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Editorial

Purification processes, ecological functions, planning and design of riparian buffer zones in agricultural watersheds

Abstract

Two international meetings on ecological engineering, with a focus on riparian buffer zones, served as the source for selected papers in this special issue: (1) an International Workshop on Efficiency of Purification Processes in Riparian Buffer Zones: Their Design and Planning in Agricultural Watersheds, jointly organised by Hokkaido University, Japan, the National Agricultural Research Center for Hokkaido Region, Japan, Civil Engineering Research Institute of Hokkaido, Japan, and the Institute of Geography, University of Tartu, Estonia, and held from 5 to 9 November 2001 in Kushiro City, Hokkaido, Japan; and (2) an International Conference on Ecological Engineering for Landscape Services and Products, jointly organised by the International Ecological Engineering Society (IEES) and Lincoln University, Christchurch, New Zealand, and held from 25 to 29 November 2001 in Christchurch, New Zealand. At these two meetings, altogether 94 oral presentations (17 from invited speakers) and 15 posters by representatives from 21 countries were presented. The editorial paper highlights trends in investigation of the purification processes in riparian buffer zones as well as planning, design and management aspects of riparian buffers regarding the wide spectrum of their ecological functions; it characterises the two international meetings which served as sources for the selected papers and briefly explains the main aspects of these papers.

Keywords: Ecological functions; Landscape planning; Nitrogen removal; Phosphorus retention; Riparian buffer zones; River restoration

1. Introduction

The riparian landscape is unique among environments because it is a terrestrial habitat strongly affecting and affected by aquatic environments (Malanson, 1993). The spatial structure and interaction with surrounding ecosystems are two typical features of riparian buffer zones. Naiman and Decamps (1997) highlight their role as terrestrial–aquatic ecotones, whereas Forman (1995) and Farina (2000) prefer to view them as corridors. In both the cases, the riparian zones can act as conduits, filters or barriers controlling flows of energy, matter and species in landscapes.

In many countries, riparian buffer zones are well-known measures in ecological engineering at a catchment scale (Muscutt et al., 1993; Kuusemets and Mander, 1999). However, this knowledge has to be distributed to all the areas suffering conflicts between water and habitat quality in rivers/lakes and intensive management of their catchments. On the other hand, not all the ecological functions are currently considered when planning and designing riparian buffer zones. This special issue tries to fill some gaps in both the knowledge of key purification processes and various ecological functions of riparian buffer zones as well as their planning, design and management practices.

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The aims of this editorial paper are: (1) to highlight trends in the investigation of purification processes in riparian buffer zones, and of the planning, design, and management aspects of riparian buffers regarding the wide spectrum of their ecological functions, (2) to characterise the two international meetings that served as sources for the selected papers, and (3) to briefly characterize the main aspects of these selected papers.

2. Ecological functions of riparian buffer zones

Some of the most important multifunctional elements of the ecological network are various riparian biotopes which have the following essential functions (see also Lowrance et al., 1997): (1) to filter polluted overland and subsurface flows from intensively managed adjacent agricultural fields, (2) to protect banks of water bodies against erosion, (3) to filter polluted air, especially from the local sources (e.g., big farm complexes, agrochemically treated fields), (4) to lessen the intensive growth of aquatic macrophytes via shading by canopies, (5) to improve the microclimate in adjacent fields, (6) to create new habitats in land/inland–water ecotones, and (7) to create more connectivity in landscapes due to migration corridors and stepping stones.

The following sections will highlight some important functions of riparian buffers based on the selected research results.

2.1. Filtering polluted overland and subsurface flows

This is the key function of buffer zones. Several case studies worldwide suggest that different riparian ecosystems can significantly decrease the nitrogen and phosphorus concentrations in both overland flow and in groundwater (Peterjohn and Correll, 1984; Pinay and Decamps, 1988; Knauer and Mander, 1989; Jordan et al., 1992; Vought et al., 1994; Mitsch, 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: see Pinay et al., 1993; Weller et al., 1994; nitrification: see Watts and Seitzinger, 2000; Wolf and Russow, 2000). Various biophysical conditions control the intensity of these processes; the variability of their intensity is, therefore, very high. For instance, gaseous emissions and plant uptake can vary from <1 to 1600 and from <10 to 350 kg N ha^{-1} year^{-1}, respectively (Mander et al., 1997). Thus, different processes can play a leading role in nitrogen removal.

