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REPORT

Identification of existing mitigation systems that can attenuate nitrates during high flow events from drained, agricultural fields

Project acronym: AQUISAFE 2

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Preparation of this report was financed in part through funds provided by

Berlin, Germany
2010

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Title

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Deliverable number

Aquisafe 2 D 1.1

Abstract

The project Aquisafe assesses the potential of selected near-natural mitigation systems, such as constructed wetlands or infiltration zones, to reduce diffuse pollution from agricultural sources and consequently protect surface water resources. A particular aim is the attenuation of nutrients and pesticides. Based on the review of available information and preliminary tests within Aquisafe 1 (2007-2009), the second project phase Aquisafe 2 (2009-2012) is structured along the following main components:

(i) Development and evaluation of GIS-based methods for the identification of diffuse pollution hotspots, as well as model-based tools for the simulation of nutrient reduction from mitigation zones.

(ii) Assessment of nutrient retention capacity of different types of mitigation zones in international case studies in the Ic watershed in France and the Upper White River watershed in the USA under natural conditions, such as variable flow.

(iii) Identification of efficient mitigation zone designs for the retention of relevant pesticides in laboratory and technical scale experiments at UBA in Berlin.

The following report focuses on (ii), providing an overview of existing mitigation systems that may reduce transport of agricultural pollutants to surface waters, with a particular focus on nitrate. The report is based on an extensive review of scientific literature as well as practical guidelines. The review emphasizes on systems, which can treat pollutant loads from agricultural fields with surface or tile drainage. Such mitigation systems could play an important role in intensely used agricultural areas, where existing efforts in farm or crop management are not sufficient to reach water quality goals in receiving rivers. This is typically the case for agricultural catchments with high ratio of artificial drainage, which allows an almost complete transfer of water and contaminants, particularly during high flow events.

For each identified mitigation system, its general approach, performance against nitrates and other contaminants, boundary conditions as well as expected cost are given. The systems are structured according to their place on the pathway between field and surface water into

1. systems which attempt to reduce contaminant loads in the drainage pipes and ditches (section 2),
2. systems, which can be placed between drainage system and surface water (section 3),
3. systems, which can be placed in the receiving surface water (section 4).

The review shows that there are a number of feasible options with the potential to mitigate NO_3^- pollution from drained agricultural land. The most promising approaches with high removal potential were found to be:

- controlled drainage (section 2.2),
- bioreactors at the tile level (section 2.3.2),
- reactive swales (section 2.4.2),
- constructed wetlands (section 3.2) and
- river-diversion wetlands (section 4.2.2).

Most practical experience exists for constructed wetlands with surface flow (globally) and for controlled drainage (mainly in the USA), whereas the other systems are currently at an experimental state.

For a model agricultural area, the above systems resulted in expected nitrate reduction between 14 and 82 % and cost efficiencies between 23 and 246 € kg-N^{-1}. In terms of absolute nitrate removal, (i) wood chip walls parallel to tile drains and (ii) constructed wetlands with straw as carbon source were found to be most effective. However, for both systems there are relatively few experiences so further testing will be necessary. Regarding cost efficiency, (iii) constructed surface flow wetland with low construction cost (dam) and (iv) controlled drainage are most efficient. Whereas constructed surface flow wetlands can be implemented independently, drainage control structures need to be managed by farmers, which requires their active cooperation and proper training.

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Chapter 1
Introduction

1.1 Approach

Pollution from agricultural crop land is a major issue, where transfer between field and river is fast. This is particularly the case in areas with high percentage of drainage systems, either via drainage ditches or via drain pipes/tile drains.

The following report focuses on mitigation systems that may reduce transport and fate of agricultural pollutants to surface waters, with a particular focus on nitrate (NO_3^-). However, it does not include changes in farm or crop management, but aims at systems, which could be supported independently by watershed and local authorities, associations or drinking water industry.

In the most simplistic form, there are two control factors of contaminant retention between field and surface water: (i) the attenuation rate that can be achieved for a given contaminant (e.g., denitrification rate for NO_3^-) and (ii) the residence time of the water before it enters the receiving surface water body. In drained systems, typically (i) and (ii) are minimal, by their basic and intended design, which allows an almost complete transfer of water and contaminants. The situation is even worse during high flow events, when the excess flows create an increased release of contaminants. Under such conditions available retention potential (e.g., deeper dips in drainage channels or small retention ponds) is quickly exceeded.

Mitigation systems, which have the potential to increase (i) and/or (ii) can be placed at different levels on the pathway from field to surface water:

1. they can attempt to change drainage pathways, flow dynamics and contaminant retention to reduce contaminant loads in the drainage pipes and ditches,
2. they can be placed between drainage system and surface water, typically close to the surface water and
3. they can be placed in the receiving surface water.

The mitigation systems covered in this report have been drawn from scientific and non-scientific literature. In particular, existing lists/recommendations of mitigation systems have been consulted, among them

- Identified mitigation options by the EU COST Action group 869 “Mitigation options for nutrient reduction in surface water and groundwaters”,
- Agricultural Best Management Practices by the Scottish Environment Protection Agency (SEPA),
- Phosphorus best management practices protecting water quality by the US Geological Survey and Natural Resources Conservation Service (www.sera17.ext.vt.edu),
- Best Management Practices to Control Nitrogen in the Neuse River Basin, by the North Carolina State University (Gilliam et al. 1997) and

- Best management practices to reduce diffuse pollution from agriculture by the UK Environment Agency (Browning et al. 1996).

In three chapters (chapter 2 to 4) existing mitigation systems of the three types are outlined. Description of each mitigation system is structured along the following subtitles:

- approach,
- performance against NO_3^- and other contaminants,
- important boundary conditions for implementation and maintenance and
- cost.

Finally, systems are summarized in a concluding chapter.

1.2 Mechanisms of nitrate transport and removal

Diffuse nitrogen pollution enters rivers on various pathways. Studies on four exemplary European river basins of very different character indicate a dominance of sub-surface pathway via groundwater and, if present, tile drains (Table 1.1). The share of surface runoff is surprisingly small in comparison, even in the peri-alpine Swiss Rhine basin, where relatively steep slopes are present. The general pattern remains similar even on a more local scale. For instance, maximal surface runoff contribution in a single subcatchment of the Elbe basin was 6.7 % (Arbeitsgemeinschaft für die Reinhaltung der Elbe 2001). Main variations between subcatchments were found in the same study for tile drainage (16 to 40 % contribution) and for urban diffuse sources (= mostly from leaf fall and atmospheric deposition on impervious surfaces, 3 to 11 %). It is noteworthy that diffuse phosphorus pollution for the four basins in Table 1.1 is dominated clearly by surface runoff with relative contributions between 44 and 67 %.

Table 1.1: Diffuse nitrogen sources in four large river basins and a summary of all German basins

	German Elbe basin [1]	**Bavarian Danube basin** [2]	**Bavarian Main basin** [3]	**All German basins** [4]	**Swiss Rhine basin** [5]
Catchment size [km^2]	147200	48200	20300	357112	9426
Studied years	1993-1997	2005-2007	2005-2007	2005	2001
Diffuse N sources:					
Groundwater/interflow	61 %	79 %	78 %	58 %	72 %
Tile drainage	24 %	6 %	4 %	26 %	14 %
Surface runoff (erosion + dissolved)	4 %	10 %	13 %	11 %	3 %
Atmospheric deposition	3 %	2 %	2 %	2 %	1 %
Urban diffuse pollution	8 %	4 %	4 %	4 %	11 %

[1] Arbeitsgemeinschaft für die Reinhaltung der Elbe (2001)
[2] Bayerisches Staatsministerium für Umwelt und Gesundheit (2009a)
[3] Bayerisches Staatsministerium für Umwelt und Gesundheit (2009b)
[4] Arle et al. (2010)
[5] Prasuhn (2003)

Nitrogen is spread on agricultural fields predominantly in the form of ammonium (NH_4^+) and organic nitrogen via fertilizer and manure. Organic nitrogen is typically stored in the soil and mineralized slowly to NH_4^+. In oxic, non-saturated soils NH_4^+ is rapidly oxidized to NO_3^-, given the typically high population density of nitrifying bacteria in agricultural soils. Water that flows from the field via groundwater or tile drains has to pass the soil column. As a result, most of the nitrogen in those two dominant pathways to surface waters (Table 1.1) will be in the form of NO_3^-, explaining the often very high NO_3^- levels in agricultural streams. Organic nitrogen and NH_4^+ can be transported to rivers via surface runoff. Although this pathway is of minor importance on a catchment level (Table 1.1), locally high NH_4^+ levels can lead to toxic conditions for fish. NH_4^+ is also nitrified in rivers but at much lower rates than in the soil (Pauer and Auer 2000).

The most important NO_3^- removal process is microbial denitrification. In the case of complete denitrification, NO_3^- is reduced to gaseous N_2 and therefore removed from the water phase. Requisites for denitrification are suboxic or anoxic conditions and sufficient organic carbon sources as electron donors. In surface water systems, such as rivers or surface wetlands, denitrification therefore occurs mostly at the sediment-water-boundary (Reinhardt et al. 2006). Higher denitrification rates can be reached in groundwater, given the low oxygen availability (e.g.,Dosskey 2001; Hoffmann et al. 2000). However, in groundwater organic carbon is often limiting denitrification. As a result, denitrification in groundwater typically increases towards the soil surface, where organic carbon supply from plant material is available (Gift et al. 2010; Hoffmann et al. 2000). Lab results indicate high NO_3^- retention rates when NO_3^--rich water flows through organic carbon sources, such as wood chips or straw (Chun et al. 2009; Greenan et al. 2009; Krause pers. comm. 2010). Given the above, design of denitrification sites often involves (i) increase in infiltration to enlarge anoxic flow zone and (ii) the addition of carbon sources, such as wood chips or straw (e.g., Braskerud 2002; De Haan et al. 2010; Søvik and Mørkved 2008).

Plants can also play a role in NO_3^- retention. Some studies indicate that they can support denitrification through (i) high surface for microorganisms on roots (Hunt 1995) and (ii) carbon supply from dead plant material (Reinhardt et al. 2006). The carbon supply can be of great importance in artificial systems with surface flow or infiltration (such as constructed wetlands) but may be of lesser importance in natural riparian zones, because of old carbon deposits (summarized in Vidon et al. 2009). Plants also assimilate nutrients, which can lead to high seasonal removal of NO_3^- (e.g., Liikanen et al. 2004). However, it is important to consider that plant uptake is a reversible process and a high percentage of assimilated nutrients is released during mineralization of dead plant material (Reinhardt et al. 2006). Part of the assimilated nutrients can be removed by seasonal removal of plants (Lu et al. 2006). However, cutting back of plants can also lead to a reduction of positive effects of plants on sedimentation and denitrification (Reinhardt et al. 2006; Uusi-Kämppä et al. 2000).

