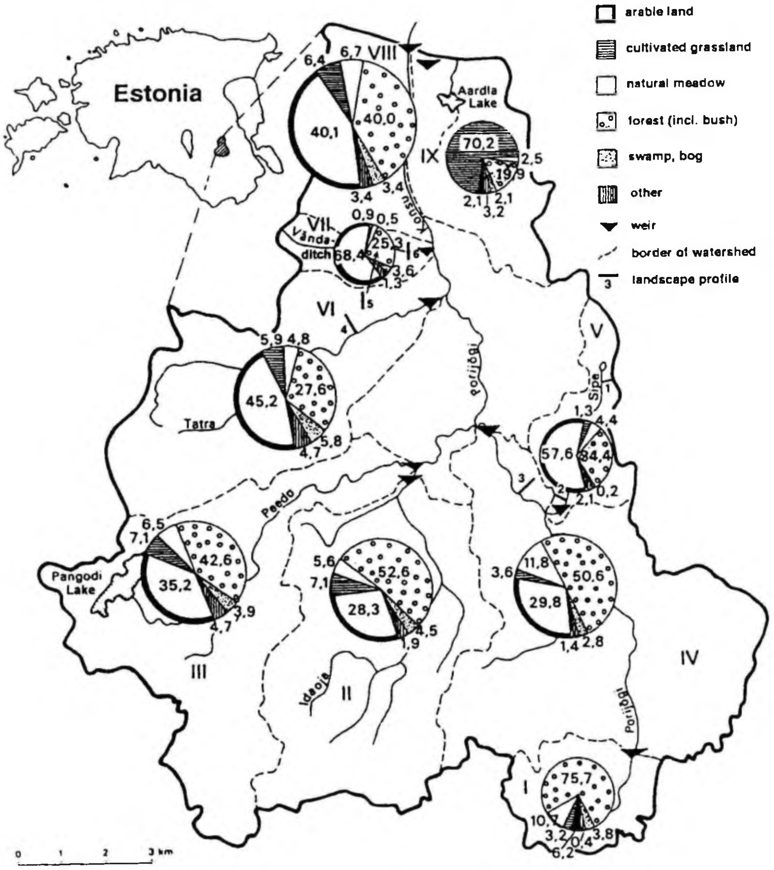

Fig. 1. Scheme of the Porijõgi River catchment area with the land-use structure in subcatchments, water sampling points (weirs), and locations of landscape profiles. Sub-catchments: I, Porijõgi upper course (Porijõgi I); II, Idaoja River; III, Peeda River; IV, Porijõgi River middle course (Porijõgi II); V, Sipeoja stream; VI, Tatra River; VII, Vända ditch; VIII, Porijõgi River lower course (Porijõgi III); IX, Aardla polder. Location of landscape profiles: 1, Sipeoja Upper; 2, Sipeoja Lower; 3, Porijõgi Middle; 4, Tatra Middle; 5, Vända Middle; 6, Vända Lower.

NO_3-N, PO_4-P, total-P and SO_4 contents.

Above-ground biomass of the herbaceous layer was determined only in two transects (Porijõgi Middle and Tatra Middle) using 1 m×1 m quadrats. Three to five quadrats were established at each location (Table 1). Below-ground root biomass was estimated from sequential coring and in-growth core samples to assess the herbaceous layer root production at the same sample locations. Three cores were taken from each harvested quadrates from April to September once every 2 months. Herb roots were separated from soil by washing. The flesh, oven-dry, and ash-free mass of both above-ground and below-

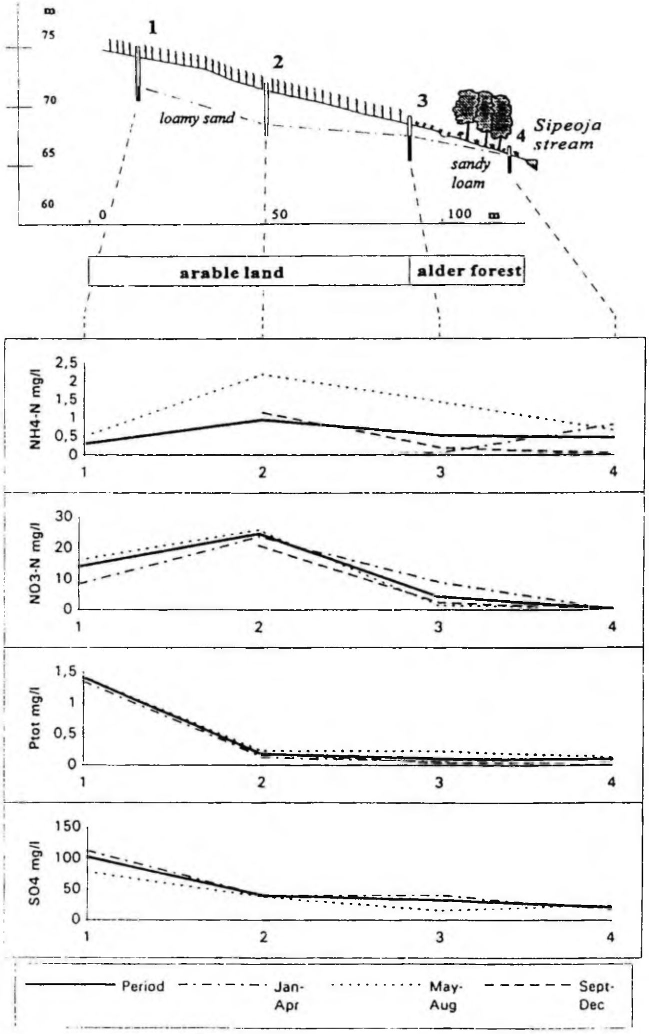

Fig. 2. Nutrient dynamics in the landscape profile of Sipeoja Lower. Crops in arable land: potato, vegetables, barley. Grey (white) alder forest (*Alnus incana*): 30–40 years old, partly with intensive herb layer growth in open sites dominated by *Cirsium oleraceum* and *Filipendula ulmaria*). 1–4, locations of piezometers and sampling sites. Average water table in piezometers is shown with black column. Lines indicate mean values of nutrient concentration in piezometers during the whole period (January 1992–August 1993) and three hydrologic periods.

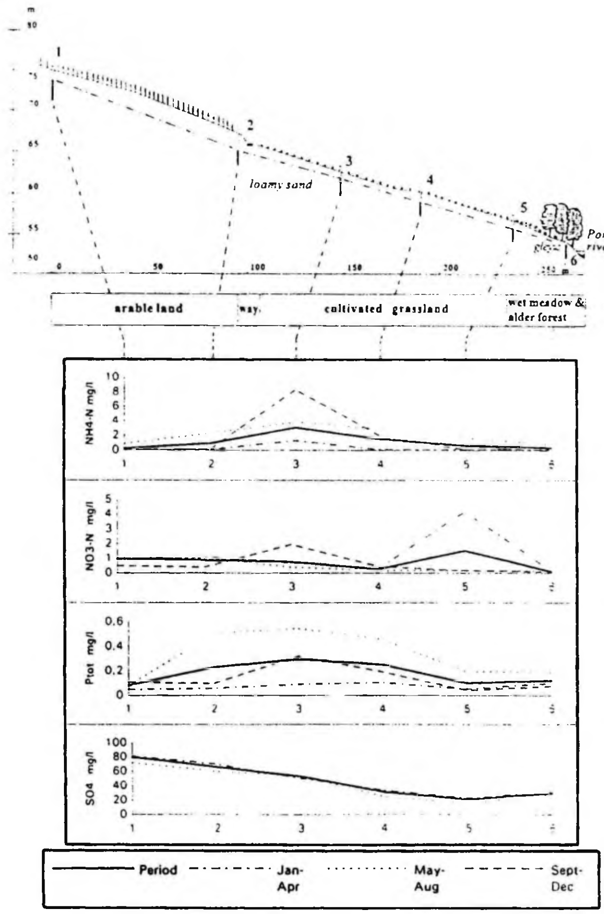

Fig. 3. Nutrient dynamics in the landscape profile of Porijõgi Middle during the period January 1992–August 1993. 1–6, locations of piezometers and sampling sites. Crops in arable land: winter rye, barley. Cultivated grassland: *Elytrigia repens* group. Wet meadow: *Filipendula ulmaria–Cirsium oleraceum–Aegopodium podagraria* association. In grey alder forest (20–30 years old), *Aegopodium podagraria* is dominating in the herb layer. For explanation of symbols see Fig. 2.

ground parts of herbs has been estimated (Persson, 1983). Soil samples were also collected from two depths (0–10 cm and 30–40 cm) at each piezometer to analyse for C, N and P contents.

Soil types in landscape profiles were recorded from soil maps (1:5000) made in the 1980s and which cover almost all of Estonia (Soil maps of Collective and State Farms of Estonia, 1987). Plant communities were described by the Braun–Blanquet method.

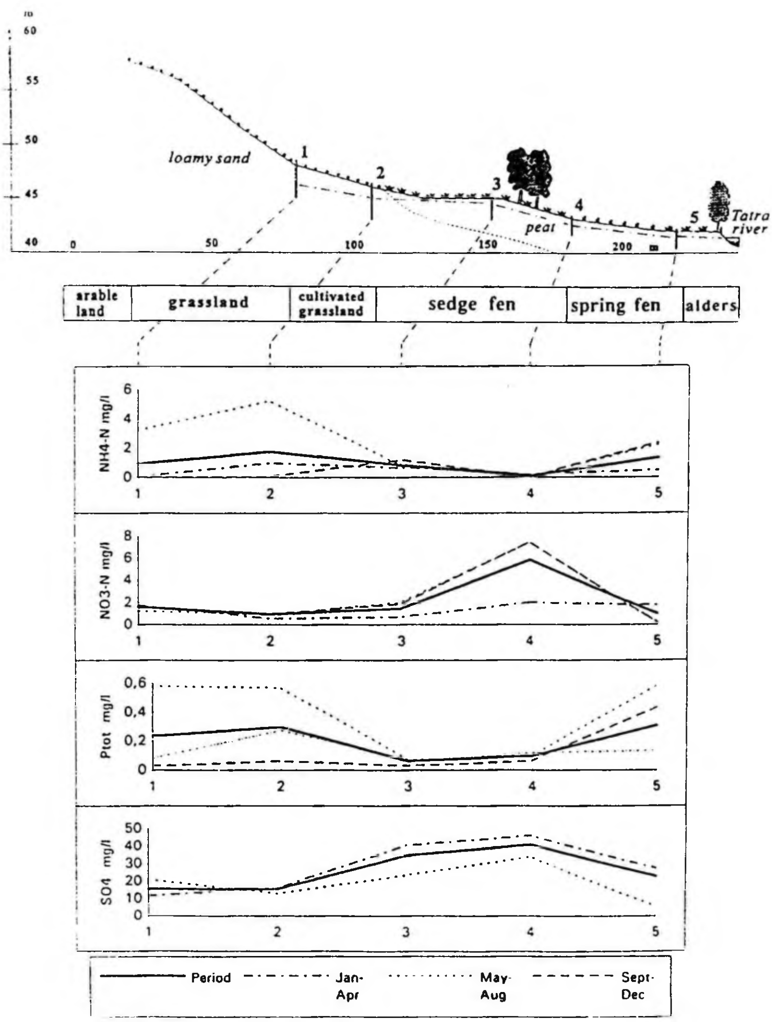

Fig. 4. Nutrient dynamics in the landscape profile of Tatra Middle. Arable land: fallow. Abandoned grassland: *Elytrigia repens* group. Cultivated grassland (with *Dactylis glomerata* dominating). Complex of valley bottom wetlands on peatland soil: a sedge fen (*Carex elata* association) with some willows (*Salix* spp.) and birches (*Betula pubescens*), a spring fen (*Sesleria coerulea* association). Alder forest strip (*Alnus incana* and *A. glutinosa*). For explanation of symbols see Fig. 2.

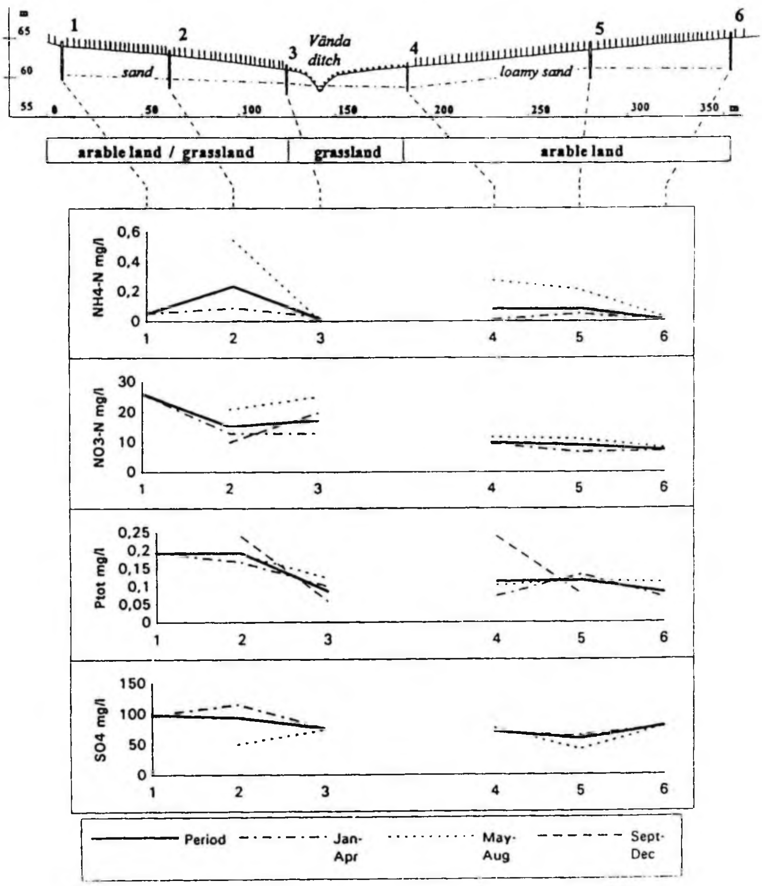

Fig. 5. Nutrient dynamics in the landscape profile of Vända Middle. Crops in arable land: barley, ripe, winter wheat. Grassland: pioneer weed vegetation (*Elytrigia repens* group). For explanation of symbols see Fig. 2.

3.2. Laboratory analyses

All water analyses were made following COMECON countries standard methods compatible with international methods for examination of water and wastewater quality (American Public Health Organization (APHA), 1981). The pH value was measured in the laboratory with glass electrode pH meter. Dissolved ammonium-N and sulphate-S were determined colorimetrically, according to Golterman et al. (1978). Nitrite and nitrate were measured in acidified samples by reduction on cadmium amalgam and colorimetry (APHA, 1981). The total phosphorus (orthophosphate phosphorus+organic phosphorus) in water samples was determined colorimetrically after conversion to orthophosphate by wet digestion with acid. Determination of dissolved orthophosphate followed colorimet-

Table 1
Herbaceous layer biomass (g DW m^{-2}) and plant uptake of nitrogen and phosphorus (g m^{-2})

Transect	Vegetation community[a]	Sampling plot no.[a]	Biomass			Nitrogen in biomass			Phosphorus in biomass		
			Above-ground	Below-ground	Total	Above-ground	Below-ground	Total	Above-ground	Below-ground	Total
Porijõgi											
Middle	Arable land (rye)	1	2948	161	3109	13.05	2.22	15.27	5.5	0.29	5.79
	Cultivated grassland	4	469	355	824	8.85	3.8	12.65	1.6	0.42	2.02
	Wet meadow	5	3090	511	3601	84.9	0.85	85.8	5.48	0.88	6.36
Tatra Middle	Grassland	1	904	778	1682	9.7	8.4	18.1	1.14	0.88	2.02
	Cultivated grassland	2	469	355	824	8.85	3.8	12.65	1.6	0.42	2.02
	Sedge fen	3	2081	2443	4524	33.3	21	54.3	3.2	2.1	5.3
	Spring fen	4	601	1690	2291	8.9	13.15	22.05	0.64	0.65	1.29
	Spring fen	5	1369	805	2174	24.4	10.9	35.3	2.44	0.67	4.06

[a] For description of vegetation community and sampling plots see Figs. 3 and 4.

rically as blue phosphomolybdate, without extraction (Golterman et al., 1978).

All plant material samples for chemical analysis were air-dried (40°C), ground, and analyzed for N and P contents: N with Kjeldahl procedure (APHA, 1981) and P photocolorimetrically (APHA, 1981).

4. Results and discussion

4.1. General influence of riparian buffers on stream water quality

The influence of riparian buffers on water quality was observed by comparison with long-term average values of nutrient contents in streams of the Porijõgi River catchment area (Table 2). All measuring points (weirs) of these streams are located 0.5–2 km downstream from the riparian ecotone transects. A dense positive correlation was found between the proportion of arable land and nutrient runoff from the sub-catchments (total inorganic nitrogen (TIN), $r=0.68$; P_{tot}, $r=0.88$; SO_4-S, $r=0.78$; Mander et al., 1989). Also, correlation between the proportion of arable land and mean nutrient concentrations in streams is positive ($r=0.82$–0.95; Table 2). A negative correlation was found between the mean buffer zone width and nutrient concentrations in streams ($r=0.72$–0.77). Both Porijõgi and Tatra are relatively unloaded small rivers that flow into valleys and have well developed riparian buffer zones (forest, meadow and fen ecosystems). Accordingly, very low average NH_4-N, NO_3-N, PO_4-P and SO_4-S concentrations in these rivers have been registered. The values are close to those recommended for Estonian rivers: 0.15 mg, 1.0 mg, 0.03 mg and 5.0 mg l^{-1}, respectively (Velner et al., 1978, cited in Mander, 1989). Relatively low standard deviations (SD) seem to suggest a stabilizing influence of natural buffer zones. Only P_{tot} content and SD value for Tatra River was relatively high (0.07 ± 0.14 mg P l^{-1}), probably as a result of uncontrolled fluxes from a pig farm close to the valley or because of orthophosphorus leaching from the peatlands in the valley bottom (Table 2, see also Fig. 4). Sipeoja stream (Sipe stream) and Vända channel are much smaller but have similar water discharge and pollution loads within the catchment area (during 1987–1991, 110–150 kg N ha^{-1}, 30–50 kg P ha^{-1} and 80–120 kg K ha^{-1}). However, Sipeoja stream has a well developed buffer zone of fens and grey alder forests/willow bushes of 50–150 m width but the Vända ditch catchment is without such compensating ecotone. Therefore, the NH_4-N and NO_3-N concentrations and SD values for Vända ditch are two to three times higher than in Sipeoja stream (Table 2).

Table 2
Long-term (1987–1992) average values and standard deviations for water discharge ($m^3\ s^{-1}$) and nutrient contents ($mg\ l^{-1}$) of the Porijõgi River catchment area. All measuring points are downstream from the riparian ecotone transects

Measuring point	Transect no. (Fig. 1)	Sub-catchment area (ha)	Percent of arable land in the sub-catchment	Mean width of buffer zone[a] (m)	No. of samples	Water discharge	NH_4-N	NO_3-N	Total P	SO_4-S
Sipeoja	2	900	57.6	50	83	0.06 ± 0.09	0.31 ± 0.19	1.40 ± 1.05	0.07 ± 0.09	11.5 ± 5.5
Porijõgi II	3	9560	29.8	200	83	0.88 ± 1.02	0.19 ± 0.13	0.84 ± 0.64	0.04 ± 0.03	6.7 ± 2.9
Tatra	4	3800	45.2	120	83	0.38 ± 0.27	0.16 ± 0.12	0.97 ± 0.65	0.07 ± 0.14	8.9 ± 4.2
Vända	5	210	68.4	2.5[b]	77	0.03 ± 0.06	0.80 ± 0.54	3.02 ± 2.00	0.08 ± 0.15	11.2 ± 5.8

[a] Mean distance of arable land from the river channel.
[b] Width of the ditch bank.
Significantly differentiated pairs of nutrient concentrations (Student's t-test, $t > 2$): NH_4-N: Sipeoja-Porijõgi II, sipeoja-Tatra, Sipeoja-Vända, Porijõgi II-Vända, Tatra-Vända; NO_3-N: all pairs; PO_4-P: no pairs; SO_4-S: all pairs.

4.2. Seasonal dynamics of groundwater quality in the riparian transects

All results are divided into four time series according to the most important hydrographical periods and presented as average values of these periods: total period (January 1992–August 1993), winter/spring (January–May), summer (May–August), and autumn/winter (September–December) (Figs. 2–5).

4.2.1. Nitrogen

Variations in the concentration of different nitrogen forms in groundwater along the landscape transects differed seasonally. The average study period (January 1992–August 1993) values of NH_4-N in all piezometers were between 0.05 and 3.0 mg l^{-1}. It was the most variable water quality parameter. The lowest values were found in sandy soils under arable land (Figs. 2, 3 and 5). This appears to be the result of relatively good aeration in light soils that guarantees good conditions for nitrification (see Peterjohn and Correll, 1984). The highest average ammonium concentrations of the whole period (2–3 mg NH_4-N l^{-1}; Figs. 3 and 4) were under the perennial cultivated grasslands that have not been plowed for at least 7 years. Similar results were described in earlier investigations on riparian buffer strips by Knauer and Mander (1989). However, several authors have pointed out the preventive role of perennial grasslands as buffers for nitrogen, including ammonium (Lowrance et al., 1984; Cooke and Cooper, 1988; Zirschky et al., 1989; Ambus and Christiansen, 1991). One reason for NH_4 leaching from grassland soils in the Porijõgi catchment is the deep humus layer (maximum depth 1.2 m; Porijõgi Middle transect; see piezometer 3, Fig. 3). This has formed on the lower slopes by accumulation during last 50–60 years owing to high rates of erosion in the upper hillslope. Relatively shallow roots of the grassland community (25–50 cm) and deep aeration in the sandy humus layer create conditions conducive to humus mineralization and ammonium leaching. This takes place mostly during the growing season (Figs. 2, 3 and 4). It was also observed that significantly higher NH_4-N concentrations in piezometer water was associated with spring fen communities (Fig. 4), owing to mineralization under anaerobic conditions in the peat. The NH_4-N concentration in groundwater from the grey alder forest was lower, averaging less than 0.2 mg l^{-1}.

Nitrite nitrogen values in transects varied very little and were always less than 0.05 mg l^{-1}. This was the same in the case of stream water quality. Therefore, this parameter has not been shown in the figures.

Transformation of nitrate nitrogen in riparian ecosystems is complex and many studies have been carried out to clarify these processes (Hussey et al., 1985; Cooke and Cooper, 1988; Pinay and Décamps, 1988; Hoffmann, 1991; Ambus and Christiansen, 1991; Groffman et al., 1991; Correll and Weller, 1992; Lowrance, 1992; Brüsch and Nilsson, 1993; Haycock and Pinay, 1993). The variation of NO_3-N concentration over the time in water samples from the Porijõgi River catchment was very small (SD less than 25% of the absolute value), and significantly less than variation in space. Very high values of NO_3-N (15–25 mg l^{-1} in the Sipeoja Lower transect and 10–35 mg l^{-1} in the Vända Middle transect) were found in piezometer water under the intensively fertilized arable lands (Figs. 2 and 5). These values exceed allowable drinking water standards (10 mg NO_3-N l^{-1}) by 1.5–3.5 times and the levels recommended for Estonian rivers by 15–35 times (Velner et al., 1978, cited in Mander, 1989). However, the NO_3-N content in the Porijõgi Middle landscape profile (1.0 mg NO_3-N l^{-1}; Fig. 3) was unexpectedly low for arable lands. Under perennial grasslands on mineral soils the average NO_3-N concentration did not exceed 2.0 mg l^{-1} and in alder forests the NO_3-N content was always less than 0.5 mg l^{-1}. One factor causing lower NO_3-N concentrations could be denitrification which can remove 0.4 g N m^{-2} h^{-1} (Lowrance, 1992) and may also be intensive during the winter (Brüsch and Nilsson, 1993; Haycock and Pinay, 1993). A significant increase of the average NO_3-N concentration in shallow groundwater has been found in the transition zone of sedge and spring fen communities in the valley bottom of Tatra River where the NO_3-N content increases by as much as 6 mg l^{-1} (Fig. 4). This is probably due to seeping deeper groundwater layers which transport nitrate to the valley bottom. This 3–5 m wide seepage zone is easily recognized in winter, when ice shields form on the frozen ground. Throughout this region (Tatra River and Vända ditch subcatchments), the groundwater is heavily polluted with nitrate (Mander et al., 1994).

The concentration of TIN (sum of NH_4-N, NO_2-N and NO_3-N) in piezometer water depends on NO_3-N concentrations. If the NO_3-N content is high (over 10 mg NO_3-N l^{-1}) then the TIN concentration is dominated by the nitrate ion (Figs. 2 and 5). In this case, the variation in TIN values between arable land and cultivated grassland was much smaller than in the various nitrogen ions. The average TIN content in piezometer water in both arable lands and cultivated grasslands varied between 1.5 and 3.5 mg l^{-1}. Within the alder forests, however, the maximal values of TIN never exceeded 1.0 mg l^{-1}.

Because of good mineralization conditions, the total nitrogen (N_{tot}; TIN + dissolved organic N) concentration in mineral soils of the transects is probably very close to the TIN concentration values (see e.g. Jordan et al., 1986; Knauer and Mander, 1989). In peatland soils the share of N_{tot} will be much higher and depends on aeration conditions but, normally, will not be more than 30% of all nitrogen forms (Nichols, 1983; Howard-Williams, 1985).

4.2.2. Phosphorus

All values of PO_4-P concentration in piezometers were about 30–40% less than P_{tot} values. The correlation between these two parameters was highly significant ($R^2 = 0.85$). Therefore, only P_{tot} values are presented in this paper. They varied on a large scale both over the time and space. The lowest variations of P_{tot} values in time occurred in the Sipeoja Lower profile (less than 20% of mean values over the study period). In the highest piezometer of this transect the average P_{tot} content reached 1.3 ± 0.31 mg l^{-1} (Fig. 2). This is probably a result of local anaerobic conditions in the soil causing a leachate of orthophosphorus. This is supported by the fact that the NO_3-N concentration in this piezometer was also lower. The P_{tot} value in the piezometer upslope from the alder buffer strip dropped to 0.2 mg P l^{-1}. Also, NO_3-N and SO_4-S mean values decreased in the same profile before the start of the buffer strip. This may be the result of small-scale changes in soil parameters.

In the rest of the transects, the P_{tot} variations over time were large. Because of relatively low concentrations, most of the differences in values were not significant (SD values differ from absolute values by 100–150%). A significant leaching of phosphorus appeared from the sandy colluvial soil under the cultivated grassland. This was most intensive during the summer period when the average P_{tot} content in piezometer water reached 0.6 mg l^{-1}. The average P_{tot} concentration in the piezometer water under the cultivated grassland for the study period was 0.25 ± 0.17 mg P l^{-1} (Figs. 3 and 4). However, under the wet meadow (the *Filipendula ulmaria–Cirsium oleraceum–Aegopodium podagraria* community) and alder forest the mean study period P_{tot} concentration and its standard deviation are significantly less: 0.10 ± 0.08 mg and 0.11 ± 0.10 mg P l^{-1}, respectively (Fig. 3).

