Ecological Engineering 103 (2017) 296–300

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Editorial

Planning and establishment principles for constructed wetlands and riparian buffer zones in agricultural catchments

ARTICLE INFO

Keywords:
Enhanced systems
Hydrological connectedness
Hydraulic retention time
Loading rate
Nitrate
Pesticides
Phosphorus

ABSTRACT

In a great number of scientific articles on water quality improvement using constructed wetlands (CW) and riparian buffers zones (RBZ) at catchment scale, contradictory results are found. In most cases this is due to underestimating or even ignoring the role of the hydrological factor for water quality improvement. It has often resulted in biased estimates of buffering systems' efficiency at catchment scale and, consequently, has caused planning and establishment failures, mistakes and inconsistencies in legislative acts and finally, it has influenced stakeholder's willingness to support these eco-technological measures. In this paper we present a short but critical overview of the potential of CWs and RBZs in water quality improvement at catchment scale and highlight the most important aspects to be considered when planning, establishing and managing these systems.

© 2016 Elsevier B.V. All rights reserved.

1. Introduction

Constructed wetlands (CW) (sometimes also referred to as created or artificial wetlands) and riparian buffer zones (RBZ) are important elements of Green Infrastructure (GI) in rural landscapes. Continuing the traditions of ecological networks and greenways in Europe (Jongman and Pungetti, 2004) in the European context GI "is a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services such as water purification, air quality, space for recreation and climate mitigation and adaptation. This network of green (land) and blue (water) spaces can improve environmental conditions and therefore citizens' health and quality of life. It also supports a green economy, creates job opportunities and enhances biodiversity" (EC, 2013). At European Union (EU) level, the Natura 2000 network constitutes the backbone of the GI. In the USA, Green Infrastructure is mostly seen as equivalent to urban stormwater treatment systems, including rain gardens, detention ponds, bioswales, riparian wetlands, and green roofs to reduce flooding, combined sewer overflows, and pollutant transport to streams and rivers (US EPA, 1990, 2013). In some cases the GI concept is extended to cover entire catchment areas (Pennino et al., 2016).

Similarly to the GI, CWs and RBZs are multifunctional, providing several provisioning, regulating, social and cultural ecosystem services (Mander et al., 2005; Verhoeven et al., 2006; Scholz et al., 2007; Borin et al., 2010; Mitsch et al., 2012, 2014; Stutter et al., 2012; Haddaway et al., 2016). In our paper, we only consider the water quality regulating services of CWs and RBZs, which is one of the most important reasons for their establishment (Correll, 2005; Mitsch and Gosselink, 2015).

This paper attempts to shed light on the many contradictory results in the literature on the efficiency and usefulness of both of these eco-technological systems. It highlights the most important factors and principles to be considered by planners, designers and stakeholders for successfully establishing and managing constructed wetlands and riparian buffer zones in agricultural landscapes.

2. Hydrology determines the efficiency of CWs and RBZs

A myriad of literature sources have been dedicated to the water purification efficiency of CWs (Mitsch and Gosselink, 2015) and RBZs (Haddaway et al., 2016). The results vary from nearly 100% removal of nutrients and pesticides to negative values (i.e., being a source of substances) (see Fisher and Acreman, 2004). Mitsch et al. (2001, 2005) found that a combined total of 20,000 km^2 of CWs and RBZs in the Mississippi-Ohio-Missouri River Basin would be necessary in the 3 million km^2 watershed to reduce the nutrient load, especially nitrate-nitrogen, to the Gulf of Mexico, by 50% and have a measureable effect on reducing the 15,000 to 20,000 km^2 hypoxia that occurs annually in the Gulf. In a recently published paper, Land et al. (2016) offered a critical analysis of 5853 literature sources on the efficiency of wetlands in nitrogen (N) and phosphorus (P) removal from nutrient rich waters other than raw sewage. After screening for relevance and critical appraisal, 93 articles covering 203 wetlands were used for data extraction. The results show that the removal rate as g m^{-2} d^{-1} of both total nitrogen (TN) and total phosphorus (TP) is highly dependent on the hydraulic loading rate

http://dx.doi.org/10.1016/j.ecoleng.2016.12.006
0925-8574/© 2016 Elsevier B.V. All rights reserved.