One disadvantage, which is often mentioned regarding denitrification zones against agricultural pollution, is the production of greenhouse gases methane (CH_4, under anaerobic, low redox conditions) and nitrous oxide (N_2O, at incomplete denitrification). Søvik et al. (2006) found significant summer emissions of 0.4 ± 0.25 mg-N_2O-N m^{-2} d^{-1} from a constructed wetland installed in agricultural settings. Jacinthe et al. (2009) measured slightly lower values between < 0.1 and 0.3 mg N_2O-N m^{-2} d^{-1} for a vegetated sand filter, which was dosed with high NO_3^- concentrations continuously. Although these rates underline that denitrification systems are indeed sources of N_2O, the results have to be put into perspective by comparing them to other agricultural sources of N_2O. For instance, IPCC (1997) estimate N_2O-emmissions from fertilizer and manure spreading of 0.0125 kg N_2O-N per kg of applied N. For a moderate fertilization of 150 kg-N ha^{-1} yr^{-1} this translates to expected emissions from agricultural cropland of ~0.5 mg-N_2O-N m^{-2} d^{-1}, which is in the same order of magnitude as in the constructed wetlands above. As a result, turning agricultural cropland into a mitigation zone to increase denitrification is not expected to lead to a major increase in N_2O emissions in most cases.

Chapter 2

Systems to mitigate drainage from fields

2.1 Removal or deterioration of artificial drainage systems

Approach of the method

Surface drainage systems as well as subsurface tile drains increase loads of nutrients and pesticides from agricultural surfaces to surface waters and are often the major source of these pollutants on catchment scale (DWA 2008; Randall and Mulla 2001). As a result, the active removal of tile drains or drainage ditches or the passive deterioration of such systems would be an effective measure to protect surface water bodies. The approach would lead to (i) a decrease in flow from fields, (ii) a rise in water tables and thus higher residence times and (iii) higher nutrient uptake, denitrification and pesticide degradation (Figure 2.1). The approach was elaborated by Schoumans in an online document of the EU COST Action group 869.

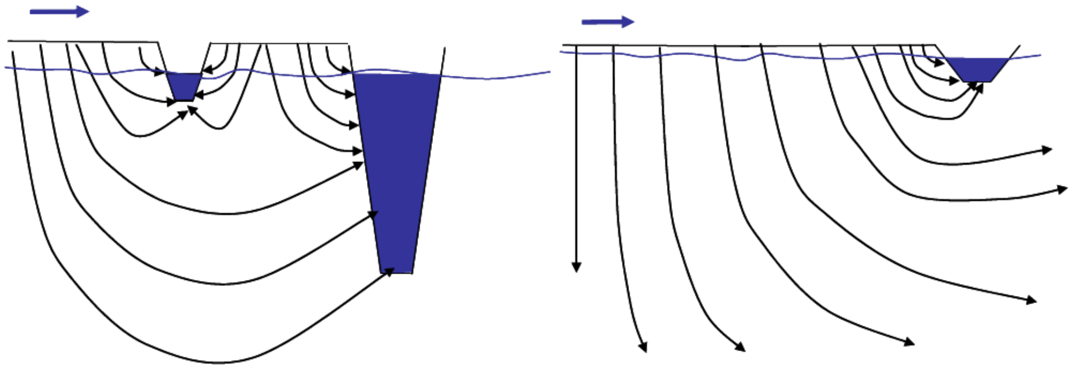


Figure 2.1: Effect of filling of drainage ditch (from Schoumans, www.cost869.alterra.nl)

Performance

We did not find any reference, where the effect of removal of a drainage system was actually monitored. Basically, one would assume that most of the negative effects of drainage systems would be removed. However, it is important to keep in mind that the agricultural use of the land will have to be changed at lower drainage.

Boundary conditions

Since artificial drainage is the basis for high crop production in many regions its removal will not be a feasible option in most cases. Active removal of drainage systems might be an option for former agricultural areas, which are no longer used agriculturally, in order to restore natural conditions with a higher water table, because of groundwater protection or similar reasons.

Cost

Leaving existing drainage systems deteriorate is a no-cost option for surfaces, which are no longer used for agricultural activities. In turn active removal of drainage systems may lead to significant but unknown cost.

2.2 Controlled Drainage

Approach of the method

As pointed out above, tile drainage and drainage channels increase the flux of water and pollutants between agricultural surfaces and surface waters. The lowering of the water level in the soil to around 1 m from the surface is necessary to allow working on the field with heavy machinery, mainly before the growing season and during harvest. However, water levels can be higher during times when no activities on the fields are necessary (depending on crop and season).

The active regulation of water levels (i) between 0.3 and 0.6 m below the surface in periods where no on-field manipulations are required and (ii) between 0.9 and 1.2 m below the surface only when machines are required on the fields, is referred to as "controlled drainage" (Fausey 2004; Gilliam et al. 1997; Helmers et al. 2008). It is typically managed by installing weirs in receiving drainage channels (Figure 2.2a). This approach works both for surface drainage (drainage ditches), as well as tile drained fields. Alternatively control shafts can be installed for single tiles (Figure 2.2b).

a)

b)

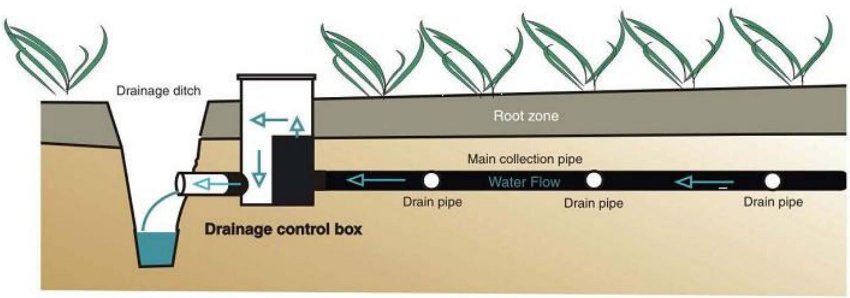


Figure 2.2: Design of controlled drainage systems
(a) Control weir on drainage ditch (from Gilliam et al. 1997)
(b) Control shaft on drainage tile (from Fouss and Sullivan 2009)

Controlled drainage systems were originally designed to increase available water for plants in dry years. However over the last decades installation of controlled drainage systems was also found to improve water quality of drainage flows. Firstly, controlled drainage lowers water loss and thus nutrient loads to receiving surface waters (Fouss et al. 2004; Gilliam et al. 1997). Secondly controlled drainage can increase denitrification in the field, because of saturation of upper soil layers and generally higher residence times in the soil (Fouss et al. 2004; Gilliam et al. 1997). Case studies for controlled drainage as a means for reduction of diffuse pollution from drained fields have been carried out mainly in the USA. In particular, the Agricultural Drainage Management Systems Task Force (ADMSTS) was founded in 2002 to promote controlled drainage to improve water quality of drainage flows (Fouss and Sullivan 2009).

Performance

Gilliam et al. (1997) indicate an average decrease in drainage flow from the fields compared to conventional drainage of around 30 % across different soil types, rainfall patterns, type of drainage system and management intensity. Case studies by Evans et al. (1991) (26 to 32 %) and Fausey (2004) (33 to 39 %) support this number, whereas works by Helmers et al. (2008) found higher 53 to 64 % reduction.

According to Gilliam et al. (1997) controlled drainage may reduce NO_3^- concentrations in drainage outflow by up to 20 percent, but organic nitrogen concentrations can be somewhat increased. Other studies mostly confirm decrease in NO_3^- concentration (Evans et al. 1991; Fausey 2004; Helmers et al. 2008), but also show specific years with increased values as a result of controlled drainage (Helmers et al. 2008). Still, as a result of increased denitrification, reductions in N loads are typically higher than for water volumes; between 40 and 45 % for total nitrogen and between 45 to 70 % for NO_3 (Drury et al. 2009; Evans et al. 1991; Fausey 2004; Helmers et al. 2008; Smeltz et al. 2005). Reduction of water and N loss is indicated to be similar for surface and tile drainage, despite different volumes and concentrations (Figure 2.3).

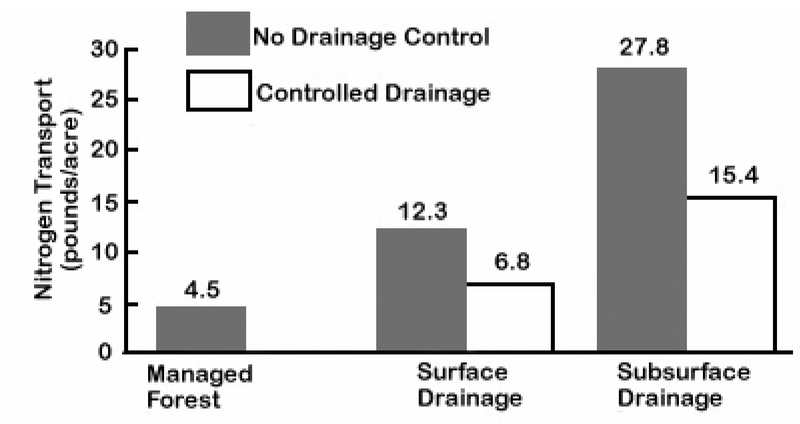


Figure 2.3: Change in total nitrogen loads from controlled drainage for 14 sites in NC, USA (from Evans et al. 1991)

According to Gilliam et al. (1997), controlled drainage is most effective in reducing drainage flow during dry years, when it may totally eliminate outflow, whereas in wet years, it may have little or no effect on total outflow. This is not confirmed by Helmers et al. (2008) who found flow reductions of 64 % for two dry years and of 58 % for two wet years. Also the average NO_3^- reduction in dry and wet years was similar with 64 % and 58 %, respectively.

For phosphorus, controlled drainage has a higher effect in surface drainage systems (~40 % reduction) than for tile drains (~20 % reduction) (Evans et al. 1991).

No information was found for other pollutants.

Regarding agricultural yield, only slight increases or decreases < 10% were found as a result of controlled drainage (Drury et al. 2009; Fausey 2004; Helmers et al. 2008).