A slight decrease of P_{tot} content in the ditch bank piezometers appeared in the Vända Middle transect, which does not have significant buffer ecosystems (Fig. 5).

A significant decrease of P_{tot} concentration within the spring fen in the Tatra Middle transect is probably a combined result of deeper groundwater seeping and mineralization of organic P to orthophosphorus which is then leached under anaerobic conditions in the fen (Fig. 4).

4.2.3. Sulphur

In comparison with other ions, the sulphate concentration showed the least variation over the experimental period. The standard deviation values of each piezometer were less than 25% of the absolute average value for the whole period. Higher sulphate concentrations were found in groundwater under the intensively fertilized arable lands on sandy soils: 100 mg S l^{-1} in the Sipeoja Lower (in the highest piezometer) and Vända Middle profiles, and 80 mg S l^{-1} Porijõgi

Middle profile (Figs. 2, 3 and 5). Under the perennial grassland vegetation downslope of arable lands the SO_4-S concentration was 20–60 mg l^{-1}. In the piezometers installed in peatland (Tatra Middle) the SO_4-S concentration was significantly higher than in adjacent mineral soils under the perennial grassland (35 mg l^{-1} vs. 15 mg l^{-1}) but was less than under the arable lands. Thus, perennial grasslands on mineral soils had lower sulphate loading to shallow groundwater.

Increasing SO_4-S values under the peatland of Tatra River valley may be the result of external sulphate transport by seeping deeper groundwater in the valley bottom (Fig. 4).

Alder forests had no significant influence on the sulphate content in the groundwater toplayer of mineral soils. Nevertheless, a slight decrease of SO_4-S content on the bank of Tatra River, where a narrow strip of alder trees on mineral alluvial sediments occur, was probably due to changing aeration conditions within the alder root-zone (Fig. 4).

4.3. Herb biomass analyses in landscape profiles

There is limited work on nutrient uptake by plants in riparian ecosystems, especially in riparian forests (Peterjohn and Correll, 1984; Fail et al., 1986). Our investigations on productivity and nutrient accumulation of herb communities in two transects of the Porijõgi catchment area showed that the production in riparian sedge fens and wet meadows is 2–5.5 times higher than in adjacent cultivated grasslands (3600–4500 g dry weight (DW) and 820–1680 g DW m^{-2}, respectively), and up to 30% higher than in the rye field (3109 g DW m^{-2}; Table 1).

Nitrogen uptake by the riparian *Carex elata* community was three to four times higher than for adjacent cultivated grassland (54 g N m^{-2} vs. 13–18 g N m^{-2}). The accumulation of nitrogen in the total biomass (above-ground + below-ground biomass) of the *Filipendula ulmaria–Cirsium oleraceum–Aegopodium podagraria* community was 85.8 g N m^{-2}, which exceeds the N uptake rate of the rye field by a factor of 5.6 and cultivated grassland by a factor of 6.8. In this riparian community, 99% of nitrogen was accumulated in above-ground biomass (Table 1).

Differences in phosphorus uptake between riparian ecosystems and cultivated land communities were significantly less but showed the advantage of riparian communities: 3.1–6.4 g P m^{-2} in riparian sedge fens and meadows and 2.0–5.8 g P m^{-2} in rye fields and grasslands. Maximum phosphorus uptake in the *Filipendula ulmaria–Cirsium oleraceum–Aegopodium podagraria* community was 6.4 g P m^{-2}. About 60–85% of the phosphorus was accumulated in above-ground biomass of riparian communities.

We do not know about further nutrient fluxes during the decomposition and mineralization of such large amounts of herb biomass. Certainly, a significant part of the nutrients will accumulate in the soil. The low nitrogen and phosphorus concentrations observed in the riparian shallow groundwater during the study period indicates that the leaching of nutrients from these communities is not an essential component. Anyway, an effective and normal way of nutrient removal from riparian zones is biomass harvesting. The calculated average annual denitrification rate within riparian meadows is about 1700 kg N ha^{-1} $year^{-1}$ (base on average denitrification rates for spring and autumn of about 20 mg N m^{-2} h^{-1}; Brüsch and Nilsson, 1993). Then, in addition, the nutrient removal via harvesting could remove 30% of annual nitrogen accumulating both in vegetation and soils. For phosphorus this rate will probably be 20%.

4.4. Perspectives

Seeping of contaminated deep groundwater within buffer zones or directly into the riverbed is a serious problem which occurs in valleys intersecting deeper groundwater layers of agricultural uplands (Fig. 4). Another problem could be that at some point in time buffer zones could become nutrient saturated (e.g. Vanek, 1991). Nevertheless, in both cases a forest or bush buffer strip on river banks seems to be an effective method to control both nitrogen and phosphorus fluxes. A positive side effect of these stands would be to protect banks against erosion. Nutrient saturation could be avoided by regular harvesting of these strips which would not cause significant nutrient losses like the clear-cutting or large-scale afforestation in the catchment if undertaken sensitively (Bormann and Likens, 1979; Smith, 1992). Thus, one possibility would be for these areas to join the network of decentralized energy forest areas.

A very important area of further investigation is the long-term role of black alder (*Alnus glutinosa*) and grey alder forests in nitrogen and phosphorus cycling. Alders (both *A. glutinosa* and *A. incana*), the common riparian trees in Europe, are nonleguminous angiosperms which possess an endophytic actinomycetal fungus in root nodules that enables them to fix nitrogen at a rate of up to 225 kg N ha^{-1} $year^{-1}$ (Wetzel, 1975). This may cause a large nitrogen surplus in the ecosystem and lead to leaching from the riparian soils (Binkley et al., 1992; Burmann and Gordon, 1984). Nevertheless, results of this study and some earlier investigations in Estonia and Germany show that alder forests have a beneficial role in filtering nitrogen and phosphorus fluxes (Mander, 1985, 1989; Knauer and Mander, 1989). For instance, nearly 100% of the phosphorus and 50% of the nitrogen was adsorbed in 10-m-wide strips of both riparian alder forests growing downslope of intensively fertilized agricultural fields (Knauer and Mander, 1989). Probably, a dense herb layer of these alder forests dominated by nettles (*Urtica dioica*) would enhance nitrogen uptake.

Many questions remain outstanding in relation to the role of alder stands in nutrient cycling. Additional studies have been initiated at the Limnology Institute, University of Lund, Sweden, and in Estonia. In Lund, the influence of the age of black alder forests on nitrogen and phosphorus cycling will be studied. In Estonia, in co-operation with several departments of the University of Tartu and Estonian Agriculture University, grey alder forest sites (one without nutrient loading, another with additional input fluxes by application of slurry and wastewater) will be analyzed for transformation and storage. The main concern is: how long will the buffering process remain if a riparian zone is vegetated with alder?

5. Conclusions

(1) A significant lower mean annual concentration of NH_4-N, NO_3-N and total-P concentration in shallow groundwater within the combination of riparian *Filipendula ulmaria–Cirsium oleraceum–Aegopodium podagraria* communities and young grey alder forest was observed (0.4 mg, 0.5 mg and 0.1 mg l^{-1}, respectively).

(2) During the vegetation assimilation period, total nutrient uptake by biomass (sum of the above-ground and below-ground biomass) of the riparian *Filipendula ulmaria–Cirsium oleraceum–Aegopodium podagraria* community was up to 86 g N m^{-2} $year^{-1}$ and 6.4 g P m^{-2} $year^{-1}$. In *Carex elata* associations the corresponding values were 54 g and 5.3 g m^{-2}.

(3) In cultivated grasslands with shallow sod layer, especially on deep colluvial soils (humus layer depth up to 1.2 m), a significant leaching of NH_4-N and PO_4-P was observed. This process intensified in summer and autumn (from May to November).

(4) Seeping of contaminated deeper groundwater at the base of valley slopes, as well as anaerobic conditions in the peatland soils increases the NO_3-N, P_{tot} and SO_4-S concentrations of groundwater within the riparian fen communities.

(5) Grey alder forests and willow bushes on stream banks are recommended as water protection strips to decrease nutrient fluxes into the stream, i.e. perennial grasslands as water- protection zones should be combined with forest/bush buffer strips.

(6) Harvesting of riparian herb communities may remove a significant fraction (25–30%) of the annual nitrogen and phosphorus accretion with riparian zones.

(7) In the Vända ditch sub-catchment, a project of ecotechnological measures to improve the water quality of this area has been worked out (Mander et al., 1994). That concept is based on principles of buffer zones and buffer strips, root-zone systems for wastewater purification and restoration of the stream channel (e.g. Petersen et al., 1992).

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References

Ambus, P. and Christiansen, S., 1991. Denitrification and other nitrogen cycling processes in a riparian meadow. Project abstracts of the Danish NPo Research Programme. C-Abstracts. No. C13a. Ministry of the Environment, National Agency of Environmental Protection, Denmark, pp. 187–202.

American Public Health Organization, 1981. Standard Method for the Examination of Water and Waste Water, 15th edn. APHA, Washington, 1134 pp.

Bingham, S.C., Westerman, P.W. and Overcash, M.R., 1980. Effect of grass buffer zone length in reducing the pollution from land application areas. Trans. ASAE, 23: 330–336.

Binkley, D., Sollins, P., Bell, R., Sachs, D. and Myrold, D., 1992. Biogeochemistry of adjacent conifer and alder–conifer stands. Ecology, 73 (6): 2022–2033.

Bormann, B.T. and Gordon, J.C., 1984. Stand density effects in young red alder plantations: productivity, photosynthate partitioning and nitrogen fixation. Ecology, 2: 394–402.

Bormann, F.H. and Likens, G.E., 1979. Pattern and Process in a Forested Ecosystem. Springer, New York/Heidelberg/Berlin, 253 pp.

Brüsch, W. and Nilsson, B., 1993. Nitrate transformation and water movement in a wetland area. Hydrobiologia, 251: 103–111.

Cooke, J.G. and Cooper, A.B., 1988. Sources and sinks of nutrients in a New Zealand hill pasture catchment III. Nitrogen. Hydrol. Process., 2: 135–149.

Correll, D.L., 1991. Human impact on the functioning of landscape boundaries. In: P. Risser, R. Naiman and M.M. Holland (Editors), The Role of Landscape Boundaries in the Management and Restoration of Changing Environments. Chapman and Hall, New York, pp. 22–34.

Correll, D.L. and Weller, D.E., 1992. Factors limiting processes in freshwater wetlands: an agricultural primary stream riparian forest. In: R.R. Sharitz and J.W. Gibbons (Editors), Freshwater Wetlands and Wildlife. CONF-8603101, DOE Symp. Ser. No. 61, USDOE Office of Scientific and Technical Information, Oak Ridge, TN, pp. 9–23.

Décamps, E., Fournier, F., Naiman, R.J. and Petersen, Jr., R.C., 1990. An international research effort on land/inland water ecotones in landscape management and restoration 1990–1996. Ambio, 19(3): 175–176.

Dillaha, T.A., Sherrard, J.H., Lee, D, Mostaghimi, S. and Shiholtz, V.O., 1988. Evaluation of vegetated filter strip as a best management practice for feed lots. J. Water Pollut. Control Fed., 60: 1231–1238.

Doyle, R.C., Wolf. D.C. and Bezdicek, D.F., 1975. Effectiveness of forest buffer strips in improving the water quality of manure polluted runoff. In: Managing Livestock Wastes. Proc. 3rd Int. Symp. on Livestock Wastes, 21–24 April 1975, ASAE, St. Joseph, MI, pp. 299–302.

Fail, Jr., J.L., Hamzah, M.N., Haines, B.L. and Todd, R.L., 1986. Above and belowground biomass, production, and element accumulation in riparian forests of an agricultural catchment. In: D.L. Correll (Editor), Catchment Research Perspectives. Smithsonian Environmental Research Center, Edgewater, MD. Smithtronian Institution Press, Washington, DC, pp. 193–224.

Golterman, H.L., Clymo, R.S. and Ohnstad, M.A.M., 1978. Methods for Physical and Chemical Analysis of Fresh Waters. Blackwell, London, 214 pp.

Groffman, P.M., Axelrod, E.A., Lemunyon, J.L. and Sullivan, W.M., 1991. Denitrification in grass and forest vegetated filter strips. J. Environ. Qual., 20: 671–674.

Haycock, N.E. and Pinay, G., 1993. Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. J. Environ. Qual., 22: 273–278.

Hoffmann, C.C., 1991. Water and nutrient balances for a flooded riparian wetland. In: Ministry of the Environment, National Agency of Environmental Protection, Denmark. Nitrogen and Phosphorus in Fresh and Marine Water. Project Abstracts of the Danish NPo Research Programme, C-Abstracts, C13-b-, pp. 203–220.

Howard-Williams, C., 1985. Cycling and retention of nitrogen and phosphorus in wetlands: a theoretical and applied perspective. Freshwater Biol., 15: 391–431.

Hussey, M.R., Skinner, Q.D., Adams, J.C. and Harvey, A.J., 1985. Denitrification and bacterial numbers in riparian soils of a Wyoming mountain catchment. J. Range Manage., 38(6): 492–496.

Jacobs, T.C. and Gilliam, J.W., 1985. Riparian losses of nitrate from agricultural drainage waters. J. Environ. Qual., 14(4): 472–478.

Jagomägi, J., Külvik, M., Mander, Ü. and Jacuchno, V., 1988. The structural-functional role of ecotones in the landscape. Ekológia (CSSR), 7(1): 81–94.

Jordan, T.E., Correll, D.L., Peterjohn, W.T. and Weller, D.E., 1986. Nutrient flux in a landscape: the Rhode River catchment and receiving waters. In: D.L. Correll (Editor), Catchment Research Perspectives. Smithsonian Environmental Research Center, Edgewater, MD. Smithsonian Institution Press, Washington, DC, pp. 57–76.

Knauer, N. and Mander, Ü., 1989. Studies on the filtration effect of differently vegetated buffer stripes along inland waters in Schleswig-Holstein. 1. Information: Filtration of nitrogen and phosphorus. Z. Kulturtech. Landentwicklung, 30: 365-376, in German, with English summary.

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

Lowrance, R., Todd. R., Fail, J., Hendrickson, Jr., O., Leonard, R. and Asmussen, L., 1984. Riparian forests as nutrient filters in agricultural catchments. BioScience, 34(6): 374-377.

Magette, W.L., Brinsfield, R.B., Palmer, R.E. and Wood, J.D., 1989. Nutrient and sediment removal by vegetated filter strips. Trans. ASAE, 32(2): 663-627.

Mander, Ü., 1985. The renovation effect of polluted surface flow in vegetated buffer strips. Acta Commentationes Universitatis Tartuensis, 701: 77-89, in Russian, English summary.

Mander, Ü., 1989. Kompensationsstreifen entlang der Ufer und Gewässerschutz. Bericht D 10, Landesamt für Wasserhaushalt und Küsten Schleswig-Holstein, Kiel, 61 pp.

Mander, Ü., Metsur. M. and Külvik, M., 1989. Störungen des Stoffkreislaufs, des Energieflusses und des Bios als Kriterien für die Bestimmung der Belastung der Landschaft. Petermanns Geogr. Mitt., 4: 233-244.

Mander, Ü., Kuusemets, V. and Treier, K., 1994. Variation in groundwater quality in South Estonian rural areas. IAHS Proc. of GQM '93: Groundwater Quality Management, Tallnin, Estonia, 6-9 September 1993. IAHS Publ. No. 220, pp. 81-93.

Mitsch, W.J., 1992. Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution. Ecol. Eng., 1: 27-47.

Mitsch, W.J. and Gosselink, J.G., 1993. Wetlands, 2nd edn. Van Nostrand Reinhold, New York, 722 pp.

Nichols, D.S., 1983. Capacity of natural wetlands to remove nutrients from wastewater. J. Water Pollut. Control Fed., 55: 495-505.

Overcash, M.R., Bingham, S.C. and Westerman, P.W., 1981. Predicting runoff pollutant reduction in buffer zones adjacent to land treatment sites. Trans. ASAE, St. Joseph, MI, pp. 24, 430-435.

Payer, F.S. and Weil. R.R., 1987. Phosphorus renovation of wastewater by overland flow land application. J. Environ. Qual., 16(4): 391-397.

Persson, H., 1983. The distribution and productivity of fine roots in boreal forests. Plant Soil, 71: 87-101.

Peterjohn, W.T. and Correll. D.L., 1984. Nutrient dynamics in an agricultural catchment: observations on the role of riparian forest. Ecology, 65: 1466-1475.

Petersen, R.C., Petersen, L.B.-M. and Lacoursière, J.O., 1992. A building-block model for stream restoration. In: P.J. Boon, P. Calow and G.E. Petts (Editors), River Conservation and Management. John Wiley, pp. 293-309.

Pinay, G. and Décamps, H., 1988. The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: a conceptual model. Regul. Rivers: Res. Manage., 2: 507-516.

Pinkowski, R.H., Rolfe, G.L. and Arnold, L.E., 1985. Effect of feedlot runoff on a Southern Illinois forested catchment. J. Environ. Qual., 14(1): 47-54.

Reddy, K.R., Campbell, K.L., Graetz, D.A. and Portier, K.M., 1982. Use of biological filters for treating agricultural drainage effluent. J. Environ. Qual., 11: 591-595.

Risser, P., 1990. The ecological importance of land-water ecotones. In: R.J. Naiman and H. Décamps (Editors), The Ecology and Management of Aquatic-Terrestrial Ecotones, Man and the Biosphere Series, UNESCO, The Parthenon Publ. Group, Paris pp. 7-21.

Schwer, C.B. and Clausen, J.C., 1989. Vegetative filter treatment of dairy milkhouse wastewater. J. Environ. Qual., 18: 446-451.

Smith, C.M., 1992. Riparian afforestation effects on water yield and water quality in pasture catchments. J. Environ. Qual., 21: 237-245.

Smith, R.G. and Schroeder, E.D., 1985. Field studies of the overland flow process for the treatment of raw and primary treated municipal wastewater. J. Water Pollut. Control Fed., 57(7): 785-794.

Soil Maps of Collective and State Farms of Estonia, 1937. Map series in the State Design Institute 'Eesti Põllumajandusprojekt'.

Thompson, D.B., Loudon, T.L. and Gerrish, J.B., 1978. Animal manure movement in winter runoff for different surface conditions. Best Management Practices for Agriculture and Silviculture, Cornell Conference, pp. 145-158.

Uusi-Kämppä, J. and Yläranta, T., 1993. Buffer strips for sediment removal from agricultural surface runoff. In: Ü. Mander et al. (Editors), Energy, Environment and Natural Resources Management in the Baltic Sea Region, 4th Int. Conf. on System Analysis, Tallinn, Estonia, 18-21 May 1993. Nordiske Seminar-og Arbejdsrapporter 653, pp. 469-478.

Vanderholm, D.H., Dickey, E.C., Jacobs, J.A., Elmore, R.W. and Spahr, S.L., 1979. Livestock feedlot runoff control by vegetative filters. EPA-600/2-79-143. US Environmental Protection Agency, 66 pp.

Vanek, V., 1991. Riparian zone as source of phosphorus for a groundwater-dominated lake. Water Res., 25(4): 409-418.

Varep, E., 1964. The landscape regions of Estonia. Publications on Geography IV. Acta et Commerstationes Universitatis Tartuensis, 156: 3-28.

Vought, L.B.-M., Lacoursière, J.O. and Voelz, N.J., 1991. Streams in the agricultural landscape? Vatten, 47: 321-328.

Wetzel, R.G., 1975. Limnology. W.B. Saunders, Philadelphia, PA, 539 pp.

Yates, P. and Sheridan, J.M., 1983. Estimating the effectiveness of vegetated floodplains/wetlands as nitrate-nitrite and orthophosphorus filters. Agric. Ecosyst. Environ., 9: 303-314.

Young, R.A., Huntrods, T. and Anderson, W., 1980. Effectiveness of vegetated buffer strips in controlling pollution from feedlot runoff. J. Environ. Qual., 9(3): 483-487.

Zirschky, J., Crawford, D., Norton, L., Richards, S. and Deemer, D., 1989. Ammonia removal using overland flow. J. Water Pollut. Control Fed., 61(7): 1225-1232.

II

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The potential role of wet meadows and grey alder forests as buffer zones

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Ü. Mander[1], K. Lõhmus[1], V. Kuusemets[1] and M. Ivask[2]

[1] *Institute of Geography, University of Tartu, 46 Vanemuise St., EE2400 Tartu, Estonia*
[2] *Environmental Protection Institute, Estonian Agricultural University, 5 Kreutzwaldi St., EE-2400 Tartu, Estonia*

Abstract
Two differently loaded riparian buffers including wet meadows *(Filipendula ulmaria-Aegopodium podagraria – Cirsium oleraceum – Urtica dioica)* and grey alder *(Alnus incana)* stands were investigated. The main inputs and outputs (atmospheric deposition, input and output in surface and subsurface flow, nitrogen fixation and denitrification) and accumulation of N and P in plant biomass, litter and soil were estimated. A 31 m wide buffer zone of wet meadow and grey alder forest removed 67% nitrogen and 81% phosphorus, while in a 51 m buffer zone, also containing a grassland strip in addition to wet meadow and alder forest, 96% N and 97% P was retained. In the riparian buffers studied, an effective retention of nutrients (34-186 kg N ha^{-1} yr^{-1} and 1.6-6.0 kg P ha^{-1} yr^{-1}) was observed even with very high input loads (276.3 and 12.8 kg ha^{-1} yr^{-1} of N and P, respectively).

INTRODUCTION
Riparian wet meadows and forests have been identified as important buffers for water bodies (Lowrance *et al.*, 1984; Peterjohn and Correll, 1984; Haycock and Pinay, 1993; Vought *et al.*, 1994). Most research only considers inputs and outputs, but among the internal processes of riparian ecosystems relevant to nutrient retention, denitrification has been investigated most intensively (Groffmann *et al.*, 1991; Lowrance, 1992; Weller *et al.*, 1994; Pinay *et al.*, 1993). Only a few studies deal with plant uptake and soil accumulation within riparian forests (Peterjohn and Correll, 1984; Lowrance *et al.*, 1984) and wet meadows (Leonardson *et al.*, 1994). The aim of this paper is to analyse the most important nutrient fluxes and pools in complex riparian buffer zones to explain their transformation and removal efficiency in differently loaded areas.

SITE DESCRIPTION
Two different riparian buffers including wet meadows and grey alder stands were selected in Estonia: one in the unpolluted Porijõgi River catchment (for area description see Mander *et al.*, 1995), the other, in the vicinity of the Viiratsi pig farm (32,000 pigs), Viljandi County. The physio-geographical conditions of the Viiratsi study site are similar to those of the Porijõgi River catchment. In both areas, transects in thalwegs were established along a topo-edaphic gradient in autumn 1993 (Fig. 1). In the Porijõgi test site the following series of riparian buffer communities, in the downhill direction, was analysed: abandoned (formerly cultivated) grassland (serves as source community) – buffer 1; wet meadow (dominated by *Filipendula ulmaria, Aegopodium podagraria, Cirsium oleraceum,* and *Urtica dioica*), 11 m – buffer 2; grey alder stand (14 yr), 20 m. In the heavily polluted Viiratsi test site the transect was established through the following communities: arable land (fertilised by pig slurry; serves as source community) – buffer 1; eutrophic grassland strip *(Elytrigia repens, Urtica dioica)* with a young grey alder stand, 11 m – buffer 2; wet meadow pattern *(Filipendula ulmaria)*, 12 m – buffer 3; grey alder forest (40 yr), 28 m. In the landscape profiles piezometers (3 rows in the Porijõgi transect and 4 rows in the Viiratsi site, with 3 replicates in each row) and study plots were established on the boundaries between plant communities. The main nitrogen (N) and phosphorus (P) cycles and

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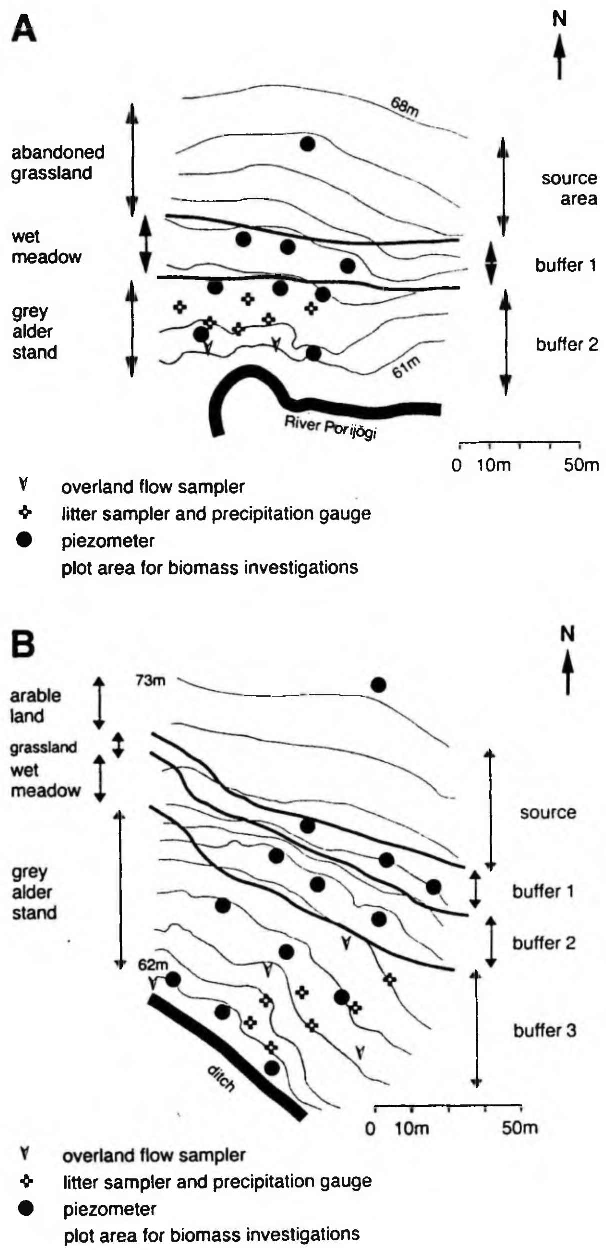
budgets have been assessed. This paper presents the main inputs and outputs (atmospheric deposition, input and output in surface and subsurface flow, nitrogen fixation and denitrification) as well as accumulation in plant biomass, litter and soil from July 1994 to July 1995.

Figure 1. Schemes of test sites: A – Porijõgi (less polluted), B – Viiratsi (heavily polluted).



