Published January 17, 2019

Journal of Environmental Quality SPECIAL SECTION

RIPARIAN BUFFER MANAGEMENT

An Assessment of the Multifunctionality of Integrated Buffer Zones in Northwestern Europe

Dominik Zak,* Marc Stutter, Henning S. Jensen, Sara Egemose, Mette V. Carstensen, Joachim Audet, John A. Strand, Peter Feuerbach, Carl C. Hoffmann, Benjamin Christen, Sandra Hille, Mads Knudsen, Jenni Stockan, Helen Watson, Goswin Heckrath, and Brian Kronvang

Abstract

Integrated buffer zones (IBZs) have recently been introduced in the Northwestern Europe temperate zone to improve delivery of ecosystem services compared with the services associated with long-established vegetated buffer zones. A common feature of all the studied IBZ sites is that tile drainage, which previously discharged directly into the streams, is now intercepted within the IBZ. Specifically, the design of IBZs combines a pond, where soil particles present in drain water or surface runoff can be deposited, and a planted subsurface flow infiltration zone. Together, these two components should provide an optimum environment for microbial processes and plant uptake of nutrients. Nutrient reduction capacities, biodiversity enhancement, and biomass production functions were assessed with different emphasis across 11 IBZ sites located in Denmark, Great Britain, and Sweden. Despite the small size of the buffer zones (250–800 m^2) and thus the small proportion of the drained catchment (mostly <1%), these studies cumulatively suggest that IBZs are effective enhancements to traditional buffer zones, as they (i) reduce total N and P loads to small streams and rivers, (ii) act as valuable improved habitats for aquatic and amphibian species, and (iii) offer economic benefits by producing fast-growing wetland plant biomass. Based on our assessment of the pilot sites, guidance is provided on the implementation and management of IBZs within agricultural landscapes.

Core Ideas

- Integrated buffer zones are a novel edge-of-field approach within riparian zones.
- Drain water and surface runoff will be trapped within a pond and charge a filter bed.
- The inclusion of trees aims to provide some of the benefits of riparian forests.

J. Environ. Qual. 48:362–375 (2019)
doi:10.2134/jeq2018.05.0216

Supplemental material is available online for this article.
Received 30. May 2018.
Accepted 21 Nov. 2018.
*Corresponding author (doz@bios.au.dk).

Eutrophication of fresh and marine waters and biodiversity loss across ecosystems as a result of global climate and land management changes constitute major environmental challenges. Despite increasing fundamental ecological knowledge, these challenges are exacerbating. An important reason for this is the pressure on agricultural land to meet the food demands of the world's increasing population and needs for energy and water (WWAP, 2015; Scanlon et al., 2017). Despite attempts over recent decades to control excess fertilizer application (Neal and Jarvie, 2005; Lam et al., 2011), diffuse pollution of watercourses remains sufficiently large to negate achieving the environmental goals of European legislation and national environmental programs (Kronvang et al., 2008; Land et al., 2016). Among the mitigation measures targeting diffuse pollution, buffer strips, commonly called vegetated buffer strips, have been introduced in some European countries since the 1980s to mitigate the deterioration of watercourses by surface runoff from intensively managed agricultural land (Stutter et al., 2012). The aim was to create a buffer between a field and an adjacent watercourse to interrupt the transport of particles and nutrients to surface waters. Since this time, numerous scientific studies and guidance documents have been published on the functioning and effectiveness of buffer systems (Mayer et al., 2007; Stutter et al., 2012; Hille et al., 2018a). Today, it is well documented that their performance is closely linked with a number of biological, hydrological, and geological factors, often demonstrating strong spatial and temporal variations (Hill, 1996; Hoffmann et al., 2009; Christen and Dalgaard, 2013).

Mitigating a range of diffuse pollutants remains the primary objective for buffers. The evidence for the effectiveness of conventional buffers, in terms of reducing diffuse pollution from sediment, P, N, pesticides, and fecal organisms, is varied. Documentation was mostly done at plot scales and over short durations, hence it is difficult to transfer across sites and to upscale

D. Zak, Dep. of Chemical Analytics and Biogeochemistry, Leibniz-Institute of Freshwater Ecology and Inland Fisheries Berlin, Müggelseedamm 301, D-12587, Berlin; D. Zak, M.V. Carstensen, J. Audet, C.C. Hoffmann, S. Hille, M. Knudsen, and B. Kronvang, Dep. of Bioscience, Aarhus Univ., Vejlsøvej 25, 8600 Silkeborg, Denmark; M. Stutter, J. Stockan, and H. Watson, The James Hutton Institute, Aberdeen, AB15 8QH, UK; H.S. Jensen and S. Egemose, Dep. of Biology, Univ. of Southern Denmark, Campusvej 55, DK-5230 Odense M, Denmark; J.A. Strand and P. Feuerbach, Dep. of Wetlands & Biodiversity, Hushållningssällskapet Halland, SE-305 96 Eldsberga, Sweden; B. Christen, G. Heckrath, Institute for Agroecology, Aarhus Univ., Blichers Allé 20, DK-8830 Tjele, Denmark. Assigned to Associate Editor Daire O Huallachain.

Abbreviations: DK, locations in Denmark; DM, dry matter; GB, locations in Great Britain; HRT, hydraulic residence time; IBZ, integrated buffer zone; ICP–OES, inductively coupled plasma optical emission spectroscopy; SE, locations in Sweden; SRP, soluble reactive phosphorus; TN, total nitrogen; TP, total phosphorus.

from the plot scale. Generally, pollution reduction (output vs. input mass loads) rates of buffers are greatest for coarser sediments and associated particulate transport of P and pesticides (Collins et al., 2009). The effectiveness, and between-site consistency of buffers, is less for the trapping and retention of fine particles and soluble N, P, and other pollutants (Syversen and Borch, 2005; Dorioz et al., 2006; Collins et al., 2009). Reasons for this include: (i) the primary design of grass buffer strips is to slow and retain particles in surface runoff, while important, subsurface pathways are ignored in such buffers (Jaynes and Isenhart, 2014); (ii) over longer periods, surface runoff can converge into preferential pathways that reduce even coarser sediment trapping (Collins et al., 2009); and (iii) P is cycled and interconverted between forms (Stutter et al., 2009). Hence, conventional grass buffers are limited in their ability to reduce the amount and rate of transport of many forms of pollutants unless they have widths much greater than 10 m (Hill, 2018). Yet the slow conveyance across or through buffers is vital to allow biogeochemical processing (sorption, biological uptake, and degradation) of pollutants in situ in the buffer (Dorioz et al., 2006; Arora et al., 2010). Innovative designs of buffers are required to address a range of belowground, surface, and aboveground transport pathways to enhance retention of particles in sculpted ground, ponds, and ditches, which preferentially slow subsurface flows and improve in situ biogeochemical processing. Addressing these issues provides opportunities to integrate aspects of multiple wider benefits in systems, including biogeochemical nutrient processing in root zones, nutrient uptake, and biomass generation (Ferrarini et al., 2017).

Studies have not only included the capability of vegetated buffer strips to regulate nutrient pollution but have also sought to demonstrate that these systems can provide additional ecosystem services (Stutter et al., 2012). The provision of such services can be built on for new buffer designs, including inter alia climate regulation through C sequestration or shading to moderate temperature extremes, supporting habitats for birds and amphibians or a diversity of pollinators, and providing opportunities for recreation and education (Mander et al., 2005; Stutter et al., 2012) or the production of biomass (Hille et al., 2018a). However, vegetated buffer strips were often considered to be inefficient for retaining dissolved nutrients since they are often bypassed by drainage pipes and are generally too dry throughout the year (Hille et al., 2018a). Therefore, a modified concept termed "integrated buffer zones" (IBZs) was recently introduced and tested to improve their services in the landscape (Strand et al., 2018; Zak et al., 2018). Even though IBZs are targeted to combine relatively small surface areas with potentially high rates of pollutant removal, they seek to improve wider ecosystem services that represent beneficial enhancements within riparian zones (Stutter et al., 2012). In general, IBZs combine two main compartments: an aquatic part, where suspended soil particles in drain water can settle, and a planted infiltration zone; together, these provide an optimum environment for anaerobic microbial processes and plant uptake (Supplemental Fig. S1). Accordingly, the major difference to the traditional design of vegetated buffer strips is that both drain water and surface runoff will be trapped within the pond and charging the infiltration zone thereafter. Recently, the concept of IBZs has been improved in Denmark and Sweden and also latterly introduced in Finland and Great Britain. So far, 14 IBZs have been established in Sweden alone, with ~20 more planned for the next 2 yr, and the concept is currently expanded in Germany as well (M. Trepel, personal communication, 8 Oct. 2018).

The present work focused on water storage, pollution control, and nutrient uptake as a subset of regulating services, biodiversity as a supporting service, and biomass production as a provisioning service (Hassan et al., 2005). The study methods and desired outcomes for each site differed and did not necessarily cover all services, mirroring different environmental priorities in Denmark, Sweden, and Great Britain. Hence, this synthesis concentrates on the key results for different aspects of services from where the best evidence exists (Table 1).

Materials and Methods

Study Sites

The IBZs under investigation were established between 2012 and 2014 at five locations in Denmark, Great Britain, and Sweden, abbreviated as DK-I, DK-II, GB, SE-I, and SE-II, respectively (Fig. 1). Both DK locations and SE-I had two IBZ sites each, GB had four sites, and SE-II had one site, resulting in a total number of 11 IBZs (Fig. 2A–2D). Generally, each IBZ consisted of two similar-sized compartments: an aquatic part called a ditch or pond, and an infiltration zone planted with trees, either *Alnus glutinosa* (L.) Gaertn. (alder, all sites) or *Salix viminalis* L. (willow, GB only) (Fig. 2). In some of the IBZ sites, an overflow system was implemented to bypass excess drainage during periods with high runoff, and water flow and levels were controlled by vertical tubes inside specially designed concrete wells, allowing overriding control of water levels if this was required by the land managers (Supplemental Fig. S1). The specific design, climate, and landscape context of the different IBZ settings are summarized in Table 2.

Water Storage

Water balances were determined to quantify water storage and the retention of P and N in the IBZs from all DK sites (IBZ_{DK}). For that purpose, water flow in the inlet and the outlet (Supplemental Fig. S1), as well as the water table, were measured every 10 min using flow meters (KROHNE) and water level recorders (MadgeTech) from July 2015 to June 2016. For the Danish sites, daily precipitation and potential evapotranspiration data were derived from the Danish Meteorological Institute using a weather data grid of 10 × 10 or 20 × 20 km (DMI, 2017), respectively. The outlet was leveled ~0.8 m above the bottom of the aquatic part to prevent an overrun of the IBZs. By regulating the water flow at the inlet and outlet, a more or less constant water level of ~0.7 m could be maintained in the aquatic part, and the infiltration zone was usually slightly inundated to enhance nitrate removal by denitrification.

In GB sites (IBZ_{GB}), ditch water height was recorded semicontinuously by Troll pressure transducers (Troll 100, In-Situ) for the purpose of investigating tradeoffs between flood water storage (i.e., free volume utilizable for runoff during storms) and ditch water levels required for filter bed action. Using GIS flow accumulation routines, we calculated that the flow paths into the IBZ_{GB} area of Ditches A, B, C, and D comprised a catchment of 9.6 ha.

Table 1. Overview of assessed ecosystem services and used indicators and metrics in different integrated buffer zones (IBZs) of Denmark (DK), Great Britain (GB) ,and Sweden (SE) (see Table 2 for IBZ code and Fig. 1 for location of sites).

Ecosystem service	Indicators and metrics†	IBZ
Water storage	Hydraulic residence time	DK-I, DK-II
	Ditch capacity for peak flow additional storage	GB
Pollution control	Removal of nitrate and TN, retention of SRP and TP	DK-I, DK-II, partly GB
	Nutrient and C stock of plants	DK-I
	Sediment retention	DK-II
Biodiversity	Terrestrial ground level and aquatic invertebrates	SE-I, SE-II, GB
	Amphibians, aquatic plants, birds, mammals	SE
	Terrestrial plants	GB
Biomass production	Tree growth or biomass yield	DK-I, GB

† TN, total N; SRP, soluble reactive P; TP, total P.