(HLR) and the inlet concentration. In contrast, the removal rate was negatively correlated with wetland area, which probably reflects the difficulty in maintaining a high hydraulic efficiency in large wetlands. Median removal rates of TN and TP were 93 and 1.2 g m^{-2} yr^{-1}, respectively, whereas the median removal efficiency for TN and TP was 37% (with a 95% confidence interval of 29–44%.) and 46% (37–44%), respectively (Land et al., 2016).

Due to the vast number of pesticides used in agriculture, it is difficult to make generalizations on their removal efficiency in CWs (Tournebize et al., 2017) and RBZs (Rasmussen et al., 2011). Vymazal and Březinová (2015) found that the removal of pesticides in CWs generally increases with the increasing value of the soil/water partition coefficient (K_{OC}). Nevertheless, the most important factors in the general efficiency of pesticides removal in both CWs and RBZs are HLR and, at the catchment scale, the proportion of the polluted water intercepted by these buffering systems (Tournebize et al., 2017).

2.1. Constructed wetlands

There are two different types of removal efficiencies when considering wetlands' performance: the individual wetland–based and the catchment-based efficiency. The first one shows removal of substances in a particular wetland and is based on differences in mass transport between the inflow and outflow. The second one is the share of substances removed by all wetlands from the runoff of particular catchment.

Arheimer and Pers (2017) carried out a modelling-based comprehensive analysis of nitrogen and phosphorus removal in more than 1500 created wetlands in southern Sweden. In their paper they show that these wetlands reduced the load to the sea by 0.2% for TN and 0.5% for TP. The range of simulated removal rates in wetlands was respectively 0.01–34 g N m^{-2} yr^{-1} and 0.001–3.7 g P m^{-2} yr^{-1}, thus, significantly lower than reported by Land et al. (2016). Possible reasons for this discrepancy are at least three. First, Land et al. (2016) analysed the temperate zone wetlands around the world, also including examples from more optimal climatic conditions than those in Sweden. Second, such a low removal rate is possibly due to the fact that a majority of these wetlands received low loads as they were not constructed in locations optimal for removing nutrients from arable land but rather to enhance biodiversity (Arheimer and Pers, 2017). Third, the sensitivity analysis carried out in the model study showed that the assumptions used in the model may have resulted in an up to fivefold underestimation of the wetland removal rate. Hence, the removal rates reported by Land et al. (2016) from field studies fall within the uncertainty range of the modelled results. In comparison, Mitsch et al. (2014) found an average retention of 38.8 ± 2.2 g N m^{-2} yr^{-1} for total nitrogen (for 14 wetland-years), 15.6 ± 2.7 g N m^{-2} yr^{-1} for nitrate–nitrogen (32 wetland-years) and 2.40 ± 0.23 g P m^{-2} yr^{-1} for total phosphorus (30 wetland-years) over more than 15 years at the well-studies two 1-ha created wetlands on the Olentangy River in central Ohio USA (Mitsch et al., 2012, 2014). Water was pumped from the river to these wetlands in proportion to the stream flow for all of these years, so the data represented variable hydrologic conditions from year to year.

Another factor which may influence the N and P removal efficiency of wetlands is their hydrological connectedness to the stream network. Marton et al. (2015) and Yu et al. (2015) demonstrated that even isolated wetlands as biogeochemical reactors can be important in terms of water quality improvement. Likewise, Whigham and Jordan, 2003 showed that in most cases isolated wetlands can temporarily become connected ones, and that there is no significant difference in water quality improvement between these two types of wetlands. However, Johnson et al. (2014) showed that one of the major predictors for total watershed N retention was the surface area of hydrologically connected newly created wetlands. Since the Swedish wetlands considered in the Arheimer and Pers (2017) paper are created wetlands and the majority of them are isolated from the stream network, this factor could also be a reason for the overall low catchment scale N and P removal efficiency of these wetlands.

