WETLAND ECOSYSTEMS

# Impact of vegetation harvesting on nutrient removal and plant biomass quality in wetland buffer zones

E. Jabłońska · M. Winkowska · M. Wiśniewska · J. Geurts · D. Zak · W. Kotowski

**Abstract** Fertiliser use in agriculture increases the non-point pollution of waters with nitrogen (N) and phosphorus (P). Wetland buffer zones (WBZs) are wetland ecosystems between agricultural lands and water bodies that protect surface waters from non-point source pollution. We assessed how vegetation harvesting within WBZs impacts their N and P removal efficiency, nutrient uptake by plants and their biomass quality. We surveyed vegetation of a spontaneously rewetted fen along a small river in Poland, and analysed plant biomass, its nutrient contents and nutrient-leaching potential and the water chemistry. Total N removal reached 34–92% and total P removal 17–63%. N removal was positively related to the initial N concentration, regardless of mowing status. We found a high N removal efficiency (92%) in the harvested site. Vegetation of mown sites differed from that of unmown sites by a higher water-leached carbon and P contents in the biomass. We found that vegetation harvesting may stimulate the overall N removal, but may increase potential biomass decomposability, which eases the recycling of plant-incorporated nutrients back to WBZ. Thus, mowing should always be followed by the removal of biomass. Neglecting already mown WBZs may temporarily lower their nutrient removal efficiency due to potentially faster decomposition of plant biomass.

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## Introduction

Agricultural activities, combustion of fossil fuels and other impacts related to a growing human population have significantly altered the cycles of elements controlling the primary production of ecosystems (e.g. Steffen et al., 2015; Battye et al., 2017). Increased food demands and fertiliser use are the causes of a growing rate of non-point pollution of waters with nitrogen (N) and phosphorus (P). While treatment and water purification can be applied to the majority of point pollution discharges, non-point sources require other approaches (e.g. Mitsch et al., 2001). One of the proposed mitigation strategies is the use of wetland buffer zones (WBZs), i.e. wetlands located in between the agricultural land and the river that capture nutrient-rich runoff water, to reduce nutrient loads in surface waters at the water-land interface (e.g. Mayer et al., 2007; Hoffmann et al., 2009; Jabłońska et al., 2020). WBZs can capture and remove nutrients through several mechanisms, including biological (microbial denitrification, plant uptake), chemical (precipitation of P), physical (sedimentation) and biochemical (humification during decomposition of organic matter), and all these processes can be further influenced by vegetation management (Zak et al., 2019). The impact of plant uptake and biomass harvesting (i.e. mowing and removal of aboveground shoots) on the nutrient export remains, however, particularly understudied. Harvesting can enhance water purification in WBZs because it removes the labile N and P that would otherwise be partially released after plant's death at the end of the growing season. Moreover, harvesting of wetland plants is instrumental for the concept of paludiculture, i.e. the climate friendly use of rewetted peatlands that combines conservation of organic carbon (by avoiding drainage-based peat mineralisation) with economic use of plant biomass (Schröder et al., 2015; Wichtmann et al., 2016). Rewetted riparian fens (i.e. minerotrophic peatlands) constitute an important type of WBZs and, due to legacies of drainage and agricultural use, as well as nutrient input from agricultural land, they are often

highly productive ecosystems (Zak et al., 2018). Therefore, they are potentially promising sites for implementation of paludiculture, thereby combining nutrient cleanup with climate change mitigation. Even in non-peat wetlands, 'wet agriculture' could contribute to combined N, P, and carbon (C) capture. Plant biomass harvested in a WBZ may be used to produce compost for agricultural use (thereby reducing use of synthetic fertilisers), potting media (reducing peat content in conventionally peat-based potting media), energy (limiting fossil fuel burning), or industrial raw materials (replacing conventional non-recyclable materials) (Wichtmann et al., 2016).

However, what still remains uncovered is how mowing impacts the overall efficiency of N and P removal from groundwater and soil in WBZs, especially through interactions with microbial, chemical, and physical properties of the soil. For N removal, plant uptake is the second most important process after the microbial mechanisms (Mander et al., 1995; Hefting et al., 2005; Mander et al., 2017). These include heterotrophic denitrification, found to be the dominant N removal process in riparian zones, as well as different types of microbial nitrate ($NO_3^-$) reduction mechanisms, such as dissimilatory nitrate reduction to ammonium ($NH_4^+$) (Hefting et al., 2003). The rates of microbial N retention and removal processes are both controlled by hydrological factors such as water residence time (Gold et al., 1998), N load (Audet et al., 2020), frequency of flooding (Bernard-Jannin et al., 2017), and by plant species composition and microbial biodiversity (Liu et al., 2017). Microbial uptake is a temporal process that is followed by the release of ammonium after microbial die-off (Rivett et al., 2008), and plants can use this excess N during their growing season. However, microorganisms can also trap N by rapid accommodation to an increased N input and thereby prevent its leaching from the ecosystem during periods of reduced or absent root uptake (Kuzyakov & Xu, 2013). The efficiency of riparian buffer zones for P removal is highly variable and depends on the type and preservation state of the riparian buffer; thus, some wetlands act as sources of P and others as sinks (Hoffmann et al., 2009; Land et al., 2016). Contrary to N, the total P load to the wetlands and wetland size was found not to have a significant effect on P retention (Audet et al., 2020). However, it has been shown that the extent and duration of the P release are connected to the P sorption capacity in

restored soils, the quantity of mobile P forms accumulated in drained soils, and to redox conditions (Zak et al., 2008). This holds true in particular for organic matter-rich soils in groundwater-fed peat-lands, i.e. fens. Due to extensive modern agricultural drainage, riparian wetlands often have significantly altered physical and chemical soil properties and hydrological regimes, so rewetting cannot restore the original hydrological and ecological functioning within a human life-time (Tiemeyer et al., 2007; Zak et al., 2018). Long-term drainage of the topsoil results in peat degradation and subsidence, after which accumulation of rainwater and flooding from the adjacent water bodies may occur. It is unlikely that rewetting of these degraded fens will restore the hydrology to predominantly groundwater-percolated peatlands, rather it will create systems fed by a mixture of groundwater, rainwater and surface water (Wheeler & Proctor, 2000). High nutrient availability in the degraded soils poses a risk of nutrient flux to surface waters after rewetting (Tiemeyer et al., 2006; Zak et al., 2018). Therefore, planting and harvesting of wetland plants (e.g. cattail, reed) are considered a valuable option to remove additional nutrients from such systems (Zak et al., 2014).