Pollution Control

Surface Runoff

At IBZ_{DK-II}, surface runoff was generated from a 5.18-ha field area adjacent to the IBZ, as calculated through the application of a 0.4-m × 0.4-m digital terrain model in ArcGIS 10.2 (ESRI, 2014). The surface runoff was redirected around the IBZ because an upstream dike was constructed to prevent any interference from surface runoff into the IBZ water and mass balances (see below). Thus, fluxes of water, suspended sediment, N, and P generated from surface runoff on the field were measured and used as a proxy for the functioning of the IBZ for collection and possible retention of sediment and nutrients from surface runoff. Surface runoff was measured at the experimental IBZ using a V-notch weir installed in an erosion gully formed in front and along the side of the IBZ ~20 m from the stream channel. Water containing suspended sediment was sampled using an ISCO sampler containing 24 glass bottles that was triggered by a magnetic contact whenever the water started flowing through the V-notch weir. Out of a total of 15 individual surface runoff events occurring during the winter period (November 2015 to April 2016), water samples representing the entire period were randomly taken at six events. Runoff water samples were analyzed for suspended sediment, total P (TP), and total N (TN) applying standard analytical methods (Hansen and Koroleff,

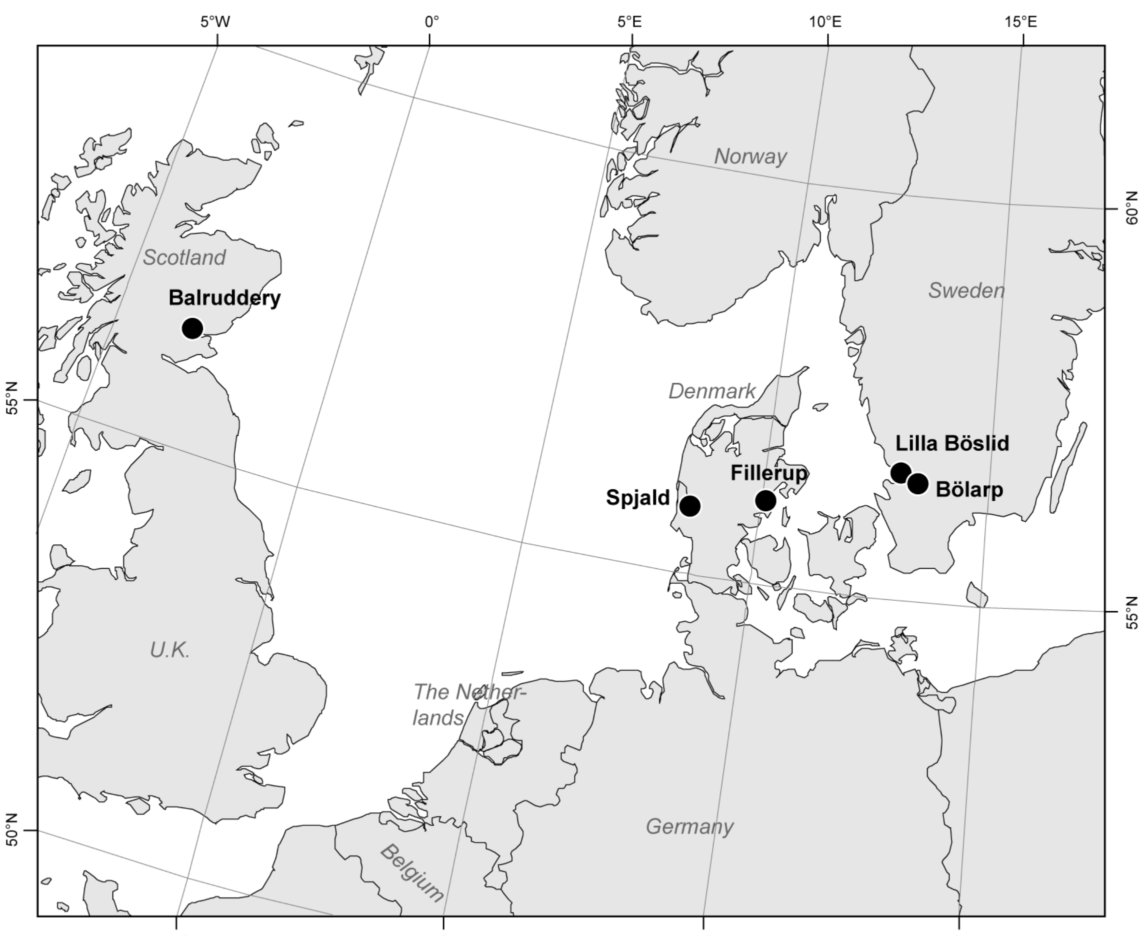

Fig. 1. Locations of integrated buffer zones (IBZs) in Denmark, Great Britain, and Sweden.

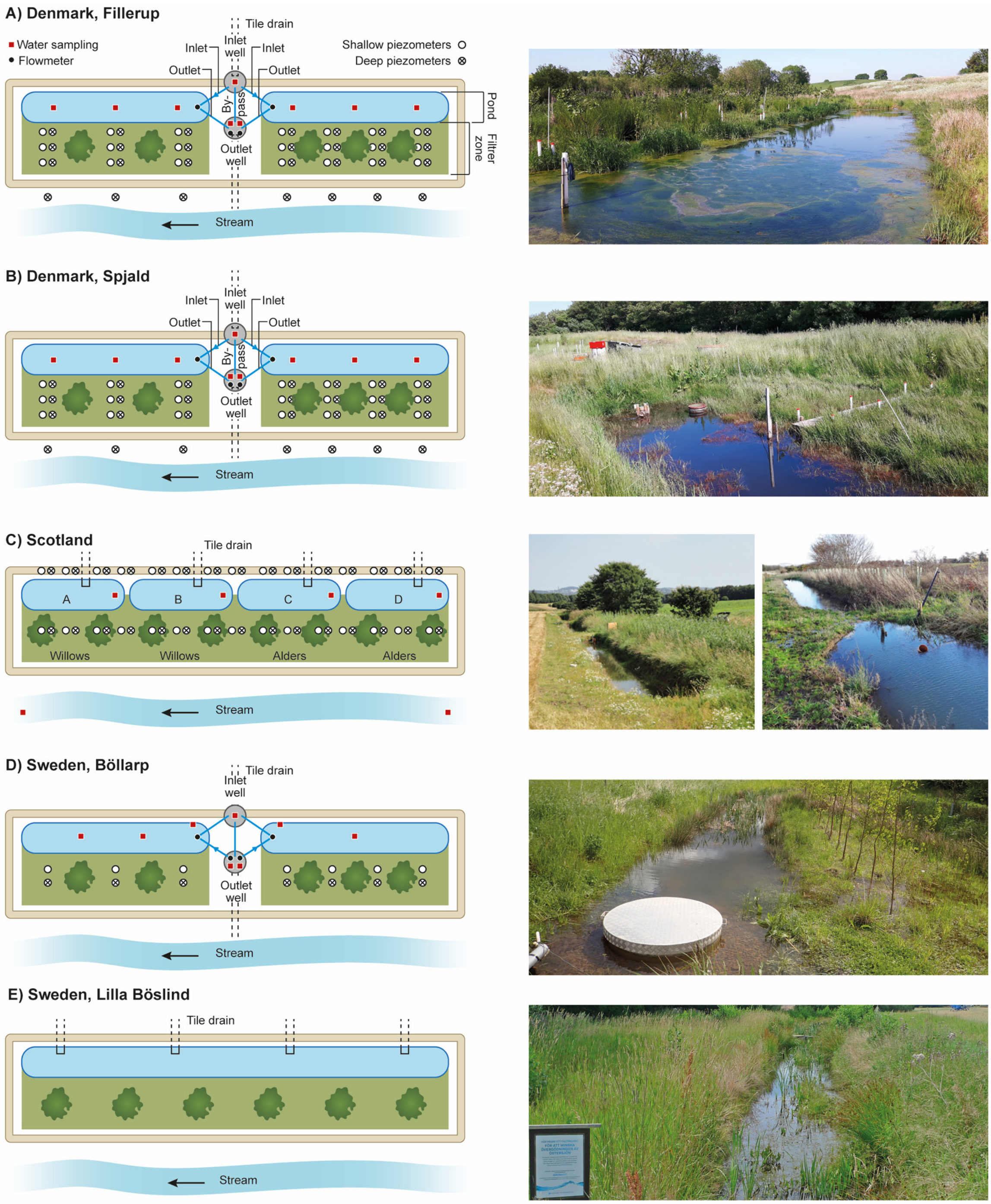

Fig. 2. Schemes from integrated buffer zones (IBZs) established in (A, B) Denmark, (C) Great Britain, and (D, E) Sweden. Further details on the specific design, climate, and landscape context of the different IBZ settings are summarized in Table 1.

2007). Flow-weighted concentrations from each event covered by water sampling were used for the unsampled surface runoff events to calculate the total export of suspended sediment, TP, and TN in surface runoff from the field during the winter period.

Tile Drainage Water

At the IBZ_{DK} sites, water samples were taken fortnightly to enable use of a mass-balance approach by sampling the inlet water, three points in each pond, the outlet water, and the piezometers in the filter bed (Fig. 2A and 2B). All surface water samples were analyzed for TP and TN using standard analytical methods (Hansen and Koroleff, 2007). For the analysis of soluble reactive P (SRP) and nitrate, the water was either filtered in the field or on the day of sampling in the laboratory using Whatman GF/C filters (pore size = 0.45 μm). Daily concentrations of nutrients in ditch and piezometers were obtained using linear interpolation between two subsequent sampling occasions (Kronvang and Bruhn, 1996). Daily loads of P and N forms were calculated by multiplying the concentration (mg L^{-1}) in the inlet by the total volume of water (L) entering the IBZs. Nutrient load and nutrient removal were calculated on a daily basis for the total IBZ basins, and also separately for the two basin compartments (pond and filter bed). Daily loads of P and N fractions for the pond and the total basin, respectively, were calculated by multiplying the concentration (mg L^{-1}) at the inlet by the total volume of water (L) entering the wetland, and for the filter bed by multiplying the concentration in the pond by the volume of infiltrating water of the filter bed. Assuming particle transport was negligible in the filter bed, only dissolved species were considered for this compartment, in the calculation of both load and removal. As a last step, the overall nutrient removal was calculated as the sum of the removal from the pond and filter bed components. Below, the combined daily data as monthly averages for both components are presented. Detailed information on the calculation of water infiltration, hydraulic residence time (HRT), and nutrient removal, as well as on standard methods for chemical analysis, can be found in Zak et al. (2018).