Storage of phosphorus in riparian buffer zones depends on the following processes: (1) soil adsorption, (2) removal of dissolved inorganic phosphorus by plant uptake, and (3) microbial immobilisation 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 (Mander et al., 1997). However, accumulated P can also be released from the wetland soils of riparian zones, especially after lowering of the input concentrations (Richardson and Marshall, 1986; Vanek, 1991).

Based on the results obtained during about 30 years, three important aspects in the buffering capacity of buffer zones can be highlighted:

(1) Removal of materials (suspended solids, nutrients, organic material, heavy metals, pesticides) has a non-linear character: in the first part of the buffer (0–5 m from the field-buffer borderline), significantly more material (20–60%) is retained than in the remote parts of the buffering ecosystem (Doyle et al., 1977; Knauer and Mander, 1989, 1990; Vought et al., 1994). The removal process can be described by the following equation (Mander, 1989):

$$C_L = (1 - e^{-kL}) \times 100\%$$

where C_L is the change in concentration (%) at distance L (m) from the buffer boundary, k the removal rate coefficient (m^{-1}); $k = (\ln C_1 - \ln C_2)/L$, where C_1 is the initial concentration at the field-buffer boundary and C_2 is the concentration at the distance L from the boundary.

(2) A strong linear regression has been found between logarithmic values of both N and P initial load (x) and mass removal (y) in buffer strips (Mander and Mauring, 1994):

$$\text{nitrogen}: \quad y = -0.194 + 0.948x \qquad (R^2 = 0.98, n = 26)$$

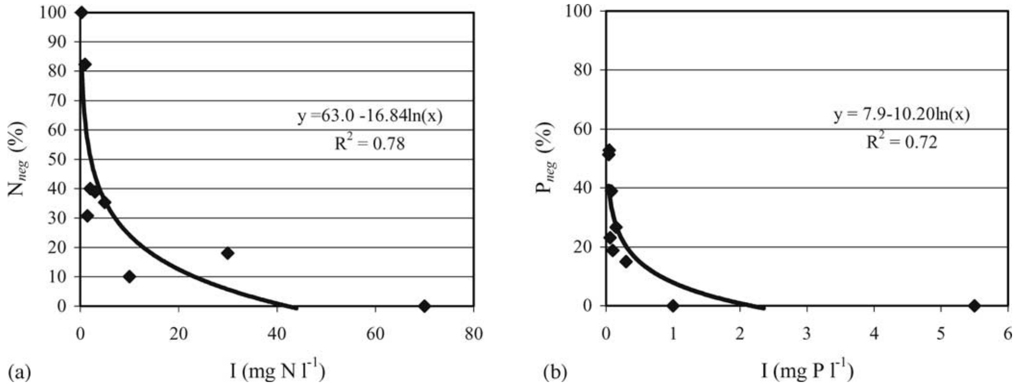

Fig. 1. The relation between input concentration (I, mg l^{-1}) and probability of negative removal of (a) nitrogen and (b) phosphorus (N_{neg} and P_{neg}, %, respectively). Adopted from Kuusemets et al., 2001.

$$\text{phosphorus}: \quad y = -0.184 + 0.967x \quad (R^2 = 0.99, n = 11).$$

However, the relative removal efficiency *y/x* for both N and P is decreasing when *x* increases. Although the correlation between *y/x* and load (log *x*) is not high, the plots clearly demonstrate the decreasing trend of *y/x* values (Mander et al., 1997).

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 rainfalls and/or intensive thaws). This phenomenon has been documented in several studies on natural buffer strips (Knauer and Mander, 1989) and has also been demonstrated in experimental plots (Dillaha et al., 1989; Magette et al., 1989).

The N and P mass removal in buffer zones can be negative when the input value is lower than a certain threshold (e.g., <0.3 mg N l^{-1}). On the other hand, the purification efficiency was always positive when the input value exceeded a certain value (5 mg N l^{-1} and 0.15 mg P l^{-1}; Fig. 1; Kuusemets et al., 2001). For instance, due to a significant decrease in agricultural intensity in Eastern Europe in the last 12 years, the nutrient losses from fields have dropped but the buffers' outflow values have not changed, i.e., being sometimes higher than inflow concentrations (Kuusemets et al., 2001).