Boundary conditions

Controlled drainage can be implemented for new systems or by adapting existing drainage systems (Fouss and Sullivan 2009). Management schemes (particularly water levels) for controlled drainage need to be adapted to local crops, soil properties and rainfall (Gilliam et al. 1997). In general, the water level from the surface should not be set below 0.3 meters at the control weir to avoid plant damage (Evans et al. 1991).

Controlled drainage can only be used on relatively flat fields of a slope of 1% or less (Fouss and Sullivan 2009). Fouss and Sullivan (2009) point out that some recent technological developments may make controlled drainage applicable to lands of greater slopes but no details are given.

Controlled drainage systems need to be actively managed by the farmers. Water levels have to be adapted slowly below 0.15 m d^{-1} to avoid bank instability (Evans et al. 1991). As a result, water levels need to be lowered at least several days prior to field activities. Moreover, adaptations are necessary at continuous (> 24h) strong rain events.

Although controlled drainage provides important water quality benefits, Gilliam et al. (1997) point out that control weirs on ditches and streams prevent their restoration as ecologically functional streams, which may also result in pollutant attenuation and provide ecological benefits in addition (see also section 4.1).

Cost

Information on cost are provided by Wossink and Osmond (no year) in an online document of the North Carolina Cooperative Extension Service. They estimate a necessary investment for installation of 1900 US$ ha^{-1} (≈ 1400 € ha^{-1}) and annual maintenance cost of 125 US$ ha^{-1} yr^{-1} (≈ 90 € ha^{-1} yr^{-1}), both numbers are for an operation time of 15 years but without being specific on type of controlled drainage. Wossink and Osmond (no year) also expect benefits for the farmers from increased yield between 200 and 1100 US$ ha^{-1} yr^{-1} (≈ 150 and 800 € ha^{-1} yr^{-1}) for a tobacco-wheat-soybean rotation. However the expected yield increase seems optimistic, since several studies on controlled drainage have not shown significant increases in agricultural yield (see above).

2.3 Mitigation systems between field surface and drainage

The following sections 2.3.1 and 2.3.2 introduce two different approaches, which attempt at placing carbon sources, mostly wood chips, in agricultural fields to increase denitrification in subsurface water.

2.3.1 In-field groundwater denitrification reactors

Approach of the method

Reaction walls were originally developed to prevent plumes from septic systems or landfills. However, several applications in agricultural fields have been documented over the past decade. Reaction walls in agricultural applications contain carbon sources and aim at treating shallow groundwater before it enters rivers or drainage ditches. Classically, walls are placed perpendicular to groundwater flow direction and are flown through laterally (Figure 2.4a). An alternative are so-called upflow reactors, which are placed in sloping shallow aquifers and use the hydrostatic pressure to create upward flow through the laterally closed reactors (Figure 2.4b). The upflow design has the advantage that a more groundwater passes through the reactor per square meter and consequently less material is needed to treat the same amount of groundwater. On the other hand, residence times are usually lower in the upflow design.

Performance

Hydraulic residence times (HRT) in reaction walls depend on groundwater flow. In a review of published values, Schipper et al. (2005) found a range between 1 to 40 d. HRT in upflow reactors depend on shallow groundwater level and vary seasonally. Van Driel et al. (2006b) measured HRT in upflow reactors between 0.3 and 6 d.

NO_3^- retention in reactor walls is between 0.2 and 50 mg-N d^{-1} L^{-1} (median around 2.4 mg-N d^{-1} L^{-1}) relative to the reactor volume (Robertson et al. 2000; Robertson et al. 2008; Schipper et al. 2005; Schipper and Vojvodic-Vukovic 2001) and around 2.5 g-N d^{-1} m^{-2} relative to the reactor surface area (van Driel et al. 2006a). Retention in upflow reactors is similar with 3.2 to 9.9 mg-N d^{-1} L^{-1} and 0.7 to 3.5 g-N d^{-1} m^{-2}. Removal rates were generally increasing with decreasing HRT and increasing NO_3^- concentrations (Schipper and Vojvodic-Vukovic 2001; van Driel et al. 2006b). NO_3^- concentrations decreased between 30 and more than 95 % in the above studies. Similar numbers are reported for vertical reaction walls used to attenuate NO_3^- from septic systems (Robertson et al. 2000).

However, it has to be noted that none of the above results are based on a total nitrogen budget. Consequently there is a significant uncertainty regarding potential leaching of NH_4^+ or organic nitrogen. Although DOC is discussed in several of the sources regarding agricultural pollution cited above, only Robertson et al. (2000) give measured concentrations, but mostly for septic systems, where inflow DOC concentrations are already at very high level. In one reaction wall with moderate DOC inflow concentrations of 5.7 mg l^{-1}, they observed a 40 % increase to 9.9 mg l^{-1}. There was no information on other contaminants.

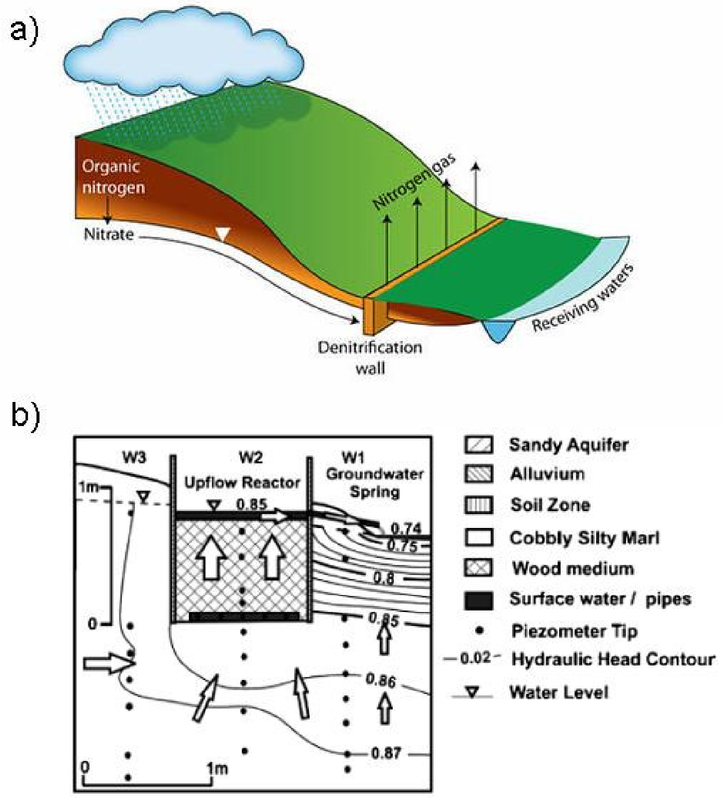


Figure 2.4: Design of denitrification reactors to treat shallow groundwater
(a) Lateral flow reactor/reaction wall (source: Schipper 2009, University of Waikato, New Zealand)
(b) Upflow reactor (source: van Driel et al. 2006b)

Long-term observations indicate that reactors maintain a high level of denitrification. Robertson et al. (2008) found a 50 % decrease in efficiency after 15 years of operation, whereas studies by Robertson et al. (2000) and Schipper and Vojvodic-Vukovic (2001) did not indicate any reduction in denitrification during 8 and 5 years of observation, respectively. Robertson et al. (2008) observed a drop in denitrification after removing the wood particles from the reactive wall, underlining the importance of the carbon source even for DOC-rich septic water.

Boundary conditions

Case studies on groundwater denitrification reactors are mostly from Canada and New Zealand. Denitrification walls are constructed in trenches perpendicular to groundwater flow within the water table. Carbon sources, such as saw dust or wood chips, are either filled directly into the trench (van Driel et al. 2006a) or as a mixture with sand or gravel (Robertson et al. 2000; Robertson et al. 2008; van Driel et al. 2006a). An alternative is the mixing with material of the present aquifer, in- or outside of the trench (Schipper et al. 2004). Upflow reactors are typically filled with 100 % of the carbon source, either using wood chips or saw dust (van Driel et al. 2006b). The hydraulic conductivity of the reactor material can also be influenced by mixing coarse wood particles with saw dust (van Driel et al. 2006a).

As outlined above, denitrification in reactor walls remains high for more than a decade. Nevertheless, the carbon source is used up during operation and needs to be replaced at one point. Robertson et al. (2000) estimated that denitrification used up below 3 % of available carbon in reactors after 6 to 8 years of operation, and that even if other carbon sinks are taken into account less than 20 % of initial carbon mass are likely to be consumed, which would lead to necessary replacement every ~20 years. Schipper and Vojvodic-Vukovic (2001) indicate that life span of carbon sources may be reduced if O_2 enters the system; however they could not detect any difference in sections of a denitrification wall, which were seasonally unsaturated.

Apart from a decrease in efficiency, gradual clogging might be an issue. In two described systems (one reactive wall, one upflow system) clogging occurred right after construction (Schipper et al. 2004; van Driel et al. 2006b). Reactive walls that did work maintained their hydraulic conductivity throughout observation. In upflow reactors, outflow pipes (Figure 2.4b) are prone to clogging from fouling and need to be cleaned at regular intervals. Unfortunately the authors do not indicate necessary time interval for such maintenance.

Precondition for the implementation of both reactive walls and upflow reactors are shallow water tables. In addition, upflow reactors require a certain slope, which allows a release pipe below the uphill water table (Figure 2.4b).

Whereas groundwater reactors can be placed in any soil, amount of treated water increases with hydraulic conductivity of the aquifer. On the other hand, it is crucial that hydraulic conductivity is clearly higher in the reactor than in the surrounding aquifer. As a result aquifers with moderate conductivity would be an ideal compromise. In case studies described by Schipper et al. (2004) hydraulic conductivity of the reactor material changed after construction (both increase and decrease was observed). In one case, hydraulic conductivity decreased more than an order of magnitude after installation, leading to an almost complete bypassing of the denitrification wall (Schipper et al. 2004). The decrease was explained by mixing of material in the trench (rather than outside) and the presence of clay minerals.

In summary, upflow reactors and denitrification walls are recommended for (i),shallow water tables (ii) fields with slope and (iii) aquifer with moderate conductivity. What is more, the filling material should be regularly replaced, and maintenance needs include regular cleaning of outflow pipes (for the outflow reactor).

Cost

Cost for denitrification walls in agricultural fields filled with mixed sawdust and local soil were indicated at 23 € per meter of a 1.5 m deep and 1.5 m wide trench containing ~50 % of sawdust and 22 € per meter of a 2.5 m deep and 3 m wide trench containing ~30 % of sawdust (Schipper 2000; cost converted from NZ$).