Biodiversity

Biodiversity was monitored in the Swedish and British sites (Table 1). The monitoring focused on five different organism groups in the IBZ_{SE} sites through surveys conducted during 2013 to 2015. Species were identified using different keys depending on organism groups, namely, aquatic invertebrates (Hynes, 1977; Macan, 1978; Cranston, 1982; Mandal-Barth, 1982; Elliott and Humpesch, 1983; Lillehammer, 1988; Savage, 1989; Edington and Hildrew, 1995; Andersen, 1996; Engblom, 1996; Jansson, 1996; Norling and Sahlén, 1997; Elliott and Mann, 1998; Dannelid and Sahlén, 2007), birds (Brown et al., 2003), mammals (Brown et al., 1982), amhibians (Cederhagen and Nilsson, 1979; Ahlén et al., 1995), and vegetation (Moeslund et al., 1990; Schou et al., 2017). The monitoring varied depending on organism group (Naturvårdsverket, 2010). Aquatic invertebrates were surveyed by time-standardized netting and by activity traps and bottle traps. Netting was done in the aquatic part in each IBZ at two to four spots on five occasions between October 2013 and May 2014. Each sampling consisted of 30 s of netting in swipes at the sediment surface and at the edges of vegetation with a D-shaped hand net (mesh size = 1.4 × 1.4 mm). This method, with relatively few and short netting periods during winter and spring, was used since it was part of a project designed to compare dominant invertebrate species between IBZs and four other small aquatic systems, and not to maximize species richness. The other aquatic systems were located within 1 km of IBZ_{SE-II}. These sites were 300 to 600 m^2 and 50 to 100 yr old and consisted of three marl pits and one "nature park" pond sampled at the same time period using the same methods. Samples from each spot were live sorted, and the invertebrates were preserved in 70% ethanol for subsequent identification and counting under a stereomicroscope (Zeiss Stemi 2000-C). Simple activity traps and bottle traps were constructed using plastic, transparent polyethylene terephthalate (PET) bottles that were cut at the top, after which the top part was inversely inserted into the bottom half, thus creating a funnel trap. The traps were located with the opening of the funnel at the sediment bottom and with the bottom part extending just above the water surface. A small hole was drilled in the trap to ensure O_2 exchange. The traps were fixed in position with bamboo sticks. Four activity traps, baited with liverwurst, were placed in one of the IBZs in Sweden on two occasions, 31 Mar. and 7 May 2014, and left in the IBZ for 24 h each time. The use of activity traps followed standard methods from the Swedish environmental monitoring program for great crested newts (*Triturus cristatus*) (Naturvårdsverket, 2005). In addition to the above surveys, a continuous species list of larger, easily identified insect species was made during visits to the IBZ. The other groups were investigated as follows:

- Amphibians: by visual survey of eggs and adults during spring and by netting and activity traps, as described above. Visual surveys were done on 16 and 23 April during daytime and by flashlight for salamanders at night on 13 May 2014.
- Birds: by three repeated visits during April to June in 2014 and 2015. The small area of the IBZ prevented use of territory maps; instead, point counts with behavior assessments (breeding criteria) were applied. In each survey, the species and number of individuals and their behavior were noted following standard Swedish monitoring program methods (Naturvårdsverket, 2012).
- Mammals: by tracks and signs on the muddy ground of the infiltration zones after inundation events during 2013 to 2015. Also, visual observations of mammals were recorded

Table 2. Selected site properties of the integrated buffer zones (IBZs) in Denmark (DK), Great Britain (GB), and Sweden (SE) (see map in Fig. 1).

Property	Fillerup, DK	Spjald, DK	Balruddery, GB	Bölarp, SE	Lilla Böslid, SE
Code	DK-I	DK-II	GB	SE-I	SE-II
Year of establishment	2014	2014	2014	2013	2012
Coordinates	55.57180° N, 10.05294° E	56.06173° N, 8.28375° E	56.28470° N, 3.07320° W	56.33502° N, 13.65927° E	56.35469° N, 12.56297° E
Altitude (m asl)	46	40	50	20	10
No. of facilities	2	2	4	2	1
Upslope field size (ha)	30	16.2	4.5	8	3.5
IBZ size (m^2)	250	350	300	330	800
IBZ/field ratio (%)	0.1	0.2	0.7	0.4	2.2
Adjacent cropping after IBZ establishment	Spruce, grain	Grain	Grain, potatoes, beans	Grain	Grain, potatoes
Annual avg. precipitation (mm)	718	910	705	850	800
Avg. temperature, January (°C)	1.6	1.9	3.2	−1.5	−1.5
Avg. temperature, July (°C)	16.6	16.5	14.7	16	16
Climate data source	Danish Meteorological Institute	Danish Meteorological Institute	Onsite meteorological station	SMHI† (maps)	SMHI (maps)
Tree species	*Alnus glutinosa*	*A. glutinosa*	*A. glutinosa*, *Salix viminalis*	*A. glutinosa*	*A. glutinosa*

† SMHI, Swedish Meteorological and Hydrological Institute.

during visits to the IBZ during the surveys of other organism groups.

- Aquatic plants: by surveys during 2013 to 2015 where a telescopic rake was used to reach and sample submerged flora to obtain a cumulative species list.

At IBZ_{GB}, pitfall trapping was used to sample ground invertebrates. Three traps were installed in a linear arrangement, 7.5 m apart, within the IBZ (two replicates within each tree type) and two adjacent riparian margins (managed by sowing with standard wildflower seed mix). The traps consisted of 7.5-cm-diam. plastic cups sunk into the ground and partly filled with ethylene glycol (with a roof for rainfall protection). The traps were emptied twice during each 28-d sampling period. Sampling was conducted four times: autumn 2015, spring and autumn 2016, and spring 2017. The abundance of Coleoptera (beetles), Araneae (spiders), Opiliones (harvestmen), and Collembola (springtails) were counted. Carabid beetles were identified to species according to Luff (2007). Invertebrates were also sampled from the vegetation and litter in the IBZ using a Vortis insect suction sampler over five separate ground areas of 0.09 m^2 for 5 s each and then bulked. Four samples were taken from each margin. Invertebrates were identified at least to order.

Biomass Production

To assess the uptake of P and N as well as C sequestration by plants, six randomly chosen plots in both the pond and the filter bed at both IBZ_{DK-I} sites were harvested at the end of the growing season in mid-September 2016. Due to litter loss before biomass yield, date shown below refer to net production of aboveground biomass and net nutrient or C uptake, respectively. For the sampling of aquatic plants in the pond, plastic cylinders with an inner area of 1 m^2 were used. The whole plant to the bottom of the pond was harvested. For the infiltration zone, a plastic frame with the same inner area was used. Furthermore, six alder trees were also removed from each IBZ_{DK-I} site, leaving the belowground biomass untouched. For the determination of dry mass, the plants were dried at 60°C for 2 d until mass constancy. Aboveground biomass from trees was divided into branches and stems, and four tree size categories were distinguished for later separate analysis of biomass and P, N, and C concentrations. Before determination of total nutrient and C contents by standard methods (Hansen and Koroleff, 2007), the dried material was homogenized using a fine-grain mill. The net P, C, and N uptake by aboveground plant biomass per unit area in the sites under investigation was calculated using P concentrations and biomass data (Zak et al., 2014).

For IBZ_{GB}, the C and nutrient stock from tree leaves were assessed. This was done separately, since the woody material represents longer-term sequestration of biomass and nutrients, whereas leaves represent annual cycling periods where the litter returns nutrient to the soils. The biomass of leaf and woody material were sampled in 2017 and analyzed as separate components to determine total aboveground biomass. Leaves were sampled from alders and willows in summer (n = 10 leaves each from 10 randomly selected trees per plot). Branches were sampled from willows (n = 10 trees per plot), but for alders, literature-derived values of N and P contents were used so as not to damage the trees at this early stage. Detailed methods and assumptions are given in Supplemental Table S1. Total C and TN concentrations were determined on 2-mg oven-dried subsamples using a Flash EA 1112 Series elemental analyzer. TP was determined following aqua-regia digest using inductively coupled plasma optical emission spectroscopy (ICP–OES). The results of standing stock (aboveground biomass only) for C, N, and P were aggregated up to area level (either t ha^{-1} or kg ha^{-1}), and uncertainties were calculated as ±95% confidence intervals from accumulated errors. For comparison with biomass, topsoil (down to the 25-cm depth) was collected and analyzed for N and P stocks. Phosphorus was extracted by modified Morgan's reagent (according to McIntosh, 1969) and analyzed in the extracts by ICP–OES. Soil-extractable nitrate was determined in a 1:5 (w/v) extract using 1 M KCl for 1 h. Before colorimetric analysis, extracts were filtrated by prewashed Whatman No. 1 papers. The calculations of P and N stock were done on volume basis using bulk density determined on triplicate soil core samples per plot.

Statistical Analysis

Average values are represented as means ± SD unless otherwise stated. For the comparison of water quality, nutrient load, and water quality between the four IBZ_{DK} sites, as well as the elemental composition of leaf material from the IBZ_{GB}, the nonparametric Kruskal–Wallis test was used, followed by a post-hoc test based on pairwise comparisons with Bonferroni adjustment of the significance level. In addition, factors controlling the nutrient removal efficiency were tested. Activity–density of carabids and total invertebrates (for both pitfall and suction sampling) from IBZ_{GB} were log transformed $\{\log[10(y + 1)]\}$ to normalize the data. To avoid pseudoreplication, means were calculated from all traps of one site. Differences between means were tested using restricted maximum likelihood with year and season as random factors (VSN International, 2011). The significance level for all statistical tests was $p < 0.05$. All tests were performed using the R software version 3.4.4 (R Development Core Team, 2018).

Results

Water Storage

In IBZ_{DK}, between 10 and 60% of the inflowing tile drainage water infiltrated through the filter bed, whereas the rest of the water left via an overflow pipe (Supplemental Table S2, Supplemental Fig. S1). The daily hydraulic load and the HRT of the aquatic part were, on average (±SD), 208 ± 121 mm and 2.7 ± 2.0 d, respectively (Supplemental Table S2). The HRT in the filtration zone was estimated to be roughly twice as high as in the ditch (as determined once during a 2- to 4-wk Br tracer experiment; Zak et al., 2018), and it should be noted that it might take several days more before the water leaving the filtration zone enters the adjacent stream.

At IBZ_{GB}, despite comparable rainfall and interception of drained sites, the existing conditions caused considerable variation in water level (Supplemental Fig. S2). The total ditch height was ~0.9 m to overtopping, and commonly Ditch A was full except in summer (water level minimum ~ 0.6 m). Ditches B, C, and D behaved differently from Ditch A, but they generally maintained their capacity and were filled at runoff events and emptied at daily to weekly intervals. Thus, the ditches maintained their capacity for temporary storage of flood runoff and settling of fine particles passing through the IBZ even if they were already filled with water. As an example, the stage change in September in Ditch A of ~0.3 m represents some 20 m^3 of storage volume.

Pollution Control

Surface Runoff

Surface runoff from a 5.2-ha field to IBZ_{DK-II} amounted to 48 mm during the winter period of 2015–2016. The loss of suspended sediment, TP, and TN amounted to, respectively, 56, 0.27, and 0.30 kg ha^{-1} during winter 2015–2016. Thus, it is estimated that a total of 292 kg suspended sediment, 1.39 kg P, and 1.57 kg N would have entered IBZ_{DK-II} from surface runoff of the field. Taking into account that 30 to 80% of sediment particles become retained, depending on hydraulic loads and vegetation growth (Braskerud, 2001), the removal for sediment and associated nutrients would account 0.5 to 1.3 kg m^{-2}, 2 to 6 g P m^{-2}, and 3 to 7 g N m^{-2}, respectively, during the winter period of 6 mo.

Tile Drainage Water

The tile drainage water discharging into the IBZ_{DK} sites showed monthly average TN concentrations of 6.2 ± 1.4 and 9.6 ± 1.0 mg N L^{-1} (Table 3), whereas the monthly average TP concentrations were comparatively low (Table 4), in particular at the two IBZ_{DK-II} sites (0.02 ± 0.01 mg P L^{-1}). The proportion of nitrate was ~90% of TN, and SRP constituted 40% of TP. In comparison with the ditch inflow water at IBZ_{GB} for a single intercepted tile drain (entering Ditch A), the monthly measured nitrate and SRP were 9.0 ± 0.3 mg N L^{-1} and 0.03 ± 0.0 mg P L^{-1}. The removal of nitrate and TN in the four IBZ_{DK} amounted to, respectively, 0.2 to 0.5 and 0.1 to 0.6 g N m^{-2} d^{-1}, yielding a monthly average removal efficiency between 23 and 37% for nitrate and between 8.1 and 38% for TN (Table 3).