MATERIAL AND METHODS

Productivity and uptake estimation

Dimension-analysis techniques (Bormann and Gordon, 1984; Rytter, 1989; Huss-Danell and Ohlsson, 1992) were used to estimate the above-ground biomass and productivity of grey alder forests. At both test sites (age 14 and 40 years) 17 and 5 model trees per plot, respectively, were felled to collect data on the following tree components: stem (wood and bark), secondary branch growth (wood and bark), primary branch growth, leaves, generative organs. The relative increments of the wood and bark of an overbark fraction were assumed to be equal. Root systems for 6 and 3 out of the sampled 17 and 5 trees respectively were excavated and divided into five fractions; stump, coarse roots: $d \geq 20$ mm, 5 mm $\leq d < 20$ mm, 2 mm $\leq d < 5$ mm and fine roots ($d < 2$ mm). Nodule mass (kg ha^{-1}) was estimated separately in June and July 1995. To estimate the below-ground production, the shoot/root ratios for tree biomass and production were assumed to be equal.

All tree components were analysed for N, P, energy and ash contents. Tree components have been estimated using the regression equation:

$$\ln y = a + b \ln dbh \qquad (1)$$

where y is the oven-dry mass of tree component (kg) and dbh – diameter at breast height (cm); all equations had very high correlation coefficients and significance ($p < 0.0001$ in all cases; Table 1).

Table 1. Parameters of regression equations (1) used in dimension analysis for estimating the mass of tree compartments (kg); r^2 – coefficient of determination, s.e.e. – standard error of estimate.

Age (years)	Tree compartment overbark (kg)	a	b	r^2	s.e.e.
14 (Porijõgi)	Stem	-2.492	2.399	0.992	0.07
	Branches	-6.064	3.123	0.925	0.31
40 (Viiratsi)	Stem	-2.406	2.354	0.984	0.14
	Branches	-3.891	2.353	0.947	0.33

The phytomass (*i.e.*, standing crop) samples were collected during the maximum flowering time of the dominant plant species (2nd and 3rd week in July; see Milner and Hughes, 1968) from all riparian plant communities. Sampling plots (six in Porijõgi and three in Viiratsi), for analysing plant cover and phytomass, were installed in typical areas of the community. Two typical patches within the wet meadow community (dominated by *Aegopodium podagraria* and *Filipendula ulmaria*) in the Porijõgi transect were analysed (see Balsberg, 1982). Above-ground biomass was collected from three replicate quadrats (1x1 m) in each community. Below-ground root biomass was collected from soil cores taken by auger (diameter 158 mm) from the depth of up to 40-50 cm in three replicates from each location.

Field experiments and laboratory analysis

Water samples were collected and groundwater depth measured once or twice a month by piezometers. Filtered soil water samples were analysed for NH_4-N, NO_2-N, NO_3-N, total Kjeldahl nitrogen (TKN), PO_4-P, total Kjeldahl phosphorus (TKP), SO_4, Fe, Ca (APHA, 1989). Soil bulk density, texture class and field capacity were determined for each 20 cm of soil profile (up to 1.5 m depth). Hydraulic conductivity was estimated by using tracer (chloride) and pumping experiments (Freeze and Cherry, 1979). Groundwater discharge was estimated on the basis of both Darcy's law and by gauging with weirs installed in groundwater seeping sites. TKN and TKP of plant samples were estimated.

The acetylene method was used to measure the N fixation rate in soil (Groffmann and Tiedje, 1989). To assess the denitrification rate acetylene as a nitrous oxide reduction inhibitor was used (Yoshinari *et al.*, 1977).

RESULTS AND DISCUSSION

Nitrogen and phosphorus budgets in differently loaded buffer zones

Nitrogen

Despite the significantly higher N loading in the Viiratsi riparian buffer zone relative to the Porijõgi complex (0.4-4.3 and 2.0-62.1 mg N l^{-1}, respectively) the output concentrations were comparable (0.4-2.1 and 0.5-3.0 mg N l^{-1}). Atmospheric N deposition in Porijõgi and Viiratsi was estimated to be 6.1 and 6.3 kg N ha^{-1} yr^{-1}, respectively. Extremely high TKN contents in the Viiratsi soil water (56.1 and 62.1 mg l^{-1}) reflect pig slurry application in the adjacent field in July and August 1994. Intensive fertilisation over many years has compacted the soil and disturbed the microfauna. Therefore the N concentration in soil water under the arable land has always been high. In the Porijõgi catchment, by contrast, the N input has decreased during the last three years since agricultural activities in the upland field ceased. The high buffering capacity in the Viiratsi study (Table 3) is the result of: (1) the large accumulation of organic nitrogen in the soil, (2) the relatively high plant uptake, (3) the relatively high denitrification value, (4) the relatively low N_2 fixation. Fluxes (2)-(4) were generally smaller than those in the Porijõgi test site (Table 2).

Table 2. Main fluxes and pools of nitrogen and phosphorus in the heavily polluted test site at Viiratsi (kg ha^{-1} yr^{-1}).

Fluxes and pools	Grassland		Wet meadow		Alder forest	
	N	P	N	P	N	P
Precipitation	6.3	3.1	6.3	3.1	6.7	1.0
Nitrogen fixation	5.5		0.8		21.0	
Input surface flow and subsurface flow	264.5	9.7	79.0	3.7	45.2	2.0
Accumulation in plant biomass	165.0	45.0	352.0	38.0	140.2	10.8
Litter	*78.2	*21.9	*201.8	*17.9	87.0	4.0
Denitrification	20.1		10.2		7.9	
Output surface flow and subsurface flow	79.0	3.7	45.2	2.0	9.0	0.4
Active soil exchange	90.2	-14.0	-119.5	-15.3	2.8	-4.2
Soil store (kg ha^{-1})	11.2	2.7	17.3	1.4	20.6	2.1

* above-ground litter, estimated values

Denitrification within the grey alder forest was 12-21 µg N m^{-2} hr^{-1} at Porijõgi and 3-14 µg N m^{-2} hr^{-1} in Viiratsi. Nevertheless, in adjacent wet meadow and abandoned grassland upslope from the forest in Porijõgi, the rate was higher (4-57 and 5-41 µg N m^{-2} hr^{-1}, respectively). The main characteristics in denitrification intensity are comparable with other investigations:

(a) most denitrifying activity was observed in spring and late summer (Struwe and Kjøller, 1990; Weller *et al.*, 1994)

(b) denitrification was faster in the upper part of the complex buffer zone (upslope from the alder forest; see also Duff and Triska, 1990; Pinay *et al.*, 1993; Weller *et al.*, 1994)

Characteristic (b) is due to significantly higher nitrate concentrations in soil water upslope. Struwe and Kjøller (1991) have found up to 100 times more denitrifying activity in slurry incubations than in black alder forest in the field.

Nitrogen uptake by grey alders is high at both sites. In Viiratsi the summary uptake is about 30% less than in Porijõgi, i.e. 140.2 and 204.8 kg N ha^{-1} yr^{-1}, respectively (Tables 2 and 3). The forest stand in Viiratsi is less dense and older than that in Porijõgi (1810 trees ha^{-1} by average age of 40 yr and 6110 trees ha^{-1} by 14 yr, respectively). In addition, only a portion of trees are grey alders; in Viiratsi 77% and in Porijõgi 86%. The N allocation in alder production indicates that most of the N accumulates in leaves (88.5 kg N ha^{-1} yr^{-1} in Viiratsi and 85.8 kg N ha^{-1} yr^{-1} in Porijõgi, in percentages of the total N uptake: 63% and 42%, respectively). We measured a relatively large amount of N in the bark of stems and branches (8.5 kg N ha^{-1} yr^{-1} in Viiratsi and 23.1 kg N ha^{-1} yr^{-1} in Porijõgi). Because the growth of trees in older stands is less than that in younger stands, the N uptake by stems and branches of old trees is lower than that of younger trees. Moreover, the annual N uptake stored in root production decreases in older stands. Due to slow retranslocation into senescing leaves in autumn (8% in Viiratsi and 14% in Porijõgi), most N is accumulated in leaf litter, half of which mineralises during the next season. Thus, a realistic N removal rate by tree uptake is about 40-50 kg less than the total annual uptake.

Atmospheric N_2 fixation in the alder stand in Viiratsi was significantly less than that in Porijõgi (i.e., 0.2-2.8 and 0.6-15 µg N m^{-2} hr^{-1}, respectively) with a maximum in July. The highest N_2 fixation values (up to 21.3 µg N m^{-2} hr^{-1} in Porijõgi and 17.9 µg N m^{-2} hr^{-1} in Viiratsi) were observed within wet meadows and grassland communities; most fixation was observed in May. The smaller N_2 fixation in Viiratsi is due to predominating N assimilation over N_2 fixation while high concentrations of mineral N are present in the root (see Troelstra *et al.*, 1992). However, our investigations show that N_2 fixation plays a less significant role in the total N budget in both study plots.

The active soil exchange (ASE) is calculated as follows (Tables 2 and 3):

$$\text{ASE} = \text{Input} - \text{Output} - \text{Accumulation in plant biomass} + \text{Litter} \qquad (2)$$

Except for the alder stand in Viiratsi, the active soil exchange of N for riparian communities was negative. Therefore, plant uptake exceeded the accumulation in soil during the study period. In comparison with the soil store of N in different communities (8-19 t ha^{-1} in Porijõgi and 11-21 t ha^{-1} in Viiratsi; Tables 2 and 3) the plant uptake and all other fluxes are small. Thus, the buffering capacity of colluvial soils with a deep humus layer, typical of riparian soils of agricultural areas, is the key factor in nutrient retention in studied buffer zones.

Table 3. Main fluxes and pools of nitrogen and phosphorus in the less polluted test site at Porijõgi (kg ha^{-1} yr^{-1}).

Fluxes and pools	Wet meadow		Alder forest	
	N	P	N	P
Precipitation	6.1	3.9	6.4	0.7
Nitrogen fixation	6.5		36.0	
Input surface flow and subsurface flow	40.0	2.5	25.6	1.8
Accumulation in plant biomass	223.0	27.0	204.8	15.1
Litter	*125.4	*12.7	82.0	4.0
Denitrification	19.3		8.5	
Output surface flow and subsurface flow	25.6	1.8	13.2	0.6
Active soil exchange	-89.9	-9.7	-76.5	-9.2
Soil store (kg ha^{-1})	7.9	1.9	19.2	19.6

* above-ground litter

Phosphorus

The P output concentration from the intensively loaded Viiratsi site is not significantly higher than that in Porijõgi, varying from 0.2 to 0.55 mg P l^{-1} and 0.08 to 0.65 mg P l^{-1}, respectively. Nevertheless, the input P values at the border of the arable land and the eutrophic *Elytrigia repens* – grassland in Viiratsi are significantly higher than those on the border of the *Filipendula-Aegopodium* wet meadow and the alder forest in Porijõgi, being 0.6-7.09 and 0.42-1.05 mg P l^{-1}, respectively. The high P concentrations at Viiratsi are caused by slurry applications to the adjacent field upslope. The high rate of P retention in the Viiratsi site can be explained by: (1) uptake by alders and (2) accumulation in the soil.

In Viiratsi, the total P uptake is about 28% less than that in Porijõgi: 10.8 and 15.1 kg P ha^{-1} yr^{-1} (Tables 3 and 2). In the older stand, half the assimilated P is utilised in leaves; in the younger stand one third. The P retranslocation from senescing leaves in autumn is significantly higher than that for N, being about 60% in both stands. The active soil exchange of P in all buffer zones was negative, *i.e.*, the plant uptake in all communities exceeded the annual accumulation in soil, as for N. However, considering the large soil store of P (1.9-2.0 t ha^{-1} in Porijõgi and 1.4-2.7 t ha^{-1} in Viiratsi; Tables 2 and 3), we suggest that in the long-term most retained P is accumulated in the soil.

In the long-term, this very high loading cannot be balanced by Fe, Al and Ca phosphate precipitation. This is, seemingly, the key process in P retention in Viiratsi. Also, some investigations suggest that permanently high N concentration in soils can cause P leaching (Andrusch *et al.*, 1992). On the other hand, our earlier investigations demonstrate that riparian alder forests are effective buffers for P (Mander *et al.*, 1995). Even in riparian wetlands P can be retained due to micro-scale oxygenation variability within the wetland and, probably, due to phosphorus inactivation by nitrate (see Ripl, 1982).

Our results suggest it is important to harvest older *Alnus incana* stands (>20 yr) earlier, due to their decreasing uptake and productivity with age.

Removal efficiency of nitrogen and phosphorus in buffer zones

Removal efficiency E (%) of N and P in riparian communities was estimated as:

$$E = 100\% * (Q_{in}C_{in} - Q_{out}C_{out}) / (Q_{in}C_{in}) \qquad (3)$$

where Q_{in} and Q_{out} = inflow and outflow values (m^3 d^{-1}), respectively; C_{in} and C_{out} = concentration values (mg l^{-1}), respectively.

The retention capacity R (kg ha^{-1} yr^{-1}) was calculated as follows:

$$R = \Sigma(Q_{in}C_{in} - Q_{out}C_{out}) / A \qquad (4)$$

where $\Sigma(Q_{in}C_{in} - Q_{out}C_{out})$ is the annual retention and A is the area of the buffer zone.

The specific removal (% m^{-1}) is defined as the removal efficiency per unit width of a buffer zone. This characteristic is useful for planning and establishing buffer communities.

Table 4. Removal efficiency (%), specific removal (% m^{-1}) and retention (kg ha^{-1} yr^{-1}) of nitrogen and phosphorus in test sites.

	Grassland		Wet meadow		Alder forest		Whole complex	
	N	P	N	P	N	P	N	P
Porijõgi (less polluted)								
Removal efficiency (%)			36	28	48	67	67	81
Specific removal (% m^{-1})			3.3	2.5	2.4	3.4	2.2	2.6
Retention (kg ha^{-1} yr^{-1})			14.4	0.7	12.4	1.2	13.1	1
Viiratsi (heavily polluted)								
Removal efficiency (%)	70	62	43	46	80	80	97	96
Specific removal (% m^{-1})	6.4	5.6	3.6	3.8	2.9	2.9	1.9	1.9
Retention (kg ha^{-1} yr^{-1})	185.5	6	33.8	1.7	36.2	1.6	56.7	2.2

The buffers investigated showed high removal efficiency and retention values (Table 4). The specific removal of N was decreased downslope which coincides with the edge effect reported in earlier papers (Knauer and Mander, 1989). According to values presented in Table 4, the 50-60 m wide complex buffer zone is able to retain and transform most of nitrogen and phosphorus entering the buffer.

CONCLUSIONS

(1) Riparian wet meadows and grey alder forests (*Alnus incana stands*) are effective buffers on stream banks and lake shores, even with very high input loads (276.3 and 12.8 kg ha^{-1} yr^{-1} of N and P, respectively).

(2) Both N and P retention in more complex sequential buffer zones consisting of different biotopes was higher than in simple sequential buffer zones of fewer biotopes: *e.g.* a 31 m wide buffer zone of wet meadow and grey alder forest removed 67% N and 81% P, within the 51 m buffer zone of grassland strip, wet meadow and alder forest 96% N and 97% P was retained.

(3) Grey alder stands provide potential as wood for fuel. From the point of view of both productivity and nutrient retention, the optimal age to harvest is 12-15 years.

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REFERENCES

Andrusch, T., Hupfer, M. and Luther, D. (1992) Chemical and microbial binding forms of phosphorus considering the availability of nitrate in sediment-water systems. Int. Revue ges., *Hydrobiol.*, **77**, 109-120.

APHA (1989) *Standard Methods for the Examination of Water and Waste Water*, 15th edition, American Public Health Organization. 17th edition. Washington.

Balsberg, A.-M. (1982) Plant biomass, primary production and litter disappearance in a *Filipendula ulmaria* meadow ecosystem and the effects of cadmium, *Oikos*, **38**, 72-90.

Bormann, B.T. and Gordon, J.C. (1984) Stand density effects in young red alder plantations: Productivity, photosynthate partitioning and nitrogen fixation. *Ecology*, 65, 394-402.

Duff, J.H. and Triska, F.J. (1990) Denitrification in sediments from the hyporheic zone adjacent to a small forested stream. *Can. J. Fish. Aquat. Sci.*, **47**, 1140-1147.

Freeze, R.A. and Cherry, J.A. (1979) *Groundwater*. Prentice Hall. 588 p.

Groffmann, P.M. and Tiedje, J.M. (1989) Denitrification in north temperate forest soils: Spatial and temporal patterns at the landscape and seasonal scales. *Soil Biol. Biochem.*., **21**, 613-620.

Groffmann, P.M., Axelrod, E.A., Lemunyon, J.L. and Sullivan, W.M. (1991) Denitrification in grass and forest vegetated filter strips. *J. Environ. Qual.*, **20**, 671-674.

Haycock, N. and Pinay, G. (1993) Nitrate retention in grass and poplar vegetated riparian buffer strips during the winter. *J. Environ. Qual.*, **22**, 273-278.

Huss-Danell, K. and Ohlsson, H. (1992) Distribution of biomass and nitrogen among plant parts and soil nitrogen in a young *Alnus incana* stand. *Can. J. Bot.*, **70**, 1545-1549.

Knauer, N. and Mander, Ü. (1989) Untersuchungen über die Filterwirkung verschiedener Saumbiotope an Gewässern in Schleswig-Holstein. 1. Mitteilung: Filterung von Stickstoff und Phosphor. *Zeitschrift für Kulturtechnik und Landentwicklung*, **30**, 365-376.

Leonardson, L., Bengtsson, L., Davidsson, T., Persson, T. and Emanuelsson, U. (1994) Nitrogen retention in artificially flooded meadows. *Ambio*, **23**, 332-341.

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

Lowrance, R.R., Todd, R., Fail, J., Hendrickson, O., Leonard, R. and Asmussen, L. (1984) Riparian forest as nutrient filters in agricultural watersheds. *BioScience*, **34**, 374-377.

Mander, Ü., Kuusemets, V. and Ivask, M. (1995) Nutrient dynamics of riparian ecotones: A case study from the Porijõgi River catchment, Estonia. *Landscape and Urban Planning*, **31**, 333-348.

Milner, C. and Hughes, R.E. (1968) *Methods for the Measurement of the Primary Production of Grassland*. IBP Handbook No. 6. Blackwell Scientific Publications, Oxford. 70 p.

Peterjohn, W.T. and Correll, D.L. (1984) Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology*, **65**, 1466-1475.

Pinay, G., Roques, L. and Fabre, A. (1993) Spatial and temporal patterns of denitrification in a riparian forest. *Journal of Applied Ecology*, **30**, 581-591.
Ripl, W. (1982) Nitrification-denitrification and phosphorus inactivation in lake sediments – a reply. *Vatten*, **38**, 363-365.
Rytter, L. (1989) Distribution of roots and root nodules and biomass allocation in young intensively managed grey alder stands on a peat bog. *Plant and Soil*, **119**, 71-79.
Struwe, S. and Kjrller, A. (1990) Seasonality of denitrification in water-logged alder stands. *Plant and Soil*, **128**, 109-113.
Struwe, S. and Kjrller, A. (1991) Denitrification in wet forest soil systems *in situ* and in slurry experiments. *For. Ecol. Manage.*, **44** (1), 41-52.
Troelstra, S.R., Wagenaar, R. and Smant, W. (1992) Growth of actinorhizal plants as influenced by the form of nitrogen with special reference to *Myrica gale* and *Alnus glutinosa*. *J. Exp. Bot.*, **43**, 1349-1359.
Vought, L.B.-M., Dahl, J., Pedersen, C.L. and Lacoursière, J.O. (1994) Nutrient retention in riparian ecotones. *Ambio*, **23**, 342-348.
Weller, D.E., Correll, D.L. and Jordan, T.E. (1994) Denitrification in riparian forests receiving agricultural discharges. In: Mitsch, W.J. (Ed.) *Global Wetlands: Old World and New*. Elsevier Science B.V. p. 117-131.
Yoshinari, T., Hynes, R. and Knowles, R. (1977) Acetylene inhibition of nitrous oxide reduction and measurement of denitrification and nitrogen fixation in soil. *Soil Biol. Biochem.*, **9**, 177-183.

III

Kuusemets, V., Mander, Ü., Ivask, M. and Lõhmus, K.
Phosphorus retention in riparian buffer zones in agricultural landscapes in Estonia.
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PHOSPHORUS RETENTION IN RIPARIAN BUFFER ZONES IN AGRICULTURAL LANDSCAPES IN ESTONIA

V. Kuusemets[1], Ü. Mander[1], M. Ivask[2] and K. Lõhmus[1]

[1] *Institute of Geography, University of Tartu, 46 Vanemuise St., EE2400 Tartu, Estonia*
[2] *Environmental Protection Institute, Estonian Agricultural University, 5 Kreutzwaldi St., EE2400 Tartu, Estonia*

INTRODUCTION

The role of vegetation in the cycling of nutrients in riparian buffer zones is still uncertain (Correll, 1997). In 1992-97, a study was conducted of nutrient concentrations and transformations in plants, shallow ground water dynamics and herbage biomass production within riparian ecotones of south-eastern Estonian moraine plain, an intensive agricultural area in southern Estonia.

STUDY AREA AND METHODS

In 1994-95, two transects were established on the thalwegs of slopes adjacent to streams, and crossing different riparian plant communities. The transects had similar physico-geographical conditions and similar plant communities, covering field, grass and woodland. The Porijõgi transect is situated in a less polluted area where agricultural activities stopped in 1992. The Viiratsi transect is situated in an area impacted by pig slurry (for area description see Mander *et al.*, 1995). Ground water and soil water samples were collected twice per month from piezometers installed on the borders of plant communities. Nitrogen and phosphorus (P) concentrations in water were analysed in the laboratory of the Estonian Agricultural University using standard methods (see Mander *et al.*, 1997). Above-ground biomass and root samples of herbage were taken from 1 m^2 plots in different plant communities once per year during the period of most intensive flowering of dominant species. The dimension-analysis techniques was used for estimating the above-ground biomass of alder (*Alnus incana*), and roots were sampled by excavation (Mander *et al.*, 1997). The phytomass, calorific, nitrogen and P content in plants were analyzed.

RESULTS

Table 1. Phosphorus variation in soil water, soil and the plant biomass in ecotones of the Porijõgi transect. Field: 150m upslope from the river; cultivated grassland I: 120m upslope from the river (dominated by *Dactylis glomerata*); cultivated grassland II: 50m upslope from the river (dominated by *Dactylis glomerata* and *Alopecurus pratensis*); wet meadow I: 25m upslope from the river (*Aegopodietum*); wet meadow II: 25m upslope from the river (*Filipenduletum*), grey alder stand (A. incana): 5m upslope from the river.

	Field	Cultivated grassland I	Cultivated grassland II	Wet meadow I	Wet meadow II	Alder stand
Total-P in soil water (0.5-2 m; mg l^{-1})	0.489	0.288	0.269	0.088	0.088	0.055
Topsoil (10 cm) P content (mg kg^{-1})		17.1	26.8	17.7	17.7	51.3
Plant biomass production (g m^{-2})	700	1163	1947	1205	2358	1730
P content in grass shoot (%)	0.31	0.27	0.18	0.20	0.20	
P content in grass roots (%)	0.25	0.24	0.22	0.33	0.24	
P assimilation by plants (g P m^{-2})	1.7	2.19	2.94	2.85	4.85	1.51

The results show a significant variation in P concentrations measured in pathways of water movement for the different plant communities. Average total P concentrations in soil water decreased in the riparian zones of the Porijõgi and Viiratsi transects during the study period (June 1994 to December 1995) from 0.489 mg l^{-1} to 0.055 mg l^{-1} and from 0.430 to 0.068 mg l^{-1}, respectively (see Tables 1 and 2).

Grass vegetation plays an important role in P retention in riparian buffer zones. The biomass production of the *Filipendula ulmaria* community was up to 2358 g m^{-2} and accumulation of P was 4.85 g P m^{-2}. This is much higher than the production and P assimilation of the grey alder (*Alnus incana*) community being 1730 g P m^{-2} and 1.51 g P m^{-2}, respectively.

In the Porijõgi test site, the 30m wide buffer zone of wet meadow and grey alder forest removed 50% of nitrogen and 80% of P, while in a 50m buffer zone in Viiratsi containing a grassland strip, a wet meadow and an alder forest, 86% N and 85% P was retained. In the riparian buffers studied, an effective retention of nutrients (34-186 kg N ha^{-1} yr^{-1} and 1.6-6.0 kg P ha^{-1} yr^{-1}) was observed even with very high input loads (276.3 and 12.8 kg ha^{-1} yr^{-1} of N and P, respectively). From the total annual inputs of 82.4 kg N ha^{-1} and 6.4 kg P ha^{-1} into a 30m wide sequential buffer strip of grey alder forest and wet meadow, only 16 and 9%, respectively, was leached into the stream.

Table 2. Phosphorus variation in soil water, soil and plant biomass in ecotones of the Viiratsi transect. Field: 100m upslope the ditch; grassland strip I (*Elytrigia repens*): 45m upslope the ditch; grassland strip II (*Urtica dioica*): 45m upslope the ditch; grey alder stand I (*A. incanae*): 20m upslope the ditch; wet meadow II (*A. incanae*): 5m upslope the ditch.

	Field	Grassland I	Grassland II	Alder stand I	Alder stand II
Total-P in soil water (0.5-1 m; mg l^{-1})	0.430	0.173	0.173	0.052	0.068
Topsoil (10 cm) P content (mg kg^{-1})	29.7	210.1	210.1	123.0	35.0
Plant biomass production (g m^{-2})		1458	1156	1060	1060
Phosphorus content in grass shoot (%)		0.47	0.49		
Phosphorus content in grass roots (%)		0.32	0.49		
Phosphorus assimilation by plants (g P m^{-2})		4.63	5.48	1.08	1.08

REFERENCES

Correll, D.L. 1997. Buffer zones and water quality protection: general principles. pp. 7-20. *In* N.E. Haycock, T.P. Burt, K.W.T. Goulding, and G. Pinay (eds) Buffer Zones, Their Processes and Potential in Water Protection. Proceedings of the International Conference on Buffer Zones, September 1996. Quest Environmental, Herts., UK.

Mander, Ü., V. Kuusemets, and M. Ivask. 1995. Nutrient dynamics of riparian ecotones: A case study from the Porijõgi River catchment, Estonia. Landscape and Urban Planning 31: 333-348.

Mander, Ü., K. Lõhmus, V. Kuusemets and M. Ivask. 1997. The potential role of wet meadows and grey alder forests as buffer zones. pp. 35-46. In N.E. Haycock, T.P. Burt, K.W.T. Goulding, and G. Pinay (eds) Buffer Zones, Their Processes and Potential in Water Protection. Proceedings of the International Conference on Buffer Zones, September 1996. Quest Environmental, Herts., UK.

IV

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Shallow Groundwater Quality and Plant Uptake of Nitrogen and Phosphorus in Complex Riparian Buffer Zones. *Journal of Environmental Quality* (*submitted*).