The removal efficiency (in% of load) of created wetlands depends on their area and volume (Tournebize et al., 2017), and is negatively correlated with the hydraulic load as demonstrated in the study by Land et al. (2016). If the wetland to catchment ratio (WCR) is too small (<1%), the water and substances retention times in the CW risk being too short for a high removal efficiency (% of load). In addition, in catchments with a low proportion of wetlands, a significant amount of water will not flow through wetlands due to the limited available wetland area and the connectedness problem. Koskiaho and Puustinen (2005) demonstrate, that in boreal conditions, the WCR should be more than 2% of the catchment area for substantial (>20%) N and P load reductions, but this figure needs to be corrected for differences in regional runoff rates. Garnier et al. (2014) showed that if the areal cover of wetlands amounts to 8% of the catchment area, it contributes to removing 50% of total annual nitrogen fluxes. Fink and Mitsch (2004) showed that a wetland that was 6.6% of a small 17 ha agricultural catchment in central Ohio reduced concentrations of nitrate–nitrite, SRP, and TP by 40, 56, and 59% respectively. However, with optimal conditions for denitrification and also for anammox (see Ligi et al., 2015), up to 50% catchment scale NO_3^- removal may be possible with lower WCR values (Tournebize et al., 2017).

Where space is limited, it may be advantageous to remove a significant proportion of pollutants by treating only part of the total volume of water, by focusing on intercepting and treating the most concentrated flows. Tournebize et al. (2017) propose two strategies depending on the quality parameters of the targeted water: in-stream and off-stream interception. One or the other of these strategies is more or less better adapted depending on the transfer mode of the targeted pollutant (e.g., nitrate or pesticide) (Passy et al., 2012; Tournebize et al., 2013). If the available area adjacent to the stream is broad enough, the CW can be located in an in-stream design, i.e. at the outlet of a collector or in an agricultural ditch by widening it, to allow all drainage/ditch water to flow through the CW. For narrow ditches, a CW could be located in a suitable area off-stream. In this case, only part of the ditch-flow is intercepted, using a control gate on the pipe connecting the ditch and CW (Tournebize et al., 2017). Based on experimental results, Tournebize et al. (2017) propose in-stream interception for NO_3^- removal, whereas the use of several small off-stream CWs in the headwater area is a more suitable strategy for pesticide removal.

Nutrient removal efficiency in CWs can also be increased by an optimal design. Channel-like surface flow (SF) CWs with aspect ratio (length to width ratio) >20:1 show the best water treatment performance (Kadlec and Knight, 1996). In quadrangular beds of SF CWs, a serpentine flow-path increases hydraulic retention time (HRT; Persson et al., 1999), thus increasing the denitrification performance (Poe et al., 2003) and removal efficiency of nutrients and pesticides (Kadlec and Knight, 1996; Passeport et al., 2013).

A smart choice of interception strategies in combination with optimal design and the use of denitrification bioreactor devices (see below) is the best way to enhance the performance of CWs. Nevertheless, NO_3^- removal with constructed wetlands requires space that is not always available. Therefore, cooperation with stakeholders is appropriate. Sometimes, due to the involvement of land use planning issues, the establishment of CWs for treating non-point pollution from large agricultural catchments with intensive agriculture can be time-consuming and the final result is often a

compromise with a much smaller area of wetland than required for the targeted treatment (Tournebize et al., 2017).

2.2. Riparian buffer zones

The best practice in terms of riverine systems is to protect and/or sustainably manage floodplains (Palmer et al., 2005; Verhoeven et al., 2006; Comin et al., 2017). In regions where agricultural land use has left the stream corridors intact, most basin drainage can move through the riparian zones of first- and second-order headwater streams with continuous riparian buffers on both sides of these streams (Correll, 2005). A combination of several parallel zones of buffer vegetation are most effective (Lowrance et al., 1984; Correll, 2005). A narrow grass strip at the upland edge traps suspended particulates and phosphorus. A wider zone of woody vegetation traps nitrate, and provides natural organic matter to the receiving waters (Correll, 2005). Also, the shading and cooling of streams under tree canopies is one of the effects of riparian buffer zones (Mander et al., 2005). Wide buffer zones have a diverse internal pattern with hot spots for enhanced nutrient removal (Vidon et al., 2010). The connectivity of buffer zones to wider river corridors and floodplains is as beneficial for nutrient removal as the connectivity of CWs (Roley et al., 2012; Kristensen et al., 2014; Poulsen et al., 2014).

In regions where agricultural use leaves only limited areas along streams, narrow unfertilized buffer strips have become important. This widely studied and implemented "edge of field" mitigation measure aims at providing an effective barrier against N, P, and sediment transfer towards the stream (Stutter et al., 2012). To ease the legislative process, these buffers have often been conceived as narrow mandatory strips along streams and rivers, across different riparian soil water conditions, bordering land uses of different types of water quality impacts, and without prescribed buffer management (Stutter et al., 2012).