(3) Complex buffer zones consisting of different sequential plant communities and soil complexes have higher purification efficiency than those with a single structure. For instance, a heavily loaded complex buffer zone consisting of grass and forest strips showed relatively low output concentrations for total N and total P, which are comparable with the output values from the unloaded catena (Kuusemets et al., 2001). Also, forest buffer strips on the stream banks between the sedge fens (or wet meadows) can remove the released phosphorus. Therefore, a combination of grasslands (wet meadows) as wider buffer zones (10–50 m) and forest/bush communities as buffer strips (5–10 m) on stream banks is the most optimal structure of riparian buffer communities (see also Vought et al., 1994). This is the same structure as is naturally formed in floodplains and on stream banks.

2.2. Shading effect

Large quantities of aquatic plants in watercourses reduce water flow and increase the risk of water flooding adjacent land. Intensive macrophyte growth causes silting of the watercourse and this, in turn, accentuates the problem. Therefore, since according to law in most countries of the temperate zones, streams and rivers are primarily managed for land drainage and for reduction of flood risk and ecologically based reasons are typically not considered, mechanical removal of aquatic macrophytes and some of the marginal vegetation once or twice a year, combined with occasional dredging of the channel, is the most common practice of watercourse management. This method very much disturbs the trophic structure of the stream ecosystem and, as reported by different authors, leads to more intensive macrophyte growth than before the treatment (Böttger, 1978). Several authors have suggested that trees and bushes on stream banks can be used as a very effective control of macrophyte growth in channels (Lohmeyer and Krause, 1975; Krause, 1977; Böttger, 1978; Binder, 1979; Dawson and Kern-Hansen, 1979; Bobrowski and Böttger, 1983). This practice gives a better effect in the long-term perspective and is even two times less expensive than the mechanical treatment of watercourses (Krause, 1977). Likewise, Dawson and Kern-Hansen (1979) refer to American studies that demonstrated that stream bank stability increased significantly to 85 and 95% of the initial value after trees and bushes were planted. Data and experience from Estonia coincide completely with the results of other studies in this respect (Mander, 1995). Fig. 2 shows significant correlation ($R^2 = 0.89$; $p < 0.001$) between the shading rate of the stream surface (S_n; relative light intensity measured at the water surface compared to that in the open) and aquatic macrophyte biomass (B) in lowland watercourses in South Estonia (Mander, 1995). An ecologically balanced level of aquatic plant biomass (about 100 g DW m^{-2}, shading rate about 0.2; Fig. 2) is recommended because too much shade is detrimental to the aquatic fauna and can lead to accentuated local accumulation of leaves (Dawson and Kern-Hansen, 1979). Leaves shed in the autumn can exert an oxygen demand locally (Slack and Feltz, 1968) but leaves are also an important food source for aquatic organisms. On the other hand, the technologically acceptable level of aquatic plant biomass (about 300 g DW m^{-2}, shading rate about 0.5; Fig. 2) means that macrophyte growth still enables water flow from drained areas and does not cause a substantial decrease in oxygen supply in critical periods (e.g., in the autumn and in morning hours). This level coincides with the results of other researchers (250 g DW m^{-2}; Jorga and Weise, 1977).

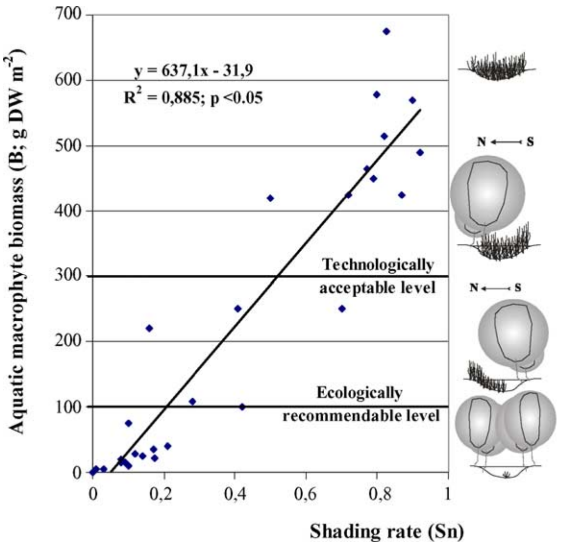

Fig. 2. Influence of shading (S_n) on the biomass of aquatic macrophytes (g DW m^{-2}) in lowland ditches of agricultural landscape in Estonia (adopted from Mander, 1995). Biomass below ecologically recommendable level causes disturbances in stream benthos ecosystem, particularly decreasing the biodiversity. Macrophyte biomass above the technologically allowable limit creates significant obstacles to water runoff.