Shallow Groundwater Quality and Plant Uptake of Nitrogen and Phosphorus in Complex Riparian Buffer Zones

***Valdo Kuusemets*, *Ülo Mander* and *Krista Lõhmus*, Institute of Geography, University of Tartu, 46 Vanemuise St, EE-51014, Tartu, Estonia**
***Mari Ivask*, Institute of Environmental Protection, Estonian University of Agricultural Sciences, 5 Kreutzwaldi St, EE-51014, Tartu, Estonia.**

ABSTRACT

A study of purification efficiency and nutrient assimilation in plants was made in two riparian buffer zones with a complex of wet meadow and grey alder (*Alnus incana*) stand. In the less polluted Porijõgi test site, the 31 m wide buffer zone removed 50 % of total nitrogen (TOTN) and 78 % of total phosphorus (TOTP), while the heavily polluted a 51 m wide buffer zone in Viiratsi retained 87% of TOTN and 84% of TOTP. Load and purification efficiency displayed a significant relationship. The TOTN removal in buffer strips was negative when the input load was less than 0.3 mg L^{-1} and the purification efficiency was always positive when the input load exceeded 5 mg L^{-1}. The purification efficiency of TOTP was positive when the input load exceeded 0.15 mg L^{-1}. Grass vegetation plays an important role in nutrient retention in riparian buffer strips. Biomass production of the *Filipendula ulmaria* community was up to 2358 g m^{-2}, assimilation of N 33.9 and of P 5.3 g m^{-2}, respectively. This is much higher than the production and N and P uptake of the grey alder (*Alnus incana*) community, being 1730, 20.5 and 1.5 g m^{-2}, respectively.

Buffer zones have many functions that improve water quality, protect air and soil, and increase biological and landscape diversity (Mander et al., 1997b). One of the main functions of buffer zones and riparian wetlands, to purify water of substances, has been widely described in many regions (Peterjohn and Correll, 1984; Lowrance et al., 1984; Pinay and Decamps, 1988; Cooper, 1990; Uusi-Kämppä and Yläranta, 1992; Haycock and Pinay, 1993; Gilliam, 1994; Vought et al., 1994). However, our knowledge concerning the water quality buffering effects of riparian zones is far from adequate (Correll, 1997). Different studies indicate that buffer zones can retain 0.0043 to 13 g N m^{-2} day^{-1} and 0.000057 to 8.67 13 g P m^{-2} day^{-1} (see Mander et al., 1997b). Because retention and efficiency rates vary greatly under different climatic and physico-geographical conditions, few proposals have been presented with design criteria for buffer zones and their establishment and management (Dillaha III and Inamdar, 1997, Mander et al., 1997b). The optimal vegetation and the most effective buffer width is still unclear (Correll, 1997). Haycock and Pinay (1993) found that forested buffer zones are more effective than grass buffer zones, while other works show good purification in grass buffer zones (Peterjohn and Correll, 1984, Groffman, 1991; Correll, 1997). Grass strips are considered sediment traps by which a large portion of nutrients, especially P is deposited from surface flow (Mitsch et al., 1979, Peterjohn and Correll, 1984; Yoyng et al., 1980). At the same time grass buffer strips can also remove dissolved nutrients (Peterjohn and Correll, 1984; Haycock and Burt, 1993; Vought et al., 1994) which shows that grass strips are an important part of buffer zones. Although the assimilation of nutrients by herbs in buffer zones has been described in only a few works, results indicate a high nutrient assimilation ability. Van Oorschot (1994) measured above-ground N and P uptake in riparian communities up to 7.1 and 1.07 g m^{-2}, respectively. Prach and Rauch (1992) estimate the N and P removal hay from a floodplain to be 15-30 and 2-4.5 g m^{-2}, respectively.

We provide an overview on the purification efficiency of complex riparian buffer zones in Estonia consisting of native riparian plant communities with emphasis on nutrient assimilation and input versus retention calculations.

METHODS
Site Description

To study buffer zone efficiency we established two transects in south Estonia with similar physico-geographical conditions. The transects are situated on slopes adjacent to streams, following of surface water flow and crossing agricultural fields and different riparian plant communities. The Porijõgi is situated in the plain of south-east Estonia (Varep, 1964). The elevation of the moraine plateau is 30 - 60 m above m.s.l., with undulated relief (slopes are usually 5-6%) and the landscape is dissected by primeval valleys. The Porijõgi transect is located on the slope of a primeval valley where agricultural activities stopped in 1992.

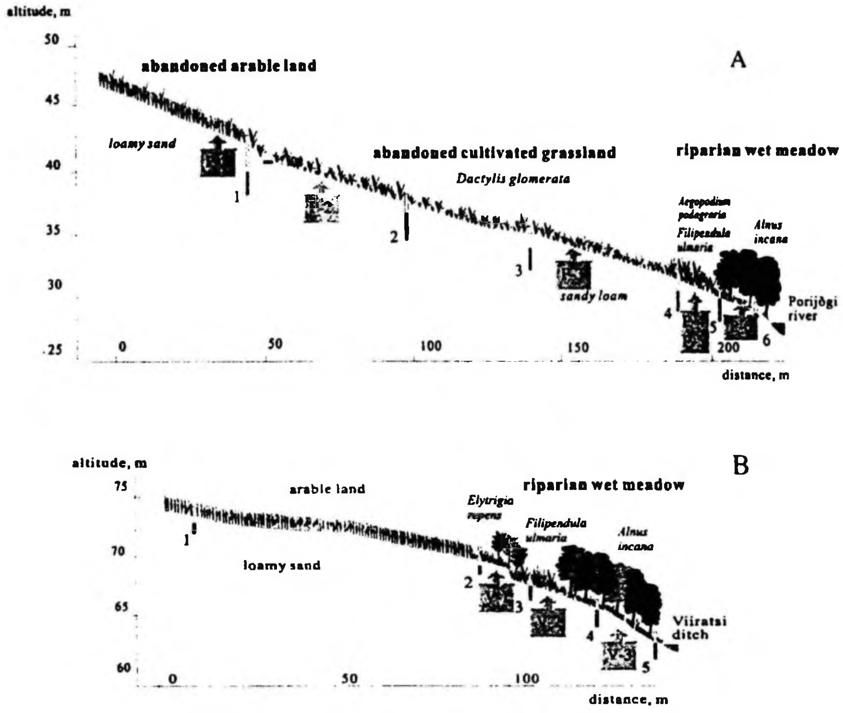

Fig. 1. Study transects in complex riparian buffer zones in South Estonia. A – Porijõgi; B – Viiratsi. 3 – shallow groundwater sampling point; phytomass sampling plots

The transect crosses following plant communities: abandoned field (cultivated last time in 1992) on planosols and podzoluvisols, abandoned cultivated grassland (mowed last time in 1993) on colluvial podzoluvisol (dominated by *Dactylis glomerata* and *Alopecurus pratensis*), an 11 m wide wet grassland on gleysol (two parallel communities, one dominated by *Filipendula ulmaria*, another by *Aegopodium podagraria*), a 20 m wide grey alder stand (*Alnus incana*) on gleysol (Fig. 1A). The Viiratsi transect is situated in the Sakala heights (Varep, 1964) consisting of moraine hills and undulated plains with a variety of glacial deposits. The transect is located on the moraine plain in the vicinity of a pig farm (with about 30 000 pigs during the study). Almost all the slurry from the pig farm is spread on the neighbouring fields and whole area is heavily impacted by pig slurry. The transect crosses following plant: field on planosols and podzoluvisols (slurry was spread in autumn 1994), an 11 m wide grassland (*Elytrigia repens-Urtica dioica*) and young alder (***Alnus incana***) trees strip on colluvial podzoluviol, 12 m wide wet grassland (***Filipendula ulmaria***) on gleysol, 28 m wide grey alder (*Alnus incana*) forest on podzoluvic gleysol (Fig. 1B).

Water Sampling and Analysing

Shallow ground water samples were collected once or twice a month from piezometers installed on the borders of plant communities (with 3 replicates on the borders of riparian communities). Samples were taken from July 1994 to December 1995. Filtered water samples were analysed for NH_4^+-N, NO_2^--N, NO_3^--N, total Kjeldahl-N, PO_4^{3-}-P, TOTP, SO_4^{2-}, Fe, Ca^{2+} in the laboratory of Estonian Agricultural University following standard methods for examination of water and wastewater quality (APHA, 1989).

Phytomass Sampling and Analysing

The phytomass (*i.e.*, standing crop) samples were collected from all riparian plant communities during the maximum flowering time of the dominant plant species (2nd and 3rd week in July; see Milner and Hughes, 1968). Sampling plots (six in Porijõgi and three in Viiratsi) were installed in typical areas of the community. The above ground biomass was collected from three replicate quadrates (1x1 m) in each community. Below ground root biomass was collected from soil cores taken by auger (diameter 158 mm) from a depth of up to 40-50 cm in three replicates from each location. Roots were washed of soil and from dried roots and aboveground biomass dry weight was measured and N and P content was analysed in the laboratory of Estonian Agricultural University. To estimate the aboveground biomass and productivity of grey alder forests, dimension-analysis techniques (Bormann and Gordon, 1984; Rytter, 1989; Huss-Danell and Ohlsson, 1992) were used. At both test sites (age 14 in Porijõgi transect and 40 years in Viiratsi transect) 17 and 5 model trees per plot, respectively, were felled to collect data on the following tree components: stem (wood and bark), secondary branch growth (wood and bark), primary branch growth, leaves, generative organs (Lõhmus et al., 1996). The relative increments of the wood and barks of an overbark fraction were assumed to be equal. Root systems for 6 and 3 out of the sampled 17 and 5 trees respectively were excavated. The dried weight of all tree components was measured and N and P content in dried biomass were analysed in laboratory of Estonian Agricultural University.

Statistical Analysis and Calculations

Kruskal-Wallis test was perfomed due to inhomogeneity of variances to analyse N and P concentration changes between measurement points and the test for Binary Sequences for load versus purification efficiency probability analyses using ***Statgraphics Plus 7.1***. The regression analyse of relation between input load and appearance of negative removal of nutrients was performed by *Microsoft Excel 97*.
Removal efficiency E (%) of N and P in buffer communities was estimated as:

$$E = 100\ \% * (Q_{in}C_{in} - Q_{out}C_{out}) / (Q_{in}C_{in}) \qquad (1)$$

where

Q_{in} and Q_{out} = inflow and outflow values ($m^3\ d^{-1}$), respectively; C_{in} and C_{out} = concentration values (mg L^{-1}), respectively.

RESULTS

The soil water nitrogen load in two transects was different. The TOTN content in Viiratsi transect reached up to 138 mg L^{-1} in the cultivated field after slurry application. In Porijõgi transect the highest TOTN load was 19.5 mg L^{-1}. The results show that there was a considerable decrease in nitrogen content through the Viiratsi transect buffer zone (see Table 1). The average TOTN decreased during the study period (1994 to 1995) from 23 mg L^{-1} in the field (transect point 1, Fig. 1B) to 3.1 mg L^{-1} at the end of the buffer zone (transect point 5, Fig. 1B). However, this decrease was not significant (Kruskal-Wallis test, $P>0.05$). There was considerable decrease in nitrogen through the first 2 m of the buffer zone where the average TOTN decreased from 23 to 14.3 mg L^{-1} in the grass community (transect point 2) and in the following 11 m wide wet grass community to 5.6 mg l^{-1} (in transect point 3). In the alder forest zone the nitrogen content decreased to 3.1 mg N L^{-1}. This change from point 2 is highly significant ($P<0.01$) compared to the values at points 4 and 5. The change was highly significant also when comparing points 3 and 4, and 3 and 5.

The nitrogen load in the Porijõgi transect was lower than in the Viiratsi transect (Table 2). The average TOTN content for study period (1994 to 1995) was 1.4 mg L^{-1} in the abandoned field (sampling point 1, Fig. 1A) which increased to 3.0 mg L^{-1}in the cultivated grassland (sampling point 2). Two meters from the edge of the abandoned cultivated grassland the TOTN content decreased to 1.9 mg L^{-1}. By the end of the buffer zone (sampling point 6) the average TOTN content was 1.5 mg L^{-1}. The only significant decrease ($P<0.05$) was between sampling points 3 to 6.

The phosphorus values were similar in both transects. The highest average TOTP content in Viiratsi transect was 0.43 mg L^{-1} in the field which decreased (not significantly) to 0.17 mg L^{-1} in first buffer zone sampling point (2, Fig. 1B). In grassland-young alder forest strip the average phosphorus content decreased highly significantly ($P<0.001$ in comparison to sampling points 1 and 2) down to 0.05 mg L^{-1}. There was a slight but statistically insignificant increase in the last part of alder forest from 0.05 to 0.07 mg L^{-1} (sampling point 5).

In the Porijõgi transect the average P content was highest in the abandoned arable land (0.49 mg L^{-1}, sampling point 1, Fig. 1A) which decreased to 0.27 mg L^{-1}in the abandoned cultivated grassland. The phosphorus content decreased significantly ($P<0.01$ in comparison with point 3 and $P<0.05$ in comparison with point 2) to 0.06 mg L^{-1}at the beginning of buffer zone (sampling point 4) increased insignificantly in the wet grassland to 0.09 mg L^{-1} and decreased to 0.06 mg L^{-1}in the alder forest.

Plant nutrient uptake in the Viiratsi transect was higher in the wet meadow *Filipendula ulmaria* association (sampling plot V-2, Fig. 1B), where an average of 21.1 g N m^{-2} yr^{-1} and 4.8 g P m^{-2} yr^{-1} was assimilated in grass (Table 1). This was higher than the nutrient uptake in the alder stand (14.0 and 1.1 g m^{-2} yr^{-1}, respectively). In the Porijõgi transect the uptake of nutrients was also highest in the wet meadow where in the *Filipendula ulmaria* association (sampling plot P-5, see Fig. 1A) assimilated 21.3 and 3.0 g m^{-2} yr^{-1} of N and P, respectively. This was higher than annual N and P uptake by the alder forest: 20.5 and 1.5 g m^{-2}, respectively. Nutrient assimilation in 1994 reached 33.9 g N m^{-2} in the Porijõgi transect (sampling plot P-5) and 5.5 g P m^{-2} yr^{-1} in the Viiratsi transect (sampling plot V-2).

The nutrient content in soil was highest in the alder *Alnus incana* community, where the nitrogen content in the Porijõgi topsoil layer (10.7 mg g^{-1}, sampling point 6, see Fig. 1) was comparable with that in Viiratsi (9.9 mg g^{-1}). The phosphorus contents were 1.0 and 5.6 mg g^{-1}, respectively.

Table 1. Nutrient variation in shallow groundwater (average ± standard error), soil and the plant biomass in the complex buffer zone in Viiratsi. Sampling points are given in brackets following Fig. 1B.

	Field (1)	Grassland (2)(V-1)	Wet Meadow (3)(V-2)	Alder Stand I (4)(V-3)	Alder Stand II (5)
TOTN in shallow groundwater (mg L^{-1})	23± 7.7	14.3± 3.6	5.6± 0.4	3.1± 0.5	3.1± 0.7
N uptake by plants (g N m^{-2})		17.5	21.1	14.0	
Topsoil (0-10 cm) N content (mg g^{-1})		2.16	6.76	9.87	
Soil (10-20 cm) N content (mg g^{-1})		1.66	5.37	5.61	
TOTP in shallow groundwater (mg l^{-1})	0.43± 0.28	0.17± 0.04	0.05± 0.005	0.05± 0.004	0.07± 0.01
P uptake by plants (g P m^{-2})		3.7	4.8	1.1	
Topsoil (0-10 cm) P content (mg g^{-1})		0.49	0.50	0.94	
Soil (10-20 cm) P content (mg g^{-1})		0.37	0.40	0.53	
Plant biomass production (g m^{-2})		1320	1015	1060	

Table 2. Nutrient variation in shallow groundwater (average ± standard error, soil and the plant biomass in the complex buffer zone in Porijõgi. Sampling points are given in brackets following Fig. 1A.

	Field (1) (P-1)	Cultivated grassland I (2)(P-2)	Cultivated grassland II (3)(P-3)	Wet Meadow I (4)(P-4)	Wet meadow II (5)(P-5)	Alder stand (6)(P-6)
TOTN in shallow groundwater (mg L^{-1})	1.4± 0.2	3.0± 0.8	3.0± 1.4	1.9± 0.3	2.3± 0.4	1.5± 0.2
N uptake by plants (g N m^{-2})	11.8	15.3	19.5	21.1†	24.1††	20.5
Topsoil (0-10 cm) N content (mg g^{-1})			2.02		2.82	10.74
Soil (10-20 cm) N content (mg g^{-1})			1.56		2.36	9.54
TOTP in shallow groundwater (mg L^{-1})	0.49± 0.3	0.29± 0.01	0.27± 0.15	0.06± 0.02	0.09± 0.03	0.06± 0.005
P uptake by plants (g P m^{-2})	2.9	2.4	3.2	3.0†	3.6††	1.5
Topsoil (0-10 cm) P content (mg g^{-1})			0.45		0.63	1.02
Soil (10-20 cm) P content (mg g^{-1})			0.35		0.53	0.9
Plant biomass production (g m^{-2})	1113	1152	1493	1748	1977	1730

† *Aegopodium podagraria*, †† *Filipendula ulmaria*

DISCUSSION

Both transects show high purification efficiency. The average removal of TOTN and TOTP in Viiratsi was 87% and 84% and in Porijõgi 50% and 78%, respectively. The water quality had already improved within the first meters of the buffer zones. In Viiratsi transect the purification efficiency of TOTN and TOTP within the first 2 m of the buffer grassland was 38% and 60%, respectively, whereas in Porijõgi transect the purification efficiency of TOTN and TOTP the 2 m from the edge of the cultivated grassland was 37% and 78%, respectively.

To analyse the relation between purification efficiency and input load we calculated the purification efficiency separately for each sampling day for every buffer strip between two sampling points. Data were divided by input load concentration into 8 classes with 11 to 36 measurements in each class (Tables 3 and 4). The Tests of Binary Sequences were performed to calculate the level of significance between accurance of positive or negative removal efficiency. The probability of negative removal was calculated by dividing the number of negative removal cases with the total measurement number (%). Comparison of removal efficiency and input load shows that the removal of TOTN was negative (*P<0.01*, Table 3) when the input was less than 1.0 mg L^{-1}. For loads between 1.0 to 5.0 mg L^{-1}, removal of TOTN showed no significant positive or negative tendency, although positive removal was more common - the probability of negative removal is 30.8 to 40.0% (*P>0.05*). For input loads greater than 5.0 mg L^{-1} the purification efficiency is significantly positive (*P<0.01*), for input loads greater than 42 mg L^{-1} purification efficiency is always positive (Fig. 2A, *P<0.001*). The relation between input load and appearance of negative removal of nitrogen (N_{neg}) is described by logarithmic regression:

$$N_{neg} = 63.0 - 16.84 \mathrm{Ln}(I_{Nmax}) \quad (2)$$
$$R^2 = 0.78, \; P<0.001$$

where

I_{Nmax} = maximum value of N input class

Table 3. The probability (Tests Binary Sequences) of TOTN positive or negative removal versus input load in buffer strips.

Input range of TOTN in mg L^{-1}	Probability of negative removal (%)	Level of significance	Number of samples (n)
0.02-0.3	100	**	11
0.3-1.0	82.4	*	17
1.0-1.5	30.8	n.s.	13
1.5-2	40.0	n.s.	15
2.0-3.0	38.9	n.s.	18
3.0-5.0	35.3	n.s.	17
5.0-8.0	10.0	**	20
8.0-70	13.3	**	

* *P< 0.05*; ** *P< 0.0*; n.s. – not significant

Table 4. The probability (Tests Binary Sequences) of TOTP positive or negative removal versus input load in buffer strips.

Input range of TOTN in mg L^{-1}	Probability of negative removal (%)	Level of significance	Number of samples (n)
0.01-0.04	51.4	n.s.	11
0.04-0.05	52.8	n.s.	36
0.05-0.06	23.8	n.s.	13
0.06-0.08	38.9	n.s.	18
0.08-0.10	18.8	n.s.	16
0.10-0.15	26.7	n.s.	15
0.15-0.3	15	**	20
0.3-6	0	**	15

* *P< 0.05*; ** *P< 0.0*; n.s. – not significant

For TOTP there is no interval for negative removal (Table 4). The input value 0.01 to 0.15 mg L^{-1} yields positive or negative removal. Positive removal is prevalent for input from 0.05 to 0.15 mg P L^{-1} – the probability of negative removal is 18.8 to 38.9% ($P>0.05$). For input greater than 0.15 mg L^{-1}, the purification efficiency of TOTP was significantly positive ($P<0.01$), for input loads greater than 2.1 mg L^{-1} purification efficiency is always positive (Fig. 2B, $P<0.01$). The relation between input load and appearance of negative removal of phosphorus (P_{neg}) is described by logarithmic regression:

$$P_{neg} = 7.9 - 10.20\mathrm{Ln}(I_{Pmax}) \quad (3)$$
$$R^2 = 0.72, \ P<0.01$$

where

I_{Pmax} = maximum value of N input class

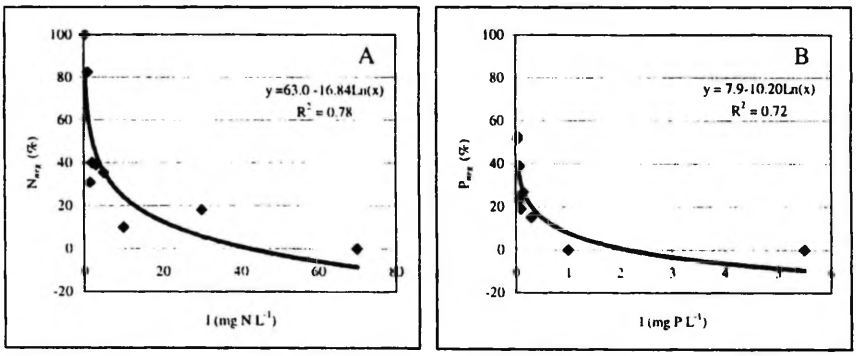

Fig. 2. The relation between input concentration (I; mg L^{-1}) and appearance of negative removal of nitrogen (A) and phosphorus (B), (N_{neg} and P_{neg}; %).

The load-retention relationship in various ecosystems has been discussed in earlier studies (Fleischer et al., 1991; Haycock and Pinay, 1993; Mander et al., 1997b). The results show strong positive correlation between nutrient load and removal. However, buffer zones have upper limits of purification and these regression formula can not been used in the planning in the case of high input values.

This analysis provides limits for buffer strips as water quality purification systems. The input load range (1.0-5.0 and 0.01-0.15 g l^{-1} of N and P, respectively) can be considered to represent natural conditions of buffer strips where water output quality also depends on natural processes taking place in the buffer. This can explain certain increase of TOTN and TOTP inside both studied buffers. However, the complex buffer zone consisting of different ecosystem strips have good purification ability. The average outflow values from both buffer zones were lower than 3.1 and 0.07 mg L^{-1} of TOTN and TOTP in Viiratsi, respectively and 1.5 and 0.06 mg L^{-1} in Porijõgi, respectively. Our estimation on denitrification intensity showed that this process does not play a substantial role in nitrogen removal (7.9-20.1 kg N ha^{-1} yr^{-1} in Porijõgi and 8.5-19.3 kg N ha^{-1} yr^{-1} in Viiratsi, Mander et al., 1997a).

The nutrient uptake in plants indicates, that grass communities play a more important role in nutrient retention than grey alder forest. This gives good opportunity to remove a portion of nutrients by grass mowing and hay cutting while felling of trees can be done with intervals of decades. The average N and P content in herbal shoots in the wet meadows was 11.6 and 1.6 g m^{-2}, respectively in the Porijõgi transect and 10.6 and 2.3 g m^{-2}, respectively in the Viiratsi transect. Cutting should be done during the maximum flowering period of the dominant species when the nutrient content in the shoot biomass is highest (Deinum, 1966). The mowed herbs should be removed after mowing to avoid rapid nutrient loss from hay (see Schaffers, et al, 1998).

The results show that complex buffer zones of grass and forest strips are very effective in N and P retention, which agrees with previous research (Schultz, et. al., 1995; Lowrance, 1991). This kind of complex can be recommended for buffer strip design where grass strips considered as sediment traps but also as important mechanism for dissolved N and P removal. Both features provide the opportunity to remove part of the nutrients from the system. In addition to efficient nutrient purification potential, forest buffer strips have many other environmentally important functions such as protection against soil erosion, filtering polluted air, canopy shading, and increasing biological and landscape diversity.

CONCLUSION

In general, buffer zones can be effective multifunctional tools to control nutrient losses from intensively used watersheds. The studied buffer strips showed good purification efficiency for N and P when input was high (>5 mg N L^{-1} and >0.15 mg P L^{-1}, respectively). At the same time the complex buffer zone that consists of different buffer strips have high purification efficiency. For instance, a heavily loaded complex buffer zone consisting of grass and forest strips showed relatively low output concentrations for TOTN and TOTP that are comparable with the output values from the unloaded transect. However, removal can be negative in case of lower input (<1 mg N L^{-1}). Some unpredictability of output concentration was observed when the initial input was intermediate. When planning and designing buffer zones, the role of grass buffer strips and wet meadows should be considered. Management of grasslands and forests can significantly decrease the load in complex buffer zones.

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REFERENCES

APHA, 1981. Standard Methods for the Examination of Water and Waste Water, 15th edition, American Public Health Organization, Washington.

Bormann, B.T. and J.C. Gordon. 1984. Stand density effects in young red alder plantations: Productivity, photosynthate partioning and nitrogen fixation. Ecology 65: 394-402.

Correll, D.L. 1997. Buffer zones and water quality protection: general principles. p. 7-20. *In* N.E. Haycock et al (ed.) Buffer Zones: Their Processes and Potential in Water Protection. Proceedings of the International Conference on Buffer Zones, September 1996. Quest Environmental, Foundation for Water Research, Oxford.

Cooper, A.B. 1990. Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. Hydrobiologia 202: 13-26.

Deinum, B. 1966. Climate, nitrogen and grass. Mededelingen Landbouwegeschool, Wageningen, 66-11: 1-91

Dillaha, T.A. III and S. P. Inamdar. 1997. Buffer Zones as Sediment Traps or Sources. p. 33-44. *In* N.E. Haycock et al (ed.) Buffer Zones: Their Processes and Potential in Water Protection. Proceedings of the International Conference on Buffer Zones, September 1996. Quest Environmental, Foundation for Water Research, Oxford.

Fleisher, S., L.Stibe, and L.Leonardson. 1991. Restoration of wetlands as a means of reducing nitrogen transport to coastal waters. Ambio, 20: 271-272.

Gilliam, J.W. 1994. Riparian wetlands and water quality. J. Environ.Qual. 23: 896-900.

Groffman, P.M., E.A. Axelrod, J.L. Lemunyon, and W.M. Sullivan. 1991. Denitrification in grass and forest vegetated filter strips. J. Environ. Qual. 20: 671-674.

Haycock, N.E and T.P. Burt.1993. The sensitivity of rivers to nitrate leaching. The effectiveness of near-stream land as a nutrient retention zone. In: Thomas, D.S.G. and Allison, R.J. (eds.). Landscape Sensitivity, Wiley, London, pp. 261-272.