At IBZ_{GB}, a time series of nitrate concentrations in the four ditches (Supplemental Fig. S3) showed considerable spatiotemporal variability. However, in early summer 2016, the nitrate concentrations in several ditches became greatly depleted relative to drain concentrations, which may be taken as a reference for their inputs.

The removal of SRP and TP in the four IBZ_{DK} amounted to, respectively, −0.3 to 5.0 and 0.3 to 6.9 mg P m^{-2} d^{-1}, amounting to a monthly average removal efficiency of −29 to 67% for SRP and 18 to 52% for TP (Table 4).

With few exceptions, nitrate and TN removal efficiencies were significantly negatively correlated to nitrate and TN loads (Fig. 3). For SRP and TP, both IBZ_{DK} locations behaved differently. Although the load and the retention were negatively correlated in the two IBZ_{DK-I} sites, an opposite relation was found for the two IBZ_{DK-II} sites (Fig. 4).

Habitat Provisioning

The overall species richness of aquatic invertebrates in the two IBZs investigated in Sweden was comparable with the other

Table 3. Nitrogen concentrations of drain water discharging the integrated buffer zones (IBZs), N load, specific N removal, N removal efficiencies, and N stock of aboveground biomass of nonwoody plants (submerged, floating, and emergent plants [SFE]) and of black alder trees (ALD) of the duplicated IBZ sites (a and b) in two locations of Denmark (see Fig. 1 for sampling locations and Fig. 2 for site details). Data on concentration and removal represent monthly averages ± SD for the monitoring period from July 2015 to June 2016. Nitrogen stock was determined at the end of growing season in mid-September (median of 12 sampling plots for SFE and of six alder trees ~6 yr old).

Parameter	DK-I a	DK-I b	DK-II a	DK-II b
Nitrate conc. (mg N L^{-1})	5.5 ± 2.0a†		9.5 ± 1.0b	
Total N conc. (mg L^{-1})	6.2 ± 1.4a		9.6 ± 1.0b	
Nitrate load (g N m^{-2} d^{-1})	1.4 ± 1.1a	1.9 ± 1.0a	1.4 ± 0.8a	1.5 ± 0.9a
Total N load (g N m^{-2} d^{-1})	1.6 ± 1.1a	2.1 ± 1.0a	1.4 ± 0.8a	1.5 ± 0.9a
Nitrate removal (g N m^{-2} d^{-1})	0.3 ± 0.2a	0.5 ± 0.2b	0.3 ± 0.2a	0.2 ± 0.1a
Total N removal (g N m^{-2} d^{-1})	0.4 ± 0.2a	0.6 ± 0.2b	0.2 ± 0.1ac	0.1 ± 0.1c
Nitrate removal efficiency (%)	30 ± 19a	37 ± 17a	23 ± 13a	25 ± 19a
Total N removal efficiency (%)	31 ± 16	38 ± 16	17 ± 11	8.1 ± 14
SFE N stock (g N m^{-2})	17.6	15.2	No values	No values
ALD N stock (g N m^{-2})	1.9	2.0	No values	No values

† Significant differences (p < 0.05) are indicated by different lowercase letters.

Table 4. Soluble reactive P (SRP) and total P (TP) concentrations of drain water discharging from the integrated buffer zones (IBZs), P load, specific P removal, P removal efficiencies, and P stock of aboveground biomass of nonwoody plants (submerged, floating, and emergent plants [SFE]) and of black alder trees (ALD) of the duplicated IBZ sites (a and b) in two locations of Denmark (see Fig. 1 for sampling locations and Fig. 2 for site details). Data on concentration and removal represent monthly averages ± SD for the monitoring period from July 2015 to June 2016. Phosphorus stock was determined at the end of growing season in mid-September (median of 12 sampling plots for SFE and of six alder trees ~6 yr old).

Parameter	DK-I a	DK-I b	DK-II a	DK-II b
SRP conc. (mg P L^{-1})	0.03 ± 0.01a†		0.01 ± 0.00b	
TP conc. (mg L^{-1})	0.05 ± 0.01a		0.02 ± 0.01b	
SRP load (mg P m^{-2} d^{-1})	6.6 ± 4.4a	9.3 ± 4.7a	1.2 ± 0.9b	1.2 ± 1.1b
TP load (mg P m^{-2} d^{-1})	12 ± 8a	17 ± 7a	4.5 ± 4.3b	4.8 ± 4.6b
SRP removal (mg P m^{-2} d^{-1})	3.3 ± 1.7a	5.0 ± 2.5b	0.2 ± 0.7c	−0.3 ± 1.0c
TP removal (mg P m^{-2} d^{-1})	5.6 ± 3.8a	6.9 ± 4.2a	0.7 ± 2.9b	0.3 ± 2.8b
SRP removal efficiency (%)	44 ± 18a	67 ± 19a	25 ± 159ab	−29 ± 449b
TP removal efficiency (%)	44 ± 10a	52 ± 12a	21 ± 32b	18 ± 29b
SFE P stock (g P m^{-2})	2.34	2.03	No values	No values
ALD P stock (g P m^{-2})	0.16	0.18	No values	No values

† Significant differences (p < 0.05) are indicated by different lowercase letters.

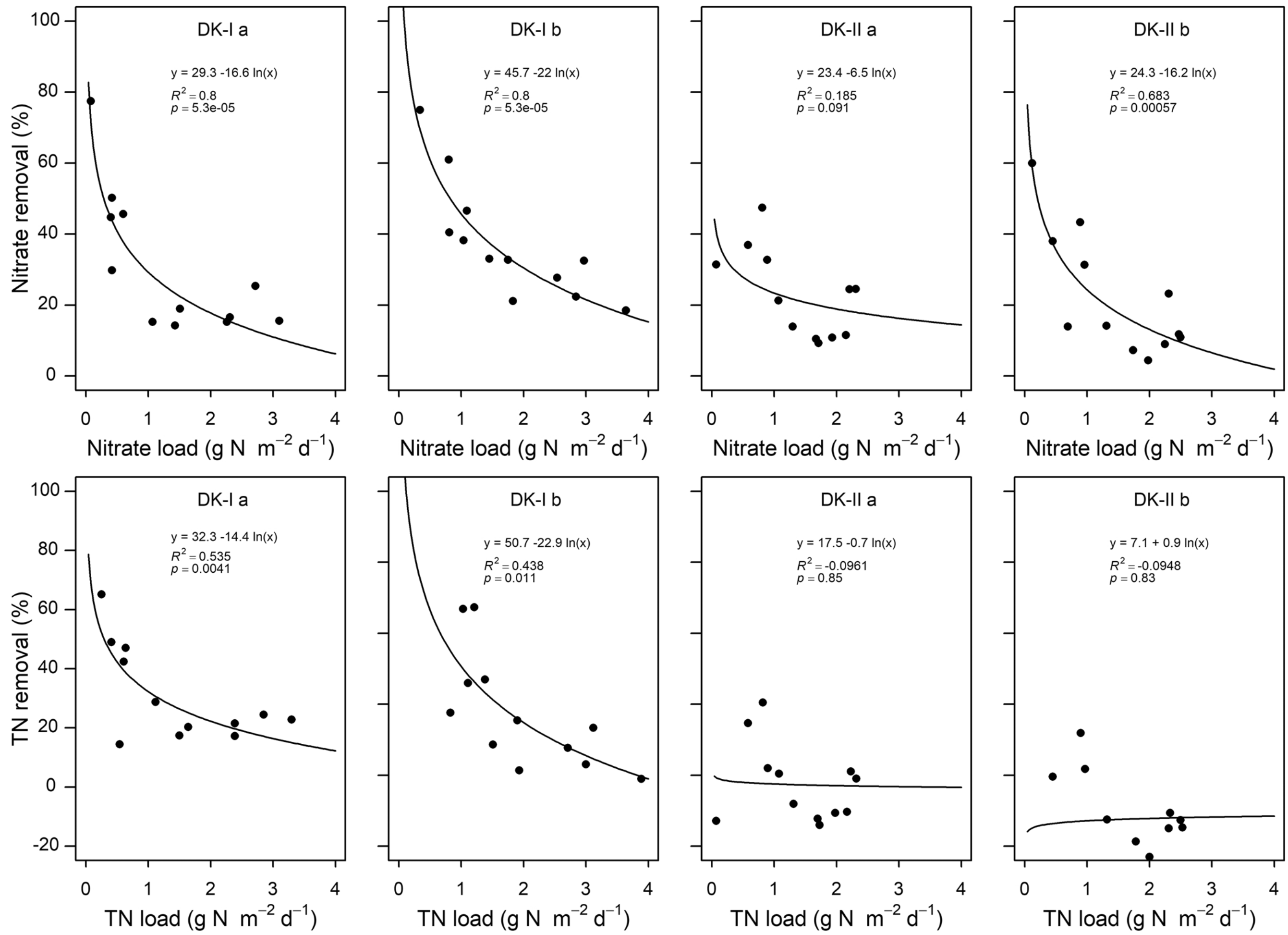

Fig. 3. Relationships between monthly averaged daily loading rate and removal efficiency for nitrate and total N (TN) of the four integrated buffer zones (IBZs) in Denmark (see Table 2 for IBZ code and Fig. 1 for location of sites).

small aquatic systems that were investigated simultaneously in Sweden. In total, 15 and 39 aquatic invertebrate species were found in the two Swedish IBZs (Table 5) compared with 24, 40, 27, and 19, respectively, in the four other investigated sites (single results are not shown, unpublished data). Altogether, 13 different plant species found in the IBZs in Sweden including widespread helophytes and hydrophytes like *Typha latifolia* L. and *Potamogeton natans* L. Regarding amphibians, the survey showed that out of the five species occurring in the region of Sweden where the IBZs were placed, *Rana temporaria* and *Bufo bufo* had established populations in IBZ_{SE-I}, and *R. temporaria*, *B. bufo*, *Lissotriton vulgaris*, and *Triturus cristatus* had established populations in IBZ_{SE-II} (Supplemental Table S3).

In IBZ_{SE-II}, at least eight species of dragonflies (Odonata) were observed either in the invertebrate sampling or by visual observation of adults (e.g., *Libellula quadrimaculata*, *Libellula depressa*, *Orthetrum cancellatum*, *Ischnura elegans*, and *Sympetrum sanguineum*). The dominant species in the invertebrate fauna in terms of individuals was the mayfly, *Cloeon dipterum*, a species adapted to small water bodies that may have ice cover and low O_2 levels. No surveys of terrestrial invertebrates were done in Swedish IBZs, but during the visits, some easily identified species were noted (e.g., four butterfly species [*Aglais urticae*, *Gonepteryx rhamni*, *Anthocharis cardamines*, and *Inachis io*] and two beetles [*Cicindela campestris* and *Potosia cuprea*]).

Apart from aquatic and semiaquatic species, the nationally red-listed skylark (*Alauda arvensis*) was found to be breeding in IBZ_{SE-II} (Supplemental Table S4). Skylark were found to breed in the narrow zone (1 m) between the farmland and the excavated aquatic part of the IBZ. Also, mammal species were recorded in the IBZs, and in addition to the six species observed directly within the IBZ, a badger was observed ~50 m outside the IBZ, likely using the IBZ habitat for foraging and shelter.