In agreement with the EU Nitrates and Water Framework Directives, in most EU countries riparian buffer strips along streams draining agriculturally used catchments are now mandatory (Buckley et al., 2012; Kronvang et al., 2014). However, the establishment of buffer zones with constant width along all the ditches and streams without considering local hydrological and soil conditions will not give the expected effect. In addition, farmers are sometimes unwilling to establish buffer strips (Buckley et al., 2012). In Denmark, for instance, many farmers were against the introduction of buffer strips as a general mitigation measure. The farmers claimed that RBZs in general are not very efficient for reducing N and P losses to surface waters, which was originally the argument behind the law from the Ministries of Environment and Food (Kronvang et al., 2014). It has been known for a long time that RBZs can act efficiently if there are no water flow bypasses (e.g. by drainage pipes directing water straight to the ditch/stream) and if they are located in lower positions at stream banks (thalwegs) where they can intercept surface runoff from adjacent fields (Lowrance et al., 1984; Mander et al., 1997; Correll, 2005).

The scientific basis for judging the best course of action in designing and placing RBZs to enhance their efficiency and multifunctionality has gradually become more substantial over the past decade (see Stutter et al., 2012). In Denmark, the concept of Intelligent Buffer Zones has been developed and implemented (Kronvang et al., 2014). This includes different innovative methods to enhance RBZs efficiency. In Sweden, the concept of "adapted buffer zones" is implemented to reduce phosphorus losses, encouraging farmers to maintain grass-covered area on field "hot spots" that are prone to erosion (e.g. around drainage wells) or flooding (JV, 2016). Similarly to CWs, there are several ways to enhance RBZ performance. To reduce the watershed export of excess nitrogen to sensitive aquatic ecosystems, denitrification must be enhanced in riparian zones, whereas the key factor is the presence of additional (external) carbon sources. Wilcock et al. (2009) demonstrated that the combined treatment of organic-rich farm wastewater with nitrate–polluted water from fertilized fields yields the best results. Likewise, phosphorus removal in riparian buffer zones can be enhanced using Fe-rich materials like ochre for better adsorption (Fenton et al., 2012) or Ca-rich materials for precipitation and filtering of phosphorus (Kirkkala et al., 2012). For instance, sand filters incorporating lime used together with buffer zones will reduce both P and the suspended solids load entering watercourses (Kirkkala et al., 2012).

A new and promising method is the use of denitrifying bioreactors. This is an approach where solid carbon substrates are added into the flow path of nitrogen rich water. These carbon substrates (often fragmented wood products) act as a C and energy source to support denitrification: the conversion of NO_3^- to nitrogen gases (Schipper et al., 2010; Warneke et al., 2011). Likewise, woodchips-filled bioreactors in the bottom of streams have been used to treat NO_3^- (Robertson and Merkley, 2009; Moorman et al., 2015).

Redirecting tile drainage as subsurface flow through the riparian buffer zones is another promising measure for enhancing denitrification. Using this method, Jaynes and Isenhart (2014) observed a significant increase in NO_3^- removal and considered this to be a promising management practice to improve surface water quality within tile-drained landscapes. It is recommended that sub-soils should be permeable and have a reasonable groundwater retention time (Correll, 2005).

It is necessary to carry out regular management of riparian buffer zones and strips, for instance the regulated harvesting of trees and the cutting of hay for bioenergy production (Mander et al., 2005; Stutter et al., 2012). Correll (2005) highlights the following maintenance measures: protecting buffer zones/strips from erosion and periodically removing sediment berms that develop, fencing out livestock, adding only native plant species and eradicating invasive ones, controlling excessive activity by wild ungulates, voles, and beavers.

3. Recommendations for the planning and implementation of CWs and RBZs

Based on the above explanations, we can conclude with some general principles for planning, establishing, and managing CWs and RBZs.

3.1. Constructed wetlands

- As a first step, a hydrological diagnosis of the catchment (quantity and seasonality of water flow and substances transport) should be carried out;
- The hydroperiod for a CW which is governed by inflow, outflow and storage capacity should be optimized, and strongly linked to the dynamics of precipitation, runoff and tile-drainage regime;
- Design for an average flow and accept removal efficiency variability due to high variation of HRT (not peak flow as flooding purposes);
- Wetlands for water quality regulation should be located downstream from hot-spots in the landscape, i.e. areas with expected high losses of nutrients or pesticides;
- A suitable interception strategy (in-stream or off-stream or a combination thereof) should be chosen;
- Hydrologically (geographically) isolated CWs should be avoided;
- Historical locations should be used for the construction of wetlands;

- Integrated CWs treating polluted agricultural runoff and farm wastewater, and carrying out multiple services have a reasonable potential for widespread implementation;
- Consider should be given to harvesting biomass on a regular basis for bioenergy or fodder production.