The vegetation of WBZs and its harvesting can interact with the above-described mechanisms of N and P removal in several ways. Plant litter supplies organic matter that stimulates denitrification and thus indirectly impacts N removal rates (Hefting et al., 2005). Therefore, harvesting, which exports the plant-incorporated N, may hamper microbial N removal due to the reduction of the labile C required by denitrifying microorganisms. Another potential interaction is caused by soil compaction due to the pressure exerted by mowing machinery. Soil compaction has been reported to stimulate denitrification by increasing water-filled pore space in the soil, thereby lowering the oxygen supply (Li et al., 2014. However, such changes might promote internal release of P due to favoured reduction of redox-sensitive iron(III)-P binding forms (Zak et al., 2018). On the other hand, the P transport to surface or ground water can be diminished due to lowered preferential flow, but this is a topic that requires further research (Wang et al., 2020).

Another important issue is how repetitive harvesting changes the quality of plant biomass, thereby affecting the residence time of nutrients in dead biomass. Unless above ground shoots are mown and removed, nutrient immobilisation in aboveground plant biomass is temporary and significant amounts of plant-immobilised nutrients are released from dead biomass during subsequent decomposition. The main factors controlling decomposition rates are diversity and biomass of invertebrate and microbial decomposers, litter quality, i.e. nutrient stoichiometry and presence of recalcitrant substances (Coûteaux et al., 1995; Leroy & Marks 2006; Zhang et al., 2008; Krishna & Mohan, 2017), microclimate (Coûteaux et al., 1995; Zhang et al., 2008), soil nitrate availability (Hefting et al., 2005), sunlight penetration [which may explain the different rates of leaf litter decomposition in forest and herbaceous communities (Foereid et al., 2018)], and oxygen availability (Greenwood 1961; Sierra et al., 2017). Specifically, persistent soil anoxia in mires can immobilise part of the plant-incorporated nutrients for a long period of time in the form of peat deposits (Wang et al., 2015). Harvesting can indirectly impact most of the mentioned factors, either by modifying litter quality (as species-specific responses to cutting or due to shifts in species composition, e.g. Kozub et al., 2019) or by changing decomposition conditions (e.g. through the mentioned soil compaction or increased insulation of the soil).

The present study is aimed at assessing how vegetation harvesting impacts the overall N and P removal, as well as nutrient uptake by plants and plant biomass quality, within a WBZ occupied by vegetation re-established spontaneously after rewetting of drained fens along a small lowland stream in Poland. There is a need to re-establish similar buffer systems in modified rural landscapes worldwide to improve the quality of surface waters and to buffer water flows to mitigate floods and droughts amplified by climate change (Mitsch et al., 2001; Jabłońska et al., 2020). An urgent question is whether combination of nutrient removal function and biomass production in wet agriculture (or paludiculture) is a viable option. More specifically: do mown and unmanaged WBZs perform differently in terms of nutrient capture and removal? We hypothesise that N and P removal by plant harvesting improves the nutrient removal efficiency in WBZ and alters the plant biomass quality toward a higher refractory character, i.e. to lower nutrient concentrations and/or higher C:N ratios.

## Materials and methods

### Study area

The study area is located in central Poland (approx. coordinates: 52° 36′ N, 21° 03′ E), more specifically in the Ciechanowska Upland physico-geographical mesoregion (Solon et al., 2018) situated in the central part of the North Mazovian Lowland—an undulating plain characterised by moraines and kame hills; it is a typical agricultural land interspersed with a few forest clusters (Fig. 1a). We analysed WBZs bordering agricultural fields along a free-flowing small river, Pokrzywnica, a 16-km-long tributary of the Narew river. Variable organic soils (peat, degraded peat, organic riverine mud) form the upper 50–75 cm of soil

layers that are underlain by sand or impermeable clay. The area used to be a fen (data obtained from the Polish Wetland Database "GIS Mokradła", http://www.gis-mokradla.info). On the basis of analyses of archival aerial photographs (obtained from the Polish Head Office of Geodesy and Cartography) as well as interviews with local farmers, we learnt that a draining channel was made probably in the beginning of XX century and the area was used then for hay mowing. After stepwise cessation of mowing in XX/XXI century, overgrowing by trees and shrubs started at some places. Since about 10 years a spontaneous restoration of high water level and wetland conditions has occurred, most probably thanks to beaver activity.

According to the Köppen-Geiger classification, the study area has a continental warm-summer humid



Fig. 1 Study area, study sites and sampling points: a location of the study area Pokrzywnica in Poland; b location of study sites m1, m2, u1 and u2 in the WBZ in the Pokrzywnica river valley; c location of sampling points and zones A, B and C (A on the border between the WBZ and agricultural field, B in the middle section of WBZ and C close to the river) within an exemplary study site
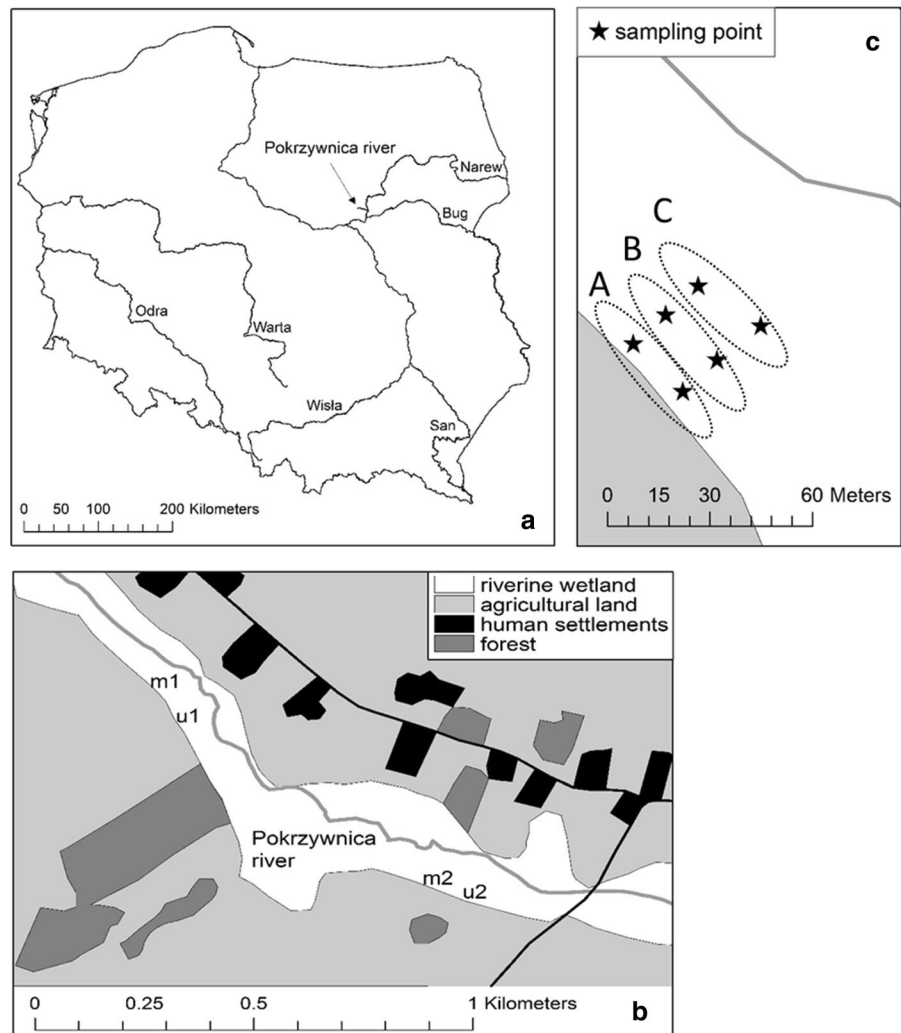
continental climate (Rubel & Kottek, 2010). Meteorological data from the nearest station, Pułtusk, recorded over the last two decades (years 2000–2019) and obtained from repository of the Polish Institute of Meteorology and Water Management—National Research Institute (http://danepubliczne.imgw.pl) using the "climate" package in R (Czernecki et al., 2020), shows a mean annual temperature level of 8.5°C, with a mean temperature in the warmest month (July) of 19.6°C and a mean temperature in the coldest month (January) of − 2.6°C. Mean annual precipitation in this period was 589 mm, with 31 mm in the driest month (April) and 92 mm in the wettest month (July). In the years 2017–2018, when the field research took place, the weather characteristics differed from the climatic means of the last two decades. Thus, in 2017, the average annual temperature was the same as the mean value for the last two decades, January was colder than the long-term mean (mean temperature in January in 2017 was − 4.2°C) and the warmest month was August with 18.6°C. Annual precipitation in 2017 was around 810 mm, with 14 mm in the driest month (January) and 149 mm in the wettest month (June). In 2018, the average annual temperature was 9.4°C, February was the coldest month (mean temperature in February 2018 was − 4.0°C) and the warmest month was July with an average temperature of 20.9°C. Annual precipitation was 527 mm, with 8 mm in the driest month (February) and 139 mm in the wettest month (July).