Simplified schemes in Fig. 2 show the typical location of forest and bushes on stream banks leading to the respective shading rate and macrophyte growth. For instance, if the water is flowing towards the East or West, higher vegetation is recommended for the southern bank (see Jorga et al., 1982). All the investigations suggest that light should be reduced to about half of that presently available in the open (S_n = 50%; Fig. 2).

Another important aspect of canopy shading is related to temperature control in rivers, which regulates conditions in the entire stream ecosystem (Chen et al., 1998).

Modelling of riparian shade has been considered by Davies-Colley and Rutherford (2005). Parkyn et al. (2005) analyse the influence of riparian plantations on the stability of banks.

2.3. Filtering effect of canopies

In the event of the occurrence of local atmospheric pollution, e.g., application of fertilizers and pesticides from planes or helicopters, the filtering of pollutants by canopies plays an important role in stream protection. During the experiments carried out in Estonia, carbamide (urea; NH_4CONH_2) solution was applied as fertilizer on summer-barley fields and adjacent alder forest buffer strips, using planes (Mander, 1995). The mean application value (50 kg ha^{-1}) was the same in the fields and the adjacent buffer strips. Water samples taken from 0.9 m^2 gauges installed at different distances from the forest edge in the field and under the tree canopies showed a significantly lower loading in the buffer strips. Although the wind direction was different, only 5–10% of the carbamide was found to reach the soil surface under the canopies when compared to the values in the open field. This experiment also demonstrates a significant edge effect: higher loading values were found directly in the vicinity of the forest edge. Although some amount of carbamide was probably absorbed by leaves, the experiment suggests that filtering of atmospheric fluxes of wind-borne pollutants is an essential function of forest buffer zones.

There are several examples in landscape ecological studies about the role of riparian buffer zones as corridors connecting fragmented communities in the homogenized cultural landscape (Forman, 1995; Farina, 2000). Some aspects of ecological networks are considered in Meier et al. (2005).

3. The International Workshop on Riparian Buffer Zones in Kushiro, Japan

An International Workshop on Efficiency of Purification Processes in Riparian Buffer Zones: Their Design and Planning in Agricultural Watersheds was organised by the international organizing committee consisting of representatives from Hokkaido University, Japan, the National Agricultural Research Center for Hokkaido Region, Japan, the Civil Engineering Research Institute of Hokkaido, Japan, and the Institute of Geography, University of Tartu, Estonia. The workshop was held from 5 to 9 November 2001 in Kushiro City which is the primary city in Eastern Hokkaido.

The objectives of this workshop were: (1) to provide participants with new and innovative ecological methods for wastewater treatment which will help reduce pollution from agricultural fields and farms and (2) to introduce the latest knowledge about riparian buffer zones to Japan and specifically to Eastern Hokkaido, which is a core food production area. Japan's largest dairy farms are situated here. At the same time, extensive salmon/trout fishery is going on downstream. Thus, there is a clear conflict between the salmon/trout development and the dairy farming, which reduces the quality of the river water. There is an urgent need for effective countermeasures against such water pollution.

During the workshop, 30 oral presentations (13 from invited speakers) and 8 posters were presented. Contributions to this conference were received from 18 countries: Japan, USA, United Kingdom, France, Germany, Canada, The Netherlands, Denmark, Norway, Finland, Sweden, Estonia, Spain, Switzerland, Poland, Romania, China, and Korea. The keynote and volunteered papers as well as the posters were on the following topics: (1) significance of riparian buffer zones on nutrient discharge to the aquatic environment, (2) purification functions in buffer zones with different types of land use, (3) analytical aspects for purification functions in buffer zones, (4) ecological functions of buffer zones, and (5) design, dimensioning, management, and planning of buffer zones.

