THE USE OF CONSTRUCTED WETLANDS FOR NITROGEN REMOVAL FROM AGRICULTURAL DRAINAGE: A REVIEW*

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Diffuse pollution from agricultural drainage is a severe problem for water quality and it is a major reason for eutrophication of both freshwaters and coastal waters. Constructed wetlands were proposed as a suitable tool for removal of nitrogen from agricultural drainage in the early 1990s. Since then constructed wetlands with free water surface have been successfully used in Europe, North America, Asia, and Australia. The predominant form of nitrogen in agricultural drainage waters is nitrate and therefore denitrification is considered as the major removal process. The literature survey of 41 full-scale constructed wetlands revealed that removed nitrogen amount varied widely between 11 and 13 026 kg N ha^{-1} per year with the median removal of 426 kg N ha^{-1} per year. A very close relationship was found between inflow and removed nitrogen loads but the overall percentual efficiency decreases with increasing inflow nitrogen load. It has also been found that nitrogen removal depends on the size of constructed wetland (W) as compared to catchment (C) area. The survey indicated that the W/C greater than 1% does not result in any substantial increase of nitrogen removal. There are still some gaps in our knowledge that need to be evaluated, namely the optimum vegetation maintenance.

drainage waters, nitrate, eutrophication, denitrification, wetlands

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INTRODUCTION

The non-point source pollution is a global problem affecting the safety of drinking water supply and aquatic habitats. Pollutants originating from agricultural runoff include sediment, nitrogen, phosphorus, pesticides, pathogens, salts, trace elements, and dissolved organic carbon (O'Geen et al., 2010). It is well known that excessive amounts of phosphorus and nitrogen may cause eutrophication of freshwaters (Figs. 1 and 2) and coastal zones (Fig. 3). It has been well established that phosphorus is the key element and limiting nutrient for algae and cyanobacteria in a majority of freshwaters (e.g. Vollenweider, 1968; Chiaudani, Vighi, 1974; Miller et al., 1974). On the other hand, it has been long known that eutrophication of estuaries and coastal waters is caused by excessive loadings of nitrogen (Fleischer et al., 1987; Baker, 1992; Nixon, 1995; Howarth et al., 1996; Howarth, Marino, 2006; Hägg, 2010).

Nitrogen is applied on agricultural fields in various forms (inorganic fertilizers, urea, manure), however the major nitrogen species in tile drainage is nitrate-N as all other forms are converted into nitrate-N via hydrolysis and nitrification. Powlson, Addiscott (2005) pointed out that urea is by far the most commonly used nitrogen fertilizer. After application it is hydrolyzed to ammonia-N within few days and ammonia-N

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is then available for plant uptake. If not uptaken by plants, ammonia-N is nitrified to nitrate within 1–4 weeks. Nitrate-N is a very mobile and chemically inert nitrogen species and, therefore, it is easily lost from the soil profile by leaching (e.g. Kladivko et al., 1991; Kovacic et al., 2000; Goswami et al., 2009; Meisinger et al., 2015). It has been shown that nitrate leaching is affected to a great extent by tillage practices – this effect is highly variable and the impact is the highest when tillage occurs shortly before a season of high water-recharge (Addiscott, Dexter, 1994; Zibilske, Bradford, 2007; Strudley et al., 2008; Meisinger et al., 2015).

Wetlands can act as filters removing particulate material, as sinks accumulating nutrients, or as transformers converting nutrients to different forms such as gaseous forms of nitrogen (Richardson, 1989). The ability of natural wetlands to retain nitrogen from freshwaters was recognized and reported since the 1970s (Mitsch et al., 1979; Richardson, 1990; Fisher, Acreman, 2004). Fisher, Acreman (2004) reviewed the available results from 54 natural wetlands in North America, Europe, Australasia, and Africa and concluded that 80% of the wetlands retained/removed nitrogen with the average retention of 67%. The major processes responsible for nitrogen removal in wetlands are denitrification (Lowrance et al., 1984; Xue et al., 1999; Poe et al., 2003; Tanner, 2004), uptake by plants, and subsequent nitrogen accumulation in the plant biomass (Jiang et al., 2007; Borin, Toccheto, 2007), sedimentation (Borin, Toccheto, 2007), and volatilization (Vymazal, 2007).