Field sampling scheme

Four study sites were established. The first pair, comprised a mown site (m1; 52° 36′ 52.5″ N, 21° 02′ 48″ E) and an unmown site (u1; 52° 36′ 48.5″ N, 21° 02′ 51.3″ E) that directly bordered fertilised arable fields of triticale and oats. The second pair, covering a mown site m2 and an unmown site u2 (52° 36′ 35″ N, 21° 03′ 19.5″ E and 52° 36′ 33.7″ N, 21° 03′ 25″ E, respectively), bordered c. 50-m-wide rarely fertilised meadows with fertilised fields lying further behind them (Fig. 1b). Soil types were peat and degraded peat ("moorsh", Okruszko 1993) in m1 site, peat and moorsh with some insertions of riverine mud in u1 site, and moorsh and riverine mud in m2 and u2 sites. Within each study site, two (c. 30 m long) transects were placed perpendicularly to the river. Each transect

consisted of three sampling points located in three different zones: A on the border between the WBZ and the agricultural fields, B in the middle section of the WBZ, and C close to the river (Fig. 1c). Riparian vegetation has been mown (and biomass removed) since at least half of the XX century until today, probably every year, within zones A to B in study sites m1 and m2, and not mown since about 80. of XX century in u1 and u2 (information based on archival aerial photographs and interviews with local farmers).

Water sampling and analysis

Two piezometers were installed at each sampling point: one with its bottom at 75 cm and one with its bottom at 125 cm (reaching the impermeable clay layer if it was shallower than 75 cm/125 cm, respectively). In total, a grid of 48 piezometers was installed in the study area. The piezometers were made of PVC pipes of 4.5 cm diameter perforated along the bottom 25 cm, and covered with filtering gauze.

The groundwater level was measured in August 2017, November 2017, April 2018 and June 2018 in each piezometer and water was pumped out following measurements. After the tubes were refilled, 100 ml water samples were taken. All samples were cooled during sampling and transportation. On the day of sampling, pH and electronic conductivity (EC) were measured using a HACH HQ40D multimeter. The samples were filtered (0.45 μm pore size) before further analysis and separated into three sets: 20 ml was analysed for total inorganic carbon (TIC) in Laboratory of Biogeochemistry and Environmental Protection at the Biological and Chemical Research Centre at the University of Warsaw 20 ml was frozen for colorimetric analysis of $NO_3^-$ and $NH_4^+$ on an Auto Analyser III system (Bran + Luebbe GmbH, Norderstedt, Germany), and 10 ml was preserved with 0.1 ml of 65% $HNO_3$ for bulk ion concentration analysis (including dissolved P) using ICP-OES (iCAP 6000, Thermo Fischer Scientific, Bremen, Germany). Total nitrogen (TN) concentration was calculated as a sum of N-$NO_3^-$ and N-$NH_4^+$. Bulk analysis of dissolved P on ICP-OES does not allow for separation into inorganic and organic fractions. However, unless sites are in a near-natural state, most of the dissolved P (> 95%) will occur as phosphate ($PO_4^{3-}$), while in natural peatlands a higher proportion (> 50% of total dissolved P) can be formed by organic P (Zak

et al., 2004). It still remains unclear to what extent organic P can be used for microbial metabolism or as a plant nutrient. Accordingly, data on P concentration may potentially partly overestimate the amount of dissolved bioavailable P. Analyses on ICP-OES and Auto Analyser were performed at the Department of Aquatic Ecology and Environmental Biology at the Radboud University in Nijmegen, the Netherlands.

### Nutrient removal in groundwater

In our analysis, removal refers to the ratio of a reduced concentration of N and P in the water phase in the WBZ to the concentration in zone A, i.e. in the zone where water coming from agricultural fields enters WBZ (Land et al., 2016). Nutrient removal from zone A to B was calculated following the equation (Eq. 1):

$$
\begin{aligned}
\text{RemAB}_{\text{TN}/\text{N-NO}_3/\text{N-NH}_4/\text{P}} \\
= \big( \big( A_{\text{TN}/\text{N-NO}_3/\text{N-NH}_4/\text{P}} - B_{\text{TN}/\text{N-NO}_3/\text{N-NH}_4/\text{P}} \big) / \\
A_{\text{TN}/\text{N-NO}_3/\text{N-NH}_4/\text{P}} \big) \times 100\%,
\end{aligned}
\tag{1}
$$

where RemAB—nutrient removal from zone A to B; $A_{\text{TN}/\text{N-NO}_3/\text{N-NH}_4/\text{P}}$—concentrations of TN, N-NO$_3^-$, N-NH$_4^+$ or P in groundwater in zone A (mg l$^{-1}$); $B_{\text{TN}/\text{N-NO}_3/\text{N-NH}_4/\text{P}}$—concentrations of TN, N-NO$_3^-$, N-NH$_4^+$ or P in groundwater in zone B (mg l$^{-1}$). In order to calculate the removal from zone A to C, the similar equation was used but with C zone values instead of B zone values. Negative values mean that the nutrient concentrations in water leaving the WBZ (i.e. in zone B or C) were higher than in the water in zone A. The calculation was performed on the averaged values of nutrient concentrations for each zone in a study site, i.e. averages for two sampling points in each zone in every study site, two piezometers at each sampling point, four measurements in each piezometer.