Haycock, N.E. and G. Pinay. 1993. Groundwater nitrate dynamics in grass and poplar vegetated riparian buffer strips during the winter. J. Environ. Qual. 22: 273-278.

Huss-Danell, K. and H. Ohlsson. 1992. Distribution of biomass and nitrogen among plant parts and soil nitrogen in a young *Alnus incana* stand. Can. J. Bot. 70: 1545-1549.

Jordan, T.E., D.L. Correll and D.E. Weller. 1993. Nutrient interception by riparian forest receiving inputs from adjacent cropland. J. of Envir. Qual. 22(3): 467-473.

Lowrance, R.R., R.Todd, J.Fail, O.Hendrickson, R.Leonard, and L.Asmussen. 1984. Riparian forest as nutrient filters in agricultural watersheds. BioScience, 34: 374-377.

Lowrance, R. 1991. Effects of buffer systems on the movement of to N and P from agriculture to streams. In: NPo-forskning fra Miljøstyrelsen. Conference Contributions. International Conference on N, P and Organic matter. Helsingør, Denmark.

Lõhmus, K., Ü.Mander, H.Tullus and K.Keedus. 1996. Productivity, buffering capacity and recourced of grey alder forests in Estonia. p 95-105. *In:* Short Rotation Willow Coppice for Renewable and Improved Environment. Swedish University of Agricultural Sciences, Uppsala.

Mander, Ü., K.Lõhmus, V.Kuusemets, and M.Ivask. 1997a. The potential role of wet meadows and grey alder forests as buffer zones. p. 35-46. *In* N.E. Haycock et al (ed.) Buffer Zones: Their Processes and Potential in Water Protection. Proceedings of the International Conference on Buffer Zones, September 1996. Quest Environmental, Foundation for Water Research, Oxford.

Mander, Ü., V.Kuusemets, K.Lõhmus and T.Mauring. 1997b. Efficiency and dimensioning of riparian buffer zones in agricultural catchments. Ecological Engineering 8: 299-324.

Milner, C. and R.E. Hughes. 1968. Methods for the Measurement of the Primary Production of Grassland. p. 70 *In*: IBP Handbook No. 6. Blackwell Scientific Publications, Oxford.

Mitsch, J.W., C.L.Dorge and J.R.Wiemhoff. 1979. Ecosystem Dynamics and a Phosphorus Budget of an Alluvial Cypress Swamp in Southern Illinois. Ecology 60(6):1116-1124.

van Oorschot, M.M.P. 1994. Plant production, nutrient uptake and mineralization in river marginal wetlands: the impact of nutrient additions due to former land-use. p. 133-150. *In* W.J. Mitsch (ed). Global Wetlands: Old World and New. Elsevier Science B.V., Amsterdam.

Peterjohn, W.T. and D.L.Correll. 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 65: 1466-1475.

Pinay, G., and H. Decamps. 1988. The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: a conceptual model. Regulated Rivers: Research and Management 2: 506-516.

Prach, K. and O. Rauch. 1992. On filter effects of ecotones. Ekologia (CSFR) vol. 11, No. 3: 293-298.

Rytter, L. 1982. Distribution of roots and root nodules and biomass allocation in young intensively managed grey alder stands on a peat bog. Plant and Soil 128: 363-365.

Schaffers, A.P., M.C. Vesseur, and K.V.Sykora. 1998. Effects of delayed hay removal on the nutrient balance of roadside plant communities. J. of Appl. Ecol.: 349-364.

Schultz, R.C., J.P. Colletti, T.M. Isenhart, W.W. Simpkins. C.W. Mize, and M.L. Thompson. 1995. Design and placement of a multi-species riparian buffer strip system. Agroforestry Systems, 29: 201-226.

Uusi-Kämppä, J. and T.Yläranta. 1992. Reduction of sediment, phosphorus and nitrogen transport on vegetated buffer strips. Agric. Sci. Finl. 1: 569-574.

Varep, E. 1964. The landscape regions of Estonia. Publications on Geography IV. Acta et Commerstationes Universitatis Tartuensis, 156: 3-28.

Vought, L.B.-M., J.Dahl, C.L.Pedersen and J.O.Lacoursiere. 1994. Nutrient Retention in Riparian Ecotones. Ambio 6: 342-348.

Young, R.A., T.Huntrods and W.Anderson. 1980. Effectiveness of vegetative buffer strips in controlling pollution from feedlot runoff. J. Environ. Qual. 9: 483-487.

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Efficiency and dimensioning of riparian buffer zones in agricultural catchments

Ülo Mander *, Valdo Kuusemets, Krista Lõhmus, Tõnu Mauring

Institute of Geography, University of Tartu, 46 Vanemuise, EE-2400 Tartu, Estonia

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Abstract

A strong linear correlation was found between the log-transformed load and retention of nitrogen and phosphorus in riparian buffer zones ($r = 0.99$ and 0.997, respectively). Analyses of N and P budgets in four riparian forests of varying age (two grey alder stands in Estonia and two riparian deciduous forests in USA) show a significant efficiency. Despite the different input load (72.9–110.4 kg N ha^{-1} year^{-1} and 2.5–3.0 kg P ha^{-1} year^{-1}), the outputs into streams from the alder stands systems were comparably low (9.0–13.2 and 0.38–0.62 kg ha^{-1} year^{-1}). The older forests from the USA showed less efficiency. Plant uptake of both N and P in younger stands was significantly higher than in older forests. Methods to determine the buffer zones' and buffer strips' width and their efficiency are presented. The testing of efficiency assessment in a watershed in Estonia demonstrated an expected efficiency of buffers. © 1997 Elsevier Science B.V.

Keywords: Nitrogen; Phosphorus; Load; Retention; Buffer zones; Buffer strips; Riparian forests

* Corresponding author. Tel.: +372 7 465819; fax: +372 7 465825; e-mail: mander@math.ut.ee

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1. Introduction

1.1. Ecological network of the territory

Riparian ecosystems are essential multifunctional elements of the worldwide ecological network (Bishoff and Jongman, 1993; The EECONET Declaration, 1993; Baldock et al., 1993). The same is valid for Estonia, where stream and ditch banks covered with grey alder (*Alnus incana* (L.) Moench.) and willow species (*Salix* spp.) are typical of rural landscapes and the ecological network during the last century. However, with the beginning of intensive amelioration and land reclamation in 1960s, the traditional mosaic of rural landscapes has changed significantly. Many grasslands, small wetlands, woodlots, trees rows, stone fences and other ecologically important landscape elements were liquidated in order to enlarge and homogenize agricultural fields. Channels and ditches were usually constructed without forest/bush vegetation. Despite the dominating mentality of Soviet 'agroindustrial' development, which did not pay attention to ecological objectives, the first experimental reaches of ditches with forest buffers were re-established in the 1970s to demonstrate the economical benefit of such ecological engineering measures (Mander, 1985).

1.2. Main functions of riparian buffers

Riparian biotopes have the following essential functions: (1) filtering of polluted overland and subsurface flow from intensively managed adjacent agricultural fields; (2) protecting banks of water bodies against erosion; (3) filtering polluted air, especially from local sources (e.g. large farm complexes, agrochemically treated fields); (4) avoiding intensive growth of aquatic macrophytes by canopy shading; (5) improving the microclimate in adjacent fields; (6) creating new habitats in land/inland water ecotones; and (7) creating more connectivity in landscapes due to migration corridors and stepping stones.

1.3. Filtering of polluted overland and subsurface flow from intensively managed adjacent agricultural fields

Several case studies worldwide suggest that different riparian ecosystems can significantly decrease the nitrogen and phosphorus concentration in both overland flow and groundwater (Peterjohn and Correll, 1984; Pinay and Decamps, 1988; Knauer and Mander, 1989; Jordan et al., 1992; Osborne et al., 1993; Vought et al., 1994).

Three biological processes can remove nitrogen: (1) uptake and storage in vegetation; (2) microbial immobilization and storage in the soil as organic nitrogen; and (3) microbial conversion to gaseous forms of nitrogen (denitrification; Weller et al., 1994). Table 1 shows the variability of the intensity of the most important processes relevant to nitrogen removal in riparian ecosystems, including those which control the transformation and fluxes of nitrogen within ecosystems (e.g., Table

Intensity of processes relevant to nitrogen removal in riparian buffer ecosystems

Process	Intensity rate (kg N ha^{-1} $year^{-1}$)	Ecosystem	Source
N mineralization	24–400	Temperate hardwoods in riparian zone	Melillo, 1981 (cited after Bowden, 1986a)
	3.7–456[a]	Pine-oak forest, swamp forest	Zak and Grigal, 1991
	46–113[b]	Riparian fen and bog	Verhoeven et al., 1990
	1.1-219[b]	Old-field site (potential buffer zone)	Robertson et al., 1988
Nitrification	38–80	Oak-beech forest	Tietema et al., 1987
	<1–100[c]	Freshwater wetlands	Bowden, 1987
	13–29[a]	Pine-oak forest, swamp forest	Zak and Grigal, 1991
	0.73–183[b]	Old-field site (potential buffer zone)	Robertson et al., 1988
	1580–1930[c]	Tidal freshwater marsh	Bowden, 1986b
Denitrification	0.16–475	Riparian deciduous forests	Weller et al., 1994; Fustec et al., 1991
	20–1600	Riparian meadows	Brüsch and Nilsson, 1993
	114–2880[d]	Grassland buffer strip	Groffmann et al., 1991
	<0.04–2895[a]	Riparian forests	Ambus and Christiansen, 1991
	2.6–2960[e]	Riparian meadow	Cooper, 1990
	1400–23 650[c]	Riparian successional forest	Klingensmith and Van Cleve, 1993
Vegetation uptake	30–220	Riparian flooded meadows	Leonardson et al., 1994
	34–90	Experimental grass and forest buffer strips	Uusi-Kämppä and Yläranta, 1996
	<10–350[c]	Freshwater wetlands	Bowden, 1987
	150–300[c]	Riparian meadow	Prach and Rauch, 1992
N fixation	5–10	Temperate hardwoods	Melillo, 1981 (cited after Bowden, 1986a)
	30–164	Riparian alder stands	Klingensmith and Van Cleve, 1993
Dissimilative reduction of nitrate to ammonium	0.02–1.2[f]	Riparian fen	Ambus et al., 1992
	5[c]	Tidal freshwater marsh	Bowden, 1986b and Bowden, 1987
Ammonia volatilization	30–56	Riparian forests	Kim, 1973 (cited after Bowden, 1986a)
	3.5–16.5	Riparian meadow	Woodmansee, 1978 (cited after Bowden, 1986a)
	95–114	Riparian grazed sheep pasture	Denmead et al., 1976 (cited after Bowden, 1986a)

[a] mg N m^{-2} day^{-1}.
[b] μg N cm^{-2} day^{-1}.
[c] g N m^{-2} $year^{-1}$.
[d] g N ha^{-1} day^{-1}.
[e] mg N m^{-2} h^{-1}.
[f] μg N m^{-2} h^{-1}.

mineralization, nitrification, nitrogen fixation, and dissimilative reduction of nitrate to ammonium). Table 1 shows that denitrification varies greatly (< 1–1600 kg ha^{-1} year^{-1}) but vegetation uptake, especially in riparian meadows, also displays considerable variation (< 10–350 kg ha^{-1} year^{-1}). Thus, different processes can play a leading role in nitrogen removal.

Phosphorus, in contrast, can be released from the wetland soils of riparian zones (Richardson and Marshall, 1986; Vanek, 1991). Table 2 presents the most important processes controlling phosphorus retention in riparian buffers.

Storage of phosphorus in riparian buffer zones depends on the following processes: (1) soil adsorption; (2) removal of dissolved inorganic phosphorus by plant uptake; (3) microbial uptake; and, in the case of peatlands, (4) incorporation of organic phosphorus into peat (Richardson, 1985). In absolute terms, soil adsorption and vegetation uptake are on a comparable level, varying from 0.1 to 236 and from 0.2 to 50 kg P ha^{-1} year^{-1}, respectively (Table 2). An interesting mechanism to control phosphorus storage in the soil and sediments is the inactivation by nitrates. Andersen (1982) demonstrated that oxidized nitrogen can buffer the redox potential of lake sediments and immobilize the phosphorus up to 16 kg P ha^{-1} year^{-1}. The same mechanism can possibly play an important role in buffer zones.

1.4. Riparian buffer zones and buffer strips

In some countries the complex structure of buffer zones is officially recommended or legislatively stated. In the USA the recommended complex buffer zone consists of three parts which are perpendicular to the stream bank or lake shore (sequentially from agricultural field to the water body): a grass strip, a young (managed) forest strip and an old (unmanaged) forest strip (Lowrance, 1991). In agricultural areas of Estonia, the preferable land-use alternative was a perennial grassland (buffer zone) in combination with a forest or bush buffer strip directly on river banks or lake shores (Mander, 1989). New laws on nature conservation and environment protection in Estonia stress the importance of riparian buffer zones and strips, and provides jurisdiction for buffer zones of protected areas and all natural/seminatural ecosystems outside protected areas. For instance, the Act on Protection of Coastal Areas (1994) and the Act on Protection Surface Waters (1994) state that all building activities are prohibited within 200 m from the coastal line and lake shores.

Although the buffering efficiency of vegetated riparian buffers has been reported by many authors for several ecosystems, there are few examples in the literature about the dimensioning of buffer zones and strips. The first well known examples come from the USA, where methods for the determination the parameters of buffer zones adjacent to feedlots and manure land treatment sites (Doyle et al., 1977; Overcash et al., 1981). Phillips (1989) presented calculations to evaluate the buffering efficiency of riparian forests along a coastal plain river.

However, calculation methods usable for landscape planning and stream restoration purposes are not much developed.

In this paper some methods to determine the buffer zones' and buffer strips' width and their efficiency are presented and results of efficiency assessment in a watershed in Estonia are demonstrated.

Table 2
Intensity of processes relevant to phosphorus removal in riparian buffer ecosystems

Process	Intensity rate (kg P ha^{-1} year^{-1})	Ecosystem	Source
Adsorption in soil and sediments	1.72–7.3	Floodplain wetland	Yarbro, 1979 (cited after Mitsch and Gosselink, 1993)
	15–38	Riparian fen	Richardson and Marshall, 1986
	36	Alluvial cypress swamp	Mitsch et al., 1979
	236[a]	Sawgrass tidal marsh	Hsieh, 1988
	4.1–28.6[a,d]	Constructed riparian wetlands	Mitsch et al., 1995
Sedimentation	5.9–130[a]	Constructed riparian wetlands	Fennessy et al., 1994
Incorporation of organic P into peat	0.05–2.4	Temperate wetlands	Richardson, 1985
Vegetation uptake	16.6–50.1	Riparian fen	Richardson and Marshall, 1986
	5–8	Experimental grass and forest buffer strips	Uusi-Kämppä and Yläranta, 1996
	200–4500[a]	Riparian meadow	Prach and Rauch, 1992
	185[a]	Sawgrass tidal marsh	Hsieh, 1988
	8.7	Alluvial cypress swamp	Mitsch et al., 1979
	2.6–16.6[b]	Constructed riparian wetlands	Mitsch, 1992
Microbial uptake	2.3–14.4	Riparian fen	Richardson and Marshall, 1986
P inactivation by NO_3^-	0.14–16[f]	Eutrophic lake sediments	Andersen, 1982
Release	0.13–0.30	Aspen-birch forest	Timmons et al., 1977
	0.46	Riparian forested bog	Verry and Timmons, 1982
	26–42	Riparian fen	Richardson and Marshall, 1986
	7.3–1044[c]	Riparian forested wetland	Clausen and Johnson, 1990

[a] g P m^{-2} year^{-1}.
[b] mg P m^{-2} week^{-1}.
[c] kg P ha^{-1} day^{-1}.
[d] Retention mostly based on adsorption in sediments.
[e] Calculated after the depth and bulk density of sediments.

2. Material and methods

2.1. Multi-site analysis

To analyze the influence of pollution load on nutrient retention in riparian ecosystems, about 50 literature sources of international distribution were compared. In addition, the results of some Estonian studies were taken into consideration. The following types of buffer ecosystems were analyzed: (1) riparian forest buffer strips (in some cases wetland forests), (2) shelterbelts, (3) hedges, (4) riparian grassland buffer strips, (5) grassy slopes for wastewater treatment (e.g., from feedlots), (6) grassland strips in strip-cropping. The majority of chemical budget studies was available only for the spring and vegetation period. Only a few studies were made over a whole year. Nonetheless, the average values of collected data were used for this analysis. The nutrient input and output values of ecosystems were calculated by multiplying the nutrient concentration and water discharge, averaged over the study period, and dividing by the area. The input of nutrients into the system is defined as the nutrient (pollution) load (in g m^{-2} day^{-1}).

The nutrient retention capacity in g m^{-2} day^{-1} was calculated as follows:

$$R = \Sigma(Q_{in}C_{in} - Q_{out}C_{out})/A \quad (1)$$

where $\Sigma(Q_{in}C_{in} - Q_{out}C_{out})$ is the daily retention, A is the area of the buffer zone, Q_{in} and Q_{out} are the inflow and outflow values (m^3 day^{-1}), respectively, C_{in}, and C_{out} are the concentration values (mg l^{-1}), respectively.

Removal efficiency E (%) of N and P in riparian communities was estimated as:

$$E = 100^*(Q_{in}C_{in} - Q_{out}C_{out})/(Q_{in}C_{in}) \quad (2)$$

Statgraphics 7.1 was used for statistical analysis.

2.2. Estonian riparian forest comparisons

To analyze buffering efficiency in riparian forests, two riparian buffers with grey alder stands of different ages were selected in southern Estonia: a 20 m strip with a 14-year-old stand in the Porijõgi River catchment and a 28 m strip (40 year) in Viiratsi, Viljandi County (a territory of a large pig farm). In both areas, landscape transects in thalwegs were established along a topoedaphic gradient. Groundwater wells, overland flow samplers, precipitation gauges, litter traps, and study plots to analyze various plant and soil parameters were established on both the upper, middle and lower parts of the grey alder stands. The main nitrogen and phosphorus flows and budgets have been assessed and the methods are described more precisely in earlier papers (Mander et al., 1994, 1997).

Data on nutrient flows in two riparian forests in USA (Peterjohn and Correll, 1984; Lowrance et al., 1984; Fail et al., 1986) are used to compare their buffering efficiency with Estonian grey alder stands.

2.3. Dimensioning of riparian buffers

Calculations presented in this paper are based on hydrological models which depend on capacity of buffer strips to infiltrate overland flow, and are supplemented with parameters significant for absorption and cation exchange capacity of soils. Basically, this method is comparable to the widely used universal soil loss equation (USLE; see Meyer and Wischmeier, 1969). However, the length-slope factor in this equation, which is a purely empirical relationship, does not account for changes in either surface flow or erosion processes (Moore and Burch, 1986). Therefore, a specific slope length factor has been induced into our equation. It depends on overland flow concentration in lower parts of the relief, e.g., in thalwegs.

The width of riparian buffer strips depends on soil and relief conditions of the adjacent landscape, and ranges normally between 5 and 50 m. It can be determined on the base of maps of reclaimed areas in 1:2000 scale with detailed topographic and soil data (Fig. 1) using the following formula (Mander, 1995):

$$P = t\, q\, f i^{1/2} / (m\, K_i\, n) \qquad (3)$$

where P is the optimal width of forest/bush buffer strip (m); t is the time variation coefficient (from days to min; $t = 0.00069$); q is the mean intensity of overland flow during the thawing period (mm day^{-1}, for Estonia $q = 8.4$); f is the specific slope length (m); i is the mean slope in the catchment ($i = \tan\alpha$); m is the roughness coefficient of the surface in the catchment (mean value for plowed fields, 1.0; for intensively managed grasslands, 1.1; and for natural meadows, 1.2); K_i is the water infiltration within the buffer strip during the spring (mm min^{-1}, mean value over different soil types varies normally between 0.1 and 1.0); and n is the soil adsorption capacity in the buffer strip.

In heterogeneous topographic areas (e.g. moraine-hilly landscapes), the specific slope length f is defined as:

$$f = F/l \qquad (4)$$

where F is the elementary catchment area of a gully (m^2); and l is the width of a gully immediately on the bank of a stream or on the lake shore (m). This approach applies in a complex glacial relief or when buffer strips discharge to receiving waters through a gully. For homogeneous slopes, the f value is calculated as the distance (m) from the watershed border to the stream bank (lake shore).

The soil adsorption capacity n is calculated as:

$$n = \ln Is_X / \ln Is_{\text{coarse sand}} \qquad (5)$$

where $\ln Is_x$ is the specific area of the investigated soil type (m^2 g^{-1}); and $\ln Is_{\text{coarse}}$ sand is the specific area of the coarse sand (m^2 g^{-1}).

Mean value of the integrated soil parameter $k (k = K_i * n)$ for main soil types is: coarse sand, 1.00; fine sand, 0.80; loamy sand, 0.61; sandy loam, 0.53; sandy clay loam and loam, 0.43; clay loam and sandy clay, 0.33; and clay, 0.21.

Fig. 1 presents a fragment of a 1:2000 map with isohypses and soil investigation data that was used for the dimensioning of the width of riparian buffer strips using the formulae presented above.

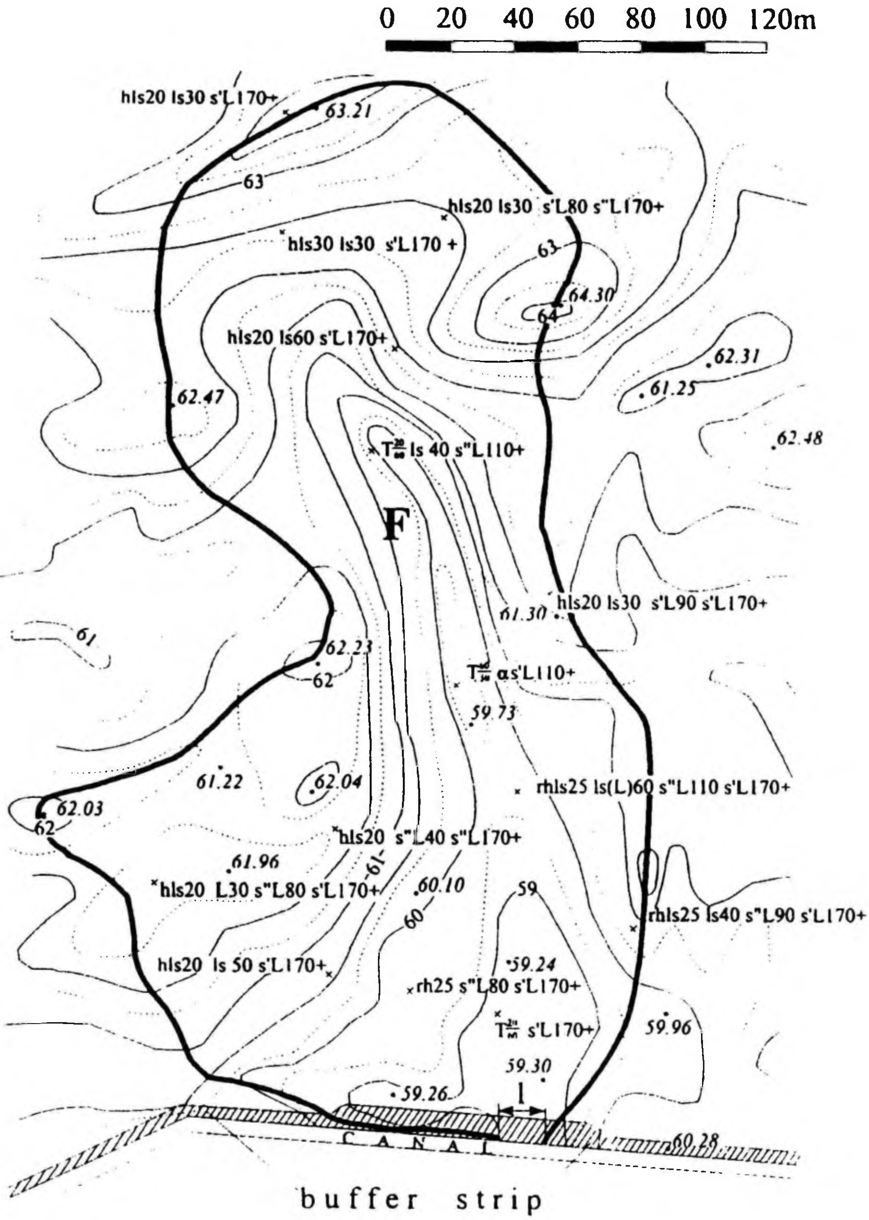

Fig. 1. A portion of a topographical map with elevations (m, asl), isohypses and soil data (horizons' depth in cm; h, humus; r, gravel; ls, loamy sand; s'L, light sandy loam; s"L, middle sandy loam; L, loam; T, peat with numbers of depth and mineralization rate), presenting an elementary watershed (used as arable land) in an agricultural landscape. The optimal recommended width (P) of the buffer strip is calculated using the Eq. (3). F is the area of the elementary watershed, l is the width of the watershed 'mouth' on the bank of a channel (m). Eq. (4) gives the specific slope length (f) of the elementary watershed. The parameters i, m, K_i, n, and k (Eq. (3) and Eq. (5)) are determined as mean values using the soil, land-use and topographic data of the map.

In addition to detail soil and topographical maps, we used soil and hydrogeological data from the Department of Soil Science and Department of Land Reclamation, both Estonian Agricultural University (Mander, 1985, 1989).

2.4. *Efficiency assessment of designed buffer zones and buffer strips*

The methods described in this paper served as a basis for designing of buffer zones and buffer strips in several river catchments in Estonia. To assess the efficiency of designed buffer zones and buffer strips, a simplified assessment method, based on catchments, has been developed (Mander, 1995). The method depends on the use of soil and topographic land use maps (from 1:2000 to 1:10 000) normally available for detail planning. Buffer zones are defined as perennial (extensively used) cultivated grasslands along the watercourses (normally, 50–200 m wide). Buffer strips are forest or bushes (5–50 m wide) growing directly on the stream banks. In simplified presentation, the formulae are as following:

$$E_a = (Y_{BZ} + Y_{BS})/(X_C A_C\ 100\%) \tag{6}$$

where E_a is the assessed summary buffering efficiency of designed buffer zones and strips (%), Y_{BZ} is the buffering efficiency of designed buffer zones (kg year^{-1}), Y_{BS} is the buffering efficiency of designed buffer strips (kg year^{-1}), X_C is the mean annual nutrient runoff from the catchment (kg ha^{-1} year^{-1}) and A_C is the area of the catchment (ha).

$$Y_{BZ} = X_C\, A_{BZ}\, (1 - \delta) \tag{7}$$

where A_{BZ} is the area of designed buffer zone (former arable land converted into perennial grassland; ha), δ coefficient showing the average decrease of nutrient runoff from perennial grasslands when compared to arable land ($\delta_{\text{arable land}} = 1.0$); this value can vary in a large range; in the case study catchments presented in this paper, the δ values for N and P have taken 0.5 and 0.7, respectively (Mander, 1995).

$$Y_{BS} = X_C\, K(e/100\%) \sum_{i=1}^{n} (m_i\, Am_i) \tag{8}$$

where K is the coefficient showing the share of nutrients and organic matter transported with overland flow; in the case of studied catchments, the K value for N and P is 0.5 and 0.8, respectively; e is the average buffering effect of forest/bush buffer strips (% m^{-1}); in a buffer strip of 1 m width, the e value for N is 3 and for P 4%; for buffer strips wider than 10 m, the nonlinear retention function must be used; m_i is the width of a designed buffer strip (m); Am_i is the area of the elementary catchment of a designed buffer strip of mi width (i.e. the area from which overland flow entering the buffer strip will be formed; ha); the value of Am_i will be derived from the topographic land use maps.