The main objective of this study is to evaluate nitrogen removal efficiencies of constructed wetlands designed to treat agricultural drainage waters.

Fig. 2. Cyanobacterial water bloom in a drinking water reservoir Švihov, Czech Republic
Photo: author

Constructed wetlands for nitrogen removal from agricultural drainage waters

The use of constructed wetlands to remove nitrogen from agricultural drainage waters was first proposed during the late 1980s (Mitsch, 1992; van der Valk, Jolly, 1992). Mitsch (1992) set up several design principles that could be applied to wetland construction for non-point source pollution control: (a) minimum maintenance, (b) use of natural energies, (c) multiple objectives, (d) landscape-friendliness, (e) systems as ecotones, (f) design for function, not for form, (g) no over-engineered systems, and (h) time necessary for full development. Constructed wetlands designed to remove nutrients from drainage waters could be positioned either ‘in-stream’ or ‘off-stream’,

Fig. 1. Filamentous algae, a typical result of eutrophication. Břehovský Creek, South Bohemia
Photo: author

Fig. 3. Eutrophication of a coastal water in Baltic Sea at Kuressaare, Saaremaa, Estonia
Photo: author

however, the in-stream configuration is applicable only in small streams (Mitsch, 1992). A special type of 'in-stream' wetlands designed to mitigate agricultural drainage pollution are naturally overgrown vegetated ditches that have recently been used (e.g., Moore et al., 2010; Kröger et al., 2012). In general, shallow ponds fed by small streams with high concentrations of nitrate or directly by agricultural drainage can also effectively remove nitrate in the anoxic zones near the bottom, especially during the growing season. Knight (1992) pointed out that besides the primary purpose of retention of nutrients and sediments, constructed wetlands designed for non-point pollution amelioration have the ancillary or secondary benefits such as biomass production, secondary production of fauna, food chain and habitat diversity or aesthetic/recreational/educational human uses.

Vast majority of constructed wetlands for nutrient removal from agriculture drainage waters have been designed as free water surface systems (FWS CWs) with deeper inflow section for sedimentation of particles which may be present in drainage ditches from the surface runoff after heavy rain events (Figs. 4 and 5). The deeper part is usually 1–2 m deep and often is modified to allow for sediment removal. The vegetated beds are usually shallow trenches with no specific requirements for bottom soil. The major function of the soil is to provide rooting medium for macrophytes. Water depth is usually between 10 and 50 cm (Kadlec, Wallace, 2009; Vymazal, 2013). Most processes responsible for nitrogen removal in FWS CWs occur in the water column and in the layer of decaying plant material on the bottom. The processes in the water column are aerobic, the anoxic and anaerobic processes may occur within the litter layer (Vymazal, 2007).

There is a wide variety of plants that have been used for CWs treating agricultural drainage (Table 1), however, the most commonly used species are *Typha latifolia* and *Phragmites australis*. Other more frequently used species are *Phalaris arundinacea, Scirpus lacustris, Glyceria fluitans, Sparganium erectum, Carex* spp. *or Polygonum lapathifolium.* The plants are usually not harvested as decaying plant material provides important source of organics for denitrification as the concentration of organics in agricultural drainage waters is often low as compared to nitrate concentrations. The emergent macrophyte species are quite often accompanied by submerged species which occur mostly voluntarily. Emergent species produce more organics in the sediment but submerged species may provide more surface for denitrifying bacteria (Weisner et al., 1994). Both emergent and submerged macrophytes provide a substrate for the growth of periphytic algae which contribute to removal of nitrogen as well (Vymazal, Kröpfelová, 2008).