### Vegetation survey

At each sampling point, the percentage cover of all species was recorded within $2 \times 2$ m quadrats in August 2017 and the total aboveground biomass of all vascular plants was sampled from an area of 0.14 m$^2$ in July 2018 (just before mowing on the mown areas). Biomass samples were dried (in 70°C for 24 h) and weighed.

### Nutrient uptake and nutrient-leaching potential of plant biomass

The dried biomass was cut in 5 cm long pieces, after which the sampled material was homogenised in a fine-grain mill. The total P content in the samples was determined as soluble reactive P. Molybdenum blue method was used after an acid digestion procedure where 10 mg of dry sample was mixed with 2 ml 10 M H$_2$SO$_4$, 4 ml 30% H$_2$O$_2$ and 20 ml de-ionised water at 160°C for 2 h (Murphy & Riley, 1962). The N and C contents in the samples were determined using a CN elemental analyzer (Vario EL; Elementar, Mt. Laurel, New Jersey, USA). Net nutrient uptake was calculated from nutrient concentrations and biomass data (Zak et al., 2014). The nutrient and C uptake by plants were calculated using the following equation (Eq. 2):

$$
\text{NCUP} = \text{Pl}_{\text{N}/\text{P}/\text{C}} \times \text{DM},
\tag{2}
$$

where NCUP is nutrient and C uptake by plants (kg ha$^{-1}$), Pl$_{\text{N}/\text{P}/\text{C}}$ is the concentrations of N, P or C, respectively, in dried plant biomass (kg kg$^{-1}$ dry mass) and DM is the dry mass of the aboveground biomass of plants (kg ha$^{-1}$) sampled from an area of 0.14 m$^2$ (see above). Due to possible litter loss before biomass harvest, the data refer to the net production of aboveground biomass and the net nutrient uptake, respectively.

To imitate situation when cut biomass is left in the field, and asses the differences in short-term nutrient-leaching potential of cut biomass from regularly mown versus long abandoned/unmanaged WBZs, we performed an experiment on the collected, dried and cut biomass samples, following a slightly modified approach of that presented in Shumilova et al. (2019). To estimate N and P leaching, 5 g of the dried plant material was inserted in 250 ml large dark glass bottles. The bottles were filled with 200 mg l$^{-1}$ NaCl solution that acted as a leaching solution to imitate the ionic strength found in groundwater. The bottles were covered with caps and left on a 100 RPM platform shaker in dark climatic chambers for 24 h. In the next step, the solution was filtered through 0.45 μm pre-rinsed glucose-acetate filters and the amounts of soluble reactive P and dissolved N were measured. Soluble reactive P concentrations were determined using the molybdenum blue method, developed by Murphy & Riley (1962), by spectrophotometer (Cary

60 UV–Vis; Agilent Technologies, USA). Dissolved N and dissolved organic C were analysed with a CN elemental analyzer (TOC-L, Shimadzu, Kyoto, Japan). C, N and P analysed in this experiment are further referred to as "water-leached C, N, P".

Analyses of the nutrient content in plant biomass and a nutrient-leaching experiment were carried out at the Department of Chemical Analytics and Biogeochemistry of the Leibniz-Institute of Freshwater Ecology and Inland Fisheries in Berlin, Germany.

### Data analysis

Water chemical properties were averaged for two sampling points in each zone in every study site, two piezometers at each sampling point, four measurements in each piezometer. Ordination analyses for phytosociological relevés and vegetation data in relation to the chemical parameters were performed using Canoco5 (Šmilauer & Lepš, 2014). Detrended correspondence analysis (DCA) was performed for phytosociological relevés as the gradient length for the first DCA axis exceeded 4 SD. Detrended Cross-Correlation Analysis (DCCA) was performed for phytosociological relevés using the concentrations of N, P and C in harvested biomass as explanatory variables, and N, P and TIC in water as supplementary variables. The nutrient content in the harvested biomass was statistically compared between mown and unmown sites using a non-parametric Wilcoxon test performed in R (R Core Team, 2016). Samples from zones A to B in study sites m1 and m2 were treated as "mown" and the rest of the samples as "unmown" in these comparisons.

## Results

### Vegetation characteristics

The vegetation characteristics of the study sites are summarised in a phytosociological table in Appendix 1—Supplementary Material. Four groups of relevés could be distinguished on the basis of species composition. The dominant species in the first, second and third group was *Carex gracilis*, while the fourth group was dominated by *Phragmites australis*, *C. riparia* and *C. acutiformis*. The first group consisted of relevés made in zones A and B from the mown site m1. The vegetation there differed from the vegetation in the other groups by the presence of poor fen species (Cl. *Scheuchzerio-Caricetea*, O/All. *Caricion nigrae*), species of flooded pastures (O/All. *Agropyro-Rumicion crispi*) and bryophytes (*Brachythecium rutabulum*, *Calliergonella cuspidata*). The first group lacked riverside tall herbs (All. *Filipendulion*) that were present in the remaining three groups of relevés. The second group included sampling points in zones A and B from study sites m2 and u2, as well as sampling points in zone C from site m1. There, the vegetation was characterised by the presence of species typical of flooded pastures (O/All. *Agropyro-Rumicion crispi*) and sporadic occurrence of species of alder swamp forests and willow shrubs (Cl. *Alnetea glutinosae*). The third group included sampling points from zones B and C in study sites m2 and u2 where the most homogenous stands of *C. gracilis* occurred. The fourth group consisted of sampling points from zones A, B and C from study site u1 where vegetation was dominated by reeds and tall sedge species (Cl. *Phragmitetea*, All. *Magnocaricion*) with a considerable share of ruderal species (Cl. *Artemisietea vulgaris*).

The above-mentioned groups of relevés are illustrated by a DCA graph (Fig. 2) The first and the fourth group were the most distinct, while the second and third group largely overlapped, with the latter forming a more compact cluster in the DCA. The main variation of riparian vegetation in the study sites (expressed along the first ordination axis in Fig. 2) probably represented the flooding gradient, spanning from communities dominated by *Phragmites australis* and the sedges *Carex riparia* and *C. acutiformis* close to the rivers, to grass-dominated meadows in drier habitats. Communities of *Carex gracilis* occupied an intermediate position along the gradient. The second important gradient (along the second ordination axis) was visible mainly in drier habitats and may indicate the impact of mowing—spanning from a mown brown moss fen with *Carex nigra* to unmown communities with *Phalaris arundinacea* and *Juncus effusus*.