3.2. Riparian buffer zones

For wider buffer zones (larger areas available):

- Preference should be given to 3-zone RBZs: from field to stream – grass strips, younger forest/brush, and older forest stands;
- Grass should be cut and younger forest stands should be harvested for bioenergy production;
- Floodplains should be protected by using buffer zones/strips at field edges;
- Regulating services (e.g., biodiversity support) of the whole landscape should be enhanced, in order to connect the network of RBZs with wider floodplains and river corridors

For narrow buffer strips (available area limited)

- Establish Buffer strips must be established on thalwegs and at the edge of sloping fields with low soil permeability, where overland flow and groundwater can be intersected;
- Water flow bypasses should be avoided;
- Tile drainage should be redirected as subsurface flow through the RBZ;
- Ditch bottoms should be widened at drainage pipe outflow sites (horseshoes or bioswales measure) to enhance denitrification;
- Denitrifying bioreactors should be used in combination with P-sorbing filter media;
- In degraded riparian zones, soils of the appropriate porosity and organic carbon content should be preferred;
- The buffer surface should be contoured to avoid concentrated storm flows and periodically remove sediment berms that develop

Finally, some general rules for the establishment of both CWs and RBZs must be highlighted:

- Cooperation with farmers and other stakeholders is imperative, the transfer of the most recent know-how about the best practices for managing buffer strips is necessary for their success;
- Always consider and prefer The multifunctionality principle should always be considered and preferred, even if services for regulating water quality are less favorable in a particular location;
- Constructed wetlands and buffer zones should not be considered as an excuse to increase loading in upstream fields. Good management of the uplands is essential and hydrologically connected constructed wetlands with effective buffer zones along the streams draining the basin will complete the task of water quality protection (Correll, 2005).

Acknowledgements

This study was financially supported by AESN, LifeARTWet (ENV/F/000133), and PIREN Seine. Also, financial support was given by the Estonian Research Council (grant no. IUT2-16), and by the EU through the European Regional Development Fund (Centres of Excellence ENVIRON and EcolChange).

References

Arheimer, B., Pers, B.C., 2017. Lessons learned? Effects of nutrient reductions from constructed wetlands in 1996–2006 across Sweden. Ecol. Eng. 103, 404–414.

Borin, M., Passoni, M., Thiene, M., Tempesta, T., 2010. Multiple functions of buffer strips in farming areas. Eur. J. Agron. 32, 103–111.

Buckley, C., Hynes, S., Mechan, S., 2012. Supply of an ecosystem service-Farmers' willingness to adopt riparian buffer zones in agricultural catchments. Environ. Sci. Policy 24, 101–124.

Comin, F.A., Sánchez-Pérez, J.M., Español, C., Carranza, F., Sauvage, S., Antigüedad, I., Zabaleta, A., Martinez-Santos, M., Gerino, M., Yao, J.M., Bodoque, J.M., Ladera, J., Yela, J.L., Teissier, S., Bernard-Jannin, L., Sun, X.L., Navarro, E., Pinelli, E., Chamsi, O., Neves, R., Brito, D., Ruiz, E., Uriarte, J., Jiménez, J.J., García, M., Barcos, A., Sorando, R., 2017. Floodplain capacity to depollute water in relation to the structure of biological communities. Ecol. Eng. 103, 301–314.

Correll, D.L., 2005. Principles of planning and establishment of buffer zones. Ecol. Eng. 24, 433–439.

European Commission (EC), 2013. Building a Green Infrastructure for Europe. Publications Office of the European Union, Luxembourg, 24 pp.

Fenton, O., Kirwan, L., Oí hUallachaín, D., Healy, M.G., 2012. The effectiveness and feasibility of using ochre as a soil amendment to sequester dissolved reactive phosphorus in runoff. Water Air Soil Pollut. 223, 1249–1261.

Fink, D.F., Mitsch, W.J., 2004. Seasonal and storm event nutrient removal by a created wetland in an agricultural watershed. Ecol. Eng. 23, 313–325.