Probably the highest number of constructed wetlands for nutrient removal from agricultural drainage waters was built in Sweden (Fleischer et al., 1994; Jansson et al., 1994; Arheimer, Wittgren, 2002; Arheimer, Pers, in press). According to Arheimer, Pers (in press), during the period 1996–2006, a total of 1574 wetlands were constructed with a total surface area of 4135 ha. In the following period 2009–2011, 564 wetlands were created with a total surface area of 2468 ha. The investment cost during the period 1996–2011 amounted to 130 million EUR.

Nitrogen removal in constructed wetlands

The inflow load of constructed wetlands treating agricultural drainage waters included in the survey of

Fig. 4. Free water surface constructed wetland for treatment of tile drainage planted with *Typha latifolia* at Rodstenseje, Denmark
Photo: author

Fig. 5. Free water surface constructed wetland for treatment of agricultural drainage planted with *Phragmites australis*. Laluzea, Spain
Photo: author

41 constructed wetlands varies within several orders of magnitude between 11 and 47 272 kg N ha^{-1} per year. The amount of removed nitrogen from these wetlands varies considerably as well, between 11 and 13 026 kg N ha^{-1} per year (Table 1) with the median removal of 426 kg N ha^{-1} per year. This is in a good agreement with results obtained by Tonderski et al. (2015) who reported an average annual nitrogen removal in constructed wetlands built for treatment of agricultural drainage between 298–336 kg N ha^{-1} per year for the period 2007–2013. As compared to these wetlands, constructed wetlands built with the primary purpose of biodiversity increase, the nitrogen removal varied only between 32 and 46 kg N ha^{-1} per year. The major reason for this difference is the fact that biodiversity wetlands were not optimally designed for nutrient removal (e.g. flow path, retention time). The authors also calculated the potential nitrogen removal up to 1000 kg N ha^{-1} per year in the case of high inflow loads.

Nitrogen removal efficiency is highly variable and the major factors that affect removal efficiency are inflow load and the ratio between the drained catchment (C) and constructed wetlands (W) surface areas (Fisher, Acreman, 2004; Koskiaho, Puustinen, 2005; O'Green et al., 2010). Concerning the effect of the W/C ratio, Koskiaho, Puustinen (2005) found a close relationship (R^2 = 0.77) when analyzing results from agricultural wetlands in Finland, Sweden, and the USA. The authors pointed out that for 20% N removal, the W/C ratio should be at least 2% while 50% removal can be achieved with the W/C ratio > 7%. Tanner et al. (2010) reported that 40% nitrate removal can be achieved with the W/C ratio 5% and further increase of the W/C ratio would not bring any substantial removal of nitrate. The W/C ratio effect on nitrogen removal in surveyed systems (Table 1) yields a similar relationship to that reported by Tanner et al. (2010). The W/C ratio

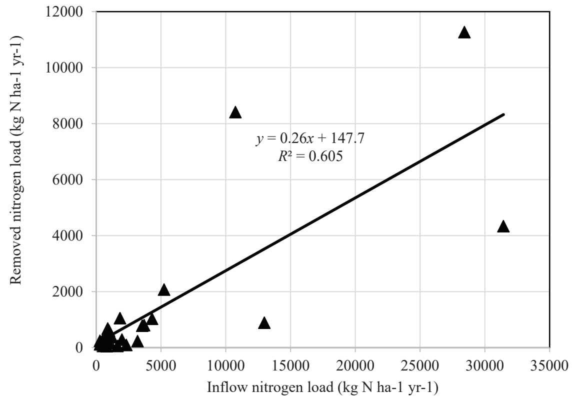

Fig. 7. Relationship between inflow and removed nitrogen loads (kg N ha–1 per year) in constructed wetlands treating agricultural drainage waters

necessary for 40% removal of total nitrogen (TN) is only 1% and further increase of the W/C ratio does not result in increased TN removal (Fig. 6). However, the relationship is quite weak (R^2 = 0.116). This may be caused by the fact that the 'drained catchment area' reported in the literature may not necessarily include in some cases only drained agricultural fields.