Plant biomass per area in the mown sites was significantly lower than in the unmown sites (Fig. 3), whereas we did not find any differences in either the molar C:N ratio or the N:P ratio between mown and unmown sites (Fig. 3). The median molar C:N ratio was about 40 and the median mass N:P ratio about 7.
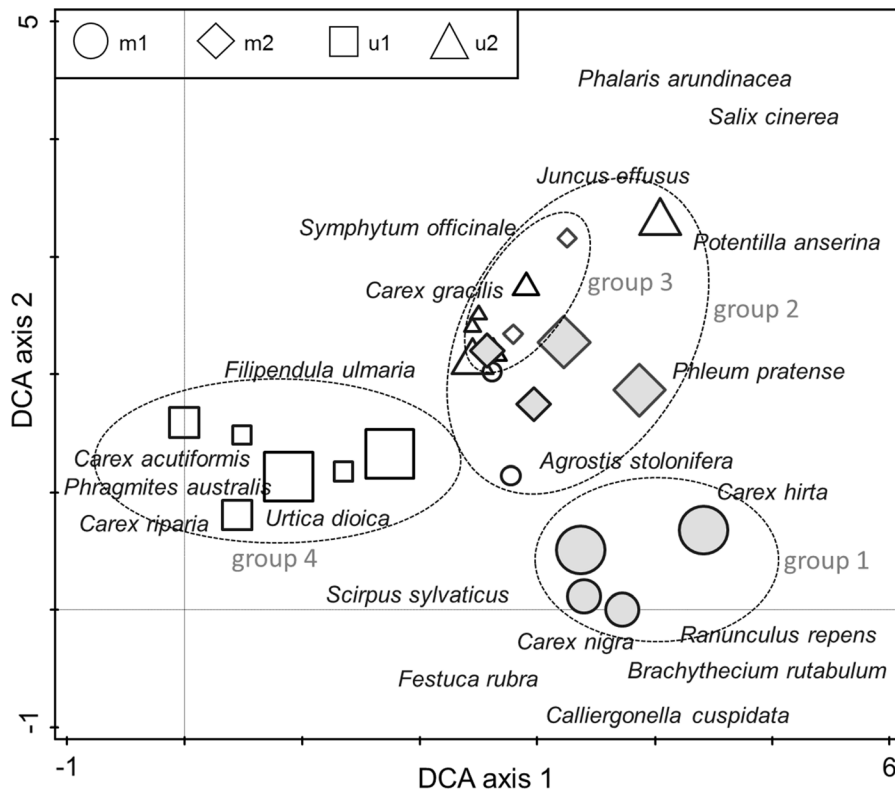
**Fig. 2** DCA of phytosociological relevés. Eigenvalues: 1st axis—0.697, 2nd axis—0.411. m1, m2, u1, u2—study sites. Grey symbols—sampling points located in the mown sites. White symbols—sampling plots located in the unmown sites. Sampling points in zone A—large symbols, in zone B—medium symbols and in zone C—small symbols. The location of a species is indicated by the location of its name; only the 20 most important species are shown. Groups 1–4 correspond to the groups of relevés in the phytosociological table in Appendix 1—Supplementary Material
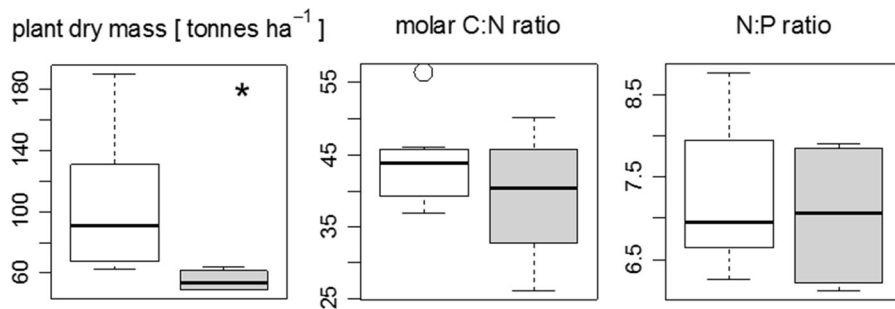


**Fig. 3** Median, first and third quartiles and 1.5 interquartile range for the dry weight of harvested biomass, molar C:N ratio and mass N:P ratio in plant biomass. Grey boxes gather sampling points located in the mown sites, white boxes—in the unmown sites. Differences significant at $p < 0.05$ in Wilcoxon test are marked with an asterisk

Nutrient uptake and leaching by plants

Biomass characteristics depended on the location of the wetland in the river valley and on the harvesting regime (Fig. 4). Communities of *Phragmites australis* and the sedges *Carex riparia* and *C. acutiformis*, located close to the river (medium and small symbols in Fig. 4), were characterised by a higher aboveground biomass and C content in the biomass, as well as lower total N and P contents and lower water-leached N and P contents in the biomass. The vegetation of zone A, situated most distantly from the river bank (large
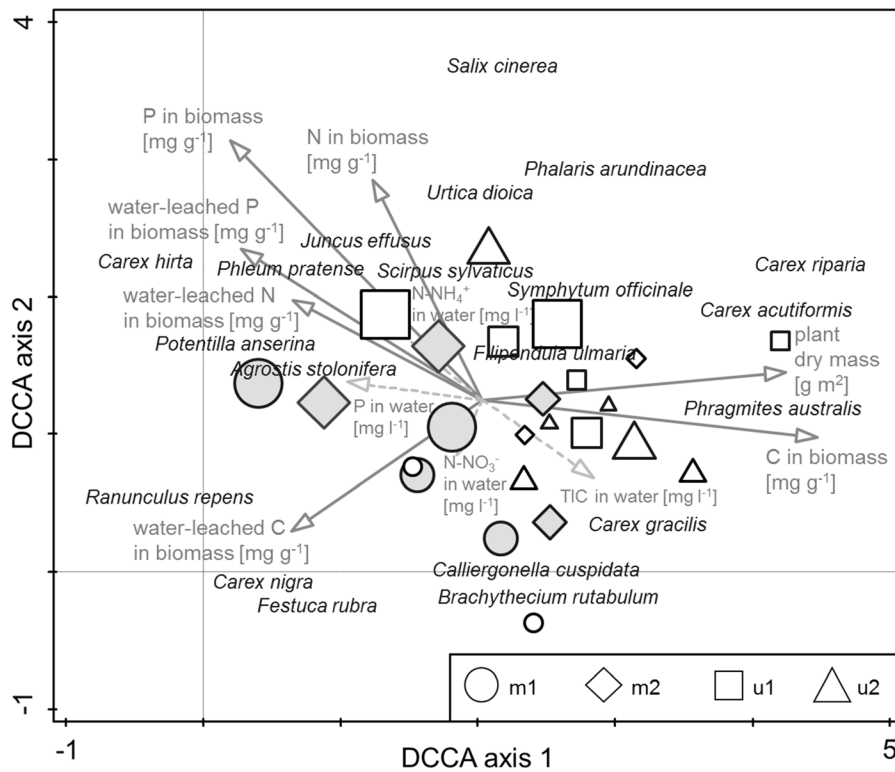
**Fig. 4** DCCA of phytosociological relevés and concentrations of N, P and C in harvested biomass (explanatory variables) as well as N, P and TIC in water (supplementary variables). Explanatory variables account for 46.7% of the total variation. Eigenvalues: 1st axis—0.508, 2nd axis—0.356. m1, m2, u1, u2—study sites. Grey symbols—sampling points located in the mown sites. White symbols—sampling plots located in the unmown sites. Sampling points in zone A—large symbols, in zone B—medium symbols, in zone C—small symbols. Location of a species is indicated by the location of its name; only the 20 most important species are shown