In IBZ_{GB}, pitfall trapping showed that carabid activity–density ($F = 6.17_{18}$, $p = 0.009$) and species richness ($F = 6.19_{18}$, $p = 0.009$) were significantly higher in the controls (adjacent grass margins) than at the ground level of the tree zone in the IBZ margins (Supplemental Table S5). There was no significant difference between alder and willow buffer systems respectively (log mean activity–density = 0.12 ± 0.06 [SEM] and 0.11 ± 0.04; species richness = 0.29 ± 0.13 and 0.33 ± 0.14). Total invertebrates, sampled by pitfall trapping, showed significant differences between the controls and the IBZ buffers ($F = 5.00_{18}$, $p = 0.038$; Supplemental Table S5), but not between alder and willow plots (log means = 1.09 ± 0.12 and 1.07 ± 0.11 respectively). However, these showed a diverse invertebrate community containing several trophic levels including parasitoids and the terrestrial stages of aquatic species (Supplemental Table S6). Like the invertebrates, plant species richness and diversity showed no difference between alder and willow plots. Plant species richness

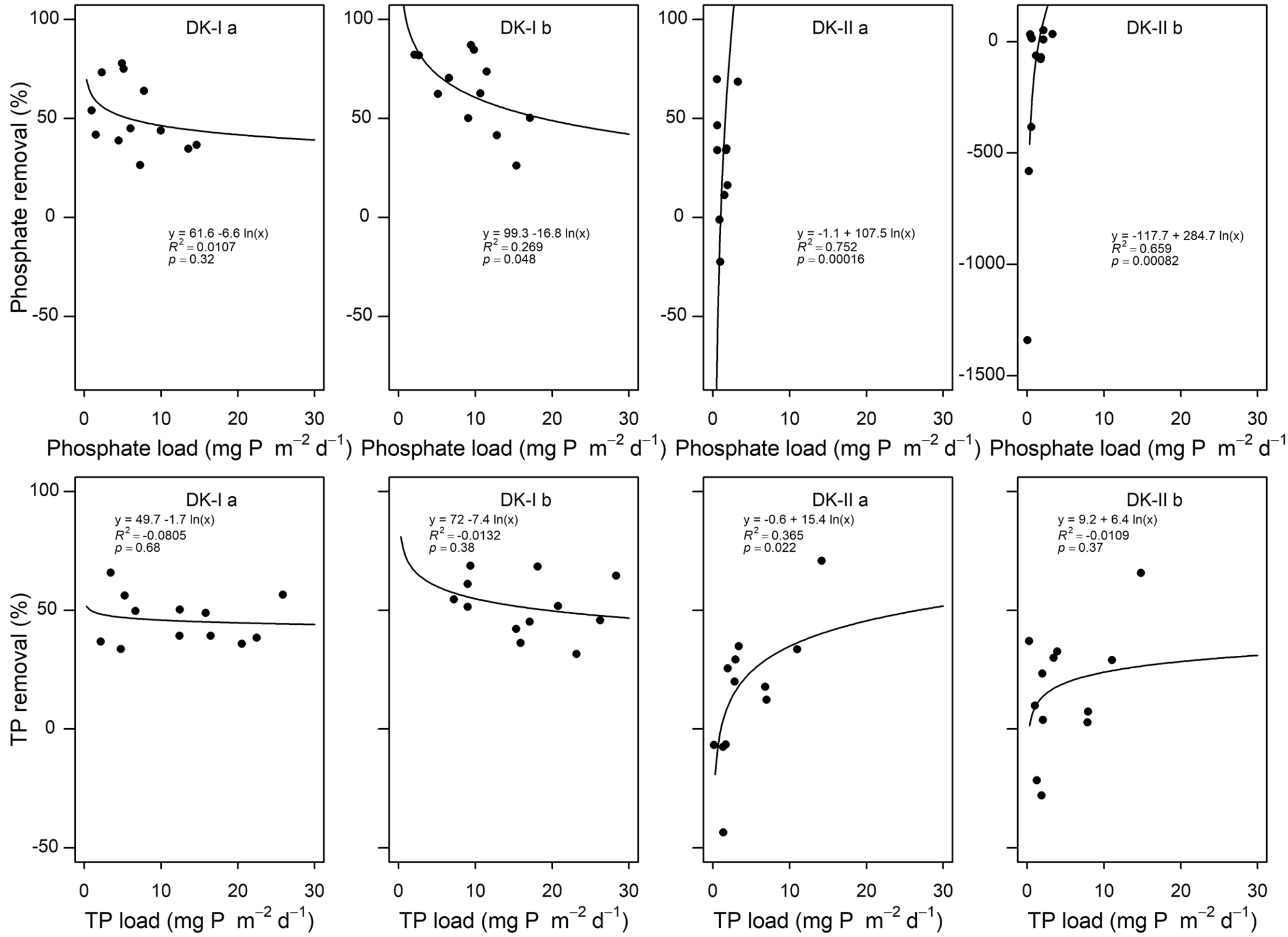

Fig. 4. Relationships between monthly averaged daily loading rate and removal efficiency for phosphate and total P (TP) of the four integrated buffer zones in Denmark (see Table 2 for IBZ code and Fig. 1 for location of sites).

averaged 15.25 (± 2.18), 9.50 (± 0.65), and 12.00 (± 1.83) in the alder, willow, and control buffers. Shannon diversity averaged 1.74 (± 0.05), 1.51 (± 0.11), and 1.44 (± 0.03) in the same buffers, respectively. An overview of plant species from Swedish and British IBZ sites can be found in Supplemental Table S7.

Biomass Production

In IBZ_{GB}, 205 and 198 willows (Plots A and B, respectively, out of 240 planted in each) and 36 and 55 alders (Plots C and D, out of 60 in each) had established by July 2017, 28 mo after planting. Shade from isolated mature trees and possibly deer browsing led to a plot of stronger and another of weaker tree growth in both the duplicate willow and alder plots. Full compositional data (Supplemental Table S1) show that the measured site-specific stem and branch N and P contents of willows were much greater than those of alder (although statistical differences were not possible, as the latter were based on literature values). Conversely, leaf material (all measured onsite in July) had slightly greater C, pronounced greater N contents in alder than in willow leaves, and no difference in P contents ($p < 0.001$, $p < 0.001$, and $p = 0.9$, respectively; $n = 20$ per tree type). Accordingly, the molar C/N ratios of alder litter (19–21) were smaller ($p < 0.001$) than for willow litter (24–26); however, both were in the optimal range of 20 to 30 for microbial decomposition (Neely et al., 1991). The most productive willow plot achieved 40 t dry matter (DM) yield ha^{-1} (13% leaf mass), and the weaker plot achieved 17 t DM ha^{-1} (18% leaf mass, Table 6). This was much greater than for alders, which achieved 2 and 10 t DM ha^{-1} (of 18 and 12% leaf, respectively). Additionally, the willows underwent a management cut after 12 mo (standard practice to encourage denser shooting) that yielded 6 and 2 t DM ha^{-1} from Plots A and B, respectively.

The calculated standing aboveground biomasses of C, N, and P are compared with the soil topsoil stocks to 25-cm depth for IBZ_{GB} sites (Table 6). The strongest willow plot demonstrated 19 t C ha^{-1}, 458 kg N ha^{-1}, and 70 kg P ha^{-1} in the aboveground biomass after the 2 yr. The strongest alder plot exhibited 5 t C ha^{-1}, 70 kg N ha^{-1}, and 4 kg P ha^{-1}. Total standing aboveground biomasses of C in the willow plots were 27 and 11% of those in

Table 5. Species numbers for different investigated organism groups in integrated buffer zones in Great Britain (GB) and Sweden (SE-I and SE-II). For size of the sampling area, see Table 1.

Organism	GB (Balruddery)	SE-I (Bölarp)	SE-II (Lilla Böslid)
	no.		
Aquatic invertebrates	8	15	39
Amphibians	not determined	2	4
Birds	not determined	not determined	10
Mammals	not determined	2	6
Aquatic plants	30	7	13
Total species no.	38	26	72

the topsoil (0–25 cm) but much smaller in the alder plots (2 and 8%). Total standing aboveground biomasses of N in the willow plots were 337 and 236% of those in the topsoil (considering the KCl-extracted nitrate) and 50 and 109% of those in the alder plots. Total standing aboveground biomasses of P in the willow plots were 163 and 103% of those of the topsoil and 6 and 17% of those in the alder plots. There were large differences in the tree wood and leaf partitioning of N and P between willows and alders. Regarding willows, N and P were mainly stored in stem and branch material, whereas in alders, N and P were dominantly partitioned in leaves.

In IBZ_{DK-I}, the seasonal net N uptake by nontree aboveground biomass in both the pond and the filter bed was estimated on the basis of the N stock to be ~17 g N m^{-2} (Table 3). This equaled ~10% of the TN removal of 182 g N m^{-2} over 12 mo for both IBZ_{DK-I} sites. The N stock of ~4-yr-old alder trees accounted only for 2 g N m^{-2} (20 kg N ha^{-1}), a negligible amount despite it being a long-term storage compared with the nontree biomass. According to P stock, the average seasonal net uptake of P by plants amounted to 2.2 g P m^{-2} (22 kg P ha^{-1}, Table 4), which was in the range of annual P retention for the four IBZ_{DK} sites (0.6–3.08 g P m^{-2}).

Discussion

Assessment of ecosystem services from IBZs requires an interdisciplinary perspective that recognizes mutual interdependence of hydrologic, chemical, and biological processes, which may vary strongly in space and time. As shown by our results from the first IBZs in Northwestern Europe, inclusion of IBZs in vegetated buffer strips will be a valuable option for mitigating nutrient runoff. Moreover, the IBZs will create additional valuable ecosystem services such as flood attenuation by increasing the water storage in agricultural landscapes, thereby providing habitats for amphibians and wetland plants, and can be used for biomass production as well. A common feature of all the studied IBZ sites is that tile drains previously discharging directly into the streams are now intercepted within the aquatic part of the IBZ, thereby enhancing HRT and the potential for biogeochemical processing.

Water Storage

Natural flood management is becoming increasingly important as the climate change has been documented to increase precipitation in Northwestern European countries due to expected more extreme and intense rainfall and runoff events (Arnell, 1998; Groisman et al., 2005; Andersen et al., 2006). Therefore, methods for conservation and restoration of rivers and floodplains, assisting flood management practitioners in obtaining various flood management objectives such as creating water storage in the landscape that can delay and desynchronize peak runoff in streams and rivers, are becoming an important issue in Northwestern Europe (Wilkinson et al., 2013; Collentine and Futter, 2018). Our results from IBZ_{DK} and IBZ_{GB} show that these features can, indeed, delay inflowing tile drainage water and also assist in storing part of the surface runoff from adjacent fields, especially if the IBZ is optimized for this requirement via an outlet flow control (Supplemental Fig. S1). Such storage would be beneficial for natural flood management and in times of peak sediment transfer from tile drains or surface runoff. As an example, for IBZ_{GB}, a 20-mm rain event could generate 200 m^3 of runoff. For comparison, the capacity in Ditch A would itself take 10% of this runoff. Thus, the capability of IBZs for water storage should not be overestimated, even though we expect that the unsaturated soil between the IBZ and the stream will also act as additional water buffer depending on the water saturation and antecedent conditions; however, this needs to be better quantified.

Pollution Control

Sediment and nutrients derived from soil erosion and surface runoff from adjacent fields are important sediment and P sources that negatively affect the surface water quality in many landscapes (Renard et al., 1996; Kronvang et al., 2007). As soil erosion processes are spatially and temporally highly variable, a targeted approach for mitigation efforts is often needed. Among other measures, riparian buffer strips have been prioritized by a number of European countries as tools to prevent sediment and sediment-associated substances such as P from entering surface waters (European Commission, 2016). However, the

Table 6. Tree composition and standing biomass for aboveground components only in the four plots in Great Britain. Values given are means with uncertainty ranges (95% confidence interval ranges) for calculated accumulated errors. For comparison with biomass yields, the topsoil (to the 25-cm depth) areal stocks of C, N, and P are given. All values are measured at the plots unless stated and are presented as dry matter (DM). The percentage attributed to leaf is shown, since this only represents a temporary store due to litter recycling.