In addition, a Citizens Forum brought together specialists and local stakeholders to have a brainstorming session for solving the conflicts between the fishery and the dairy farmers. Riparian buffer zones and various constructed wetlands were considered as the most sustainable solutions for Hokkaido.

A workshop excursion was organised to the Kushiro Marsh, the Kushiro Catching Station, the Tokachi Region Branch of the National Salmon Resources Center and the Shibetsu River nature restoration project area, all in Eastern Hokkaido, Japan. Kushiro Marsh, the first Japanese Ramsar registered wetland with an overall area of 183 km^2, is located in a suburb of Kushiro City. Recently, Kushiro Marsh has been receiving undesirable nutrient input and sand from the surrounding agricultural area. The Shibetsu River nature restoration project area includes floodplain wetland construction and reconstruction of meanders (Fig. 3).

4. The IEES Conference on Ecological Engineering for Landscape Services and Products, Christchurch, New Zealand

An International Conference on Ecological Engineering for Landscape Services and Products was jointly organised by the International Ecological Engineering Society (IEES) and Lincoln University, Christchurch, New Zealand. The Conference was held from 25 to 29 November 2001 in Christchurch, New Zealand.

During the conference, 64 oral presentations (4 from invited speakers) and 7 posters were presented.

Fig. 3. The Shibetsu River nature restoration project area in Eastern Hokkaido, Japan. *Left*: Chashikotu Gate floodplain wetland construction site. *Right*: River meandering reconstruction site with planting experiments (see Uchida and Tazaki, 2005).

Fig. 4. Wetland and riparian ecosystems from New Zealand. *Left*: Restored wetlands in Christchurch. *Right*: Pristine riparian fern forest in the Franz Josef Glacier area.

Contributions to this conference were received from 12 countries: New Zealand, Australia, USA, Sweden, Germany, Canada, Austria, The Netherlands, Norway, Switzerland, Estonia, and Fiji. In addition to the plenary, contributed paper and poster sessions, 7 workshops were held on the following topics: (1) wetlands and aquatic systems for wastewater and stormwater management, (2) integrated riparian engineering, (3) ecological engineering for sustainable regional development: Banks Peninsula case study, (4) urban ecotechnology and infrastructure, (5) ecological engineering education, (6) ecological engineering and decision-making, and (7) planning an ecological farm-park for the city of Christchurch.

A conference excursion was organised to visit several water, waste, and landscape management sites in Christchurch and its suburbs: (1) the coastal dune restoration project area, (2) the wetlands and aquatic systems for wastewater and stormwater management, (3) the waste composting facility, (4) the semi-natural wetland restoration sites (Fig. 4, left), and (5) the Wigram wastewater retention basin.

5. This special issue

This special issue consists of 15 selected papers presented at two international scientific meetings: (1) at the Workshop on Efficiency of Purification Processes in Riparian Buffer Zones: Their Design and Planning in Agricultural Watersheds held from 5 to 9 November 2001 in Kushiro City, Japan, and (2) at the IEES

Conference on Ecological Engineering for Landscape Services and Products held from 25 to 29 November 2001 in Christchurch, New Zealand.

The thematic issues of this special issue concentrate on: (1) efficiency of purification processes in riparian buffer zones, (2) nutrient fluxes in buffer zones and the effect of buffer zones on the processes of nutrient discharge and soil erosion, (3) methods for measuring intensity of relevant purification in buffer zones, (4) ecological functions of riparian buffer zones, (5) landscape services of riparian buffers, and (6) methods for designing, planning, and managing riparian buffer zones.

In his overview paper, Correll (2005) presents basic principles for planning and design of riparian buffer zones. Knowles (2005) explains how losses of fixed nitrogen (N) can occur in riparian zones by the activity of denitrifying bacteria associated with methane-oxidizing (methanotrophic) bacteria. Several methanotrophs catalyze N cycle processes that can occur in riparian buffer zones, including nitrification and nitrogen fixation. Methanotrophs can produce nitric and nitrous oxides during oxidation of ammonium (nitrification) but they cannot carry out denitrification. However, there is a good evidence that denitrifying bacteria can be associated with methanotrophs and can use simple carbon compounds released by the methanotrophs as substrates for the denitrification reactions and for growth. In his paper, Knowles (2005) gives evidence that denitrifiers isolated from methanotrophic gel-stabilized oxygen gradient systems can use methanol, formaldehyde and formate, all methane oxidation intermediates, to support their denitrification. Such denitrification associated with methanotrophs can release dinitrogen and so contributes to losses of fixed nitrogen, and may also produce the important atmospheric trace gases (nitric and nitrous oxides). Data presented also show that some methanotrophs produce nitrogen oxides, including nitrite, nitric oxide, and nitrous oxide, during growth on nitrate. Assimilatory reduction of nitrate appears to be a requirement for the release of these products.