symbols in Fig. 4), was characterised by a lower plant dry mass per area and a lower C content in the biomass, as well as higher total N and P contents and higher water-leached N and P contents in the biomass. The vegetation of the mown sites differed from that of the unmown sites mainly by a higher water-leached C content in the biomass (Figs. 4, 5) but also by a higher water-leached P content in the biomass (Fig. 5). The vegetation of the unmown sites contained more N and C per hectare than in the mown sites (Fig. 5).

Groundwater chemistry

At the sampling points located further from the river bank and closer to the agricultural fields, higher $N\text{-}NO_3$, P and sulphur concentrations in the groundwater were measured, while higher pH, EC as well as calcium, magnesium, silicon and TIC concentrations were measured closer to the river (Fig. 4, Appendix 2—Supplementary Material). The water level fluctuated over the seasons, with a higher water level in autumn and spring where flooding events occurred in zones B and C (Appendix 2—Supplementary Material). See also Appendix 2—Supplementary Material for more detailed results of chemical analyses of water. We did not find any statistically significant impact of mowing on the analysed chemical properties of water (Appendix 2—Supplementary Material).

Nutrient removal in groundwater

Total N removal in the study sites reached 34–92% within the c. 15-m-wide zone AB located closest to the field (Table 1, Fig. 6) and 16–90% within the whole analysed buffer zone AC (Table 1), although for the separate N fractions (nitrate and ammonium) the removal was negative in some study sites (Table 1). In case of total N, as well as nitrate and ammonium, it
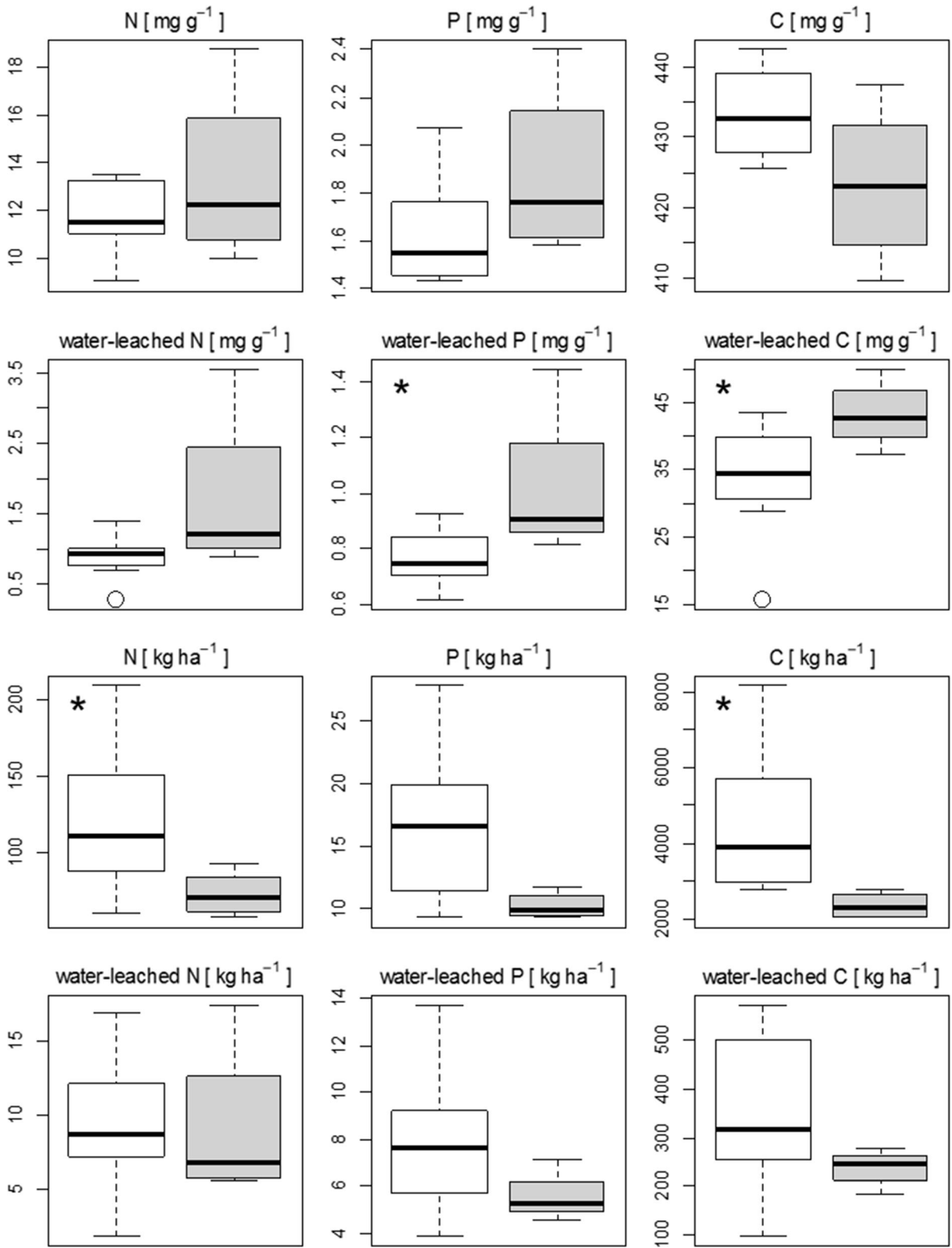
◄**Fig. 5** Median, first and third quartiles and 1.5 interquartile range for N, P and C contents in the harvested biomass. Grey boxes gather sampling points located in the mown sites. White boxes—in the unmown sites. Differences significant at $p < 0.05$ in Wilcoxon test are marked with an asterisk

seems that the higher the initial N content in the groundwater, the higher is the N removal efficiency (Fig. 6). Nitrate was the dominant N fraction in zone A in study sites m1 and u1, whereas ammonium dominated in study sites m2 and u2 (Appendix 2—Supplementary Material). The total amount of N in zone A in sites m1 and u1 was higher than in sites m2 and u2.