Parameter	Plot A, willows	Plot B, willows	Plot C, alders	Plot D, alders
No. trees per plot	205	198	36	55
Total biomass DM (t ha^{-1})	40 (32–48)	17 (7–28)	2 (1–3)	10 (6–13)
% attributed to leaf	13	18	18	12
Total biomass C (t ha^{-1})	19 (18–19)	8 (7–9)	1 (1–1)	5 (4–5)
% attributed to leaf	13	18	18	12
Total biomass N (kg ha^{-1})	458 (339–579)	201 (137–266)	16 (12–21)	70 (59–81)
% attributed to leaf	26	32	63	52
Total biomass P (kg ha^{-1})	70 (61–78)	30 (25–37)	1 (1–2)	4 (4–5)
% attributed to leaf	17	27	74	61
Topsoil C stock (t ha^{-1})†	70 (58–83)	67 (52–84)	65 (54–76)	65 (54–78)
Topsoil KCl extract nitrate (kg N ha^{-1})	136 (108–166)	85 (48–125)	32 (19–48)	64 (36–97)
Topsoil MM‡ extract P (kg P ha^{-1})	43 (32–56)	29 (17–43)	16 (10–23)	23 (18–29)

† C was estimated as 0.5 loss on ignition, and site-specific soil dry bulk density values were used in scaling C, N, and P to areal stocks.

‡ Modified from Morgan's reagent (according to McIntosh, 1969).

establishment of indiscriminate vegetated buffer strips along most water bodies has been questioned. In Denmark, the action has been highly controversial, mainly due to "insufficient compensation" of farmers and doubts on their "positive effect on the aquatic environment" (Thorsøe et al., 2017). Consequently, there is a need for new targeted approaches and guidance models for effective mitigation planning; these could include minimal buffering in low-risk locations, to IBZ designs in higher-risk locations, or to suit specific multifunction outcomes. Surface runoff, through delivery pathways, can effectively be intercepted by an installed IBZ. We anticipate that an IBZ will be much more effective for trapping and longer-term retention of sediment and sediment-associated nutrients than traditional vegetated buffer strips installed along watercourses; however, this needs further investigation. For the optimal placement of buffer zones as sediment traps, a national modeling study was conducted. The Water and Tillage Erosion Model (WaTEM) tool was used, which allows the estimation of sediment redistribution in the landscape at fine spatial resolution, including delivery to streams (Van Oost et al., 2000). A total of nearly 20,000 sites was mapped where annual sediment loss from fields was in excess of 250 kg, comparable with the example study from IBZ_{DK-II}. A simplified calculation shows that it would last several hundred years before the aquatic part of the IBZ_{DK-II} would become completely filled with sediment. This calculation is based on the pond area and depth of about 175 m^2 and 0.8 m, respectively, a bulk density of the sediment of about 1.6 g cm^{-3} (Hillel, 1980), and an estimated annual sediment removal of 1.34 kg m^{-2} (this study) equating to a siltation rate of 0.15 m^3 yr^{-1} or a sediment growth rate of approximately 1 mm yr^{-1}.

The N concentrations in IBZ_{DK} and IBZ_{GB} inlet tile drainage water were in the upper range of water quality records for agricultural drain waters (Hill, 2018). However, the P concentration was relatively low, and especially in IBZ_{DK-II}, this was probably linked to a surplus of Fe in the catchment soils, as implied by larger deposits of Fe ochre at the bottom of the receiving stream. The 8 to 38% removal efficiency documented for TN and that of 18 to 52% for TP in the mass balance for the two IBZ_{DK} zones in this study are in the same range as shown for constructed wetlands (Ross et al., 2016). Nitrate removal in dry buffers shows wide variations, ranging from net removals of up to 90% to net releases of N (Uusi-Kämppä et al., 2000; Zhang et al., 2009). Recently, the efficiency of dry buffer zones for N removal has been questioned in many studies (Heinen et al., 2012; Kahle et al., 2013; Thorsøe et al., 2017; Hille et al., 2018a). Our study showed that IBZs will provide an average nitrate removal efficiency of 23 to 37% and a TP removal efficiency of 18 to 52%. However, higher N loads were associated with a decrease in the removal efficiency (Fig. 3) so that their functioning may potentially be increased by modification of soil substrates, improving the redox conditions or increasing the HRT (Vymazal, 2011). Thus, it is well known that denitrification by heterotrophic, facultative, aerobic bacteria depends on a supply of nitrate as electron acceptors and available organic C as electron donors. During the random sampling, we found that ∼30% of dissolved organic C (i.e., ∼5 mg C L^{-1}) can be denoted as bioavailable, as it is related to biopolymers and low-molecular-weight substances (Zak et al., 2018), resulting in an accessible molar dissolved organic C/nitrate N ratio of <1. Accordingly, denitrification was probably C limited, at least in the initial part of the filtration zone where surface water was first infiltrating and nitrate concentrations were mostly >5 mg N L^{-1}. The removal capability for nitrate will expectedly improve naturally with increasing growth of trees, since both the infiltration capability and the C availability, inter alia by root exudates, should increase over time (Senbayram et al., 2012). An opposite trend might occur for phosphate if the sorption capacity of soils becomes exhausted over time or unfavorable conditions exist from the beginning. Thus, for IBZ_{DK-I}, a higher P load reduced the removal efficiency (Fig. 4), or as shown for IBZ_{DK-II}, there was even negative phosphate removal (Table 4), as proposed to result from buffer P saturation (Stutter et al., 2009). In such cases, harvesting plants and/or adding substrates having a high P-binding capacity might be recommended; however, the usage of redox-sensitive compounds like Fe(III) hydroxides might be counterproductive if redox potential declines strongly during warm summer periods (Zak et al., 2014).

Our study showed considerable nutrient retention in the willow plots within several years of planting. A further choice comes from deciding on the bioenergy and water quality benefits of woody versus herbaceous biomass buffers. Applying primary data, Fortier et al. (2015) demonstrated that 9 yr of annual herbaceous biomass (natural colonization) gave similar N but twice as high P offtakes from a single harvest of 9-yr poplar (three hybrid poplar clones: *Populus deltoides* W. Bartram ex Marshall × *P. nigra* L. [DxN-3570], *P. canadensis* Moench × *P. maximowiczii* A. Henry [DNxM-915508], and *P. maximowiczii* × *P. balsamifera* L. [MxB-915311]) growth (2222 stems ha^{-1} in a 4.5-m-wide buffer). This was supported by Ferrarini et al. (2017), who found that woody biomass was better for N uptake, whereas herbaceous buffers were better at trapping sediment runoff; also, greater P removal was observed for cumulative annual herbaceous harvests than a single woody material harvest at identical timescales. Rosa et al. (2017) observed a 64% reduction in shallow groundwater nitrate at the edge of hybrid willow plots compared with corn (*Zea mays* L.) crop plots, but a 35% increase in soluble P. Fortier et al. (2015) observed double the supply rates of N and P to ion exchange soil solution samplers in herbaceous versus woody biomass buffers; this may increase the herbaceous biomass nutrient uptake but constitutes a leaching risk at some sites outside of growth periods. The choice between woody versus herbaceous biomass buffers requires consideration and guidance concerning local water quality goals, on-farm motivations, funding, and management cost benefits.

Biodiversity

Edge habitats and ecotones are generally species-rich habitats that are in decline in the landscape, particularly due to highly intensive farming practices (Kleijn et al., 2009; Tscharntke et al., 2012; Hille et al., 2018b). Since IBZs, like all riparian buffers, are challenged by nutrient-enriched water, they are accessible for colonization by species adapted to eutrophic conditions. However, IBZs are likely to have a greater moisture in the filter bed than vegetated buffer strips due to the infiltration of pond water and shading from trees, and thus they will provide an attractive habitat to riparian species in particular. Accordingly, IBZs might, although they are engineered, exhibit functions more similar to those of natural riparian areas than vegetated buffer strips that are often infrequently wet. Although IBZs will constitute only a small part of a riparian corridor, compared with vegetated buffer strips, they could contain a higher species diversity and function as important stepping stones for species dispersal, as shown for amphibians and small wetlands (Semlitsch and Bodie, 1998). Furthermore, they could function as source habitat for invertebrates such as parasitoids and thus be indirectly beneficial to crop health (Van Lenteren et al., 2018). Possibly, the fact that the small water bodies in IBZs are free from fish makes them particularly attractive for amphibians, since fish predation on eggs and juveniles is a major negative factor constituting a serious threat to amphibian populations (Hartel et al., 2007). Carabid abundance and richness were likely higher in the controls due to (i) the presence of weed matting, which slowed the development of ground level vegetation and prevented above- and belowground interactions (e.g., burrowing behavior of some beetles, feeding, etc.), and (ii) short time for the buffers to establish compared with the controls, which have been managed the same way for many more years. Over a longer period, we would expect more species to colonize, particularly those more characteristic of woodland habitat. The lack of differences in biodiversity between the two tested tree species means that the choice of tree species to be planted can be made on the basis of other factors such as biomass production and nutrient sequestration or practical management aspects. It should be noted that our study period, covering the establishment of trees 28 mo after planting, is considered to represent a very short time period over which to expect changes in habitat and biodiversity, as evidenced by absence, for instance, of woodland-specialist ground beetles (e.g., see Stockan et al., 2014). Additionally, no biodiversity measurements were made at tree canopy level or in the remaining residual streamside habitat. It is highly likely that bankside species will benefit from the augmented protection that IBZs provide against detrimental effects of agricultural practices.

Biomass Production

The inclusion of trees in IBZs aims to provide some of the benefits of riparian forests (shading, leaf litter to aquatic systems, increased surface roughness, bank stabilization, and habitat diversity; e.g., Sweeney and Newbold, 2014, but not assessed here) together with the stabilization of nutrient levels due to their interception within the buffer by the fast-growing planted tree species. The latter process contributes to more persistent sequestering of P that, unlike N, has no gaseous removal pathway and is known to accumulate in buffers, leading to leaching of soluble P (Stutter et al., 2009). The biomass uptake shown here for alder and willow varied between strong and weaker duplicate plots of each species (associated with shading from isolated existing mature trees), but the standing biomass yields (per ha) for the willow plots were 17 to 40 t total DM, 8 to 19 t C, 201 to 458 kg N, and 30 to 70 kg P over the 16-mo growth period after cutting to the ground at 12 mo (to encourage shooting). These figures compare favorably with 55 to 194 t DM, 25 to 91 t C, 277 to 782 kg N, and 20 to 105 kg P over 9 yr recorded in hybrid poplar bioenergy buffer plots at four sites of former cropland in the United States (Fortier et al., 2015). These yields were considerably better for willow than for alder, and alder had much greater proportions of N and P stored in leaves than in stems and branches (Table 6), leaves being a temporary nutrient store that is returned annually to the soil as litter (Fortier et al., 2015). Christen and Dalgaard (2013) made the distinction between the overall lower DM yields that are often attained with short-rotation forestry (optimal 12–20 yr, but possible harvest cycles of 4–10 yr), which is used to manage species like alder, and short-rotation coppice (1–5 yr harvest cycles); their review showing a short-rotation forestry production capacity ranging from 5 to 8 t ha^{-1} yr^{-1} (with alder being the top of the range) compared with a short-rotation coppice up to 14 to 16 t ha^{-1} yr^{-1} (willow and poplar).

Production of biomass can provide a financial benefit and incentive for land managers to implement buffers (Rosa et al., 2017), which they often regard as land taken out of production. Fast-growing tree species can speed up the provision of tree-associated services in former agricultural land compared with the decades required for natural forests to establish (Fortier et al., 2015). This highlights the issue of tradeoffs between how beneficial processes are interrupted by planting, harvesting, and management as opposed to more natural riparian forest stands. However, in a review, Ferrarini et al. (2017) provided positive primary study evidence that biomass "bioenergy" buffers contribute, in the longer term, to soil C sequestration, groundwater N removal, nutrient runoff reduction and soil erosion mitigation, soil health, aboveground biodiversity, and biomass energy yield. Ferrarini et al. (2017) highlighted that there is a lack of data on these factors from a short-term perspective (0–3 yr after establishment), and the present study is thus a contribution to the elucidation of this.

Conventional biomass stands with applied fertilizer have been compared with the use of woody biomass buffers where fertilizer is not applied due to the landscape position and nutrient retention goals (Styles et al., 2016). Yet a high biomass production in bioenergy buffers can be expected in the long term due to elevated nutrient supply due to interception of field runoff and tile drain water.