2.3.2 Bioreactors at the tile level

Approach of the method

The approaches under 2.3.1 are not directly applicable to tile drained fields, although the problematic is similar. Three options have been described in literature, how bioreactors can be combined with tile drains:

(i) Denitrification walls as in 2.3.1 but parallel to tile drains (Figure 2.5 a)
(ii) Wood-filled trenches, which are built perpendicular to tile drains. Tile drainage water is partly diverted through the trench and leaves the field through a lower, parallel tile drain (Figure 2.5b).
(iii) Simple wood-filled box, which is installed at the end of tile drains (Figure 2.5c).

Performance

Jaynes et al. (2008) studied the performance of approach (i) in comparison to parallel standard tile systems over a period of five years. The system efficiently removed between 39 and 63 % of annual NO_3^- loads, with an average of 54 %. From the removal, the authors calculated an average NO_3^- retention rate per volume of reactor of ~0.6 mg N L^{-1} d^{-1}, in the lower range of the rates observed for groundwater denitrification walls. The efficiency did not show a clear correlation with annual precipitation, since lowest removal was found in the driest and the second wettest year. However, NO_3^- concentrations in tiles with denitrification walls increased with flow rate (in contrast to control) indicating effect of lower residence times. Similar to the groundwater systems, no decrease in removal was found over the five year observation.

Simple end-of-tile reactors (iii) were studied by Blowes et al. (1994) and Robertson et al. (2000). Blowes et al. (1994) filled reactors with 70 % sand and 30 % carbon sources, bark mulch in one case and equal shares of bark mulch, wood chips and leave compost in the other case. HRT were kept relatively high between 1 and 6 days treating 10 to 60 L d^{-1}, while most of the tile flow during rain storms bypassed the reactors. In contrast, Robertson et al. (2000) used 100% coarse wood mulch, which allowed much lower HRT between 3 and 7 hours, treating 800 to 2000 L d^{-1}. NO_3^- retention was >98 % for high HRT systems with a (probably NO_3^--limited) removal rate of ~0.4 mg-N L^{-1} d^{-1} (Blowes et al. 1994). The much shorter HRT still led to a high removal of 58 % of total NO_3^- load and retention rates of 5 mg-N L^{-1} d^{-1} at 2 to 5 °C and 15 to 30 mg-N L^{-1} d^{-1} at 10 to 20 °C (Robertson et al. 2000). Both high and low HRT systems led to an increase in DOC concentration from 3.6 to 4.5 mg L^{-1} (~20 % increase) and from 2.9 to 4.3 mg L^{-1} (~30 % increase), respectively. The values for the low HRT system are based on a seven years observation time. As for the above systems, Robertson et al. (2000) found no decrease in efficiency over the seven years of observation. Unfortunately no single high flow events are shown in the study, although flow variation is included in the analysis.

Wood-filled trenches (ii) are currently monitored by the US Universities of Minnesota and Illinois. However, no studies on performance are available yet.

As for the groundwater systems, only NO_3^- and in some cases NH_4^+ were considered for the above NO_3^- budgets leaving uncertainty regarding leaching of organic nitrogen.

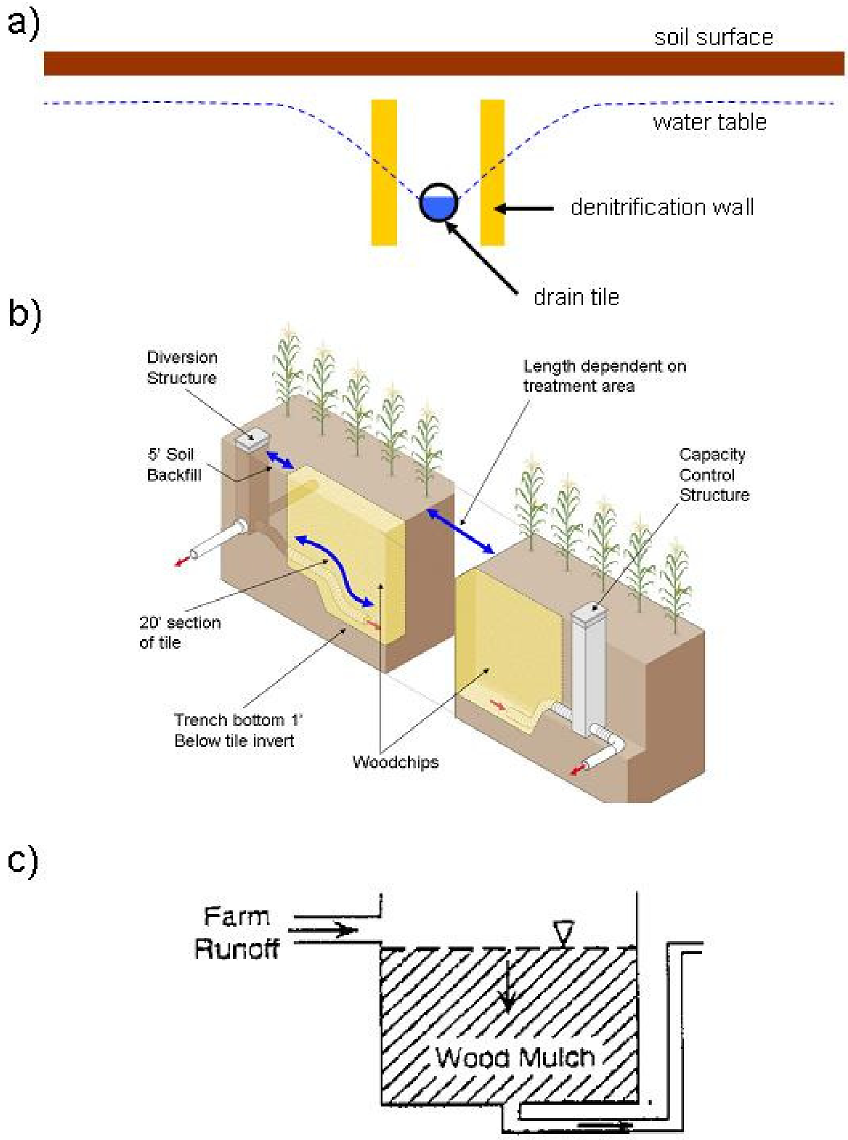


Figure 2.5: Design of denitrification reactors to treat tile drainage water
(a) Reaction wall parallel to tiles
(drawing based on description in Jaynes et al. (2008))
(b) Reactive trenches between tiles (source: Cooke and Chun 2010)
(c) End-of-tile bioreactor (source: Robertson et al. 2000)

Boundary conditions

Since tile drain systems are very common in North America, all of the above case studies are from the USA or from Canada.

While systems (i) and (iii) are relatively simple to implement, (ii) requires a sound hydraulic layout to make sure flow direction is from upstream to downstream tile drain. For that a certain slope between tile drains is required (downstream tile must be lower than upstream tile).

As for the systems in 2.3.1 replacement of carbon sources will be necessary but probably only after several decades. An issue with clogging with sediments was reported for end-of-tile reactors by Blowes et al. (1994) in the inlet pipe connecting the tile drains with the reactor.

One important boundary condition for the construction of any of the above approaches is the known location of tile drains. Particularly methods (i) and (ii) may lead to a damaging of tile drains if their location is unknown. For (iii) it is important to choose potent tile drains, so monitoring one year ahead may be necessary.

Although not mentioned by any author, it seems that (iii) could be well combined with tile-specific controlled drainage (Fig 2.2b).

Cost

Although no information was available, system (i) is likely to create similar cost to denitrification walls under 2.3.1. For system (ii), control structures are necessary in addition to the actual trench. In a fact sheet by the University of Minnesota (Minnesota no year), a cost of 3,200 US$ (~2,400 €) was estimated for one system at the end of a drainage system, covering ~3 ha. This translates into a specific cost of only 1,000 US$ ha^{-1} (~750 € ha^{-1}) of treated field surface but one has to keep in mind that nitrate retention efficiency remains to be assessed.

2.4 Drainage diversion

2.4.1 Measures at field boundaries

Similar to riparian buffers strips (see section 4.1) without agricultural crops can be placed between fields along topographic isolines. The idea of such field boundaries is the reduction of surface runoff through infiltration, the reduction of erosion through sedimentation and the potential nutrient uptake by plants. Moreover field boundaries may be areas of increased denitrification, thus reducing NO_3^- loads in subsurface water (EU COST Action group 869, http://www.cost869.alterra.nl). Most common field boundaries are hedgerows, which are part of traditional agricultural landscapes in many parts of Europe (Figure 2.6).

Performance

Available literature on field boundaries focuses on hedgerows. Caubel et al. (2003) and Ghazavi et al. (2008) showed that soil surrounding hedgerows is drier than on agricultural fields, because of increased evapotranspiration and rainfall interception of leaves. Ghazavi et al. (2008) conclude that the reduced soil water content may lead to (i) less erosion from surface runoff via increased infiltration and (ii) reduced sub-surface flow via delayed soil wetting at the start of the rain season. If hedgerows cover important areas (the authors estimate up to 20 % of agricultural surface could be covered by hedgerows, which may be unrealistic in most cases), they can also impact hydrology on a catchment scale with a 20 to 40 % increase in evapotranspiration and 2 to 6 % rainfall interception by leaves (Ghazavi et al. 2008). However, even at these high estimated hedgerow densities, flow would be mostly reduced during growth season. If main rain season does not match growth season, flow reduction during critical pollutant loads to surface waters might be limited.


Figure 2.6: Hedgerow landscape in Brittany, France (source: SMEGA 2010)

Regarding pollutant loads to surface waters there is no quantitative information on the effect of hedgerows. Caubel-Forget et al. (2001) measured NO_3^- concentrations in soil water up- and downslope of a hedge and in a parallel field without hedge. They found up to 90 % lower NO_3^- and more than double chloride concentrations around the hedgerow. They concluded that the results are most likely due to higher evaporation (chloride) and denitrification (NO_3^-). The findings indicate a denitrification potential around hedgerows, similar to riparian buffers. However, since bypassing of the hedge on preferential flowpaths or via deeper groundwater is likely to occur and groundwater flow is unknown, no quantification of overall effect of hedgerows is possible.

Whereas effects of hedgerows on flows from the fields and on water quality remain uncertain, hedgerows have an important role for biodiversity, forming a habitat and acting as corridors for a large number of species.

Boundary conditions

For water protection, field boundaries should be planned along topographic isolines, respectively perpendicular to expected surface or sub-surface flows. In the studies above hedgerows were combined with a parallel ditch upslope, but the effect of the ditch was not discussed. Regarding water or nutrient retention, it is important that a potential ditch should not be connected to the river.