P removal efficiency in the study sites reached 17–63% within the c. 15-m-wide zone AB closest to the field, and 16–77% within the whole analysed buffer zone AC (Table 1). P removal efficiency showed no clear relation to the initial P content in the groundwater (Fig. 6).

## Discussion

### Reliability of estimations

We hypothesised that plant harvesting will enhance the nutrient removal efficiency due to the removal of part of the nutrients from the site with harvested biomass, and also significantly alter plant biomass quality in terms of potential decomposability. Indeed,

our results point out that harvesting altered nutrient uptake and thus plant biomass quality, whereas no evident improvement of nutrient removal efficiency appeared in the mown WBZ sites compared with the unmanaged WBZs. Before discussing our results, we acknowledge that the nutrient removal recorded in our relatively recently rewetted WBZs on organic soils, generally falls within the ranges of nutrient removal WBZ functioning described in other studies. Thus, the rate of N removal in our WBZ was 16-92% (Table 1), which corresponds to the median total N removal efficiency of 37% reported in a review by Land et al. (2016). The rate of P removal in our WBZ was 17–77% (Table 1), which corresponds to the range of 32–93% reported in a review by Hoffmann et al. (2009), as well as to the median total P removal efficiency of 46% recorded by Land et al. (2016). Most nutrients were removed already within the first c. 15 metres of our WBZ (the AC removal rates were not consistently higher than the AB removal rates, Table 1), which is in accordance with Mayer et al. (2007) who stated that 15-m-wide buffers may remove an average of 60% of the total N load, and in many cases even more than 95%.

### The impact of mowing on plant uptake and biomass quality

The nutrient uptake by plants as well as biomass quality were influenced more strongly by vegetation harvesting than by species composition. Our indirect

**Table 1** Total nitrogen (TN), nitrate (N-NO$_3^-$), ammonium (N-NH$_4^+$) and phosphorus (P) removal between zone A (located on the border of a wetland buffer zone and next to an agricultural field) and B (located c. 15 m towards the river) of a wetland buffer zone and between zone A and C (located c. 30 m towards and closer to the river)

| Site | mown | | unmown | |
| --- | --- | --- | --- | --- |
| | m1 | m2 | u1 | u2 |
| TN removal AB (%) | 92.56 | 61.09 | 74.68 | 34.66 |
| TN removal AC (%) | 90.21 | 33.60 | 81.40 | 16.10 |
| N-NO$_3^-$ removal AB (%) | 99.41 | − 64.72 | 95.90 | 17.79 |
| N-NO$_3^-$ removal AC (%) | 99.02 | − 255.48 | 97.59 | 63.40 |
| N-NH$_4^+$ removal AB (%) | − 43.74 | 67.00 | − 73.27 | 44.63 |
| N-NH$_4^+$ removal AC (%) | − 84.84 | 35.35 | − 32.02 | 11.67 |
| P removal AB (%) | 17.56 | 56.08 | 63.46 | 51.37 |
| P removal AC (%) | 56.22 | 16.79 | 54.13 | 77.47 |

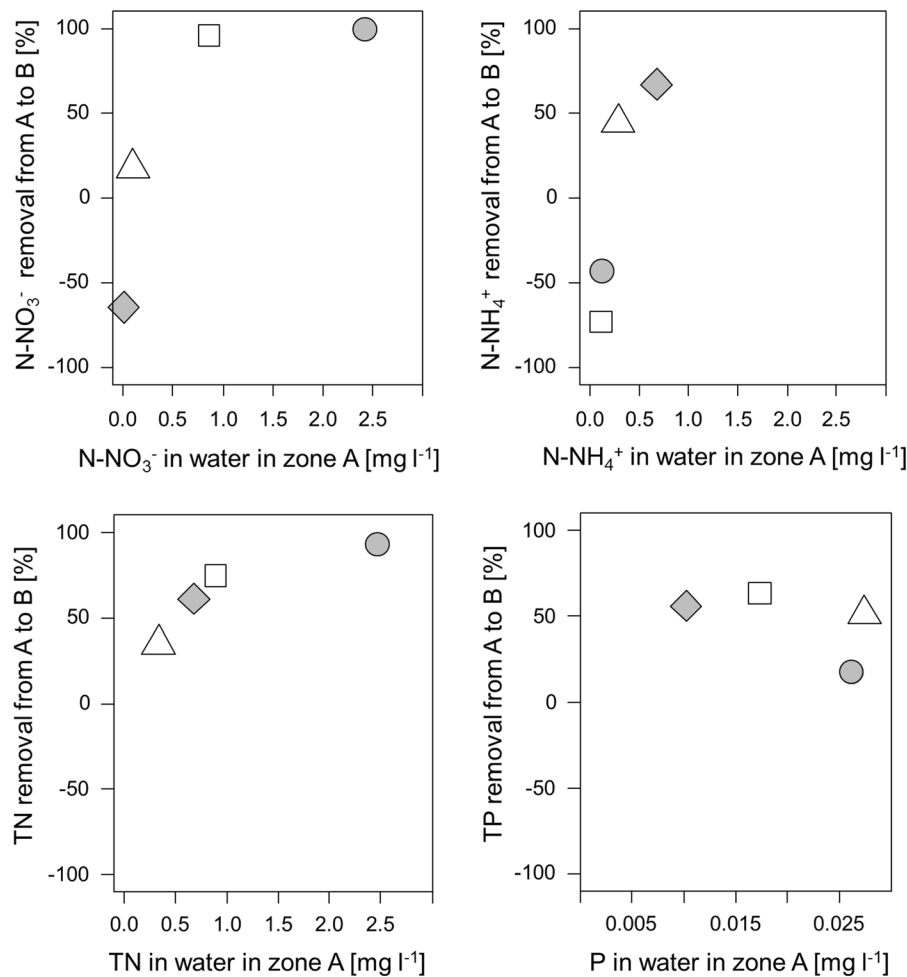Vegetation was mown from zone A to B in study sites m1 and m2