Conclusions

Integrated buffer zones are designed and targeted to combine greater functionality and provision of multiple ecosystem services, over relatively small surface areas, compared with excessive widths that may be required with a conventional grass buffer. This is of economic interest for farmers, as it means that environmental goals can be met without unduly sacrificing land for crop production. For IBZ_{DK-I}, a previous study showed that a doubling of the IBZ size is recommendable to optimize nutrient removal (Zak et al., 2018). Alternatively, IBZ performance could be improved by using other filter substrates or by optimizing the management. There are still some uncertainties about the best management of IBZs to optimize their multifunctionality. Generally, removal of infilled sediment in the ditch might be necessary, however, within longer timespans of several decades, depending on several factors such as the frequency of storm events and the localized landform, soil type, slope, and field use. Harvesting of tree biomass is aimed to take place at time intervals of approximately 10 to 15 yr, but rather than clear cutting the whole filter bed, a rotational approach is recommended, both from the perspective of water pollution control but also for the benefit of biodiversity. Ferrarini et al. (2017) cite issues of harvesting smaller, fragmented areas and machinery access as management issues for biomass buffers compared with larger bioenergy stands. Stakeholder feedback to our demonstration sites indicated that such biomass production was more attractive where uses were on farm and biomass-fueled grain driers were cited. The harvesting of nontree vegetation at the end of the growing season will substantially reduce the P remobilization, but such an effort has to be subsidized by national management programs. Removal of aboveground biomass might be in conflict with nitrate removal, since the labile C pool for denitrifiers also becomes diminished. Hence, to optimize the provision of services, it is important to consider the multifunctionality of the IBZs when developing management guidelines, taking the timing of harvesting into account.

Supplemental Material

Figures showing functional schemes for the IBZ (Supplemental Fig. S1), IBZ_{GB} water levels, and nitrate concentrations (Supplemental Fig. S2 and S3) are available online. Tables on biomass data and nutrient stock from IBZ_{GB} (Supplemental Table S1); water flows and HRT from IBZ_{DK} (Supplemental Table S2); and species information from IBZs in Sweden and Great Britain (Supplemental Tables S3–S7) are also available online.

Acknowledgments

We want to thank Henrik Stenholt, Niels Bering Ovesen, Marlene Venø Skjærbæk, Dorte Nedergaard, Carsten Brian Nielsen, Rikke Holm, and Carina Lohman for support with laboratory and field work. Tinna Christensen and Ane Kjeldgaard are acknowledged for graphical assistance and Anne Mette Poulsen for editing the text. Martina Saldner did the invertebrate sampling in the Swedish IBZs. The investigations were supported by the Strategic Research Foundation–Innovation Fund Denmark project "BUFFERTECH—Optimization of Ecosystem Services Provided by Buffer Strips Using Novel Technological Methods" (Grant no. 1305-0017B) and from the Nordic Centre of Excellence BIOWATER, funded by Nordforsk under Project no. 82263. Biodiversity studies in Sweden were supported by Brita and Sven Rahmns Foundation. The IBZ_{SE-II} (Sweden) and IBZ_{DK-I} (Denmark) sites were part of the project "Integrated Buffer Zones" financed by BalticSea2020. The GB site was supported by both BUFFERTECH and the Scottish Government's Rural and Environment Science and Analytical Services Division. The IBZ design is based largely on the design of Peter Feuerbach, Halmstad, Sweden.

References

Ahlén, I., C. Andrén and G. Nilsson. 1995. Sveriges grodor, ödlor och ormar. ArtDatabanken, Uppsala and Naturskyddsföreningen, Stockholm. AB Bocktryck, Helsingborg.

Andersen, H.E., B. Kronvang, S.E. Larsen, C.C. Hoffmann, T.S. Jensen, and E.K. Rasmussen. 2006. Climate-change impacts on hydrology and nutrients in a Danish lowland river basin. Sci. Total Environ. 365:223–237. doi:10.1016/j.scitotenv.2006.02.036

Andersen, N.M. 1996. *Heteroptera Gerromorpha*, semiaquatic bugs. In: A. Nilsson, editor, Aquatic insects of North Europe. Apollo Books, Stenstrup, Denmark. p. 77–90.

Arnell, N.W. 1998. Climate change and water resources in Britain. Clim. Change 39:83–110. doi:10.1023/A:1005339412565

Arora, K., S.K. Mickelson, M.J. Helmers, and J.L. Baker. 2010. Review of pesticide retention processes occurring in buffer strips receiving agricultural runoff. J. Am. Water Resour. Assoc. 46:618–647. doi:10.1111/j.1752-1688.2010.00438.x

Braskerud, B.C. 2001. The influence of vegetation on sedimentation and resuspension of soil particles in small constructed wetlands. J. Environ. Qual. 30:1447–1457. doi:10.2134/jeq2001.3041447x

Brown, R., J. Ferguson, M.J. Lawrence, and D. Lees. 2003. Tracks and signs of the birds of Britain and Europe. 2nd ed. Christopher Helm, London.

Brown, R.W., M.J. Lawrence, and J. Pope. 1982. Spår och kännetecken hos Europas djur. Ica Bokförlag, Västerås, Sweden.

Cederhagen, T., and G. Nilsson. 1979. Grod-och kräldjur i Norden. Fältbiologerna, Sollentuna, Sweden.

Christen, B., and T. Dalgaard. 2013. Buffers for biomass production in temperate European agriculture: A review and synthesis on function, ecosystem services and implementation. Biomass Bioenergy 55:53–67. doi:10.1016/j.biombioe.2012.09.053

Collentine, D., and M. Futter. 2018. Realising the potential of natural water retention measures in catchment flood management: Trade-offs and matching interests. J. Flood Risk Manag. 11:76–84. doi:10.1111/jfr3.12269

Collins, A.L.H., G. Zhang, Y. and J. Whitehaed. 2009. Mitigating diffuse water pollution from agriculture: Riparian buffer strip performance with width. CAB Rev. 4:039. doi:10.1079/PAVSNNR20094039

Cranston, P.S. 1982. A key to the larvae of the British *Orthocladiinae* (*Chironomidae*). Publ. 45. Freshwater Biol. Assoc., Ambleside, UK.

Dannelid, E., and G. Sahlén. 2007. Trollsländor i Sverige: En fälthandbok. Entomologiska föreningenn, Stockholm.

DMI. 2017. Vejr Klima Hav. Danmarks Meteorol. Inst., Copenhagen. http://www.dmi.dk/vejr/arkiver/vejrarkiv/ (accessed 23 Apr. 2018).

Dorioz, J.M., D. Wang, J. Poulenard, and D. Trevisan. 2006. The effect of grass buffer strips on phosphorus dynamics- A critical review and synthesis as a basis for application in agricultural landscapes in France. Agric. Ecosyst. Environ. 117:4–21. doi:10.1016/j.agee.2006.03.029

Edington, J.M., and A.G. Hildrew. 1995. Caseless caddis larvae of the British Isles. Sci. Publ. 53. Freshwater Biol. Assoc., Ambleside, UK.

Elliott, J.M., and U.H. Humpesch. 1983. A key to the adults of the British *Ephemeroptera*. Sci. Publ. 47. Freshwater Biol. Assoc., Ambleside, UK.

Elliott, J.M., and K.H. Mann. 1998. A key to the British freshwater leeches. Sci. Publ. 40. Freshwater Biol. Assoc., Ambleside, UK.

Engblom, E. 1996. *Ephemeroptera*, mayflies. In: A. Nilsson, editor, Aquatic insects of North Europe. Apollo Books, Stenstrup, Denmark. p. 13–53.

ESRI. 2014. ArcGIS desktop help 10.2 Geostatistical Analyst. ESRI. http://resources.arcgis.com/en/help/main/10.2/index.html (accessed 27 Dec. 2018).

European Commission. 2016. Direct payments 2015-2020. Decisions taken by member states: State of play as at June 2016. Information note. Eur. Comm., Brussels.

Ferrarini, A., P. Serra, M. Almagro, M. Trevisan, and S. Amaducci. 2017. Multiple ecosystem services provision and biomass logistics management in bioenergy buffers: A state-of-the-art review. Renew. Sustain. Energy Rev. 73:277–290. doi:10.1016/j.rser.2017.01.052

Fortier, J., B. Truax, D. Gagnon, and F. Lambert. 2015. Biomass carbon, nitrogen and phosphorus stocks in hybrid poplar buffers, herbaceous buffers and natural woodlots in the riparian zone on agricultural land. J. Environ. Manage. 154:333–345. doi:10.1016/j.jenvman.2015.02.039

Groisman, P.Y., R.W. Knight, D.R. Easterling, T.R. Karl, G.C. Hegerl, and V.N. Razuvaev. 2005. Trends in intense precipitation in the climate record. J. Clim. 18:1326–1350. doi:10.1175/JCLI3339.1

Hansen, H.P. and F. Koroleff. 2007. Determination of nutrients. In: Grasshoff, K., et al., editors, Methods of seawater analysis. Wiley, New York. p. 159–228.

Hartel, T., N. Szilárd, D. Cogalniceanu, K. Öllerer, C.I. Moga, and L. Demeter. 2007. The effect of fish and aquatic habitat complexity on amphibians. Hydrobiologia 583:173–182. doi:10.1007/s10750-006-0490-8

Hassan, R., R. Scholes, and N. Ash. 2005. Ecosystems and human well-being: Current state and trends. Island Press, Washington, DC.

Heinen, M., I.G.A.M. Noij, H.I.M. Heesmans, J.W. Groenigen, P. Groenendijk, and J.T.N.M. Thissen. 2012. A novel method to determine buffer strip effectiveness on deep soils. J. Environ. Qual. 41:334–347. doi:10.2134/jeq2010.0452

Hill, A.R. 1996. Nitrate removal in stream riparian zones. J. Environ. Qual. 25:743–755. doi:10.2134/jeq1996.00472425002500040014x

Hill, A.R. 2018. Landscape hydrogeology and its influence on patterns of groundwater flux and nitrate removal efficiency in riparian buffers. J. Am. Water Resour. Assoc. 54:240–254. doi:10.1111/1752-1688.12606

Hille, S., D.K. Andersen, B. Kronvang, and A. Baattrup-Pedersen. 2018a. Structural and functional characteristics of buffer strip vegetation in an agricultural landscape: High potential for nutrient removal but low potential for plant biodiversity. Sci. Total Environ. 628–629:805–814. doi:10.1016/j.scitotenv.2018.02.117

Hille, S., D. Graeber, B. Kronvang, G.H. Rubæk, N. Onnen, E. Molina-Navarro, et al. 2018b. Management options to reduce phosphorus leaching from vegetated buffer strips. J. Environ. Qual. doi:10.2134/jeq2018.01.0042 (in press).

Hillel, D. 1980. Fundamentals of soils physics. Acad. Press, New York.

Hoffmann, C., C. Kjaergaard, J. Uusi-Kämppä, H. Hansen, and B. Kronvang. 2009. Phosphorus retention in riparian buffers: Review of their efficiency. J. Environ. Qual. 38:1942–1955. doi:10.2134/jeq2008.0087

Hynes, H.B.N. 1977. Adults and nymphs of the British stoneflies (*Plecoptera*). Sci. Publ. 17. Freshwater Biol. Assoc., Ambleside, UK.

Jansson, A. 1996. *Heteroptera Nepomorpha*, aquatic bugs. In: A. Nilsson, editor, Aquatic insects of North Europe. Apollo Books, Stenstrup, Denmark. p. 91–104.

Jaynes, D.B., and T.M. Isenhart. 2014. Reconnecting tile drainage to riparian buffer hydrology for enhanced nitrate removal. J. Environ. Qual. 43:631–638. doi:10.2134/jeq2013.08.0331

Kahle, P., S. Schoenemann, and B. Lennartz. 2013. Effect of vegetated buffer strips on nitrate inputs into surface waters in drained lowland catchments. Hydrol. Wasserbewirtsch. 57:60–68.