Revsbech et al. (2005) present the microzonation and coupling of the oxidative and reductive transformations in various environments at different scales (soil of flooded riparian zones, the rhizosphere of riparian plants, biofilms at solid surfaces in the river and the surface layer of sediments) and give results of studies that have recently been carried out with ^{15}N enrichment methods and microsensors for NH_4^+, NO_2^-, NO_3^- and N_2O. The exact Microsensor analysis of gradients in sediments and biofilms have shown that nitrate production takes place in an aerobic surface zone that has a maximum thickness of a few millimeters in most shallow water sediments and may be as thin as 100 μm in biofilms from very eutrophic environments. In the anoxic zone, denitrification is also concentrated in a zone of a few millimetres at most, and typically half of the nitrate produced by nitrification is denitrified while the other half escapes to the water. The supply of nitrate from above is primarily controlled by the oxic layer acting as a diffusion barrier and, therefore, denitrification is generally a linear function of the nitrate concentration in the water. The overlying water is thus a much more important source of nitrate for denitrification if the concentration is high. The rate and location of denitrification is also affected by bioturbating animals, benthic microphytes, plants and bacteria performing dissimilatory nitrate reduction to ammonium (DNRA).

Glindemann et al. (2005) give a review of previously published and unpublished results of research into the occurrence of phosphine (PH_3) in the environment in the form of matrix bound phosphine in (riparian) soils, aquatic sediments and sludges (ng kg^{-1} to $\mu g\,kg^{-1}$ range) and free phosphine in formed biogases (ng m^{-3} to $\mu g\,m^{-3}$ range) and in the atmosphere (pg m^{-3} to ng m^{-3} range). The reviewed data support the hypothesis of the existence of a small gaseous link in the phosphorus (P) cycle which could become important over the long-term period. Matrix-bound phosphine in soils can be interpreted as a stationary state concentration of phosphine between production and consumption. This phosphine turnover within the soil may be important even if the stationary state concentration (matrix bound phosphine) is small. Under such circumstances, a slow migration process of phosphine in the interstitial gas sphere of soils is possible. Such a process would influence the balance of P in agricultural and wetland soil. The detection of easily oxidizable phosphine as a ubiquitous trace gas in the atmosphere can be interpreted as the residue of an important turnover of phosphine between widely distributed emission sources and sinks such as soils and sediments. The atmosphere can carry gaseous P to remote places.

Hefting et al. (2005) present results of a study conducted within the framework of the research project NICOLAS funded by the EU. Forested and herbaceous riparian buffers were selected in six participating European countries: France, Switzerland, The Netherlands, Spain, Poland, and Romania (R). Generally, plant uptake and denitrification are considered to be the most important processes responsible for N retention and mitigation in riparian buffers. In many riparian buffers, however, nutrients taken up by plants remain in the system only temporarily and may later be gradually released by mineralization. Still, plants increase the residence time of nutrients considerably by reducing their mobility. Over a 2-year study period, the plant production, N uptake and N retention were significantly higher in the forested buffer sites compared to the herbaceous buffer sites. However, in herbaceous buffers, periodic harvesting of herbaceous biomass contributed considerably to the N retention. No relationship between lateral N loading and plant productivity or N uptake was observed; this indicated that plant growth was not N limited. In the winter period, decaying leaf litter had a small but significant role in N retention in a majority of the riparian ecosystems studied. Moreover, no responses to the climatic gradient were found. The annual N retention in the vegetation and litter compartment was found to be substantial, accounting for 13–99% of the total N mitigation.