In the case of hedges, planting new hedges means a significant effort, since trees and shrubs should be planted at 0.45 m distance, preferably in two or more parallel rows (Conservation volunteers Northern Ireland: http://www.cvni.org). In the US 220 shrubs are recommended for 300 m of hedge (http://www.awqa.org/pubs/CostEstimates/HedgerowPlanting.pdf). If farm animals have access to adjoining fields hedges may have to be protected by fencing or by planting a 75 % proportion of thorny plants (Conservation volunteers Northern Ireland: http://www.cvni.org).

Since a large hedge network still exists across Europe but is declining, preservation of existing hedges should be a major focus there.

Cost

Field boundaries reduce the surface which can be used for agricultural production. Fiener and Auerswald (2003) estimated a revenue of 686 € ha^{-1} yr^{-1} for a classical crop rotation in Southern Germany (winter wheat - corn - winter wheat - potatoes).

Cost for hedgerow planting are estimated in the US at 800 to 2300 US$ (656 to 1900 €) per 100 m of a hedge with a width of 2.4 m, including maintenance during the first five years (http://www.awqa.org/pubs/CostEstimates/HedgerowPlanting.pdf). Given the comparably high cost for plantation, the lost revenue above is negligible for hedgerow planting.

2.4.2 Reactive swales and grassed waterways

Approach of the method

The common idea behind this heterogenic group of measures is the modification of pathways of peak runoff from agricultural fields to reduce sediment and dissolved pollutant loads to receiving water bodies. In the case of “grassed waterways” flow pathways are vegetated with grass (SEPA 2009) or naturally occurring vegetation (Fiener and Auerswald 2003; Kröger et al. 2007). Flow pathways can either be (i) preferential flowpaths along talweg between fields (Figure 2.7a; e.g., Fiener and Auerswald 2003) or (ii) along artificial drainage channels (FAO 1986; Kröger et al. 2007).

Reactive swales attempt extending the pollutant removal by transforming existing drainage ditches into basically elongated constructed wetlands (Figures 2.7b, c). Designs can be adapted from stormwater bioretention swales (e.g., EPA 1999) or wastewater constructed wetlands (e.g., Kadlec and Wallace 2009). However, only one application of a reactive swale in an agricultural drainage ditch is described in the literature (Robertson and Merkley 2009). Robertson and Merkley (2009) constructed an infiltration zone at the bottom of an existing drainage ditch, diverting part of the runoff through the zone with a pressure gradient (Figure 2.7c). Based on experience with denitrification walls (see section 2.3) they chose wood chips as a filter material.

a)

b)

c)

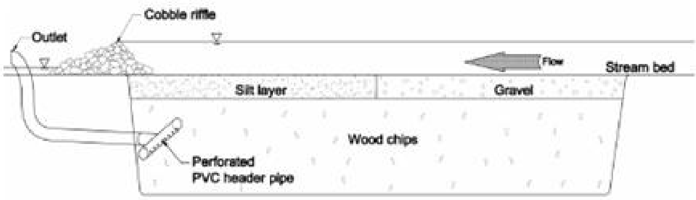


Figure 2.7: Design of grassed waterways and reactive swales
(a) Grassed waterway in the USA along talweg (source: USDA, www.wi.nrcs.usda.gov)
(b) Vertical infiltration swale in France (source: SMEGA 2010)
(c) Horizontal bioreactor design in drainage ditch in Canada (source: Robertson and Merkley 2009)

Performance

Vegetated swales

Vegetating preferential flow paths and drainage ditches has the prior goal to prevent erosion from the flow path itself (SEPA 2009). In addition, flow from the fields can be reduced via evaporation and infiltration and sediment loads can be kept back by plants. According to SEPA (2009) dissolved nutrients in the flow can be reduced by plant uptake or denitrification in infiltrating water. Whereas reduction of erosion on the flow path is well documented, data on effects on total sediment or nitrogen loads are sparse.

Fiener and Auerswald (2003) compared two parallel field sites, one with and one without grassed waterway along topographic thalweg for seven years. They found an average runoff reduction of 39 % and sediment reduction of 82 % in the grassed waterway compared to the agriculturally used thalweg. They further found that inorganic nitrogen decreased in the soil of the grassed waterway during the first year after implementation by 84 % and concluded that NO_3^- in sub-surface flow may be significantly reduced by the measure. However, the effect is only hypothetical and was not quantified. Kröger et al. (2007) found significant NO_3^- reductions of 22 and 61 % along the length of two 400-500 m long, vegetated drainage ditches. They observed high variability between removal and leaching of NO_3^- over the two-year observation period without a clear pattern. Moreover monitoring was not continuous, which may result in an error in calculated NO_3^- removal or leaching. Since removal along the drainage ditch is variable and not fully understood, results of Kröger et al. (2007) should be transferred with care. Nevertheless, findings indicate that vegetation of existing drainage ditches may be beneficial for water quality of receiving rivers.

Reactive swales

Robertson and Merkley (2009) attempted to enhance the retention capacity of drainage ditches further by constructing wood chip infiltration zones in existing ditches (Figure 2.7c). They found average removal rates between 0.3 g m^{-2} d^{-1} (at 3 °C) and 5.3 g m^{-2} d^{-1} (at 14 °C) for the water that passed the woodchip section. These removal rates are in the same range as for denitrification walls (section 2.3) and clearly higher than in surface flow wetlands (section 3.2). As for other woodchip reactors, removal rates increased with temperature and flow rates. However, it is important to note that hydraulic residence times in the woodchip zone were comparably high at 1.4 d on average (average flow of 24 L min^{-1}). At higher flow rates carbon limitation is likely to occur (Robertson and Merkley 2009). Overall impact of the infiltration zone on the water quality in the ditch cannot be assessed, since flow in the drainage ditch was not measured. However, the authors estimate that the 20 m long reactor (volume = 40 m^3) could treat the tile water of 3.3 ha of agricultural crop land.

NH_4^+ concentrations increased slightly to 0.09 mg-N L^{-1} in the outlet, moreover organic nitrogen was detected at 0.8 mg-N L^{-1}. Unfortunately, organic N was not measured at the inlet and the above removal is for NO_3^- only. Surprisingly DOC did not increase during passage but stayed at comparably high 10.9 mg L^{-1}. The only significant leaching compound was total phenols with an average of 4.8 µg L^{-1} in the outlet, but well below toxicity level throughout the two-year study. According to the authors, the occurrence of phenolic compounds is common in wood leachate.

Boundary conditions

Literature on grassed waterways and reactive swales is mostly from North America.

Vegetated waterways are relatively simple to implement if slope is < 20 % (FAO 1986). Vegetation can either be through sowing (typically grass) or through natural vegetation. Sowing of one grass species allows uniform flow pattern, which minimizes erosion. Moreover grassed waterways can be used to turn tractors or as access paths during dry season. A disadvantage is that a fine seedbed is required during sowing, which is prone to erosion (Fiener and Auerswald 2003).

The waterways must be inspected frequently during the first two rainy seasons after completion. Any minor breaks in the channels or structures should be repaired immediately (FAO 1986). Moreover, classical grassed waterways require regular cutting of the grass to reach a preferred length of ~10 cm (FAO 1986). However, Fiener and Auerswald (2003) found that waterways left to natural plant succession without annual cutting were as effective as the classical approach.

In contrast to the vegetated waterways, reactive swales need to be carefully planned, considering hydraulic head. For the design by Robertson and Merkley (2009) a flexible outlet is suggested, which allows adaptation of flow through the reactor zone. For construction, flow through the ditch was stopped and the bioreactor was implemented within one day. An alternative design is shown in Figure 2.7b, where infiltration is vertical towards a drain pipe at the bottom of the reactor.

The main issue with infiltration reactors is clogging with silt sediments which are transported in the drainage ditch. Flow through the pilot site by Robertson and Merkley (2009) decreased after 1 year of operation. Removal of silt on the gravel inlet to the reactor led to an abrupt increase in flow, but to a lower level than initially observed. Depending on sediment loads in a drainage ditch sediment traps may have to be considered.

Cost

For vegetated waterways no construction is necessary. If waterways are along talweg major cost is from land lost for agricultural production. Fiener and Auerswald (2003) estimated cost of 686 € ha^{-1} yr^{-1} of grassed waterway for a classical crop rotation in Southern Germany (winter wheat - corn - winter wheat - potatoes). The cost is similar for sowed or naturally vegetated waterways. If existing drainage ditch is left to be naturally vegetated as in Kröger et al. (2007) no major cost is expected.

Construction cost of reactive swales is expected similar to those of subsurface flow wetlands. For instance the 100 m^2 infiltration ditch in Figure 2.7c without monitoring shafts was estimated at 12.000 €, which amounts to 120.000 € per km of reactive swale.

Chapter 3
End of drainage mitigation systems

3.1 Riparian buffers

Approach of the method

In the most general form, riparian buffers are areas along watercourses with farming restrictions. In the following section we further limit “riparian buffers” to their effect in reducing diffuse pollution from agricultural fields to surface waters, whereas potential direct “cleaning effects” of the river are dealt with in section 4.1.

As indicated exemplarily in Figure 3.1, riparian buffers vary significantly in width (2 to > 20 m), plant cover (grass to forest) and management (intensive to no harvesting of plants; fencing or no fencing out of animals). Water from fields flows across buffers both via sub-surface and surface pathway.

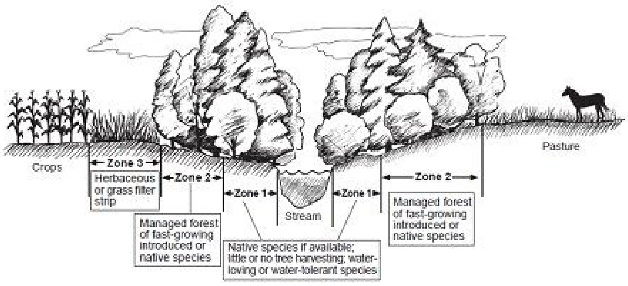
Riparian buffers have been described to reduce the amount of sediment, phosphates, NO_3^- and herbicides reaching the watercourse from agricultural fields. As a result, riparian buffers are currently planned in many European countries to reach good chemical quality of agriculturally impacted water bodies according to the EU Water Framework Directive.

For NO_3^- retention the sub-surface pathway is of major importance. However, as outlined below, riparian buffers are likely to be bypassed during high flow via (i) surface runoff, (ii) preferential flowpaths with low denitrification potential or (iii) drainage tiles and ditches. As a result riparian buffers will not be covered in maximal detail here. Nevertheless, they were added for the sake of completion.