A very close relationship has been reported between inflow and removed nitrogen loads. Saunders, Kalff (2001) reported a highly significant relationship (R^2 = 0.82, P < 0.001) for 23 wetlands in the USA, Canada, and Denmark. A similar result was observed by Mitsch et al. (2001). The results included in our survey yielded a moderate relationship (R^2 = 0.61) (Fig. 7). At the same time it is important to realize that the removal efficiency decreases with increasing inflow load. Saunders, Kalff (2001) observed a very strong (R^2 = 0.82) indirect correlation between inflow nitrogen loading and nitrogen percentual removal. Similar trend but less marked (R^2 = 0.353)

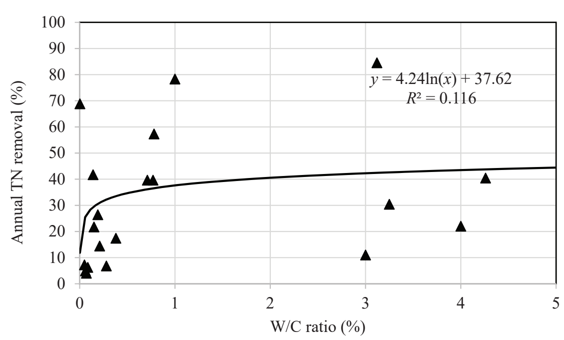

Fig. 6. Dependence of total nitrogen (TN) removal efficiency (in %) of the constructed wetland on the constructed wetland (W)/drained catchment (C) area ratio (in %)

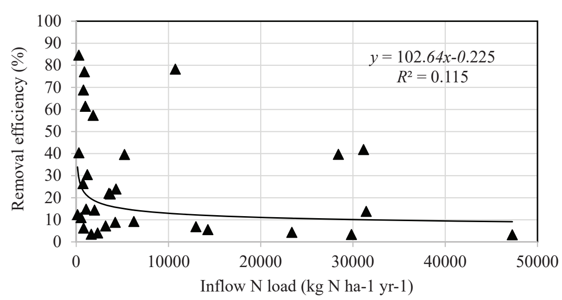

Fig. 8. Relationship between inflow nitrogen load (kg N ha–1 per year) and treatment efficiency (%) in constructed wetlands treating agricultural drainage waters

Table 1. The amount of nitrogen removed in constructed wetlands treating agricultural drainage waters. Systems are ranked according to the removed load. Part 1.