In her study conducted in the Southern Norwegian agricultural landscape, Syversen (2005) studied design criteria which optimise the effect of buffer zones: (1) buffer zone width, (2) amount of surface runoff water entering the buffer zone, (3) seasonal variation, and (4) vegetation type. The simulation experiments were short-term experiments carried out over a few days in 1992 and 1993. In the natural runoff experiments, volume proportional mixed samples were collected after each runoff period during 1992–1999. The results show significantly higher removal efficiency (in %) by 10 m wide buffer zones compared to 5 m widths, however, the specific retention (per m^2) is higher in 5 m buffers. Retention efficiency between summer and autumn varied depending on the measured parameter (particles, P, and N), and there were no significant differences in removal efficiency between summer and winter. The results show no significant differences between forest buffer zones and grass buffer zones regarding their retention efficiency for N and P, however, retention efficiency for particles was significantly higher in forested buffers. Average removal efficiencies from both simulated and natural runoff experiments varied from 60–89%, 37–81% and 81–91% for P, N and particles, respectively.

Uusi-Kämppä (2005) has studied the retention of agricultural P by 10-m wide grass buffers (systematically mowed) and buffers under natural vegetation (not managed) for 10 years in southwestern Finland. The results were compared with those from 70-m long plots without buffers. Surface waters were directed into a collector trench on each plot. The highest losses of all P fractions were measured in spring when the buffer vegetation had not yet started to grow. The mean annual total phosphorus (TP) loss from the grass buffer and natural vegetation buffer plots ($0.7\,\mathrm{kg\,ha^{-1}}$) was 40% lower than the TP loss from the non-buffer plots ($1.2\,\mathrm{kg\,ha^{-1}}$). However, the loss of molybdate-reactive P (RP) was 70% higher from the natural vegetation buffer plot than from the other plots. On the soil surface (0–2 cm), in the same plots the concentration of Olsen-P was high ($55.9\,\mathrm{mg\,l^{-1}}$). The high loss of RP from the natural vegetation buffer plot was most likely due to P leaching from the soil surface and decaying grass residues in spring.

Nagasaka et al. (2005) and Hatano et al. (2005) studied the impact of agricultural land use on soil erosion and stream water quality in Hokkaido, Japan. Nagasaka et al. (2005) found that the concentration of suspended sediments was consistently higher in cultivated catchments, where gully expansion causes two to three times more landslides than occur in forested catchments. Sediment from gullies contributed about 34% of the total sediment in the cultivated catchment. There has been increasing erosion and sedimentation on the valley floor over the past 20 years both because of the expansion of land under cultivation and because the mechanization of agriculture since the 1960s has reduced the infiltration capacity of cropland, making it easier for erosion to occur when it rains. Most of the finer sediment is transported to the sea, where it affects coastal ecosystems, while the coarser sediment such as sand remains in the stream and fills the spaces between gravel on the streambed. This eliminates habitat suitable for fish and invertebrates; the density of macro-invertebrates in cultivated catchments is only 10–20% of that in forested catchments. Effective stream restoration and the establishment of riparian buffers are considered to be a sustainable way to decrease the erosion intensity and to protect rivers. In 1999–2000, Hatano et al. (2005) conducted a study to determine the impact of farm N budgets on stream water quality in an experimental livestock farm of 457 ha in the Kepau River watershed in Shizunai, Southern Hokkaido, Japan. Grasslands and maize fields accounted for 33% of the farm's total area. As calculated on the basis of the farm's land management records, livestock was supplied with 15.2 t N year^{-1} from agricultural lands which made the farm 81% self-sufficient. Livestock excreta produced 17.2 t N year^{-1}, of which 4 t N year^{-1} was lost, probably by ammonia volatilization during decomposition. Apart from manure, the major N inputs were 9.1 t N year^{-1} of chemical fertilizers, 6.4 t N year^{-1} of atmospheric deposition and 12.6 t N year^{-1} biological N fixation. The major outputs were uptake by forest vegetation of 11.0 t N year^{-1}, denitrification of 1.5 t N year^{-1} and livestock feed production. Consequently, the annual surplus N on the whole farm was estimated to be 12.7 t N year^{-1}, which corresponds to 28 kg N ha^{-1} of the agricultural land. Although the average N concentration of stream water below the farm was 2.8 mg N l^{-1}, the maximum concentration recorded during the snowmelt season was 13.5 mg N l^{-1}. The large N load during rainfall and snowmelt could be ascribed to open ditches which collect tile drainage and surface runoff from the fields, discharging it directly to the river and bypassing the forested riparian zone.