**Fig. 6** Nitrogen removal between zone A (located on the border between the wetland buffer zone and the agricultural field) and B (located c. 15 m towards the river) in relation to the nitrogen content of the groundwater in zone A. Symbols for study sites as in Figs. 1 and 2. Grey symbols—mown sites, white symbols—unmown sites

ordination analysis of the species composition in the study sites revealed clearly differentiated groups of relevés (Fig. 2); however, these clusters disappeared in the direct ordination when applying chemical parameters of the plant biomass (Fig. 4). Still, in the direct ordination, mowing remained an important grouping factor, differentiating the relevés across a gradient of water-leached C in the biomass and weight of plant biomass per area unit (Fig. 4). The above-ground biomass was higher in unmown sites than in mown sites (Fig. 3), which was the main reason for the higher N, C (significant difference) and P (clear tendency) stock in plant biomass per hectare in the unmown compared with the mown sites (Fig. 5). The amounts of water-leached N, P and C in plant biomass

per hectare did not differ significantly between mown and unmown sites (Fig. 5). We found no significant differences between mown and unmown sites regarding N, P and C concentration in the biomass (Fig. 5) or the molar C:N ratio (Fig. 3). The median molar C:N ratio was about 40 in both groups. At all sampling points, the dominant species were sedges or reed. These are all high-C and low-N plants, which explains their relatively high C:N ratio compared with grass-dominated vegetation (Scheffer & Aerts, 2000; Wang et al., 2015). Also, both the mown and unmown sites appeared to have N-limited vegetation (N:P ratio about 7, c.f. Koerselman & Meuleman, 1996; Olde Venterink et al., 2003). The difference that we found between mown and unmown sites was that the

concentrations (mg g$^{-1}$) of water-leached C, P (significant difference) and N (clear tendency) in plant biomass were higher in the mown than in the unmown sites. Accordingly, mowing might potentially enhance tissue decomposability. P concentrations in plant tissues and P leaching can be species specific (Zak et al., 2014). As the investigated sites have been managed by mowing for decades, we cannot say whether the observed difference in plant biomass quality was an effect of mowing or rather a precondition for farmers to choose plots suitable for hay production. Nevertheless, in the current situation a possible cessation of mowing may possibly cause increased decomposition and higher N and P leaching than in sites left unmown for a long time. However, plants may interact with microorganisms in the uptake of inorganic N; for instance, microbial immobilisation can prevent leaching of inorganic N from riparian ecosystems in winter when the nutrient uptake by plants is arrested (Kuzyakov & Xu, 2013).

The impact of mowing on N removal

We found a high N removal efficiency in the mown site m1 (Table 1), which might be explained either by N removal with hay or due to soil compaction by the mowing machinery, which may stimulate denitrification by lowering the oxygen content in soil pores (Li et al., 2014). However, this does not have to be a result of mowing, as N removal in the unmown sites was also substantial. We observed a clear relation between N removal efficiency and the initial N content in groundwater (Fig. 6) regardless of the mowing status. Similarly, Addy et al. (1999) recorded no significant difference in N removal efficiency between forested and mown riparian buffers. We found that the higher the initial N concentration, the higher was the amount of N removed in the WBZ, which is consistent with the findings of other studies (Land et al., 2016; Weisner et al., 2016). However, the opposite relation found for the percentage of N removal relative to the initial N concentration is not in line with the results of other investigations. Most of the existing reviews report a negative relation between percentage N removal efficiency and the initial N load (Fisher & Acreman, 2004), whereas our results identify a positive relation (Fig. 6). The reason for this might be that we analysed relatively low ranges of N concentrations where nitrate rather than C limits the denitrification rates.

Similar results have previously been reported by Hunter et al. (2009) who found a negative relation of total N removal efficiency (%) within a wide range of N loads (0–500 g m$^{-2}$ year$^{-1}$) but a positive relation within a lower range of N loads (0–60 g m$^{-2}$ year$^{-1}$).

The impact of mowing on P removal

Also for P, the impact of mowing on removal efficiency within the WBZ was not clear. The concentration of P in the groundwater reaching the WBZ was low, which may explain the low removal rates. However, special attention should be paid to the impact of mowing on P removal in a WBZ created by rewetting of formerly drained fens. In our case, rewetting was probably a result of the damming activity of beavers that led to flooding of degraded peat soils. P is mobilised in situ due to the rewetting of highly decomposed peat (Van de Riet et al., 2013) and it cannot be permanently immobilised and removed from the ecosystem by microbial activity as it has no gaseous form that could escape to the atmosphere. Mowing and dead biomass removal might be therefore a prominent mechanism of P removal in rewetted peatlands. P uptake by wetland vegetation during the growing season mitigates the high P mobilisation in rewetted peat (Zak et al., 2014), but a large part of the P in the aboveground biomass will be released after dieback of plants through leaching and subsequent decomposition if the vegetation is not mown and removed. Zak et al. (2014) estimated that even if the aboveground biomass is removed every year, it would still take 20–50 years to exhaust the large pool of plant-available P coming from highly decomposed rewetted peat soils. The in situ mobilised P may increase the total P load in the groundwater and surface water that leaves the WBZ and reaches the river (Stutter et al., 2009), especially in iron-poor fens (Zak et al., 2017). Interestingly, site m1, exhibiting the highest rates of N removal, had the poorest P removal. This might be due to the preservation there of a deeper peat layer after drainage. This allows for more stable waterlogging (but not flooding) and high availability of C for microbial N removal; the risk of P mobilisation, however, is also higher due to rewetting, and that may cause an overall less efficient P removal.

## Assessment of catchment effects on N removal in WBZ

We endeavoured to assess the efficiency of N load removal by a mown WBZ in an agricultural catchment. A simplified calculation reveals that 1 ha of a properly functioning summer-mown WBZ could purify water coming from c. 17.5 ha of fertilised arable fields directly bordering the WBZ. We assumed that: (1) the average amount of fertilisers applied to cereal fields is 100 kg N ha$^{-1}$ year$^{-1}$ and the nutrient loss from the fields is 20% of the applied amount, i.e. 20 kg N from a hectare of agricultural land; (2) microbially mediated N removal in a properly functioning unmown WBZ may reach c. 80% (as in our u1 site; Table 1); (3) by harvesting biomass, c. 70 kg N is removed from 1 hectare of WBZ in summer (Fig. 5), and this removal can cover the remaining 20% of the N removal (which is in accordance with Hefting et al. (2005) who found that annual N retention in herbaceous riparian vegetation represented c. 30% of the annual N removal processes in their investigation site). For a WBZ bordering an unfertilised meadow (m2 and u2), the situation is opposite, its efficiency being much lower as the water coming to the WBZ already has much lower N concentrations (with almost no nitrate). In this case, the meadows probably already act as efficient buffers and the role of riparian wetlands is therefore of secondary importance.

## Conclusions

Our case study revealed that vegetation harvesting may have a two-sided effect on nutrient removal in WBZs. On the one hand, it seems to stimulate the overall removal, especially of N; however, due to the small number of study localities, our results are only indicative. On the other hand, mowing might increase potential biomass decomposability, which eases the recycling of plant-incorporated nutrients back to the water. This suggests that mowing should always be followed by removal of cut biomass and that the longer the time between cutting and collecting hay, the more nutrients may be leached. Moreover, neglecting already mown WBZs may temporarily lower their nutrient removal efficiency due to potentially faster decomposition of biomass of mowing-adapted plants. We envision a scenario with large-scale restoration of

WBZs along rivers in Poland and other European countries (Jabłońska et al., 2020). The present study shows that this scenario should consider paludiculture or wet agriculture as a complementary activity to rewetting and, if managed properly, this should result in more effective nutrient removal in combination with climate change adaptation and mitigation initiatives.

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