Locality	Wetland area (A) (ha)	Catch-ment area (B) (ha)	A/B (%)	Removed load (kg N ha^{-1} per year)	Concentra-tion (mg/l)		Remo-val (%)	Dominant vegetation	Reference
					in	out			
Finland	0.48	90	0.53	11	8.40			*Typha latifolia, Juncus filiformis, J. effusus, Alopecurus pratensis, Eleocharis palustris, Bidens sp.*	Koskiaho et al. (2003)
USA	1.30	14	9.30	17				*Eleocharis obtuse, Ludwigia palustris, Schoenoplectus americanus*	Jordan et al. (2003)
Sweden	2.10	100	2.10	17				not specified	Strand and Wiesner (2013)
Norway	0.087	103	0.085	50	1.60	1.50	6.3	*Glyceria fluitans, Carex* spp.	Braskerud (2002)
Norway	0.09	148	0.06	56	3.21	3.10	4.9	*Scirpus lacustris, Acorus calamus, Phragmites australis, Typha latifolia, Equisetum fluviale, Glyceria fluitans*	Braskerud (2002)
Finland	60	2 000	3.00	57	3.10	2.76	11	*Phragmites australis, Lysimachia thyrsiflora, Lythrum salicaria, Peucedanum palustre, Carex aquatica, Scuttelaria galerigulata*	Koskiaho et al. (2003)
Norway	0.0345	50	0.07	93	3.49	3.35	4.0	*Scirpus lacustris, Acorus calamus, Phragmites australis, Typha latifolia, Equisetum fluviale, Glyceria fluitans*	Braskerud (2002)
USA	0.16	3.76	4.26	117	15.60	9.30	40.4		Kovacic et al. (2006)
Norway	0.084	22.1	0.38	159	5.14	4.38	17.4	*Sparganium erectum, Phragmites australis, Phalaris arundinacea, Myosotis scorpioides*	Braskerud (2002)
Korea	0.886	465	0.19	195	7.20	5.30	26.4		Maniquiz et al. (2011)
Australia	0.045	90	0.05	230	2.77	2.57	7.2		Raisin et al. (1997)
USA	0.78	25	3.12	241	7.50	1.16	84.5	not specified	Larson et al. (2000)
USA	0.30	5	6.00	245				*Phragmites japonica, Typha angustifolia, T. orientalis, Miscanthus sinensis, Zizania caduciflora, Nelumbo nucifera, Oenanthe javanica*	Kovacic et al. (2000)
Finland	0.60	12	5.00	280	9.80			*Phragmites australis, Schoenoplectus validus, Rorippa nasturtiumaquaticum, Paspalum distichum, Persicaria lapathifolia*	Koskiaho et al. (2003)
Norway	0.046	21.9	0.21	285	5.14	4.40	14.4	*Sparganium erectum, Phragmites australis, Phalaris arundinacea, Myosotis scorpioides*	Braskerud (2002)
USA	0.80	25.6	3.13	298					Kovacic et al. (2000)
Taiwan	13.65			308					Wu et al. (2010)

Table 1. The amount of nitrogen removed in constructed wetlands treating agricultural drainage waters. Systems are ranked according to the removed load. Part 2

Locality	Wetland area (A) (ha)	Catch-ment area (B) (ha)	A/B (%)	Removed load (kg N ha^{-1} per year)	Concentra-tion (mg/l)		Remo-val (%)	Dominant vegetation	Reference
					in	out			
Italy	0.32	6	5.33	359				*Phragmites australis, Typha latifolia*	Borin and Toccheto (2007)
USA	0.40	12.3	3.25	367	18.40	12.80	30.4		Kovacic et al. (2006)
Sweden	1	300	0.33	374				not specified	Strand and Wiesner (2013)
USA	0.60	15	4.00	426					Kovacic et al. (2000)
USA	1.63	71 500	0.0023	542	2.40	0.75	68.8	*Scirpus sp., Typha sp.*	Beutel et al. (2009)
Sweden	0.22	60	0.37	576				not specified	Strand and Wiesner (2013)
Sweden	0.80			580					Arheimer et al. (2004)
Korea	5	20	25.00	605	3.50	1.35	61.4	*Phragmites australis, Typha angustifolia*	Kim et al. (2010)
USA	150	2 000	7.50	681	10.89	2.50	77.0	*Typha latifolia, Schoeno-plectus californicus*	Diaz et al. (2012)
USA	0.60	15	4.00	780	14.10	11.00	22.0		Kovacic et al. (2006)
Sweden	0.75	380	0.20	791				not specified	9,6
Norway	0.12	80	0.15	800	8.03	6.29	21.7	*Phalaris arundinacea, Glyceria fluitans, Typha latifolia, Sparganium erec-tum, Phragmites australis*	Blankenberg et al. (2008)
USA	4.50	1 620	0.28	888	5.70	5.31	6.8	*Typha latifolia, Polygonum lapathifolium*	Diaz et al. (2012)
Sweden	0.40	650	0.062	989				not specified	Strand and Wiesner (2013)
Sweden	0.28	200	0.14	1 003				not specified	Strand and Wiesner (2013)
Sweden	3			1 030	4.60	3.50		not specified	Arheimer and Wittgren (2002)
USA	3.30	425	0.78	1 050	8.20	3.50	57.3	*Nyssa biflora, Acer ru-brum, Salix nigra*	Hunt et al. (1999)
Spain	1.32	22	6.00	1 200*	20.00			*Phragmites australis, Typha latifolia, Scirpus lacustris, Carex divisa*	Moreno-Mateos et al. (2010)
Sweden	0.65	880	0.074	1 524				not specified	Strand and Wiesner (2013)
USA	2.50	324	0.77	2 072	16.17	9.77	39.6	*Typha latifolia, Polygonum lapathifolium*	Diaz et al. (2012)
Sweden	0.40			4 336	8.70	7.50		not specified	Arheimer and Wittgren (2002)
New Zealand	0.026	2.6	1.00	8 410	34.90	7.59	78.3	*Glyceria declinata, Holcus lanatus, Typha orientalis*	Tanner et al. (2005)
USA	2.30	324	0.71	11 267	16.17	9.76	39.6	*Polygonum lapathifolium*	Diaz et al. (2012)
Denmark	0.14	100	0.14	13 026	23.50	13.70	41.7	*Typha latifolia*	Cochran et al. (2016)