Performance

There is a large range of literature available on the topic of riparian buffers. A review by the US EPA of more than 60 published studies indicates variable N removal between -25 % and > 99 %, with average removal of 33 % and 90 % in surface and subsurface flow systems, respectively (Mayer et al. 2005). These findings are confirmed by the extensive review of Dosskey (2001) who found NO_3^- retention of -115 to 28 % in surface and > 90 % in subsurface flow. NO_3^- removal is mostly the result of denitrification, whereas plant uptake will lead to seasonal variations. Also the type of plant cover does not have a significant impact on NO_3^- removal (Figure 3.2a). Although very narrow buffers seem to be less efficient, the studies collected by Mayer et al. (2005) do not show a clear relationship between width and nitrogen removal (Figure 3.2b).

a)



b)

c)

Figure 3.1: Examples of riparian buffers
(a) Three zones of riparian buffers, as suggested by USDA (source: USDA 1997)
(b) Grassed buffer in Switzerland (source: V. Prasuhn)
(c) Fenced buffer with planted trees in Scotland (source: A. Matzinger)

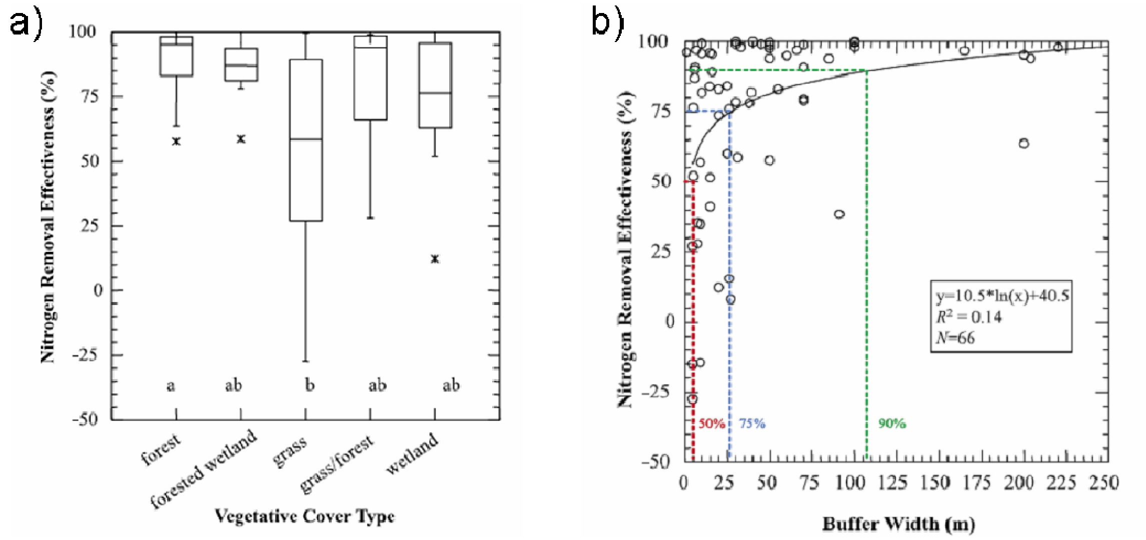


Figure 3.2: Observed NO_3^- removal efficiency (source: Mayer et al. 2005)
(a) for different vegetation cover
(b) dependent on buffer width

The above studies on subsurface flow are typically based on transects of shallow groundwater sampling points. Whereas the denitrification potential in groundwater below riparian buffer strips is unchallenged, the significance of the local results for the catchment scale is less clear for the following reasons:

- Denitrification occurs not only in riparian buffers but also in the saturated soils below crop areas. As a result, denitrification in riparian buffers should be compared to the situation before buffer implementation (Dosskey 2001). In one study, where this comparison was made a reduction in groundwater NO_3^- concentration of 35 % was found as a result of buffer strip implementation, clearly lower than the often described removal > 90 % (Clausen et al. 2000).
- A second critical point is the actual water pathway. Most studies focus on sites with low to moderate subsurface flow speeds. During high flow events, which often lead to high pollution loads, water is likely to enter the receiving streams via surface runoff or preferential subsurface pathways, not covered in local piezometer studies (Shabaga and Hill in press). For instance, Angier et al. (2005) found that upwelling zones within studied riparian buffers are responsible for most of the water and NO_3^- transport to an adjacent stream. Depending on local hydraulic conductivity and groundwater table, subsurface flow from fields can also bypass completely below or around riparian buffers (Heinen et al. 2010).
- In many agriculturally used areas, fields are drained by tiles or by drainage ditches, which typically bypass riparian buffers. As a result buffer efficiency is expected to be much lower in such areas, although preliminary studies indicate that riparian buffers may still have a removal effect for the groundwater that does not pass through artificial drainage and by reducing spraying at the water edge (B. Lennartz, pers. comm).

The limiting points above have led to serious doubts by experts (discussion at EU COST Action Group 869 meeting on riparian buffers in 2010), whether riparian buffers have a major impact on pollutant loads to streams on a catchment level. The general suggestion was that buffer implementation should have a stronger focus on various biodiversity benefits, since the extent of pollution abatement is uncertain.

Apart from denitrification, riparian buffers can also lead to the degradation of pesticides, but observations are highly variable and overall efficiency is questionable for similar reasons as for NO_3^- (Reichenberger et al. 2007). Similar to grassed waterways and hedgerows, riparian buffers can also retain sediments and sediment-bound substances such as phosphorus (Dosskey 2001). On the long run, the efficiency against particles and particle-bound substances can only be kept high if riparian buffers are managed, e.g., by regular removal of sediments and plant material.

Boundary conditions

Riparian buffers can be basically located along any stream adjacent to agricultural fields. They have been implemented in most industrial countries today and typically there are national or regional guidelines on how to implement them. However, given the uncertainty on performance, these guidelines vary significantly in

- Required/recommended buffer width (e.g., Switzerland: 6 m; USA: 29m (USDA), 15 m (GA, NC); Norway: 6 to 21 m (depending on region))
- Vegetation (grass (e.g., Switzerland) versus specific sowing or tree planting (e.g., Scotland))
- Fencing (e.g., general practice in the UK and Scandinavia, no fencing in central Europe)
- Management (e.g., regular cutting of grass in Switzerland, no or low management in UK because of fences)

In most of the guidelines, biodiversity aspects are only considered to a minor extent. Given the above limitations regarding pollution abatement, the effect of buffers on biodiversity and specifically on ecological networking functions should be stressed. For instance, this networking aspect is promoted in France within the national program “Trame verte et bleue”.

Cost

The implementation of buffer strips is comparably inexpensive. In the most simple case, a zone around the river is defined, where agriculture is restricted. For the fencing of a buffer strip a price of 4 to 5 € per meter of fencing is expected in Scotland by SEPA (2009). The Scottish recommendation also includes initial sowing of specific seeds at 50 €/ha of buffer strip. If trees are planted this may lead to significant additional cost, estimated between 370 to 1100 €/ha for the US state Maryland (Lynch and Tjaden no date).

However, at least on the long run, loss in crop yield and potential buffer management are expected to be the dominant cost. Loss in yield is estimated by case studies at 190 € ha^{-1} yr^{-1} in the US (Lynch and Tjaden no date), 686 € ha^{-1} yr^{-1} in Germany and 880 € ha^{-1} yr^{-1} in Scotland (SEPA 2009). Typical maintenance consists of mowing, which leads to cost between 20 and 110 € ha^{-1} yr^{-1} according to the Maryland case study. This can be significantly higher at 600 € ha^{-1} yr^{-1} (SEPA 2009) if buffer is fenced and thus access limited.

3.2 Wetland construction and restoration

Approach of the method

In the following section we discuss wetlands positioned at the interface between agricultural (subsurface and surface) drainage system and surface water. Wetlands situated in other conditions (e.g., in-stream) will be discussed in section 4.

Wetlands are land areas that stay wet during part of the year. Wetland plants are able to grow in saturated soil and are adapted to change in chemical, physical and biological conditions that occur during flooding (Kadlec and Wallace 2009).

Wetlands can be (i) natural, (ii) restored or (iii) constructed. Natural wetlands (i) have had low human intervention. In many areas former wetlands have been drained and no longer act as pollution sinks. Restoration (ii) refers to the recreation of structure and function of such former wetlands, often in topographic sinks. Constructed wetlands (iii) are entirely created wetlands. As restoration typically requires some sort of construction, (ii) and (iii) are not clearly distinguished in the literature and will be considered together in the following.

Constructed wetlands are widely used for mitigation of urban waste water, urban storm water, industrial waste water, as well as agricultural runoff. In the following we will focus only on wetlands in agricultural settings.

Wetland design varies widely, as showed in Figures 3.3 to 3.5:

- Complexity: from a simple pond to a succession of deep ponds, over-flow passages, shallow ponds and/or subsurface flow paths.
- Shape: Square, elongated or with an irregular shape often adapted to the topography (Figure 3.4). The compactness index (distance between inlet and outlet, divided by the area) is often used for the shape description,
- Hydrological characteristics: Kadlec and Wallace (Kadlec and Wallace 2009) distinguishes 3 types: free water surface wetlands, where water flows above the bed and open water zones exist; horizontal subsurface wetlands (as in Figure 3.5), where water is kept below the surface of the bed and flows horizontally from the inlet to the outlet; and vertical flow wetland, where water is distributed across the surface of a bed and percolates vertically through the plant root zone.

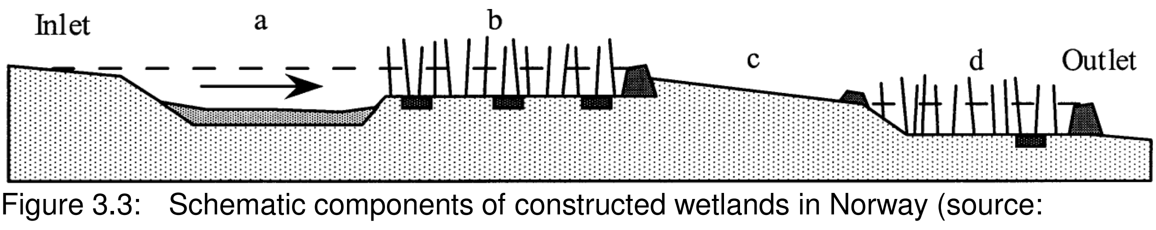


Figure 3.3: Schematic components of constructed wetlands in Norway (source: Braskerud 2002): (a) sedimentation basin, (b) wetland filter, (c) overflow zone and (d) outlet basins; often low dams separate the components.

