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

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 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 km² of CWs and RBZs in the Mississippi-Ohio-Missouri River Basin would be necessary in the 3 million km² 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 km² 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.
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(2015) found that the removal of pes-
ticides in CWs generally increases with the increasing value of
the soil/water partition coefficient (Koc). 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 propor-
tion 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 con-
sidering wetlands’ performance: the individual wetland–based and
the catchment–based efficiency. The first one shows removal of sub-
stances 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 com-
prehensive 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). Possi-
ble 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 possi-
bly due to the fact that a majority of these wetlands received low
loads as they were not constructed in locations optimal for remov-
ing 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 Glentangy 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 effi-
ciency of wetlands is their hydrological connectedness to the
stream network. Marton et al. (2015) and Yu et al. (2015) demon-
strated 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 wet-
lands 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 condi-
tions, 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. Pink 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 deni-
trification and also for anammox (see Ligi et al., 2015), up to 50% catchment scale NO₃⁻ removal may be possible with lower WCR
values (Tournebize et al., 2017).

Where space is limited, it may be advantageous to remove a sig-
ificant 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 strate-
gies 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 trans-
fer 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 NO₃⁻ 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 per-
formance (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. Never-
theless, NO₃⁻ removal with constructed wetlands requires space
that is not always available. Therefore, cooperation with stake-
holders 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; Verhoef 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 sediments 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 (thalweg) 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 (Fentoni 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 NO₃⁻ 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 NO₃⁻ (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 NO₃⁻ 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 down-stream 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 limited);

• 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 intercepted;
• 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 bio-reactors 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).

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