*Nitrate nitrogen only.

was reported by Fisher, Acreman (2004) for natural wetlands or by Turner (1999) for wetlands designed for water quality improvement in Louisiana. Our survey (Fig. 8) resulted in low correlation ($R^2 = 0.115$), however, the decreasing trend is apparent.

Other important parameters affecting the removal of nitrogen are hydraulic loading rate, hydraulic retention time, and plant uptake. Longer residence time provides a greater opportunity for sediment-water contact, thereby promoting retention processes such as denitrification and sedimentation (Svendsen, Kronwang, 1993; Windolf et al., 1996; Jordan et al., 2003). However, the residence time is difficult to calculate for agricultural drainage waters because of highly variable flow of the drainage (Woltemade, 2000). It has been generally agreed that hydraulic retention time should be longer than two days to achieve any substantial nitrate removal (Hey et al., 1994; Phipps, Crumpton, 1994; Kovacic et al., 2000; Beutel et al., 2009). However, the excessive long retention time may cause adverse effects such as increased release of dissolved organic matter or increased salinity due to evapotranspiration (Diaz et al., 2008). Hydraulic loading rate, calculated by dividing the flow rate by the wetland surface area, is very seldom calculated for agricultural drainage waters due to difficulties with a constant drainage flow. The fluctuation of the flow during the year is usually so great that any 'average' hydraulic retention time is just misleading.

Plants are an indispensable part of the wetlands but they are seldom taken into consideration when removal mechanisms in constructed wetlands are evaluated. Due to vigorous growth of wetland plants, the amount of nitrogen sequestered in the aboveground biomass is usually high and commonly amounts to 40–60 g N m^{-2}, i.e. 400–600 kg N ha^{-1} for tall species such as *Phragmites australis* or *Typha latifolia* (Vymazal et al., 1999; Vymazal, Kröpfelová, 2008). As already mentioned, the predominant form of nitrogen in agricultural drainage waters is nitrate and therefore denitrification is always considered as the major removal process. Due to lack of ammonia in drainage waters, nitrate is taken up by plants as well. In order to remove nitrogen sequestered in the biomass, the plants must be harvested. However, vegetation in constructed wetlands with free water surface is usually not harvested (Vymazal, 2013) and, therefore, most of nitrogen is released back to water during plant decay and subsequent decomposition (e.g. Dinka et al., 2004; Longhi et al., 2008). The question remains whether the biomass should be harvested to enhance removal of nitrogen because decaying biomass is a source of organics necessary for denitrification. So far, there have been no studies that would critically compare the benefits of harvesting and provision of organics for decomposition. This gap concerning vegetation maintenance still needs to be filled in.

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