The most problematic part of the calculations is the estimation of the mean annual value of nutrient runoff from catchments (X_C). This approach is rather simplified and based on the mean annual fertilization intensity and soil structure of the catchment. However, in some case study catchments the calculated N and P runoff values fit well to measured values (Mander et al., 1989). In addition, if the aim is only to estimate buffering efficiency of designed buffer zones and strips (E_a), the X_C value will be eliminated (Eqs. (6–8)).

This method to estimate the buffer zones' and buffer strips' efficiency was tested in the Porijõgi River catchment, Estonia (Mander, 1995). Two sub-catchments, the Sipe stream and the Vanda ditch sub-catchments, of the whole watershed have been chosen for the efficiency assessment. Both of them are relatively small, well calibrated, and have been intensively cultivated. Likewise, there are problems with groundwater pollution by nitrates in both areas (Mander et al., 1994). Especially, the Vända ditch sub-catchment has been used as the experimental polygon for the implementation of stream restoration and ecological engineering measures.

3. Results and discussion

3.1. Load-retention relationship

The data of nitrogen load and hydraulic load in buffer strips show predominately low values. From 26 published cases of nitrogen retention in buffer strips, the nitrogen load was lower than 2 g m^{-2} day^{-1} in 19 cases, and the hydraulic load below 100 mm day^{-1} in 21 cases (Table 3). For phosphorus the data distribution is more symmetrical. However, of 21 studies, the hydraulic load was below 100 mm day^{-1} in 12, but in nine cases the phosphorus load was lower than 0.01 mg P m^{-2} day^{-1}, and in ten cases between 0.01 and 1.5 mg P m^{-2} day^{-1}. Due to the skewed data distribution, the regression analysis was made with logarithmic values of load and retention.

The intensity of retention processes in forest and grassland buffer strips is different when compared with wetlands. Buffer strips studied were mostly in mineral soils and water residence time is lower than in wetlands. The high retention efficiency of buffer strips depends mainly on the heterogeneity of the loading events, i.e., the best results occur when the polluted water from adjacent fields enters buffers in short events (e.g., during intensive rain-falls and/or intensive thaw). This phenomenon has been documented in several studies on natural buffer strips (Knauer and Mander, 1989), and has also been demonstrated in experimental plots (Yousef et al., 1987; Dillaha et al., 1988; Magette et al., 1989). In some research and experiments on transects through different buffer strips, both overland and subsurface flow was analyzed, but most research has dealt only with overland flow.

The regression between logarithmic values of nitrogen load and removal in buffer strips is linear (Fig. 2(A))

$$y = -0.194 + 0.948x \ (r = 0.99, n = 26) \tag{9}$$

There are relevant differences between the removal of different nitrogen forms in buffer strips. Typically, the proportion of organic nitrogen increases with low nitrogen inputs in overland flow and soil water passing through the buffer strip (Peterjohn and Correll 1984). At the same time the concentration of total nitrogen

Table 3
Nitrogen and phosphorus budgets in riparian buffer ecosystems

References	Country	Buffer ecosystem	Hydraulic load (mm day^{-1})	Load (g m^{-2} day^{-1})	Retention capacity (g m^{-2} day^{-1})	Removal efficiency (%)	Area (m^2)	N and P form	Time	Comments
Abernathy et al., 1985	South Carolina, USA	Cultivated grassland on slope as wastewater application site	33	6.14	4.57	74	1687	TKN	1 year	Raw sewage
			33	7.65	3.77	49	220	—		Lagoon effluent
Bingham et al., 1980	North Carolina, USA	Cultivated grassland buffer-strip on slope	49	2.99	2.13	71	156	TKN	2 years	Dairy manure application site
Brüsch and Nilsson, 1993	Denmark	Riparian meadow (wetland) of Rabis stream	22	0.19	0.11	58	1600	NO_3-N	2 years	Denitrification study in situ
Dillaha et al., 1988	Virginia, USA	Grassland buffer strip on slope, Chesapeake Bay catchment area	150	6.47 2.29	4.0 1.26	62 52	50	TKN Total-P	1 year, summer	Feedlot runoff 9.1 m wide strip, (overland flow)
Hoffmann, 1991	Denmark	Flooded riparian wetland	49	0.187	0.135	72	1370	Total-N	1.5 years	Groundwater and overland flow

Table 3 (continued)

References	Country	Buffer ecosystem	Hydraulic load (mm day^{-1})	Load (g m^{-2} day^{-1})	Retention capacity (g m^{-2} day^{-1})	Removal efficiency (%)	Area (m^2)	N and P form	Time	Comments
Jacobs and Gilliam, 1985	North Carolina, USA	Riparian forest, Middle Coastal Plain	0.08	0.624	0.616	99	1000	NO_3-N	3 years	Surface and subsurface flow from arable land
Knauer and Mander, 1989 (partially unpublished data)	Schleswig-Holstein, Germany	Alder wood (*Alnus glutinosa*), riparian buffer strip	103	0.62 0.053	0.59 0.047	95 89	540	Total-N Total-P	1 year, summer and spring	Overland flow from agricultural upland
			0.9	0.0084 0.00017	0.0070 0.00017	83 100	540	Total-N Total-P	—	Subsurface flow from agricultural upland
		Floodplain wetland (*Cirsio-Polygonetum*) at a small lake	62	0.28 0.0465	0.27 0.0372	99 80	650	Total-N Total-P	—	Overland flow from intensively managed pasture on steep slope
		Natural grassland buffer strip	19 0.35	0.15 0.0027	0.145 0.0022	97 83	660 660	Total-N	—	Overland flow from arable land subsurface flow

Lowrance et al., 1984	Georgia, USA	Little River floodplain hardwood riparian forest	0.79	0.11	0.09	84	3400	Total-N	1 year	Non-point pollution sources
Magette et al., 1989	Maryland, USA	Cultivated grassland	20	27.2 20.0	13.0 8.67	48 43	50	Total-N Total-P	1 year, summer	Surface runoff
Mander, 1985	Estonia	*Salix*-*Carex*-wetland meadow, buffer zone	57	1.64 1.336	1.41 1.128	83 84	550	TIN PO_4-P	Summer and spring, mean of 3 years	Overland flow retention in experimental sprinkled plots and snowmelt retention
		Grey alder buffer strip (*Alnus incana*)	105	0.76 0.68	0.62 0.62	82 92	150	TIN PO_4-P	–	–
		Cultivated grassland on slope, buffer zone	49	0.85	0.745	88	150	TIN	1 year, summer	Simulated overland flow on plots, manure application site
Mitsch et al., 1979	Illinois, USA	Alluvial cypress-tupelo swamp	39.7	0.0096	0.0087	90	300 000	Total-P	1 year	Surface flow with a 5-d flood and swamp groundwater

Table 3 (continued)

References	Country	Buffer ecosystem	Hydraulic load (mm day^{-1})	Load (g m^{-2} day^{-1})	Retention capacity (g m^{-2} day^{-1})	Removal efficiency (%)	Area (m^2)	N and P form	Time	Comments
Mitsch et al., 1995	Illinois, USA	Constructed riparian wetlands:								
		EW 3	456	0.0085	0.0056	67	23 300	Total-P	3 years	High-flow wetland
		EW 5	1206	0.0064	0.0052	79	18 700		—	—
		EW 4	342	0.0014	0.0012	89	23 400		—	Low-flow wetland
		EW 6	1304	0.0031	0.0028	94	34 500		2 years	—
Payer and Weil, 1987	Maryland, USA	Grassland-strip (*Phalaris arundinacea*)	28	0.0148	0.01	67	1380	Total- P	1 year	
Pell and Nyberg, 1989	Sweden	Grassland-strip	67	0.58	0.48	83	5	Total-P	1 year	
Peterjohn and Correll, 1984	Maryland, USA	Riparian deciduous forest at Rhode River watershed	0.95	0.005	0.0043	87	59 000	Total-N	1 year	Surface flow
				0.0009	0.0008	87		Total-P		
			0.22	0.014	0.012	86	59 000	Total-N		Subsurface flow
				2.5×10^{-5}	-5.7×10^{-5}	−299		Total-P		
Pinay and Decamps, 1988	France, River Garonne	Riparian deciduous forest, R. Louge	6	0.15	0.05	33	50 000	NO_3-N	1 year	Dentrificaton calculated

Schwer and Clausen, 1989	Vermont, USA	Grassland buffer-strip on slope	4.2	0.21 0.345	0.2 0.304	95 88	276 20 480	Total-N Total-P	2 years	Dairy wastewater
Smith and Schroeder, 1985	California, USA (near Davis)	Grassland slopes for wastewater treatment	10	0.37	0.21	56	20 480	TKN	1 year, summer and winter	Ammonia volatilization, raw or primary treated wastewater
Todd et al., 1983	Georgia, USA	Riparian deciduous forest	0.57	0.0016	0.0007	42	3400	Total-P	1 year	Subsurface flow
Young et al., 1980	Minnesota, USA	Grassland buffer strip downslope of a feedlot	73	1.708 0.945	1.019 0.587	60 62	111	TKN Total-P	1 year, summer	Simulated rainfall, overland flow
Yousef et al., 1987	Florida, USA	Roadside grassy swales:							Five experiments in summer	Simulated highway runoff on grassy slopes
		Maitland	354	0.669 0.1721	0.341 0.1084	51 63	106	Total-N Total-P		
		EPCOT	398	0.437 0.1330	0.179 0.0558	41 42	340	Total-N Total-P		
Zirschky et al., 1989	Texas, USA	Grassland slopes for wastewater treatment	40	1.14	0.24	21	6300	TKN	Over 1 year	Ammonia volatilization, raw wastewater

TKN: total Kjeldahl nitrogen.

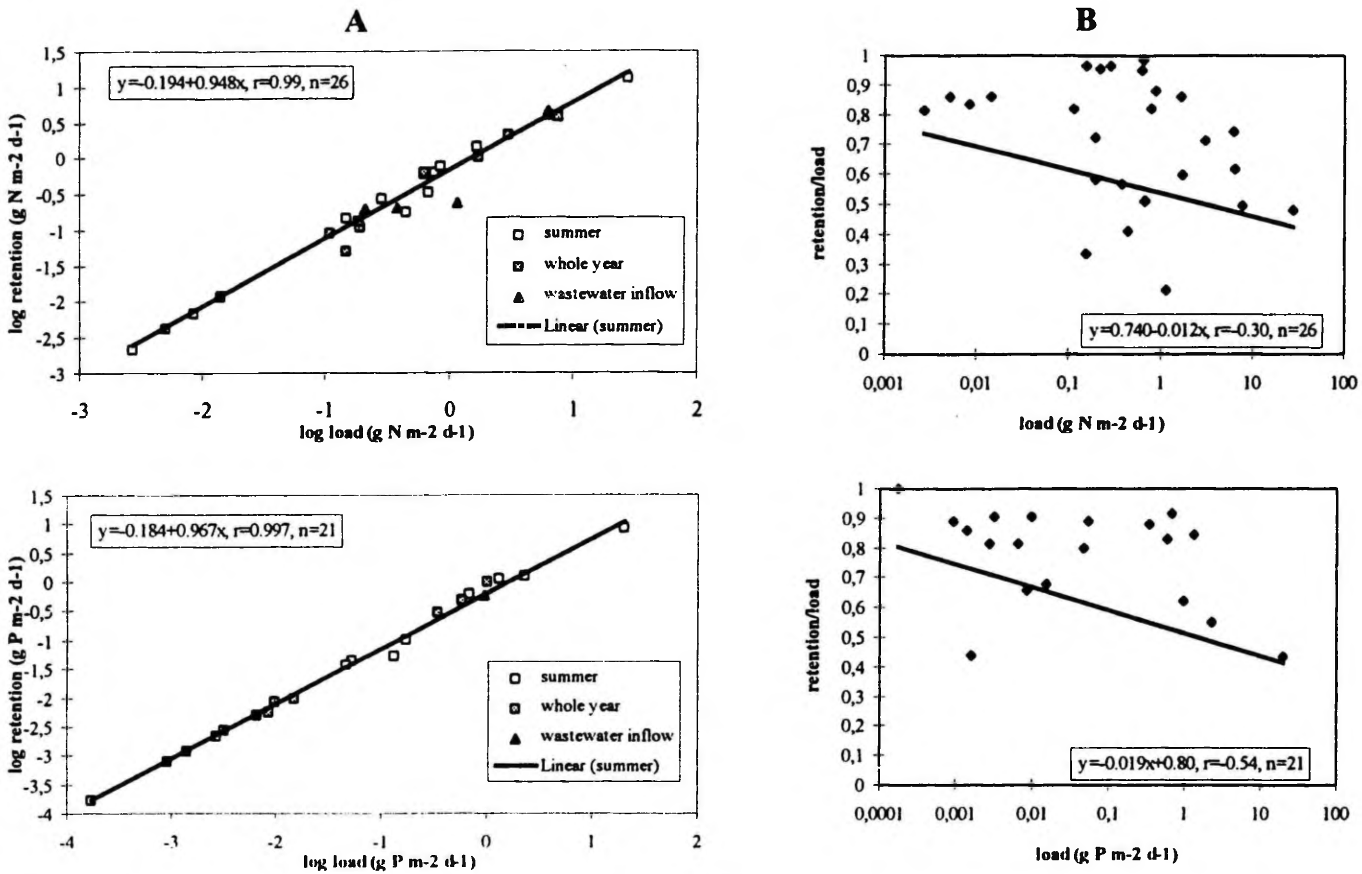

Fig. 2. Load-retention relationship of nitrogen and phosphorus in riparian buffer ecosystems. (A) Correlation between the load and retention. (B) Relative removal efficiency (retention load i.e. y/x; see text) versus logarithmic load values.

decreases. In the case of very high inputs of organic nitrogen and ammonia, the nitrification processes cause an increase in nitrate nitrogen output (Dillaha et al., 1988). The proportion of nitrate nitrogen also increases when the overland flow passes through a buffer strip with leguminous plants, which can fix the atmospheric nitrogen (Young et al., 1980; James et al., 1990).

A similar to nitrogen relationship was registered for phosphorus in buffer strips with a close linear correlation between the log-values of load (x) and retention (y):

$$y = -0.184 + 0.967x \ (r = 0.997, n = 11) \tag{10}$$

Eq. (10) was calculated mainly on the results of the vegetation period (Fig. 2(A)). However, the data from short-term events during summer with extremely high input load show the continuing high retention capacity for phosphorus (Magette et al., 1989). Some literature research described the share of groundwater transport of phosphate phosphorus in output from buffer strips (Peterjohn and Correll, 1984; Knauer and Mander, 1989). It demonstrates that phosphorus can be absorbed more intensively from the overland flow than from the subsurface water.

Significant hydraulic load seems to influence the retention capacity to a very small degree. High retention values were registered both by high and low hydraulic loads. In buffer strips the influence of hydraulic load was more significant when compared with wetlands (Mander and Mauring, 1995). On the other hand, the nitrogen retention in buffer strips was more significantly influenced by hydraulic load than the phosphorus retention. Nevertheless, increasing hydraulic load caused a slight decrease of nutrient retention both in wetlands and buffer strips.

The relative removal efficiency y/x for both N and P is decreasing when x increases. Fig. 2(B) presents the relative removal efficiency curves for N and P. Although the correlation between y/x and load (log x) is not high the plots clearly demonstrate decreasing trend of y/x values.

The load-retention relationship in various ecosystems has been discussed in several studies. Fleischer et al. (1991) proposed that the water pollution problems caused by nitrogen loading from non-point pollution sources can be solved by wetlands, because no limits had been found on the retention capacity of nitrogen in wetland ecosystems. In fact, as shown above, the relative removal efficiency is fairly limited. Thus, the 'endless' increase of retention capacity is a myth. Also, presentation in the same plot (load vs. retention) of various ecosystems with very different nutrient concentration dynamics only shows the main trends and does not give information about retention dynamics in individual wetlands. Arheimer and Wittgren (1994) showed a significant difference between two individual wetlands with different seasonal hydrological and hydrochemical dynamics. Therefore, they suggested that comparison of nutrient retention from different studies and extrapolation of results from one region to another, is only possible if detailed background data is available. A similar scheme of load-retention relationship should work for buffer strips as well. On the other hand, studies on individual riparian buffers show a very strong positive correlation between nitrogen load and removal (Haycock and Pinay, 1993). However, this relationship could not be automatically used for planning and design of riparian buffers. It should be supported by detailed data for every individual planning area.

3.2. Efficiency of riparian buffer strips: results of some case studies

Riparian forest ecosystems often serve as buffer strips but different forests show a large variation in nutrient removal. The most intensive nutrient removal was observed in bushes and young forest stands. This is due to intensive nutrient uptake by plants and high microbiological activity of soils (Bormann and Gordon, 1984). On the other hand, in undisturbed mature ecosystems, like old wetlands and other climax ecosystems, the imports and exports of nutrients are in balance (Valiela and Teal, 1979; Devito et al., 1989). Among the internal processes of riparian ecosystems relevant to nutrient retention, denitrification has been investigated most intensively (Groffmann et al., 1991; Lowrance, 1991; Weller et al., 1994; Pinay et al., 1993). However, only a few studies deal with the whole nutrient budget of riparian buffers. These are, for instance, investigations on alluvial cypress swamps (Mitsch et al., 1979), riparian forests (Peterjohn and Correll, 1984; Lowrance et al., 1984; Fail et al., 1986) and wet meadows (Leonardson et al., 1994).

In the following, buffering efficiency of two grey alder forests of different ages in Estonia (Mander et al., 1995, 1997) and two older riparian forests in the USA (Peterjohn and Correll, 1984; Lowrance et al., 1984) are compared (Table 4).

3.2.1. Nitrogen

Despite the higher input in grey alder forests the outputs are at comparable level in all of four test sites. In all cases, more than half of the input enters the buffer via overland and subsurface flow. Due to symbiotic fixation of atmospheric N_2, the nitrogen load is higher in alder stands. The accumulation in tree biomass in younger stands is higher because of their higher productivity and uptake rate. However, the flux via litter is proportionally higher in older stands. The buffering capacity of nitrogen in riparian forests is mainly the result of plant uptake which decreases as the stands become older. In comparison with the soil store of nitrogen in different communities (8–19 t ha^{-1} in Porijõgi and 11–21 t ha^{-1} in Viiratsi; Mander et al., 1997), the plant uptake and all other fluxes are small. Thus, the buffering capacity of colluvial soils with a deep humus layer, typical of riparian soils of agricultural areas, is a key factor in nutrient retention in studied alder forests.

3.2.2. Phosphorus

The inputs and outputs of phosphorus were on a comparable level, except for the Little River test site (Lowrance et al., 1984; Fail et al., 1986) where both have higher values caused by higher atmospheric deposition rate. In this case, the atmospheric input exceeds the input by overland and subsurface flow. As for nitrogen, the biggest flux is plant uptake that is lower in older stands. Considering the large soil store of phosphorus (1.9–2.0 t ha^{-1} in Porijõgi and 1.4–2.7 t ha^{-1} in Viiratsi; Mander et al., 1997), we suggest that in the long-term most retained phosphorus is accumulated in the soil.

In the long-term this very high loading cannot be balanced by Fe, Al and Ca phosphate precipitation. This is seemingly the key process in phosphorus retention

Table 4
Nitrogen and phosphorus flux and cycling in riparian forests (kg ha^{-1} $year^{-1}$) in four test sites

Flux and cycling	Porijõgi Estonia (age 14 years)[a]		Viiratsi Estonia (age 40 years)[a]		Rhode River MA, USA (age > 50 years)[b]		Little River GA, USA (age > 50 years)[c]	
	N	P	N	P	N	P	N	P
Input, included	110.4	2.5	72.9	3.0	53	2.14	51.8	5.6
Precipitation	6.4	0.7	6.7	1.0	14	0.14	12.2	3.5
Nitrogen fixation[d,e]	36		21.0				10.6	
Overland and subsurface flow	68.0	1.8	45.2	2.0	39	2.0	29.0	2.1
Transformation								
Accumulation in tree biomass	204.8	15.1	140.2	10.8	77	10	22.3	1.4[f]
Litter	82.0	4.0	87.0	4.0	62	7.8	26.8	4.1
Denitrification[d]	8.5		7.9				31.5	
Active soil exchange (ASE)[e]	−34.1	−9.2	2.8	−4.2	28.8	−0.79	11.8	4.4
Output into stream	13.2	0.62	9.0	0.38	9.2	0.73	13.0	3.9

[a] Mander et al., 1997.
[b] Peterjohn and Correll, 1984.
[c] Lowrance et al., 1984; Fail et al., 1986.
[d] Estimated on the base of four measurements during October 1993–September 1994 (fixation in soil by non-symbiotic bacteria) and after Rytter et al., 1991; Huss-Danell et al., 1991 (fixation by nodules).
[e] Calculated; '+', accumulation in soil; '−', mineralization, vegetative uptake and transformation by biota.
ASE is the input-output-accumulation in plant biomass—denitrification + litter.
[f] Above-ground storage only.

Table 5
Removal efficiency, specific removal and retention of nitrogen and phosphorus in grey alder test sites (Mander et al., 1995)

	Porijõgi		Viiratsi	
	N	P	N	P
Removal efficiency (%)	81	67	80	81
Specific removal (% m^{-1})	4.1	3.4	2.9	2.9
Retention (kg ha^{-1} $year^{-1}$)	20.9	1.2	36.2	1.6

in alder stands. Some investigations also suggest that permanently high nitrogen concentrations in soils can cause phosphorus leaching (Andrusch et al., 1992). On the other hand, our earlier investigations demonstrate that riparian alder forests are effective buffers for P (Mander et al., 1995). Even in riparian wetlands P can be retained due to micro-scale oxygenation variability within the wetland and, probably, due to phosphorus inactivation by nitrate (Andersen, 1982).

Our results suggest it is important to harvest older *Alnus incana* stands (> 20 years) earlier, due to their decreasing uptake and productivity with age.

3.2.3. Removal efficiency of nitrogen and phosphorus in buffer zones

Removal efficiency E (%) of N and P from the groundwater and overland flow in riparian communities was estimated according to formula 2. The retention capacity R (see Eq. 1) was calculated in kg ha^{-1} $year^{-1}$.

The specific removal (% m^{-1}) is defined as the removal efficiency per unit width of a buffer zone. This characteristic is useful for planning and establishing buffer communities.

In Table 5, values of the removal efficiency, specific removal and retention for studied riparian grey alder forests are presented. Despite the lower vegetation uptake of N in the older alder stand in Viiratsi, the retention is higher than in the younger stand in Porijõgi. Due to smaller width the specific removal is higher in the Porijõgi transect.

We observed a significantly higher efficiency values in the complex sequential buffer strip that includes a grassland strip, wet meadow and grey alder stand, than in single ones (Mander et al., 1997). The specific removal of N was decreased downhill which coincides with the edge effect reported in earlier papers (Doyle et al., 1977; Mander, 1985; Knauer and Mander, 1989; Vought et al., 1994). The 50–60 m wide complex buffer zone was able to retain and transform most of the nitrogen and phosphorus entering the buffer.

Some of the results of this work which are considered while designing the buffer zones and buffer strips are presented in the Table 6. The relatively high estimated summary efficiency (E) of buffer zones and buffer strips in the Vända ditch sub-catchment (14.0 and 24.4 for N and P, respectively) is due to high percentage of additional buffer zones (67% of the total area of buffer zones) and buffer strips (87.5%, correspondingly). Buffer zones have shown 1.5–3 times higher intensity than the buffer strips, however, the nutrient removal capacity in planned buffer strips is many times higher (964 and 85.7 kg ha^{-1} $year^{-1}$ for N and P, accordingly). The very high estimated nitrogen removal can be explained by both high nitrate concentration in seeping groundwater (30–50 mg NO_3 l^{-1}; Mander et al., 1994) and high denitrification. In the Sipe stream sub-catchment, the percentage of additional buffer zones and strips is 9.3 and 5.0% of their total area, respectively. This is one of the reasons why the estimated total efficiency (E) of additional buffers is lower than in the Vända sub-catchment, being 11.3 and 15.3%, correspondingly (Table 6).

In the last 3 years, a significant decrease in nutrient losses from this area has occurred. This is, again, caused by the lower intensity of fertilization and land cultivation (Mander and Palang, 1994).

Table 6
Estimated efficiency of existing (1990) and planned buffer zones and buffer strips in two sub-catchments of the Porijõgi River, Estonia (after Mander, 1995)

Sub-catchment	Sipe stream	Vända ditch
Area (ha)	900	220
Arable land in 1990 (ha)	518	150
Buffer zones in 1990 (ha)		
Existing	420	56
Planned	22	112
Total	442	168
Buffer strips (ha)		
Existing	7.8	0.2
Planned	0.8	1.4
Total	8.6	1.6
Arable land to be converted (ha)	23	113
Mean annual nutrient loss from arable land (kg ha^{-1} $year^{-1}$)		
N	6.3	12.1
P	0.57	1.23
Efficiency of buffer zones (t $year^{-1}$)		
N	0.38	2.85
P	0.035	0.18
Efficiency of buffer strips (t $year^{-1}$)		
N	0.33	1.35
P	0.052	0.12
Total efficiency (t $year^{-1}$)		
N	0.71	4.2
P	0.087	0.3

4. Conclusion

Both our investigations and studies on riparian ecosystems drawn upon in this paper demonstrated significant nutrient removal from the incoming material fluxes. The correlation between the log-transformed load and retention for nitrogen and phosphorus in buffer ecosystems was linear and very high. Even when the input concentrations were extremely high, the buffers were able to remove both nitrogen and phosphorus. Many different natural and semi-natural ecosystems play an essential role in controlling material fluxes through landscapes of which bushes, young forest stands and wet grasslands show the most intensive nutrient removal. This is due to intensive nutrient uptake by plants and high microbiological activity and adsorption capacity of soils. Therefore, the mosaic landscape pattern with different buffering ecosystems is able to control even high nutrient fluxes. However, harvesting from buffer zones and buffer strips is needed in order to remove the accumulated material and to keep the buffers in a young succession stage. Many new perspectives on the use of buffer ecosystems make their maintenance and establishment attractive from both ecological and economic point of view.