Kronvang, B., H.E. Andersen, C. Børgesen, T. Dalgaard, S.E. Larsen, J. Bøgestrand, et al. 2008. Effects of policy measures implemented in Denmark on nitrogen pollution of the aquatic environment. Environ. Sci. Policy 11:144–152. doi:10.1016/j.envsci.2007.10.007

Kronvang, B. and A.J. Bruhn. 1996. Choice of sampling strategy and estimation method for calculating nitrogen and phosphorus transport in small lowland streams. Hydrol. Process. 10:1483–1501. doi:10.1002/(SICI)1099-1085(199611)10:11<1483::AID-HYP386>3.0.CO;2-Y

Kleijn, D., F. Kohler, A. Báldi, P. Batáry, E.D. Concepción, Y. Clough, et al. 2009. On the relationship between farmland biodiversity and land-use intensity in Europe. Proc. Biol. Sci. 276:903–909. doi:10.1098/rspb.2008.1509

Kronvang, B., N. Vagstad, H. Behrendt, J. Bøgestrand, and S.E. Larsen. 2007. Phosphorus losses at the catchment scale within Europe: An overview. Soil Use Manage. 23:104–116. doi:10.1111/j.1475-2743.2007.00113.x

Lam, Q.D., B. Schmalz, and N. Fohrer. 2011. The impact of agricultural best management practices on water quality in a North German lowland catchment. Environ. Monit. Assess. 183:351–379. doi:10.1007/s10661-011-1926-9

Land, M., J.T.A. Verhoeven, W.J. Mitsch, W. Grannli, A. Grimvall, C.C. Hoffmann, et al. 2016. How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. Environ. Evid. 5. doi:10.1186/s13750-016-0060-0

Lillehammer, A. 1988. Stoneflies (*Plecoptera*) of Fennoscandia and Denmark. Fauna entomologica Scandinavica, Vol. 21. Scandinavian Sci. Press.

Luff, M.L. 2007. The Carabidae (ground beetles) of Britain and Ireland. Handbooks for the identification of British insects, Vol. 4. Part 2. Field Studies Counc., Shrewsbury, UK.

Macan, T.T. 1978. A key to the British fresh- and brackish-water gastropods. 4th ed. Sci. Publ. 13. Freshwater Biol. Assoc., Ambleside, UK.

Mandal-Barth, G. 1982. Småkryp i sötvatten. Fältbiologerna, Sollentuna, Sweden.

Mander, Ü., Y. Hayakawa, and V. Kuusemets. 2005. Purification processes, ecological functions, planning and design of riparian buffer zones in agricultural watersheds. Ecol. Eng. 24:421–432. doi:10.1016/j.ecoleng.2005.01.015

Mayer, P.M., S.K. Reynolds, Jr., M.D. McCutchen, and T.J. Canfield. 2007. Meta-analysis of nitrogen removal in riparian buffers. J. Environ. Qual. 36:1172–1180. doi:10.2134/jeq2006.0462

McIntosh, J.L. 1969. Bray and Morgan soil extractants modified for testing acid soils from different parent materials. Agron. J. 61:259. doi:10.2134/agronj1969.00021962006100020025x

Moeslund, B., B. Løjtnant, H. Mathiesen, L. Mathiesen, A. Pedersen, N. Thyssen, et al. 1990. Danske vandplanter: Vejledning i bestemmelse af planter i søer og vandløb. Danmarks Miljøundersøgelse, Silkeborg, Denmark.

Naturvårdsverket. 2005. Inventering och övervakning av större vattensalamander (*Triturus cristatus*). Ver. 1:0, 2005-04-21. Naturvårdsverket, Stockholm.

Naturvårdsverket. 2010. Handledning för miljöövervakning, Sötvatten, Bottenfauna i sjöars litoral och vattendrag- tidsserier. Ver. 1:1, 2010-03-01. Naturvårdsverket, Stockholm.

Naturvårdsverket. 2012. Fåglar, revirkartering, generell metod. Ver. 1:1, 2012-06-21. Naturvårdsverket, Stockholm.

Neal, C., and H.P. Jarvie. 2005. Agriculture, community, river eutrophication and the Water Framework Directive. Hydrol. Processes 19:1895–1901. doi:10.1002/hyp.5903

Neely, C.L., H.B. Beare, W.L. Hargove, and D.C. Coleman. 1991. Relationships between fungal and bacterial substrate-induced respiration, biomass and plant residue decomposition. Soil Biol. Biochem. 23:947–954. doi:10.1016/0038-0717(91)90175-J

Norling, U., and G. Sahlén. 1997. *Odonata*, dragonflies. In: A. Nilsson, editor, Aquatic insects of North Europe. Apollo Books, Stenstrup, Denmark. p. 13–66.

R Development Core Team. 2018. R: A language and environment for statistical computing. R Found. Stat. Comput., Vienna.

Renard, K.G., G.R. Foster, G.A. Weesies, D.K. McCool, and D.C. Yoder. 1996. Predicting soil erosion by water: A guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). Agric. Handb. 703. U. S. Gov. Print. Office, Washington, DC.

Rosa, D.J., J.C. Clausen, and Y. Kuzovkina. 2017. Water quality changes in a short-rotation woody crop riparian buffer. Biomass Bioenergy 107:370–375. doi:10.1016/j.biombioe.2017.10.020

Ross, J.A., M.E. Herbert, S.P. Sowa, J.R. Frankenberger, K.W. King, S.F. Christopher, et al. 2016. A synthesis and comparative evaluation of factors influencing the effectiveness of drainage water management. Agric. Water Manage. 178:366–376. doi:10.1016/j.agwat.2016.10.011

Savage, A.A. 1989. Adults of the British Hemiptera *Heteroptera*: A key with ecological notes. Sci. Publ. 50. Freshwater Biol. Assoc., Ambleside, UK.

Scanlon, B.R., B.L. Ruddell, P.M. Reed, R.I. Hook, C. Zheng, V.C. Tidwell, and S. Siebert. 2017. The food-energy-water nexus: Transforming science for society. Water Resour. Res. 53:3550–3556. doi:10.1002/2017WR020889

Schou, J., B. Moeslund, L. Båstrup-Spohr, and K. Sand-Jensen. 2017. Danmarks vandplanter. Biologisk Forening Nordvestjylland, Thisted, Denmark.

Semlitsch, R.D., and J.R. Bodie. 1998. Are small, isolated wetlands expendable? Conserv. Biol. 12:1129–1133. doi:10.1046/j.1523-1739.1998.98166.x

Senbayram, M., R. Chen, A. Budai, L. Bakken, and K. Dittert. 2012. N_2O emission and the $N_2O/(N_2O + N_2)$ product ratio of denitrification as controlled by available carbon substrates and nitrate concentrations. Agric. Ecosyst. Environ. 147:4–12. doi:10.1016/j.agee.2011.06.022

Stockan, J.A., J. Baird, S.J. Langan, M.R. Young, G.R. Iason, S.R. Leather, et al. 2014. Effects of riparian buffer strips on ground beetles (Coleoptera, Carabidae) within an agricultural landscape. Insect Conserv. Divers. 7:172–184. doi:10.1111/icad.12043

Stutter, M.I., W.J. Chardon, and B. Kronvang. 2012. Riparian buffer strips as a multifunctional management tool in agricultural landscapes: Introduction. J. Environ. Qual. 41:297. doi:10.2134/jeq2011.0439

Stutter, M.I., S.J. Langan, and D.G. Lumsdon. 2009. Vegetated buffer strips can lead to increased release of phosphorus to waters: A biogeochemical assessment of the mechanisms. Environ. Sci. Technol. 43:1858–1863. doi:10.1021/es8030193

Strand, J.A., L. Feuerbach-Wengel, and P. Feuerbach. 2018. Multi-functionality and holistic approach when ecologically improving an agricultural stream: A case study introducing integrated buffer zones as a landscaping tool in the project LIFE-Goodstream. In: V.G. Sychev and L. Mueller, editors, Novel methods and results of landscape research in Europe, Central Asia and Siberia. Vol. 5. Landscape planning, management and rehabilitation. All-Russian Res. Inst. Agrochem., Moscow. p. 141–145.

Styles, D., P. Börjesson, T. D'Hertefeldt, K. Birkhofer, J. Dauber, P. Adams, et al. 2016. Climate regulation, energy provisioning and water purification: Quantifying ecosystem service delivery of bioenergy willow grown on riparian buffer zones using life cycle assessment. Ambio 45:872–884. doi:10.1007/s13280-016-0790-9

Sweeney, B.W., and J.D. Newbold. 2014. Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: A literature review. J. Am. Water Resour. Assoc. 50:560–584. doi:10.1111/jawr.12203

Syversen, N., and H. Borch. 2005. Retention of soil particle fractions and phosphorus in cold-climate buffer zones. Ecol. Eng. 25:382–394. doi:10.1016/j.ecoleng.2005.06.005

Thorsøe, M.H., M. Graversgaard, and E. Noe. 2017. The challenge of legitimizing spatially differentiated regulation: Experiences from the implementation of the Danish Buffer Zone Act. Land Use Policy 62:202–212. doi:10.1016/j.landusepol.2016.12.030

Tscharntke, T., Y. Clough, T.C. Wanger, L. Jackson, I. Motzke, I. Perfecto, et al. 2012. Global food security, biodiversity conservation and the future of agricultural intensification. Biol. Conserv. 151:53–59. doi:10.1016/j.biocon.2012.01.068

Uusi-Kämppä, J., B. Braskerud, H. Jansson, N. Syversen, and R. Uusitalo. 2000. Buffer zones and constructed wetlands as filters for agricultural phosphorus. J. Environ. Qual. 29:151–158. doi:10.2134/jeq2000.00472425002900010019x

van Lenteren, J.C., K. Bolckmans, J. Kohl, J. Ravensberg, and A. Urbaneja. 2018. Biological control using invertebrates and microorganisms: Plenty of new opportunities. BioControl 63:39–59. doi:10.1007/s10526-017-9801-4

Van Oost, K., G. Govers, and P. Desmet. 2000. Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. Landsc. Ecol. 15:577–589. doi:10.1023/A:1008198215674

VSN International. 2011. GenStat for Windows. 14th ed. VSN Int. Hemel Hempstead, UK.

Vymazal, J. 2011. Constructed wetlands for wastewater treatment: Five decades of experience. Environ. Sci. Technol. 45:61–69. doi:10.1021/es101403q

Wilkinson, M.E., P.F. Quinn, and C.J.M. Hewett. 2013. The Floods and Agriculture Risk Matrix: A decision support tool for effectively communicating flood risk from farmed landscapes. Int. J. River Basin Manage. 11:237–252. doi:10.1080/15715124.2013.794145

WWAP. 2015. The United Nations world water development report 2015. Water for a sustainable world. United Nations World Water Assess. Progr., Paris.

Zak, D., J. Gelbrecht, S. Zerbe, T. Shatwell, M. Barth, A. Cabezas, et al. 2014. How helophytes influence the phosphorus cycle in degraded inundated peat soils: Implications for fen restoration. Ecol. Eng. 66:82–90. doi:10.1016/j.ecoleng.2013.10.003

Zak, D., B. Kronvang, M.V. Carstensen, C.C. Hoffmann, A. Kjeldgaard, S.E. Larsen, et al. 2018. Nitrogen and phosphorus removal from agricultural runoff in integrated buffer zones. Environ. Sci. Technol. 52:6508–6517. doi:10.1021/acs.est.8b01036

Zhang, Y.-C., C.P. Slomp, H.P. Broers, H.F. Passier, and P.V. Cappellen. 2009. Denitrification coupled to pyrite oxidation and changes in groundwater quality in a shallow sandy aquifer. Geochim. Cosmochim. Acta 73:6716–6726. doi:10.1016/j.gca.2009.08.026