Fisher, J., Acreman, M.C., 2004. Wetland nutrient removal: a review of the evidence. Hydrol. Earth Syst. Sci. (HESS) 8 (4), 673–685.

Garnier, G., Billen, G., Vilain, G., Benoit, M., Passy, P., Tallec, G., Tournebize, J., Anglade, J., Billy, C., Mercier, B., Ansari, P., Azougui, A., Sebilo, M., Kao, C., 2014. Curative vs preventive management of nitrogen transfers in rural areas: lessons from the case of the Orgeval watershed Seine River basin, France. J. Environ. Manage. 144, 125–134.

Haddaway, N.R., Brown, C., Eggers, S., Josefsson, J., Kronvang, B., Randalls, N., Uusi-Kämppä, J., 2016. The multifunctional roles of vegetated strips around and within agricultural fields. A systematic map protocol. Environ. Evid. 5, 18.

JV, 2016. Adapted Buffer Zones. Swedish Board of Agriculture (In Swedish) http://www.jordbruksverket.se/amnesomraden/stod/jordbrukarstod/miljoersattningar/skyddszoner/villkor.4.6c64aa881525004b53bdcd1f.html.

Jaynes, D.B., Isenhart, T.M., 2014. Reconnecting tile drainage to riparian buffer hydrology for enhanced nitrate removal. J. Environ. Qual. 43, 631–638.

Johnson, T.A.N., Kaushal, S.S., Mayer, P.M., Grese, M.M., 2014. Effects of stormwater management and stream restoration on watershed nitrogen retention. Biogeochemistry 121, 81–106.

Jongman, R.H.G., Pungetti, G. (Eds.), 2004. Cambridge University Press, Cambridge, 345 pp.

Kadlec, R.H., Knight, R.L., 1996. Treatment Wetlands. CRC Press/Lewis Publishers, Boca Raton, FL (893 pp).

Kirkkala, T., Ventelä, A.-M., Tarvainen, M., 2012. Long-term field-scale experiment on using lime filters in an agricultural catchment. J. Environ. Qual. 41, 410–419.

Koskiaho, J., Puustinen, M., 2005. Function and potential of constructed wetlands for the control of N and P transport from agriculture and peat production in boreal climate. J. Environ. Sci. Heal. A 40 (6–7), 1265–1279.

Kristensen, E.A., Kronvang, B., Wiberg-Larsen, P., Thodsen, H., Nielsen, C., Amor, E., Friberg, N., Pedersen, M.L., Baattrup-Pedersen, A., 2014. 10 years after the largest river restoration project in Northern Europe: hydromorphological changes on multiple scales in River Skjern. Ecol. Eng. 66, 141–149.

Kronvang, B., Baattrup-Pedersen, A., Hoffmann, C.C., Andersen, H.E., Larsen, S.E., Hille, S., Rubæk, G., Heckrath, G., Dalgaard, T., Olsen, S.B., Gertz, F., Wiborg, I.A., Jensen, H., Møller, H.B., Stutter, M., 2014. Experiences gained from 2 years with mandatory 10 m buffer strips along all Danish watercourses: do we know enough to establish BSs and evaluate their ecosystem services? In: Oral Presentation Held at The SWS Europe Chapter Meeting, Huesca, September.

Land, M., Granéli, W., Grimvall, A., Hoffmann, C.C., Mitsch, W.J., Tonderski, K.S., Verhoeven, J.T.A., 2016. How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. Environ. Evid. 5, 9.

Ligi, T., Truu, M., Oopkaup, K., Nõlvak, H., Mander, Ü., Mitsch, W.J., Truu, J., 2015. Genetic potential of N_2 emission via denitrification and ANAMMOX from the soils and sediments of a created riverine treatment wetland complex. Ecol. Eng. 80, 181–190.

Lowrance, R., Todd, R., Fail, J.Jr., Hendrickson, O., Leonard, R., Asmussen, L., 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience 34 (6), 374–377.

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

Mander, Ü., Hayakawa, Y., Kuusemets, V., 2005. Purification processes, ecological functions, planning and design of riparian buffer zones in agricultural watersheds. Ecol. Eng. 24 (5), 421–432.

Marton, J.M., Creed, I.F., Lewis, D.B., Lane, C.R., Basu, N.B., Cohen, M.J., Craft, C.B., 2015. Geographically isolated wetlands are important biogeochemical reactors on the landscape. Bioscience 5, 408–418.