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References

Abernathy, A.R., Zirschky, J., Borup, M.B., 1985. Overland flow wastewater treatment at Easley, S.C. J. Water Pollut. Control Fed. 57, 291–299.

Ambus, P., Christiansen, S., 1991. Denitrification and other nitrogen cycling processes in a riparian meadow. Project abstracts of the Danish NPo Research Programme. C-Abstracts. Number C13a. Ministry of the Environment, National Agency of Environmental Protection, Denmark, pp. 187–202.

Ambus, P., Mosier, A., Christensen, S., 1992. Nitrogen turnover rates in a riparian fen determined by ^{15}N dilution. Biol. Fertil. Soils 14, 230–236.

Andersen, J.M., 1982. Effect of nitrate concentration in lake water on phosphate release from the sediment. Water Res. 16, 1119–1126.

Andrusch, T., Hupfer, M., Luther, D., 1992. Chemical and microbial binding forms of phosphorus considering the availability of nitrate in sediment-water systems. Int. Rev. ges. Hydrobiol. 77, 109–120.

Arheimer, B., Wittgren, H.-B., 1994. Modeling the effects of wetlands on regional nitrogen transport. Ambio 23, 378–386.

Baldock, D., Beaufoy, G., Bennett, G., Clark, J., 1993. Nature Conservation and New Directions in the Common Agricultural Policy. Report for the Ministry of Agriculture, Nature Management and Fisheries, the Netherlands. Institute for European Environmental Policy, HPC, Arnhem, p. 224.

Bingham, S.C., Westerman, P.W., Overcash, M.R., 1980. Effect of grass buffer zone length in reducing the pollution from land application areas. Trans. ASAE, St. Joseph, Mich. 23, 330–336.
Bishoff, N.T., Jongman, R.H.G., 1993. Development of Rural Areas in Europe: The Claim for Nature. Netherlands Scientific Council for Government Policy. Sdu uitgeverij, Plantijnstraat, The Hague, p. 206.
Bormann, B.T., Gordon, J.C., 1984. Stand density effects in young red alder plantations: Productivity, photosynthate partitioning, and nitrogen fixation. Ecology 65, 394–402.
Bowden, W.B., 1986a. Gaseous nitrogen emmissions from undisturbed terrestrial ecosystems: an assessment of their impacts on local and global nitrogen budgets. Biogeochemistry 2, 249–279.
Bowden, W.B., 1986b. Nitrification, nitrate reduction, and nitrogen immobilization in a tidal freshwater marsh sediment. Ecology 67, 88–99.
Bowden, W.B., 1987. The biogeochemistry of nitrogen in freshwater wetlands. Biogeochemistry 4, 313–348.
Brüsch, W., Nilsson, B., 1993. Nitrate transformation and water movement in a wetland area. Hydrobiologia 251, 103–111.
Clausen, J.C., Johnson, G.D., 1990. Lake level influences on sediment and nutrient retention in a lakeside wetland. J. Environ. Qual. 19, 83–88.
Cooper, B., 1990. Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. Hydrobiologia 202, 13–26.
Denmead, O.T., Freney, J.R., Simpson, J.R., 1976. A closed ammonia cycle within a plant canopy. Soil Biol. Biochem. 8, 161–164.
Devito, K.J., Dillon, P.J., Lazerte, B.D., 1989. Phosphorus and nitrogen retention in five Precambrian shield wetlands. Biogeochemistry 8, 185–204.
Dillaha, T.A., Sherrard, J.H., Lee, D, Mostaghimi, S., Shanholtz, V.O., 1988. Evaluation of vegetated filter strip as a best management practice for feed lots. J. Water Pollut. Control Fed. 60, 1231–1238.
Doyle, R.C., Stanton, G.C., Wolf, D.C., 1977. Effectiveness of forest and grass buffer strip in improving the water quality of manure polluted runoff. ASAE Paper No 77–2501, St Joseph, MI. p. 32.
Fail, Jr., J.L., Haines, B.L., Todd, R.L., 1986. Riparian forest communities and their role in nutrient conservation in an agricultural watershed. Am. J. Alternative Agric. 2, 114–121.
Fennessy, M.S, Brueske, C.C., Mitsch, W.J., 1994. Sediment deposition patterns in restored freshwater wetlands using sediment traps. Ecol. Eng. 3, 409–428.
Fleischer, S., Stibe, L., Leonardson, L., 1991. Restoration of wetlands as a means of reducing nitrogen transport to coastal waters. Ambio 20, 271–272.
Fustec, E., Mariotti, A., Grillo, X., Sajus, J., 1991. Nitrate removal by denitrification in alluvial ground water: role of a former channel. J. Hydrol. 123, 337–354.
Groffmann, P.M., Axelrod, E.A., Lemunyon, J.L., Sullivan, W.M., 1991. Denitrification in grass and forest vegetated filter strips. J. Environ. Qual. 20, 671–674.
Haycock, N., Pinay, G., 1993. Nitrate retention in grass and poplar vegetated riparian buffer strips during the winter. J. Environ. Qual. 22, 273–278.
Hoffmann, C.C., 1991. Water and nutrient balances for a flooded riparian wetland. In: Nitrogen and Phosphorus in Fresh and Marine Water. Project Abstracts of the Danish NPo Research Programme, C-Abstracts, C13-b. Ministry of the Environment, National Agency of Environmental Protection, Denmark, pp. 203–220.
Hsieh, Y.P., 1988. Dynamics of carbon, nitrogen and phosphorus cycling in a sawgrass tidal marsh with special reference to the aboveground primary production. J. Environ. Qual. 17, 676–681.
Huss-Danell, K., Lundquist, P.-O., Ohlsson, H., 1991. N_2 fixation in a young *Alnus incana* stand, based on seasonal and diurnal variation in whole plant nitrogenase activity. Can. J. Bot. 70, 1537–1544.
Jacobs, T.C., Gilliam, J.W., 1985. Riparian losses of nitrate from agricultural drainage waters. J. Environ. Qual. 14, 472–478.
James, B.R., Bagley, B.B., Gallagher, P.H., 1990. Riparian zone vegetation effects on nitrate concentrations in shallow groundwater. In: Proceedings of the 1990 Chesapeake Bay Research Conference, Baltimore, Maryland, December, 1990, pp. 1–14.
Jordan, T.E., Correll, D.L., Weller, D.E., 1992. Nutrient interception by riparian forest receiving inputs from adjacent cropland. J. Environ. Qual. 22, 467–473.

Kim, C.M., 1973. Influence of vegetation types on the intensity of ammonium and nitrogen dioxide liberation from soil. Soil Biol. Bioch. 5, 163–166.
Klingensmith, K.M., Van Cleve, K., 1993. Denitrification and nitrogen fixation in floodplain successional soils along the Tanana River interior. Alaska. Can. J. For. Res. 23, 956–963.
Knauer, N., Mander, Ü., 1989. Untersuchungen über die Filterwirkung verschiedener Saumbiotope an Gewässern in Schleswig-Holstein. 1. Mitteilung: Filterung von Stickstoff und Phosphor. Z. f. Kulturtechnik und Landentwicklung 30, 365–376.
Leonardson, L., Bengtsson, L., Davidsson, T., Persson, T., Emanuelsson, U., 1994. Nitrogen retention in artificially flooded meadows. Ambio 23, 332–341.
Lowrance, R., 1991. Effects of buffer systems on the movement of to N and P from agriculture to streams. In: NPo-forskning fra Miljøstyrelsen. Conference Contributions. International Conference on N, P and Organic matter. Helsingør, Denmark.
Lowrance, R.R., Todd, R., Fail, J., Hendrickson, O., Leonard, R., Asmussen, L., 1984. Riparian forest as nutrient filters in agricultural watersheds. BioScience 34, 374–377.
Magette, W.L., Brinsfield, R.B., Palmer, R.E., Wood, J.D., 1989. Nutrient and sediment removal by vegetated filter strips. Trans. ASAE, St. Joseph, Mich. 32, 663–727.
Mander, Ü., 1985. The renovation effect of polluted surface flow in vegetated buffer strips. Acta et commentationes Universitatis Tartuensis 701, 77–89. (in Russian, with summary in English).
Mander, Ü., 1989. Kompensationsstreifen entlang der Ufer und Gewässerschutz. Landesamt für Wasserhaushalt und Küsten Schleswig-Holstein. Bericht D10, p. 61.
Mander, Ü., Metsur, M., Külvik, M., 1989. Störungen des Stoffkreislaufs, des Energieflusses und des Bios als Kriterien für die Bestimmung der Belastung der Landschaft. Petermanns Geographische Mitteilungen 4, 233–244.
Mander, Ü., Kuusemets, V., Treier, K., 1994. Variation in groundwater quality in South Estonian rural areas. IAHS Publ. 220, 81–93.
Mander, Ü., Mauring, T., 1995. Nitrogen and phosphorus retention in natural ecosystems. In: Ryszkowski, L., Balazy, S. (Eds.), Functional Appraisal of Agricultural Landscape in Europe (EUROMAB and 21 INTECOL Seminar 1992). Research Centre for Agricultural and Forest Environment. Pol. Acad. Sci. Poznan 1994, pp. 77–94.
Mander, Ü., Palang, H., 1994. Changes of landscape structure in Estonia during the Soviet period. Geojournal 33, 45–54.
Mander, Ü., 1995. Riparian buffer zones and buffer strips on stream banks: dimensioning and efficiency assessment from catchments in Estonia. In: Eiseltova, M., Biggs, J. (Eds.), Restoration of Stream Ecosystems. IWRB Publication No. 37, Slimbridge, Gloucester, UK, pp. 45–64.
Mander, Ü., Kuusemets, V., Ivask, M., 1995. Nutrient dynamics of riparian ecotones: a case study from the Porijõgi River catchment, Estonia. Landsc. Urban Plann. 31. 333–348.
Mander, Ü., Lohmus, K., Kuusemets, V., Ivask, M., 1997. The potential role of wet meadows and grey alder forests as buffer zones. In: Haycock, N.E., Burt, T.P., Goulding, K.W.T., Pinay, G. (Eds.), Buffer Zones, their Processes and Potential in Water Protection. Proceedings of the International Conference on Buffer Zones, September 1996. Part II. Oxford, Quest Environmental, Foundation for Water Research, pp. 35–46.
Melillo, J.M., 1981. Nitrogen cycling in deciduous forests. In: Clark, F.E., Rosswall, T. (Eds.), Terrestrial Nitrogen Cycling. Ecological Bulletin (Stockholm), pp. 427–442.
Meyer, L.D., Wischmeier, W.H., 1969. Mathematical simulation of the process of soil erosion by water. Trans. ASAE 12, 754–758.
Mitsch, W.J., 1992. Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution. Ecol. Eng. 1, 27–47.
Mitsch, W.J., Gosselink, J.G., 1993. Wetlands, 2nd ed. Van Nostrand Reinhold, New York, p. 722.
Mitsch, W.J., Dorge, C.L., Wiemhoff, J.R., 1979. Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in southern Illinois. Ecology 60, 1116–1124.
Mitsch, W.J., Cronk, J.K., Wu, X., Nairn, R.W., Hey, D.L., 1995. Phosphorus retention in constructed freshwater riparian marshes. Ecol. Applic. 5, 830–845.
Moore, I.D., Burch, G.J., 1986. Physical basis of the length-slope factor in the universal soil loss equation. Soil Sci. Soc. Am. J. 50, 1294–1298.

Osborne, L. L, Kovacic, D.A., 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biol. 29, 243–258.

Overcash, M.R., Bingham, S.C., Westerman, P.W., 1981. Predicting runoff pollutant reduction in buffer zones adjacent to land treatment sites. Trans. ASAE, St. Joseph, Mich. 24, 430–435.

Payer, F.S., Weil, R.R., 1987. Phosphorus renovation of wastewater by overland flow land application. J. Environ. Qual. 16, 391–397.

Pell, M., Nyberg, F., 1989. Infiltration of wastewater in a newly started pilot-filter system: I. Reduction of organic matter and phosphorus. J. Environ. Qual. 18, 451–457.

Peterjohn, W.T., Correll, D.L., 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 65, 1466–1475.

Phillips, J.D., 1989. Nonpoint source pollution control effectiveness of riparian forests along a coastal plain river. J. Hydrol. 110, 221–237.

Pinay, G., Decamps, H., 1988. The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: a conceptual model. Regulated Rivers: Res. Manage. 2, 507–516.

Pinay, G., Roques, L., Fabre, A., 1993. Spatial and temporal patterns of denitrification in a riparian forest. J. Appl. Ecol. 30, 581–591.

Prach, K., Rauch, O., 1992. On filter effects of ecotones. Ekologia (CSFR) 11, 293–298.

Richardson, C.J., 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228, 1424–1427.

Richardson, C.J., Marshall, P.E., 1986. Processes controlling movement, storage, and export of phosphorus in a fen peatland. Ecol. Monogr. 56, 279–302.

Robertson, G.P., Huston, M.A., Evans, F.C., Tiedje, J.M., 1988. Spatial variability in a successional plant community: Patterns of nitrogen availability. Ecology 69, 1517–1524.

Rytter, L., Arveby, A.S., Granhall, U., 1991. Dinitrogen acetylene fixation in relation to nitrogen fertilization of gray alder *Alnus incana* L. (Moench.) plantations in a peat bog. Biol. Fertil. Soils 10, 233–240.

Schwer, C.B., Clausen, J.C., 1989. Vegetative filter treatment of dairy milkhouse wastewater. J. Environ. Qual. 18, 446–451.

Smith, R.G., Schroeder, E.D., 1985. Field studies of the overland flow process for the treatment of raw and primary treated municipal wastewater. J. Water Pollut. Control Fed. 57, 785–794.

The EECONET Declaration, 1993. Declaration of the Conference 'Conserving Europe's Natural Heritage Towards a European Ecological Network', Maastricht, 9–12 November 1993. Ministry of Agriculture, Nature Management and Fisheries.

Tietema, A., Duysings, J.J.H.M., Verstraten, J.M., Westerveld, J.W., 1987. Estimation of actual nitrification rates in an acid forest soil. In: Hutchinson, T.C., Meena, K.M. (Eds.), Effects of Atmospheric Pollutants on Forests, Wetlands and Agricultural Ecosystems. NATO ASI Series, Proc. of ARW, Toronto (Canada), 1985. Springer, Berlin, pp. 190–197.

Timmons, D.R., Verry, E.S., Burwell, R.E, Holt, R.F., 1977. Nutrient transport in surface runoff end interflow from an aspen-birch forest. J. Environ. Qual. 6, 188–192.

Todd, R., Lowrance, R., Hendrickson, D., Asmussen, L., Leonard, R., Fail, J., Herrick, B., 1983. Riparian vegetation as filters of nutrients exported from a coastal plain agricultural watershed. Int. Symp. Nutrient Cycling in Agricultural Ecosystems (Athens, 1980). Special Publ. 23, 485–493.

Uusi-Kämppä, J., Yläranta, T. 1996. Effect of buffer strips on controlling soil erosion and nutrient losses in southern Finland. In: Mulamoottil, G. et al. (Eds.), Wetlands: Environmental Gradients, Boundaries, and Buffers. CRC Press, Lewis, Boca Raton, pp. 221–235.

Valiela, I., Teal, J.M., 1979. The nitrogen budget of a salt marsh ecosystem. Nature 280, 652–656.

Vanek, V., 1991. Riparian zone as source of phosphorus for a groundwater-dominated lake. Water Res. 25, 409–418.

Verhoeven, J.T.A., Maltby, E., Schmitz, M.B., 1990. Nitrogen and phosphorus mineralization in fens and bogs. J. Ecol. 78, 713–726.

Verry, E.S., Timmons, D.R., 1982. Waterborne nutrient flow through an upland-peatland watershed in Minnesota. Ecology 63, 1456–1467.

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

Weller, D.E., Correll, 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.

Woodmansee, R.G., 1978. Additions and losses of nitrogen in grassland ecosystems. BioScience 28, 448–453.

Yarbro, L.A., 1979. Phosphorus Cycling in the Creeping Swamp Floodplain Ecosystem and Exports from the Creeping Swamp Watersheds. Ph.D. Dissertation, University of North Carolina, Chapel Hill.

Young, R.A., Huntrods, T., Anderson, W., 1980. Effectiveness of vegetated buffer strips in controlling pollution from feedlot runoff. J. Environ. Qual. 9, 483–487.

Yousef, Y.A., Hvitved-Jacobsen, T., Wanielista, M.P., Harper, H.H., 1987. Removal of contaminants in highway runoffflowing through swales. Sci. Total Envir. 59, 391–397.

Zak, D.R., Grigal, D.F., 1991. Nitrogen mineralization nitrification and denitrification in upland and wetland ecosystems. Oecologia 88, 189–196.

Zirschky, J., Crawford, D., Norton, L., Richards, S., Deemer, D., 1989. Ammonia removal using overland flow. J. Water Pollut. Control Fed. 61, 1225–1232.

VI

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Eecotechnological measures to control nutrient losses from catchments.
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ECOTECHNOLOGICAL MEASURES TO CONTROL NUTRIENT LOSSES FROM CATCHMENTS

Valdo Kuusemets and Ülo Mander

University of Tartu, Institute of Geography, 46 Vanemuise St., EE-2400 Tartu, Estonia

ABSTRACT

Ecotechnological measures allow to use different natural and seminatural ecosystems to control nutrient losses from intensively used watersheds. The most effective means are buffer strips, buffer zones and constructed wetlands. In southern Estonia a 31 m wide buffer zone of wet meadow and grey alder forest removed 50 % nitrogen and 78 % phosphorus, while in a 51 m buffer zone, also containing a grassland strip in addition to wet meadow and alder forest, 87 % N and 84 % P was retained. The outflow of total – N was 4.9 and outflow of total – P was 4.8 lower in well-buffered watershed in comparison to similar watershed with lower buffering ability. Three constructed wetlands in southern Estonia were studied. The removal efficiency was 76 – 84% for BOD_5, 39 – 70% for total-N, 73 – 83% for total-P. All results show that compared to other seasons the winter performance was not reduced.

KEYWORDS

Constructed wetlands, non-point pollution, nutrient control, point pollution riparian buffer zones.

INTRODUCTION

Control of nutrient losses from catchments needs proper management of point and non-point pollution. For management of point pollution there are many technical (conventional) solutions that are in many cases expensive and do not solve all environmental problems in long term perspective. Two principle ways exist in the control of non-point pollution from rural catchments: (1) adoption of "Best Environmental Practices" by optimisation of fertilisers use, crop rotation, and soil cultivation methods in a manner that maintains the nutrient equilibrium in the soil; (2) using various mitigation measures as buffering ecosystems (*e.g.*, riparian buffer strips and buffer zones, natural and constructed wetlands, hedgerows, shelterbelts) to intercept and transform nutrient fluxes from agricultural lands to water. These two classes of measures can be classified as ecotechnological measures for pollution control that can be used in certain conditions also for management of point pollution despite of conventional technologies. Under the term "ecotechnology" (i.e., ecological engineering) we consider the engineering in the sense that it involves the design of this natural environment using quantitative approaches and basing our approaches on basic science; it is technology with the primary tool being self-designing ecosystems (Mitsch and Jørgensen, 1989). According to principles of ecological engineering, the most effective way to reduce pollution load is to decrease pollution at source (often called as "the beginning-of-pipe-principle", Mitsch and Jørgensen, 1989). This principle is applied in a large variety of organic agricultural systems. However, certain losses of nutrients occur even from the biodynamic farming, the most extremely equilibrium-orientated agricultural system (Granstedt, 1990; Van Mansvelt and Mulder, 1993), which needs measures applied. One of the most ecologically sound solutions for purifying wastewater by degree of treatment intensity and the degree of recycling are constructed wetlands (Guterstam, 1991). Constructed wetlands for wastewater treatment show good purification ability (Geller *et al.*, 1991; Heeb and Züst, 1991; Jenssen *et al.*, 1993; Mander and Mauring, 1997; Brix, 1998) likewise natural, seminatural and artificial buffering ecosystems that provide a great interest in terms of controlling the nutrient fluxes (Lowrance *et al.*, 1984; Peterjohn and Correll, 1984; Knauer and Mander, 1989; Haycock and Pinay, 1993; Vought *et al.*, 1994). Riparian buffer strips and buffer zones (Mander *et al.*, 1997b) as multifunctional elements of rural landscapes can be also defined as one of the most ecologically sound solutions for control of water quality. Constructed wetlands and buffer zones can purify water from various pollutants. Mainly, purification of suspended solids, organic matter, nitrogen and phosphorus have been considered in literature. Kadlec and Knight (1996) describe the main purification processes as following. Suspended solids are purified by: (1) sedimentation and trapping; (2) chemical precipitation. Main purification processes for organic matter are: (1) microbial decomposition and respiration; (2) chemical precipitation and sedimentation. Nitrogen is removed by: (1) uptake and storage in vegetation; (2) microbial immobilization and storage in the soil as organic nitrogen; (3) microbial conversion to gaseous form of nitrogen (denitrification); (4) ammonia volatilization. Phosphorus is removed by: (1) sedimentation of particulate phosphorus and chemical precipitation; (2) soil sorption; (3) removal of dissolved inorganic phosphorus by plant uptake; (4) microbial immobilization and storage in the soil as organic phosphorus.

The main objective of this paper is to analyse efficiency of riparian buffer zones and constructed wetlands in some agricultural watersheds in Estonia.

MATERIAL AND METHODS

Site description

The study area is the Porijõgi River basin that lies within the southeast Estonian moraine plain 5-10 km south of Tartu (58°23'N; 26°44'E). The moraine plain is 30-60 m a.s.l. with undulated relief, and is dissected by primeval valleys. The southern part lies on the northern slope of Otepää heights, which consists of moraine hills and kames. A more detailed description of the study area is given in an earlier study (Mander *et al.*, 1995).

In 1994-95, two transects were established on the thalwegs of slopes adjacent to streams, and crossing different riparian plant communities. Transects had similar physico-geographical conditions with similar plant communities, covering field, grass, wet meadow and woodland. The Porijõgi transect is located in the central part of Porijõgi catchment with lower pollution load, where agricultural activities stopped in 1992. The Viiratsi transect is situated in an area impacted by pig slurry from the Viiratsi pig farm. A more detailed description of transects is given in Mander *et al.* (1997a). In the landscape profiles piezometers (3 rows in the Porijõgi transect and 4 rows in the Viiratsi site, with 3 replicates in each row) and study plots were established on the boundaries between plant communities.

Also, following constructed wetlands for wastewater treatment were studied:

- sand/plant filter system in Põlva, established in 1992 (90 m^2, wastewater flowrate 2 m^3 d^{-1}, area loading 3.8 g BOD m^2 d^{-1}, water detention time 15-25 d);
- seminatural wet meadow covered with *Phalaris arundinacea*, established in 1989 (later called as "*Phalaris* -slope"; 2400 m^2, 130 m^3 d^{-1}, 1 g BOD m^2 d^{-1}, 20-40 d);
- drainage channel with intensive macrophyte growth that was not constructed for wastewater purification and receives wastewater from a cascade of four sedimentation ponds (one anaerobic and three aerobic ponds) for wastewater treatment (later called as "bioditch"; 140 m^2, 125m^3 d^{-1}, 40 g BOD m^{-2} d^{-1}, 0.5 d).

A more detailed description of these constructed wetlands is presented in an earlier study (Mander and Mauring, 1997).

Field experiments and laboratory analysis

Stream discharge was measured and water samples were taken for analysing in laboratory once per month from 16 sampling points from Porijõgi River catchment including 5 closing sampling points of subwatersheds. On transects soil water samples were collected once or twice a month from piezometers installed on the borders of plant communities during June 1994 to December 1995. Soil water discharge was estimated on the basis of both Darcy's law and by gauging with weirs installed in soil water seeping sites. Water samples from inlet and outlet of studied constructed wetlands were taken and water flow was measured since 1989 (*Phalaris*-slope, Rakke) and since 1993 (Põlva) once a month. The surface water from Porijõgi River and wastewater from constructed wetlands was analysed in the South-Estonian Laboratory of Environment Protection, Tartu. Filtered water samples were analysed for NH_4-N, NO_2-N, NO_3-N, total-N, PO_4-P, total-P and organic matter concentrations (on the base of BOD_5) following standard methods for examination of water and wastewater quality (APHA, 1989). Filtered soil water samples were analysed in the laboratory of Estonian Agricultural University for NH_4-N, NO_2-N, NO_3-N, total-N, PO_4-P, total-P, SO_4, Fe, Ca (APHA, 1989).

Efficiency and buffering percentage calculations

Removal efficiency E (%) of N and P in buffer communities and constructed wetlands was estimated as:

$$E = 100\ \% * (Q_{in}C_{in} - Q_{out}C_{out}) / (Q_{in}C_{in}) \qquad (1)$$

Q_{in} and Q_{out} = inflow and outflow values ($m^3\ d^{-1}$), respectively; C_{in} and C_{out} = concentration values (mg l^{-1}), respectively.

The retention capacity R in buffer zones and constructed wetlands (kg $ha^{-1}\ yr^{-1}$ and g $m^{-2}\ d^{-1}$, respectively) was calculated as follows:

$$R = \Sigma(Q_{in}C_{in} - Q_{out}C_{out}) / A \qquad (2)$$

A = area of the buffer zone or constructed wetland.

The specific removal (% m^{-1}) is defined as the removal efficiency per unit width of a buffer zone. This characteristic is useful for planning and establishing buffer communities.

The percentage of buffered stream banks was calculated as follows:

$$B = l_b/\Sigma l_t * 100\% \qquad (3)$$

l_b = length of all stream banks in the catchment (m), having buffer zone or strip between field and stream.
Σl_t = total length of stream banks in the catchment (m).

The percentage of unbuffered stream banks was calculated as follows:

$$U = l_u/\Sigma l_t * 100\% \qquad (4)$$

l_u = length of all stream banks in the catchment (m), having agricultural fields up to the bank of stream.

The percentage of stream banks with natural communities was calculated as follows:

$$N = l_n/ \Sigma l_t * 100\% \qquad (5)$$

l_n = length of all stream banks in the catchment (m), bordering with natural plant communities.

RESULTS AND DISCUSSION

Role of buffer zones

The buffers investigated showed good removal efficiency. According to values presented in Table 1, in the 31 m wide Porijõgi transect 50% of total nitrogen and 78% of total phosphorus was removed, while in the 51 m wide Viiratsi transect showed 87% and 84% removal, respectively. The lower N removal efficiency in the Porijõgi transect can be explained by lower incoming nitrogen load in soil water (average for study period 3.0 mg N l^{-1}). In Viiratsi the same value was 23 mg N l^{-1}. In both cases the most intensive retention and transformation was taken place already in the first part of the buffer. In the Porijõgi transect all phosphorus and major part of nitrogen was retained and transformed within the first 11 m wide *Filipendula-Aegopodium* wet meadow strip. Likewise, in the heavily polluted Viiratsi buffer zone first 11 m wide *Elytrigia repens*-grassland and 12 m wide *Filipendula-Aegopodium* wet meadow retained all phosphorus and main part of nitrogen entering the buffer. The specific removal of wet meadow plant communities reached 3.3% m^{-1} for N, 7.1% m^{-1} for P in Porijõgi and 5.0% m^{-1} and 5.9% m^{-1} in Viiratsi, respectively. In the Viiratsi transect grey alder stand (*Alnus incana)* did leach out some phosphorus. However, the concentrations of ground water were very low (0.07 mg l^{-1}) for such heavily polluted area and were comparable with outflow from less polluted Porijõgi River transect (0.06 mg l^{-1}). The increase of phosphorus in lower part of buffer zone can be caused due to the saturation in buffer zone.