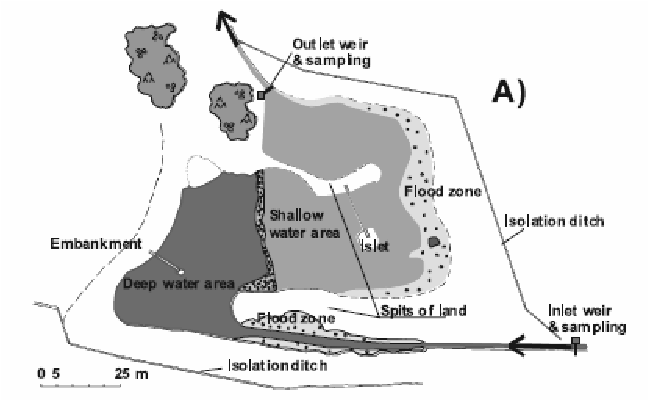


Figure 3.4: Heterogenic wetland in Hovi, Finland (source: Koskiaho et al. 2003)

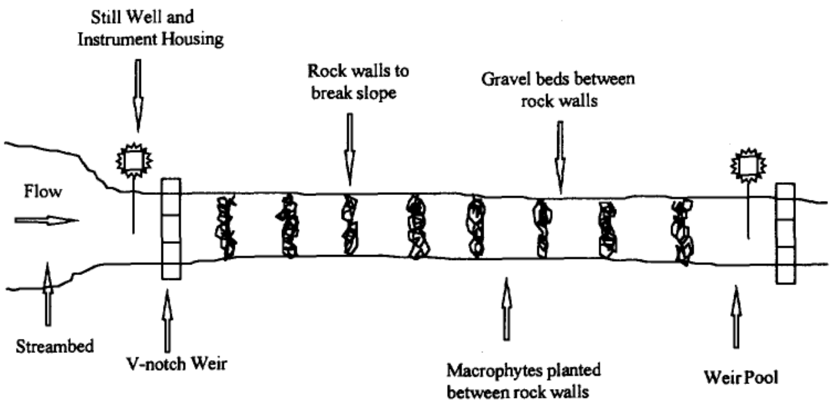


Figure 3.5: Schematic design of a small constructed ditch in Victoria, Australia (source: Raisin et al. 1997).

Based on their design, wetlands can serve different potential mitigation aims. On the one hand, loading of suspended solids is reduced by wetlands as they can act as “sediment traps” (Braskerud 2001). On the other hand, dissolved nutrient reduction relies mostly on biogeochemical cycles that can be particularly efficient in wetlands, as wetlands have higher rates of biological activities than most natural ecosystems (Kadlec and Wallace 2009).

Performance

The values presented in this part are based on an extensive literature review. All following graphs and tables are based on all or a selection of the following references: (Beutel et al. 2009; Blankenberg et al. 2008; Borin and Tocchetto 2007; Braskerud 2002; De Haan et al. 2010; Fink and Mitsch 2004; Jordan et al. 2003; Kao and Wu 2001; Koskiaho et al. 2003; Kovacic et al. 2006; Larson et al. 2000; Mangeot 2009; O'Geen et al. 2007; Poe et al. 2003; Raisin et al. 1997; Reinhardt et al. 2006; Steidl et al. 2009; Tanner et al. 2003; Tanner et al. 2005)

Wetland performance in removing total nitrogen (TN) varies strongly among published case studies (Figure 3.6). While in some experiments nearly all nitrogen input is removed (97 % in study by Larson et al. (2000)), net release of nitrogen was observed in some studies (Jordan et al. (2003), during the second experimental year). Whereas removal fractions of different forms of nitrogen are in the same range, a comparison of results from several studies show tendencies of wetlands to release NH_4^+ and highest efficiency in retaining organic forms of nitrogen (Figure 3.6, Table 3.1).

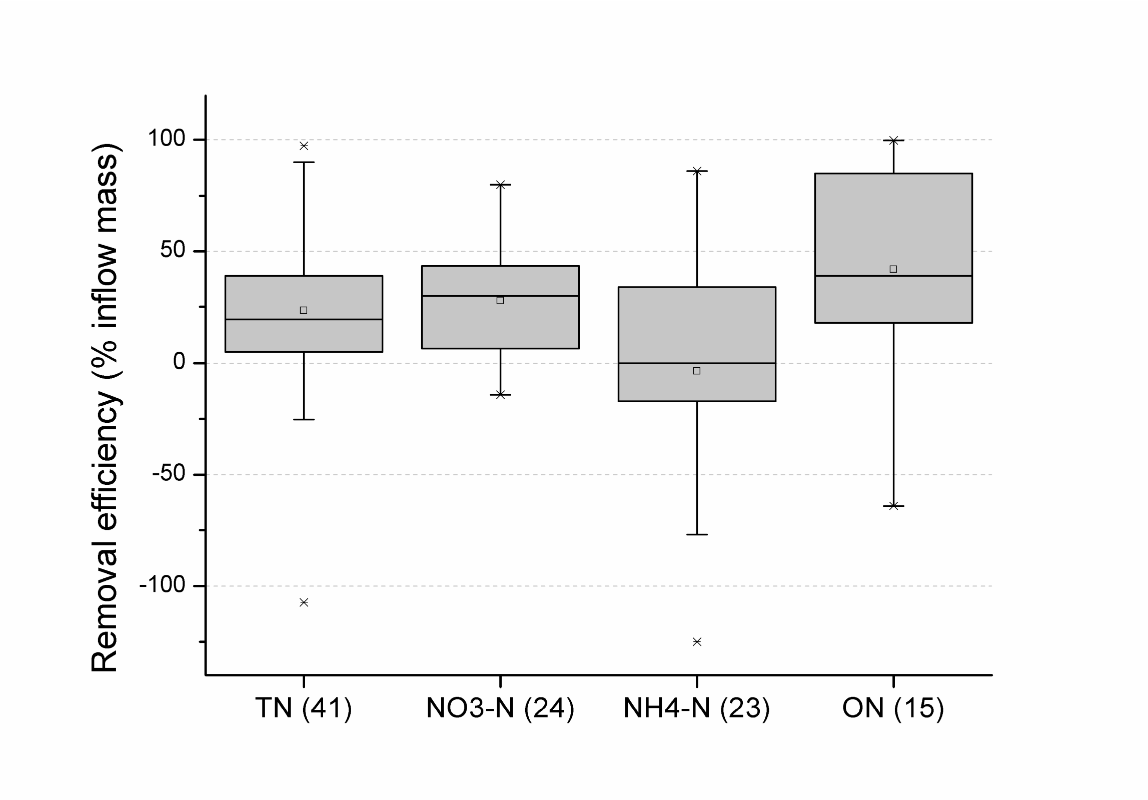


Figure 3.6: Measured removal fractions for total nitrogen (TN), NO_3^--N, NH_4^+-N and organic nitrogen (ON). The box plots represent 41 "wetland years" from upstream wetlands in temperate regions of northern Europe, USA, Australia, and New-Zealand, based on review of published case studies (see references above)

It is interesting to see whether different design actually leads to higher removal of nitrogen. Unfortunately, most studies in agricultural settings are for surface flow wetlands. Nevertheless, we compare them in Figure 3.7 to the performance with four systems with infiltration design studied by De Haan et al. (2010) and Larson et al. (2000). The comparison indicates no significant difference between the two samples ($p = 0.3$). However, De Haan et al. (2010) tested a surface flow wetland, an infiltration wetland and an infiltration wetland filled with straw as a carbon source and found TN removal of 453, 396 and 992 mg-N m^{-2} d^{-1}, respectively. Relatively low removal value in the simple infiltration wetland were explained by a mal-functioning, as infiltration layers remained aerobic. The single case study underlines the potential of infiltration coupled with a carbon source. A similar qualitative trend is described by Blankenberg et al. (2008) who compared performance of parallel test channels: highest retention among all systems was observed in 3 organic filters (shallow wetlands, with submerged vegetation or filled with straw).

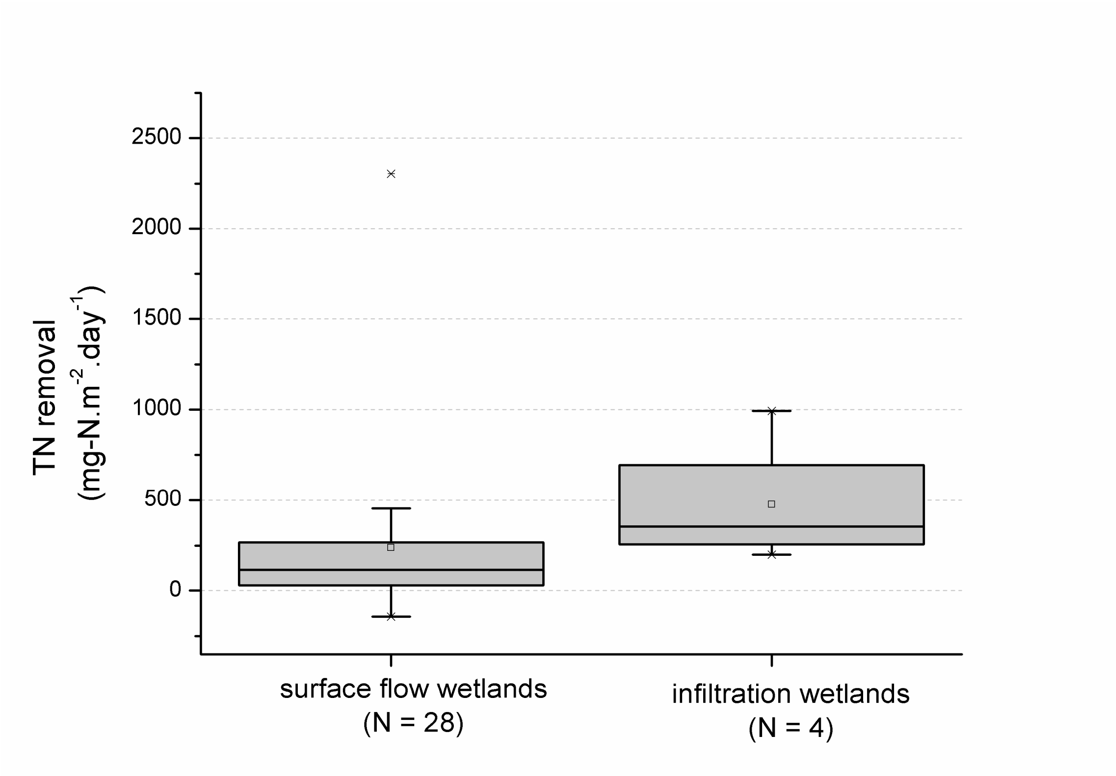


Figure 3.7: TN removal for surface and sub-surface flow wetlands (see references above)

Apart from wetland design performance against nitrogen depends on a various variables:

Residence time: Kao and Wu (2001) observed a significant amount of pollutant removal whereas Raisin et al. (1997) observed release when residence times were reduced to the order of hours. Low removal during high flow was also found for a case study in Brittany, where NO_3^- retention was reduced from ~70 % at an average residence time of 4 days to ~0 % after an intense rain event (Mangeot 2009). If we plot NO_3^- retention fractions against water residence time for all the studies found in the literature review, values are highly scattered (Figure 3.8a). Nevertheless a general tendency towards low retention or even leaching can be seen for very low residence times.