Mitsch, W.J., Gosselink, J.G., 2015. Wetlands, 5th ed. John Wiley and Sons, Hoboken, NJ, USA, 736 pp.

Mitsch, W.J., Day, J.W.Jr., Gilliam, J.W., Groffman, P.M., Hey, D.L., Randall, G.W., Wang, M., 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin: strategies to counter a persistent ecological problem. Bioscience 51, 373–388.

Mitsch, W.J., Day, J.W.Jr., Zhang, L., Lane, R., 2005. Nitrate-nitrogen retention by wetlands in the Mississippi River Basin. Ecol. Eng. 24, 267–278.

Mitsch, J.W., Zhang, L., Stefanik, K.C., Nahlik, A.M., Anderson, C.J., Bernal, B., Hernandez, M.E., Song, K., 2012. Creating wetlands: primary succession, water quality changes, and self –design over 15 years. Bioscience 62 (3), 237–250.
Mitsch, W.J., Zhang, L., Waletzko, E., Bernal, B., 2014. Validation of the ecosystem services of created wetlands: two decades of plant succession, nutrient retention, and carbon sequestration in experimental riverine marshes. Ecol. Eng. 72, 11–24.
Moorman, T.B., Tomer, M.D., Smith, D.R., Jaynes, D.B., 2015. Evaluating the potential role of denitrifying bioreactors in reducing watershed-scale nitrate loads: a case study comparing three Midwestern (USA) watersheds. Ecol. Eng. 75, 441–448.
Palmer, M.A., Bernhardt, E.S., Allan, J.D., Lake, P.S., Alexander, G., Brooks, S., Carr, J., Clayton, S., Dahm, C.N., Shah, J.F., Galat, D.L., Loss, G., Goodwin, P., Hart, D.D., Hassett, B., Jenkinson, R., Kondolf, G.M., Lave, R., Myer, J.L., O'Donnell, T.K., Pagano, L., Sudduth, E., 2005. Standards for ecologically successful river restoration. J. Appl. Ecol. 42, 208–217.
Passeport, E., Tournebize, J., Chaumont, C., Guenne, A., Coquet, Y., 2013. Pesticide contamination interception strategy and removal efficiency in forest buffer and artificial wetland in a tile-drained agricultural watershed. Chemosphere 91, 1289–1296.
Passy, P., Garnier, J., Billen, G., Fesneau, C., Tournebize, J., 2012. Restoration of pondsin rural landscapes: modelling the effect on nitrate contamination of surfacewater (the Seine River Basin, France). Sci. Total Environ. 430, 280–290.
Pennino, M.J., McDonald, R.I., Jaffe, P.R., 2016. Watershed-scale impacts of stormwater green infrastructure on hydrology, nutrient fluxes, and combined sewer overflows in the mid-Atlantic region. Sci. Total Environ. 565, 1044–1053.
Persson, J., Somes, N.L.G., Wong, T.H.F., 1999. Hydraulics efficiency of constructed wetlands and ponds. Water Sci. Technol. 40 (3), 291–300.
Poe, A.C., Piehler, M.F., Thompson, S.P., Paerl, H.W., 2003. Denitrification in a constructed wetland receiving agricultural runoff. Wetlands 23 (4), 817–826.
Poulsen, J.B., Hansen, F., Ovesen, N., Larsen, S., Kronvang, B., 2014. Linking floodplain hydraulics and sedimentation patterns along a restored river channel River Odense, Denmark. Ecol. Eng. 66, 120–128.
Rasmussen, J.J., Baattrup-Pedersen, A., Wiberg-Larsen, P., McKnight, U., Kronvang, B., 2011. Buffer strip width and agricultural pesticide contamination in Danish lowland streams: implications for stream and riparian management. Ecol. Eng. 37 (12), 1990–1997.
Robertson, W.D., Merkley, L.C., 2009. In-stream bioreactor for agricultural nitratetreatment. J. Environ. Qual. 38 (1), 230–237.
Roley, S.S., Tank, J.L., Williams, M.A., 2012. Hydrologic connectivity increases denitrification in the hyporheic zone and restored floodplains of an agricultural stream. J. Geophys. Res.: Biogeosci. 117, G3.
Schipper, L.A., Robertson, W.D., Gold, A.J., Jaynes, D.B., Cameron, S.C., 2010. Denitrifying bioreactors – an approach for reducing nitrate loads to receiving waters. Ecol. Eng. 36, 1532–1543.
Scholz, M., Harrington, R., Carroll, P., Mustafa, A., 2007. The integrated constructed wetlands (ICW) concept. Wetlands 27 (2), 337–354.
Stutter, M.I., Chardon, W.J., Kronvang, B., 2012. Riparian buffer strips as a multifunctional management tool in agricultural landscapes: introduction. J. Environ. Qual. 41 (2), 297–303.
Tournebize, J., Passeport, E., Chaumont, C., Fesneau, C., Guenne, A., Vincent, B., 2013. Pesticide decontamination of surface waters as a wetland ecosystem service inagricultural landscapes. Ecol. Eng. 56, 51–59.
Tournebize, J., Chaumont, C., Mander, Ü., 2017. Implications for constructed wetlands to mitigate nitrate and pesticide pollution in agricultural drained watersheds. Ecol. Eng 103, 415–425.
U.S. Environmental Protection Agency (EPA), 1990. National Pollutant Discharge Elimination System Permit Application Regulations for Storm Water Discharges. Federal Register, 44 FR 47990; 1990-11-16, 103 p.
U.S. Environmental Protection Agency (EPA), 2013. Green Infrastructure Strategic Agenda, 7 p.
Verhoeven, J.T.A., Arheimer, B., Yin, C.Q., Hefting, M.M., 2006. Regional and global concerns over wetlands and water quality. Trends Ecol. Evol. 21 (2), 96–103.
Vidon, P., Allan, C., Burns, D., Duval, T.P., Gurwick, N., Inamdar, S., Lowrance, R., Okay, J., Scott, D., Sebestyen, S., 2010. Hot spots and hot moments in riparian zones: potential for improved water quality management. JAWRA 46 (2), 278–298.
Vymazal, J., Březinová, T., 2015. The use of constructed wetlands for removal pesticides from agricultural runoff and drainage: a review. Environ. Int. 75, 11–20.
Warneke, S., Schipper, L.A., Bruesewitz, D.A., Baisden, W.T., 2011. A comparison of different approaches for measuring denitrification rates in a nitrate removing bioreactor. Water Res. 45, 4141–4151.
Whigham, D.F., Jordan, T.E., 2003. Isolated wetlands and water quality. Wetlands 23 (3), 541–549.
Wilcock, R.J., Betteridge, K., Shearman, D., Fowles, C.R., Scarsbrook, M.R., Thorrold, B.S., Costall, D., 2009. Riparian protection and on-farm best management practices for restoration of a lowland stream in an intensive dairy farming catchment: a case study. N. Z. J. Mar. Freshw. Res. 43 (3), 803–818.
Yu, X.B., Hawley-Howard, J., Pitt, A.L., Wang, J.J., Baldwin, R.F., Chow, A.T., 2015. Water quality of small seasonal wetlands in the Piedmont ecoregion, South Carolina, USA: effects of land use and hydrological connectivity. Water Res. 73, 98–108.