Anbumozhi et al. (2005) summarize the results of a field monitoring study on efficiency of riparian forest buffer systems in filtering sediment, nutrients and pesticides entering from upland agricultural fields in the Tokachikawa watershed in Hokkaido, Japan, in the Cisadane, Cianten and Citamyang sub-watersheds in Indonesia and in the Cauvery watershed, India. Stream water physical property values increased from upstream to downstream, influenced by the upland and by the livestock land use activities. The greatest reductions in the impairment of water quality were observed in buffer zones located along higher order streams where the gradient is very low, leading to slow groundwater movement. The lower stream water temperature in riparian buffer zones suggests that the shading effect is most pronounced in this area of the watershed. The results demonstrate the positive impact of forest buffer zones in reducing the influence of agricultural nutrients and chemicals on surface stream waters.

Likewise, Davies-Colley and Rutherford (2005) considered the shading effect of forested riparian buffer zones to be their important ecological function. The authors outlined a simple model for predicting the proportion of incident lighting at the channel centre as a function of channel dimensions (stream width) and riparian plant character (foliage density, canopy height). The model reproduces the broad empirical trend of increasing shade with increasing canopy height/stream width ratio. Those models are useful for optimal design and allocation of riparian buffers.

Meier et al. (2005) demonstrate in their case study on Clouded Apollo butterfly distribution in Estonian river corridors that riparian buffers play an important role in the migration of species. Therefore, the riparian buffers can be considered to be the significant elements of territorial ecological networks.

Ecological effects of riparian buffer zones have also been highlighted in the paper by Nakamura and Yamada (2005). Analysing various effects of pasture development on the ecological functions of riparian forests in Hokkaido, Japan, they particularly mention the benefits of woody and leaf debris, as well as lower water temperature under tree canopies, for anadromous fish whose population density is significantly higher in buffered streams than in grassland streams without forest buffers. In addition, fine sediment a prominent by-product of agricultural development adversely impacts periphyton productivity, the density and diversity of aquatic invertebrates, fish feeding, fish spawning and egg survival. Likewise, forested streams show significantly higher water quality than non-buffered ones. The authors also examine the adequate width of a riparian buffer regarding various ecological functions.

In their paper on predictions of stream nutrient and sediment yield changes following the restoration of a forested riparian buffer in New Zealand, Parkyn et al. (2005) highlight some negative aspects of riparian tree planting on water quality and stream habitat. According to them, shading of riparian pasture grasses can lead to channel widening, and riparian shade may limit the growth of macrophytes and algae that assimilate dissolved nutrients from the water column and create a substrate for periphyton. A simple model predicted that a buffer strip alongside a small headwater stream would reduce nutrient export, while a buffer strip instigated as an isolated patch alongside a larger stream (> ca. 2.5 km^2 upstream catchment size) would increase nutrient export since the relative amount of nutrients trapped by the buffer decreases as the nutrient load present in the stream water increases. In larger, approximately 6 m wide streams, however, sufficient light may reach the streambed for plant and algal growth which, in turn, would promote instream nutrient processing. The conceptual modelling exercise showed that riparian tree planting programmes should commence in the headwaters and progress downstream to avoid nutrient yield increases. Significant sediment yield from bank stored sediment of small streams can be expected until the channel reaches the more stable, original forested width, but progressive planting may decrease the peak loads of sediment.

Regarding the optimal design and establishment of riparian buffer zones, Uchida and Tazaki (2005) present a new method for the planting of riparian buffer zones. By simply placing segments of *Phragmites australis* culms by the water, the authors were able to create an area where *P. australis* flourished without the labour, time and money usually required for young shoot production (see experimental sites in Fig. 3, right). However, care is required in the mixed planting with other hydrophytes such as *Scirpus tabernaemontani*, *Zizania latifolia* and *Typha latifolia* due to the allelopathic potential of the interspecies. Therefore, a sufficient planting interval is required in order to avoid the allelopathic inhibition of root elongation.

The results of papers in this special issue can be used for modelling purification processes, as well as for optimal planning, design and management of riparian buffer zones.

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