Table 1. Average total nitrogen and total phosphorus content for study period in soil water (mg l^{-1}), removal efficiency (%) and specific removal (% m^{-1}).

	Grassland		Wet meadow		Alder forest		Whole complex	
	N	P	N	P	N	P	N	P
Porijõgi (less polluted)								
Soil water quality (input, mg l^{-1})			3.0	0.27	1.9	0.06	3	0.27
Soil water quality (output, mg l^{-1})			1.9	0.06	1.5	0.06	1.5	0.06
Removal efficiency (%)			37	78	21	0	50	78
Specific removal (% m^{-1})			3.3	7.1	1.1	0	1.6	2.5
Retention (kg ha^{-1} yr^{-1})			14.4	0.7	12.4	1.2	13.1	1.0
Viiratsi (heavily polluted)								
Soil water quality (input, mg l^{-1})	23	0.43	14	0.17	5.6	0.05	23	0.43
Soil water quality (output, mg l^{-1})	14	0.17	5.6	0.05	3.1	0.07	3.1	0.07
Removal efficiency (%)	39	60	60	71	45	-40	87	84
Specific removal (% m^{-1})	3.6	5.5	5.0	5.9	1.6	-1.4	1.7	1.6
Retention (kg ha^{-1} yr^{-1})	185.5	6.0	33.8	1.7	36.2	1.6	56.7	2.2

In general, both buffer zones showed high nutrients retention values and should be used as nutrient control elements in watersheds. The details for designing and dimensioning buffer strips and buffer zones are given in earlier papers (Mander *et al.*, 1997b).

The impact of buffer zones to water quality was analysed in Vända, Sipe and Porijõgi upper subcatchments of Porijõgi River catchment. The subcatchments of Vända and Sipe have similar physico-geographical conditions, situating in central part of Porijõgi River catchment. Currently, 68% of Vända and 58% of Sipe subcatchments are used as agricultural land (Table 2). The Porijõgi upper subcatchment has only 6.4% agricultural land, 79% of the catchment is covered by forest. This subcatchment can be used as control area to describe natural outflow rate in the region. In Estonia, the most intensive use of agricultural lands took place at the end of Soviet period, from 1985 to 1990 (Mander and Palang, 1994). For instance, in 1987 the average fertilisation intensity was 150 kg N ha^{-1} and 60 kg P ha^{-1}. Since 1990 the use of fertilisers dropped and constituted in 1994 only 2.3% N and 0.8% P of the level in 1987. In the Porijõgi catchment, the fertilisation intensity followed the tendency for the whole Estonia. In our study, outflow values for N and P are calculated as average for 1987-90 (Table 2). The results show very high outflow of nitrogen (24.4 kg ha^{-1} yr^{-1}) and phosphorus (0.67 kg ha^{-1} yr^{-1}) from the Vända subcatchment. It is 9.0 and 7.4 times higher than the outflow from the natural Porijõgi upper subcatchment, respectively. At the same time, from Sipe subcatchment with similar intensity of agricultural use, the outflow of nitrogen and phosphorus were only 1.9 and 1.6 times higher than in the natural subcatchment, correspondingly. Although Sipe stream has 51% percentage of unbuffered stream banks, the outflow of N and P is much lower than that in the Vända subcatchment with 63% unbuffered stream banks. One of the explanations is more complex landscape pattern in Sipe subcatchment that shows better buffering capacity. Here in addition to buffer zone complex structure of landscape plays important role (45% of the 6.6 km long main stream is buffered), the main stream has meandering valley with a well-developed hyporheic zone. The Vända stream is completely straightened, has no meandering parts and only very short fragment of it has buffer zone on both banks that constitutes 7% of the total length (4.8 km) of the stream.

Table 2. Riparian buffer zones, natural communities adjacent to streams, and outflow of N and P (kg ha^{-1} yr^{-1}) from three subcatchments of the Porijõgi River.

Subcatchment (stream)	Total area (ha)	Area of agri-cultural land (ha)	Percentage of buffered stream banks (B)	Percentage of unbuffered stream banks (U)	Percentage of stream banks with natural communities (N)	Outflow of total - N (kg ha^{-1} yr^{-1})	Outflow of total - P (kg ha^{-1} yr^{-1})
Vända	220	150	6	63	31	24.4	0.67
Sipe	900	518	32	51	17	5.0	0.14
Porijõgi upper	1230	79	6	1	93	2.7	0.09

Constructed wetlands for wastewater treatment

Budgets of organic matter (BOD_5), total-N and total-P of three constructed wetlands in southern Estonia (a sand/plant filter; a combined overland flow-vertical flow root-zone system on a *Phalaris*-slope, and an aquatic macrophyte channel, a "bioditch") show their high retention capacity and removal efficiency (Table 3). Except for nitrogen, the efficiency of the sand/plant filter was found satisfactory. Based on average input and output values of BOD_5 (173 and 28 mgO l^{-1}, respectively) the purification efficiency of organic matter was 84%. For total-N and total-P the purification efficiency was 39% and 76%, correspondingly. Input values of N and P in this system were 40.5 mgN l^{-1} 10.9 mgP l^{-1}, respectively, and output values 24.8 mgN l^{-1} and 2.9 mgP l^{-1}, respectively. In the *Phalaris*-slope, 76% of organic matter (input 17, effluent 4 mgO l^{-1}), 69% of total-N (input 16. effluent 5 mgN l^{-1}) and 83% of total-P (input 4.1, effluent 0.7 mgP l^{-1}) was removed. Due to high input load, the BOD_5, total-N, and total-P values in the outlet of the bioditch were high and extremely varying: 1-62, 1.5-17.2, and 0.1-4.6 mg l^{-1}, respectively. However, even in the "overloaded" conditions the purification efficiency was high: 84% for BOD_5 (average input 10.9 and effluent 2.9 mgP l^{-1}), 70% for total-N (10.9 and 2.9 mgP l^{-1}, respectively) and 73% for total-P (10.9 and 2.9 mgP l^{-1}, respectively). To guarantee the recommendable output values from the bioditch, either a significant decrease of input load or an enlargement of the area of wetland should be achieved. All results also show that compared to other seasons the winter performance was not reduced. The retention and removal efficiency values in constructed wetlands for wastewater treatment in Estonia coincides with results obtained from literature (Table 3). Mitsch (1994) showed that restored or constructed riparian wetlands established to control non-point pollution could also achieve relatively high retention capacity. For instance, in the Des Plaines River riparian wetlands in Illinois the phosphorus removal efficiency is 63-98%. The relatively low rate of retention (0.001-0.01 g m^{-2} d^{-1}) is caused by lower input values and less activity of micro-organisms in these seminatural systems.

Table 3. Main characteristics of some constructed wetlands for wastewater treatment.

Constructed wetland	Area (m^2)	Flow rate (m^3 d^{-1})	Load (g m^{-2} d^{-1})			Retention (g m^{-2} d^{-1})			Removal efficiency (%)		
			I	II	III	I	II	III	I	II	III
Sand/plant filter (Põlva, Estonia) [1]	90	2	3.8	0.8	0.23	3.2	0.3	0.18	84	39	76
Phalaris-slope (Koopsi, Estonia) [1]	2400	130	1	0.9	0.23	0.2	0.3	0.19	76	69	83
Bioditch (Rakke, Estonia) [1]	140	125	40	4.9	1.43	12.8	1.7	0.29	84	70	73
Free water surface constructed wetlands (range)				0.1-5.6[2]	0.01-0.16[3]		0.03-3.9[2]	0.005-0.14[3]		35-75[2]	46-80[3]
Subsurface flow constructed wetlands (range)				0.1-8.6[2]	0.27-1.64[3]		0.02-5.6[2]	0.03-1.37[3]		45-75[2]	8-89[3]
Des Plaines River wetlands (USA; range) [3]					0.001-0.01			0.001-0.01		55-80	63-98

[1]- after Mander and Mauring, 1997; [2]- after Mander and Mauring, 1995; [3]- after Mitsch, 1994, in g m^{-2} yr^{-1}; I - BOD_5; II - N; III - P

In many cases, especially, by optimal hydrogeological conditions, the building and maintenance of constructed wetlands for wastewater treatment is more cost-effective than the rehabilitated conventional or prefabricated systems. The cost estimation was made for 8 small settlements with load range 100 to 710

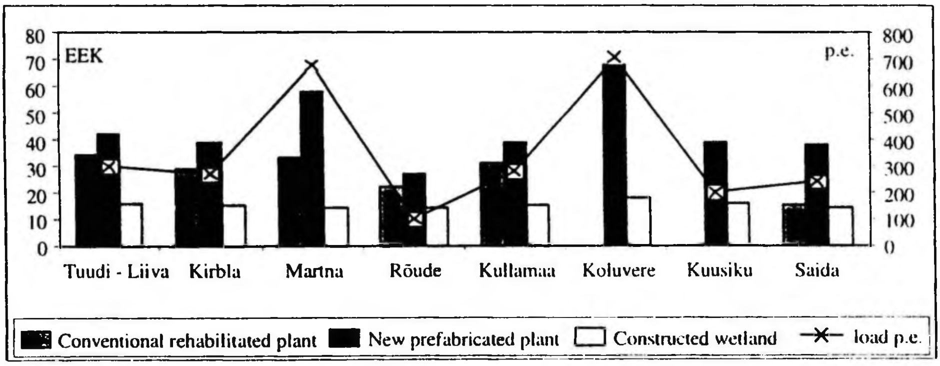

Figure 1. Comparison of annual operation and maintenance costs of three different systems for wastewater treatment in villages of the Matsalu catchment (in Estonian crowns, EEK; 1 DEM = 8.00 EEK).

population equivalent (p.e.) in the Matsalu Bay catchment, West Estonia, where three different options for every settlement were compared with equal influent-effluent quality conditions. The study was executed by the Norconsult International A.S. (Norway), Center for Soil and Environmental Research (Jordforsk, Norway) and local authorities (Ministry of Environment; Centre for Ecological Engineering Tartu) with the support of The World Bank (Norconsult, 1996). The analysis showed that building costs of constructed wetlands vary very much depending on conditions for building. In some extreme cases (karst, not protected ground water) building costs of constructed wetlands can be slightly higher than for new prefabricated treatment plants. Results of the study demonstrate that constructed wetlands are 1.5-4 times cheaper to manage and maintain than the conventional ones (Fig. 1). In Estonia, both natural and artificial wetlands have been used for purification of polluted water from agricultural fields and various point-pollution sources (Mander and Mauring, 1997). These systems have a potential for treatment wastewater from single houses, villages, small towns, tourist resorts, farms and landfills, as well as from some industrial areas of marginal location. However, in towns, settlements with central channelization, and majority of industrial objects, the wastewater is purified by various conventional (traditional) treatment plants.

CONCLUSIONS

Ecotechnological measures give good possibilities for controlling nutrient fluxes in watersheds. The most attractive multi-functional mitigation element for watershed management will be creation of buffer zones and buffer strips along ditches, channels and riverbanks. Buffer zones and buffer strips have number of positive functions. Some are important to reduce the negative environmental impacts to the water others are beneficial to the landscape and biodiversity. Buffer strips and zones allow stopping pollution at the source where it arises and reducing expenses of measures for improvement water quality in the stream.

Constructed wetlands have been used mostly for purification wastewater, also for improvement water quality in the streams, less for non-point pollution purification. Constructed wetlands analysed in this paper showed different appearance with respect to BOD, total-N and total-P. Good purification efficiency also in cold climate conditions allows widen field of use of constructed wetlands. Prototypes of all studied wetlands could be used for both main treatment systems or in conjunction with other treatment methods especially in rural areas, small settlements, single houses, motels etc. They can be used in the gullies, by the manure storage places, for purification of rainwater from technically used surfaces (manufacturing areas, roads). However, constructed wetlands are not suitable for all purposes and they can not be seen as alternative to the conventional systems in every place. The proper design criteria must be followed to guarantee required water quality standards.

Ecotechnological measures are cost-effective to control non-point pollution in intensively used watersheds.

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REFERENCES

APHA. (1989). *Standard Methods for the Examination of Water and Waste Water*, 15th edition, American Public Health Organisation. 17th edition. Washington.

Brix, H. (1998). Denmark. *Constructed Wetlands for Wastewater Treatment in Europe*. Vymazal, J., Brix, H., Cooper, P.F., Green, M.B., Haberl, R. (eds.), Backhuys Publishers, Leiden, pp 123-152.

Geller, G., Englmann, E., Haber, W., Kleyn, K., Lenz, A. and Netter, R. (1991). Bewachsene Bodenfilter zur Reinigung von Wässern - ein von Abwasser beeinflußtes System aus Boden und Pflanzen. *Wasser und Boden*, **8**, 477-495.

Granstedt, A. (1990). Case studies on the flow and supply of nitrogen in alternative farming in Sweden. *Alternative Agriculture. Swedish University of Agricultural Sciences. Research Committee for Alternative Agriculture*, Uppsala, pp 52.

Guterstam, B. (1991). Ecological engineering for wastewater treatment: Theoretical foundations and practical realities. *Ecological Engineering for Wastewater Treatment*. Etnier, C. and Guterstam, B. (eds.) Bokskogen, Gothenburg, pp 38-54.

Heeb, J. and Züst, B. (1991). Sand and plant filter systems. *Ecological Engineering for Wastewater Treatment*. Etnier, C. and Guterstam, B. (eds.) Bokskogen, Gothenburg, pp 199-207.

Haycock, N. and Pinay, G. (1993). Nitrate retention in grass and poplar vegetated riparian buffer strips during the winter. *J. Environ. Qual.*, **22**, 273-278.

Jenssen, P.D., Mæhlum, T. and Krogstad, T. (1993). Potential use of constructed wetlands for wastewater treatment in northern environments. *Wat. Sci. Tech.*, **28**, 149-157.

Kadlec, R.H. and Knight, R.L. (1996). *Treatment Wetlands*. Lewis Publishers, New York, pp 893.

Knauer, N. and Mander, Ü. (1989). Untersuchungen über die Filterwirkung verschiedener Saumbiotope an Gewässern in Schleswig-Holstein. *1. Mitteilung: Filterung von Stickstoff und Phosphor. Zeitschrift für Kulturtechnik und Landentwicklung*, **30**, 365-376.

Lowrance, R.R., Todd, R., Fail, J., Hendrickson, O., Leonard, R. and Asmussen, L. (1984). Riparian forest as nutrient filters in agricultural watersheds. *BioScience*, **34**, 374-377.

Mander, Ü. and Palang, H. (1994). Changes of landscape structure in Estonia during the Soviet period. *GeoJournal*, **33**, 45-54.

Mander, Ü., Kuusemets, V. and Ivask, M. (1995). Nutrient dynamics of riparian ecotones: A case study from the Porijõgi River catchment, Estonia. *Landscape and Urban Planning*, **31**, 333-348.

Mander, Ü. and Mauring, T. (1995). Nitrogen and phosphorus retention in natural ecosystems. *Functional Appraisal of Agricultural Landscape in Europe (EUROMAB and INTECOL Seminar 1992).* Ryszkowski, L. and Balazy, S. (eds.). Research Centre for Agricultural and Forest Environment, Pol. Acad. Sci. Poznan, pp 77-94.

Mander, Ü. and Mauring, T. (1997). Constructed wetlands for wastewater treatment in Estonia. *Wat. Sci. Tech.*, **35**, 323-330.

Mander, Ü., Lõhmus, K., Kuusemets, V. and Ivask, M. (1997a). The potential role of wet meadows and grey alder forests as buffer zones. *Buffer Zones, Their Processes and Potential in Water Protection. Proc. of the Int. Conf. on Buffer Zones.* Haycock, N.E. *et al.* (eds.). Oxford, Quest Environmental, Foundation for Water Research, pp 35-46.

Mander, Ü., Kuusemets, V., Lõhmus, K. and Mauring, T. (1997b). Efficiency and dimensioning of riparian buffer zones in agricultural catchments. *Ecological Engineering*, **8**, 299-324.

Mitsch, W.J. and Jørgensen, S.E. (1989). Ecological Engineering: An Introduction to Ecotechnology. John Wiley and Sons, New York, pp 472.

Mitsch, W.J. (1994). The nonpoint source pollution control function of natural and constructed wetlands. *Global Wetlands: Old World and New*. Mitsch, W.J. (ed.). Elsevier Science B.V. Amsterdam, pp 351-361.

Norconsult. (1996). Haapsalu and Matsalu Bays Environment Project. Point Source Water Pollution Control on the Matsalu Catchment. Final Report.

Peterjohn, W.T. and Correll, D.L. (1984). Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology*, **65**, 1466-1475.

Van Mansvelt, J.D. and Mulder, J. (1993). European features for sustainable development: a contribution to the dialogue. *Landscape and Urban Planning*, **27**, 67-90.

Vought, L.B.-M., Dahl, J., Pedersen, C.L and Lacoursière, J.O. (1994). Nutrient retention in riparian ecotones. *Ambio*, **23**, 342-348.

CURRICULUM VITAE

Valdo Kuusemets

Date of Birth 09.17.1962

Education

1981–1986	University of Tartu, Institute of Geography
1995–present	University of Tartu, Institute of Geography, Ph. D. Student

Professional experience

1986–1988	The Inspection of Water Protection Inspector of water protection
1988–1992	Environmental Research and Design Co. "ELVI-Aqua" Researcher
1992–1997	Centre for Ecological Engineering Tartu Executive director

Scientific activities

Research grants:

1995	Estonian Scientific Foundation: "Study on wastewater purification efficiency in natural and constructed ecosystems" "Using of energy forest for waste water treatment"
1996–1998	Estonian Scientific Foundation: "Study of natural wastewater purification systems"

Participation in projects:

1992–1995	"Investigations of Nutrient Cycling in Agricultural Watersheds" granted by Estonian Scientific Foundation
1993–1995	"Nitrogen Dynamic in Riparian Alder Forests" granted by the International Science Foundation
1995–1996	The World Bank EIA for "Vigala Drainage Rehabilitation Project"
1996	The World Bank project "Point Source Water Pollution Control on the Matsalu Catchment"
1997–1998	The World Bank Project "Estonian Wetlands Conservation and Management Strategy"

1997–present EU INCO-COPERNICUS Project "Research and Information on Water Purification by Means of Constructed Wetlands in the Ukraine, supported by studies in Estonia, Finland, The Netherlands and Sweden"

Professional membership:
International Ecological Engineering Society
International Association for Landscape Ecology
Estonian Geographical Society
Estonian Water Union

Scientific interest:
Landscape ecology, ecological engineering: design of buffer-strips, wastewater treatment with natural methods in constructed wetlands

Publications

Kuusemets, V., Mauring, T. 1996. Wastewater purification in a willow plantation. The case study at Aarike. In: Perttu, K. and Koppel, A. (eds.). *Short Rotation Willow Coppice for Renewable Energy and Improved Environment.* Publ. Swedish University of Agricultural Sciences, report 57, 159–165.

Kuusemets, V., Mauring, T. and Koppel, A. 1996. Domestic Wastewater Purification in Energy Forest: Case study of a Willow Plantation, Estonia. *Pre prints of 5th International Conference on Wetland Systems for Water Pollution Control held at Vienna, Austria 15 – 19 September 1996.* Vol. 1, VIII/7-1 — VIII/7-6.

Kuusemets, V., Mander, Ü., Ivask, M. and Lõhmus, K. 1998. Phosphorus retention in riparian buffer zones in agricultural landscapes in Estonia. In: R. H. Foy and Dils, R. (eds.). *Practical and Innovative Measures for the Control of Agricultural Phosphorus Losses to Water.* OECD Sponsored Workshop at Greenmount College of Agriculture and Horticulture, Northern Ireland, 16–19 June 1998, Workshop paper abstracts and posters, pp: 114–115.

Kuusemets, V. and Mander, Ü. 1999. Ecotechnological measures to control nutrient losses from catchments. *Water Science and Technology.* Accepted.

Kuusemets, V., Mander, Ü., Lõhmus, K. and Ivask. M. 1999. Shallow Groundwater Quality and Plant Uptake of Nitrogen and Phosphorus in Complex Riparian Buffer Zones. *Journal of Environmental Quality.* Submitted.

Mander, Ü., Koppel, A., **Kuusemets, V.** and Mauring, T. 1996. Energy forest use in wastewater treatment in Estonia. In: Staudenmann, J., Schönborn, A., Etnier, C. (eds.). Recycling the Resource: Ecological Engineering for Wastewater Treatment. *Environmental Research Forum* 5–6, pp. 189–197.

Mander, Ü., Kull, A. and **Kuusemets, V.** 1998. Land use change and nutrient flows in a rural catchment. In: Dover, J. W. and Bunce, R. G. H. (eds.). *Key Concepts in Landscape Ecology.* UK IALE, Colin Cross Printers Ltd., Garstang, pp. 103–109.
Mander, Ü., Kull, A., Tamm, V., **Kuusemets, V.** and Karjus, R. 1998. Impact of climatic fluctuations and land use change on runoff and nutrient losses in rural landscapes. *Landscape and Urban Planning*, 41, 3–4, 229–238.
Mander, Ü., Kull, A. and **Kuusemets, V.** 1999. Nutrient flows and land use change in a rural catchment: A modelling approach. *Landscape Ecology.* Accepted.
Mander, Ü., Kull, A., **Kuusemets, V.** and Tamm, T. 1999. Nutrient run-off dynamics in a rural catchment: influence of land-use changes, climatic fluctuations and ecotechnological measures. *Ecological Engineering.* Accepted.
Mander, Ü., **Kuusemets, V.** and Treier, K. 1994. Variation in groundwater quality in South Estonian rural areas. In: *Groundwater Quality Management* (Proceedings of the GQM 93 Conference held at Tallinn, Sept. 1993). *IAHS Publ.* No 220, 81–93.
Mander, Ü., **Kuusemets, V.** and Ivask, M. 1995. Nutrient dynamics of riparian ecotones: A case study from the Porijõgi River catchment, Estonia. *Landscape and Urban Planning*, 31, 333–348.
Mander, Ü., **Kuusemets, V.**, Järvet, A., Häberli, K., Nõges, T., Tuvikene, A. and Mauring, T. 1997. Ecological engineering for wastewater control in agricultural catchment areas: Three case studies from Estonia. In : C. Etnier and B. Guterstam (eds.). *Ecological Engineering for Wastewater Treatment.* New York, CRC/Lewis, pp. 263–286.
Mander, Ü., **Kuusemets, V.**, Lõhmus, K. and Mauring, T. 1997. Efficiency and dimensioning of riparian buffer zones in agricultural catchments. *Ecological Engineering*, 8, 299–324.
Mander, Ü. and **Kuusemets, V.** 1998. Ecotechnological measures to minimize nutrient losses from agricultural watersheds. In: O. Van Cleemput, S. Haneklaus, G. Hofman, E. Schnug and A. Vermoesen (eds.). *Fertilization for Sustainable Plant Production and Soil Fertility.* Procedings of the 11th International World Fertilizer Congress, September 7–13, 1997, Gent, Belgium. CIEC, FAL, Braunschweig-Völkenrode, Vol. II, pp. 526–537.
Mander, Ü. and **Kuusemets, V.** 1998. Non-point pollution control by riparian buffer strips: Dimensioning and efficiency assessment. In: *Proceedings of the 3rd International Conference on Diffuse Pollution, Edinburgh, Scotland, 31 August – 4 September 1998*, pp. 136–143.

Mander, Ü., Lõhmus, K., **Kuusemets, V.** and Ivask, M. 1997. The potential role of wet meadows and grey alder forests as buffer zones. In: Haycock, N. E., Burt, T. P., Goulding, K. W. T. and Pinay, G. (eds.). *Buffer Zones, Their Processes and Potential in Water Protection*. Proceedings of the International Conference on Buffer Zones, September 1996. Part II. Oxford, Quest Environmental, Foundation for for Water Research, pp. 35–46.

Mander, Ü., Tamm, V., **Kuusemets, V.** and Karjus, R. 1996. Possible influence of climate change on runoff and nutrient losses in rural landscapes: Porijõgi River catchment, Estonia. In: S. Obenauf und J. Rogasik (Hrsgb.). 1. Workshop Klimaveränderung und Landbewirtschaftung — Landwirtschaft als Verursacherin und Betroffene. *Landbauforschung Völkenrode* (Sonderheft) 165, 105–122.

Paal, J., Ilomets, M., Fremstad, E., Moen, A., Børset, E., **Kuusemets, V.**, Truus, L., Leibak, E. 1998. Estonian Wetlands Inventory 1997. Eesti Loodusfoto, Tartu, pp. 166.

CURRICULUM VITAE

Valdo Kuusemets

Sünniaeg 17.09.1962

Haridus

1981–1986 Tartu Ülikool, geograafia osakond
1995– Tartu Ülikool, geograafia instituut, loodusgeograafia ja maastikuökoloogia õppetool, doktorant

Teenistuskäik

1986–1988 Tartu Territoriaalne Vee Kasutamise ja Kaitse Valitsus, veekaitseinspektor
1988–1992 Keskkonna Uurimise ja Projekteerimise Ettevõte "ELVI-Aqua", insener
1992–1997 Tartu Ökoinsenerikeskus, tegevdirektor

Teadusorganisatsiooniline ja -administratiivne tegevus

Eesti Teadusfondi grantide hoidja:
1995 "Tehis- ja looduslike ökosüsteemide reoveepuhastusvõime uurimine"
"Energiavõsa kasutmine reovee puhastuses"
1996–1998: "Looduslähedaste reoveepuhastussüsteemide uurimine"

Osalemine projektides:
1992–1995 ETF grant "Valgala põllumajandusmaastikul: aineringe ja selle mõjutamine insenerökoloogiliste vahenditega"
1993–1995 Rahvusvahelise Teadusfondi grant "Lämmastiku ringe veekoguäärsetes lepikutes"
1995–1996 Maailmapanga keskkonnamõjude hinnang projektile "Vigala kuivendussüsteemide korrastamine"
1996 Maailmapanga projekt "Matsalu valgala punktreostusallikate reostuskontroll"
1997–1998 Maailmapanga projekt "Eesti märgalade kaitse ja majandamise strateegia"
1997– EL INCO-COPERNICUS'e projekt "Tehismärgalade veepuhastuse uurimine Ukrainas koos toetavate uuringutega Eestis, Soomes, Hollandis ja Rootsis"

Erialaliitude liige:

Rahvusvaheline Ökotehnoloogia Ühing
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