Ülo Mander[a,b,*]
[a] *Department of Geography, Institute of Ecology and Earth Sciences, University of Tartu, Vanemuise 46, 51014, Tartu, Estonia*
[b] *Hydrosystems and Bioprocesses Research Unit, National Research Institute of Science and Technology for Environment and Agriculture (Irstea), 1 rue Gilles de Gennes, 92761, Antony, France*

Julien Tournebize
Hydrosystems and Bioprocesses Research Unit, National Research Institute of Science and Technology for Environment and Agriculture (Irstea), 1 rue Gilles de Gennes, 92761, Antony, France

Karin Tonderski
Department of Physics, Chemistry and Biology (IFM), Linköping University, 58183, Linköping, Sweden

Jos T.A. Verhoeven
Ecology and Biodiversity, Utrecht University, Padualaan 8, 3584 CH, Utrecht, The Netherlands

William J. Mitsch
Everglades Wetland Research Park, Kapnick Center, Florida Gulf Coast University, Naples, 34112 USA

* Corresponding author at: Department of Geography, Institute of Ecology and Earth Sciences, University of Tartu, Vanemuise 46, 51014, Tartu, Estonia.
E-mail addresses: ulo.mander@irstea.fr, ulo.mander@ut.ee (Ü. Mander).

29 November 2016

10 December 2016
Available online 30 December 2016