Seasonality: NO_3^- retention rate is typically higher in summer than in fall in winter (Beutel et al. 2009; Koskiaho et al. 2003; Poe et al. 2003); this can be explained by different temperatures and the influence of vegetation.

Inflow concentration: As for other denitrifying systems (e.g., section 2) higher N removal is expected at higher NO_3^- concentrations at the inflow. The same tendency is found for reviewed literature, though scatter is again high (Figure 3.8b)

Wetland construction: Soon after construction, high uptake from first plants may result in overestimation of long-term wetland removal rate (Kadlec and Wallace 2009).

Wetlands can also retain other contaminants than nitrogen (Table 3.1). Total phosphorus retention in wetland varies widely, from 1% (Ulén et al. 2004) to 88% (Higgins et al. 1993) and phosphate retention from -19% (Koskiaho et al. 2003) to 89% (Ydstebø et al. 2000). Release of stored phosphorus is particularly expected if redox conditions change (e.g., during change from dry to wet conditions) and in older wetlands, where P balance reached equilibrium (Reinhardt et al. 2005; Steidl et al. 2009).

Pesticide retention fractions vary from 0- 57% measured for Metalaxyl (Blankenberg et al. 2007) to 87- 91% of Metolachlor (Moore et al. 2001). However, variations of retention fractions are observed for different years: Metamitron removal rate was 7% in 2001 and 58% in 2000 (Braskerud and Haarstad 2003).

In addition to pollutant removal, wetlands can have a number of benefits, including flood and erosion control, groundwater recharge, as well as increase in biological diversity.

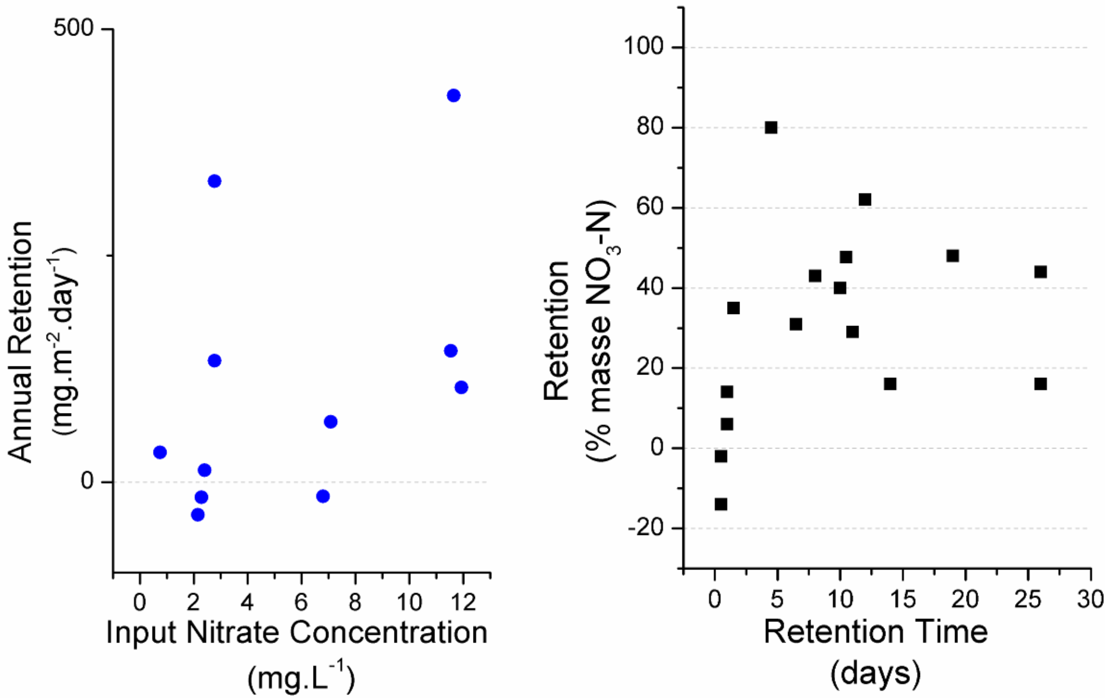


Figure 3.8: Published NO_3^- retention versus residence time and NO_3^- concentration at the inflow (see references above)

Table 3.1: Areal removal rates for different nutrient parameters, based on a review of published case studies [1]

Parameter	Mean	Standard Deviation	Minimum	Maximum	N total
		[mg m^{-2} $year^{-1}$]			
Total Nitrogen	232.5	438.3	-142.4	2303.5	29
Nitrite+Nitrate-N	185.4	502.6	-123.2	2493.0	25
NH4-N	74.2	282.5	-17.0	1342.4	23
Organic Nitrogen	276.1	525.7	-4.4	1919.9	13
Total Suspended Solids	9816.7	13955.3	-492.8	38714.7	11
Total Organic Carbon	44.0	62.2	0.5	139.6	6
Total Phosphorus	78.5	204.9	-13.7	1150.6	32
Organic Phosphorus	1.7	1.5	-0.4	3.8	6
Phosphate-P	-0.5	2.6	-8.8	1.1	13

[1] see references above

Boundary conditions

Nitrogen is removed mainly by nitrification-denitrification processes (see section 1.2). As carbon is needed for the growth of denitrifying bacteria, organic carbon sources within the wetland provide better conditions for nitrogen removal. Therefore, old and vegetated wetlands are found to be more effective against NO_3^- (Mitsch and Bouchard 1998). However, release of organic material from decaying plants may considerably alter this effectiveness (Reinhardt et al. 2006; Steidl et al. 2009).

Large exchange surfaces between plants, water and different substrates improve development of aerobic and anaerobic microbial populations. Thus open surface areas and vegetated areas are often combined in wetlands in order to enhance treatment efficiency (Koskiaho et al. 2003).

Wetlands treating agricultural runoff must be adapted for extremely varying flow conditions, as the loads depend on rain and irrigation. As high NO_3^- removal fractions are favored by high water residence time (Chavan et al. 2008), wetland area is an important design criteria. Kovacic et al. (2000) proposed a wetland area pro catchment area ratio of 5% for 46% mass NO_3^- retention, whereas according to (Hey et al. 1994b), water quality enhancement would be substantial with a 2% ratio. Reinhardt et al. (2006) suggest a residence time above 7 days to reach high nitrogen removal in surface flow wetlands.

As flow paths within the wetland and removal rates will depend on site and upstream watershed, previous studies are essential for planning (pollutant loadings, hydrology, soil characteristics, topography, land use and drainage systems).

As wetland ecosystems are strongly dependent on inflow characteristics, dramatic changes in these characteristics can significantly alter their mitigation ability. Therefore, the wetland must only receive agricultural runoff and post-treatment point pollution. Moreover, degradation of wetlands can lead to a decrease in its ability to remove pollution and even to the release of nutrients (EPA 2005).

In the case of waters with high loads of suspended solids, a sedimentation pond built upstream of the wetland can prevent clogging (for subsurface wetland) and land accretion/loss in storage volume (surface wetlands).

Wetland long-term management is needed to ensure continued success (EPA 2005). Particularly for long term removal of phosphorus, removal of sediments is required when wetland starts to leach phosphorus (Reinhardt et al. 2005; Steidl et al. 2009). Moreover it is important to identify changes in watershed (land use, hydrology and water quality), to prevent erosion and sedimentation and to controll exotic species.

Cost

Compared with constructed wetland for urban or industrial sewage treatments, wetlands for treatment of agricultural drainage water generally require lower initial investments and maintenance. Nevertheless, wetland restoration or construction costs vary strongly depending on the complexity, used materials and construction site. In order to embrace as many possibilities as possible, we will give three different examples:

1. Restoration of seasonal marsh, area of 4047 m^2 and 0.6 m depth; excavation of basin and swales for entry and exit: cost US $ 9 000 (~6 700 €), which corresponds to an average price of 1.66 € m^{-2} (Zentner et al. 2003)

2. A constructed surface floe wetland, 200 m^2 with an average depth of 1m. Excavation, inflow and outflow pipes, géomembrane: 10 300 € (= 51.5 € m^{-2}) (pers. Comm. H. Rustige)

3. An infiltration wetland, 50*2m, filled with gravel covering a drain pipe. Excavation, géomembrane, gravel bed, drainage pipes and inflow and overflow pipes: 12 200 € (= 135 € m^{-2}) (pers. Comm. H. Rustige),

Those costs do not include monitoring needs, planting, land acquisition, studies before construction and long-term management.

Other studies report lower prices, which may be caused by the greater scale of the constructions. Building costs reported by Söderqvist (2002) vary between US $ 0.04 (~0.03 €) and US $ 10.12 (7.6 €) per square meter of wetland.

Those costs may be balanced partly by monetary benefits. EPA (2005) reports that natural products can be harvested from wetland areas, such as fish, shellfish, blueberries, timber, wild rice, medicines derived from wetland soil and plants. The catch of wetland-dependent species of fish is valued at US $ 15 billion per year in all USA (EPA 2005). Recreational value of wetland can be added too, for hunting, birdwatching or photographing wildlife. However, the above monetary benefits may be highly site-specific.

Chapter 4
In-river mitigation systems

4.1 Stream restoration

Approach of the method

Human activities in watersheds and directly on streams lead to a number of hydrological perturbations, such as river incision. Moreover, pulsed hydrology often observed in those streams and degraded physical habitat are associated with water quality degradation (Shields Jr et al. 2010). Most stream restoration projects are not focused on chemical water quality improvement, but are made with other purposes: bank stabilization, bank erosion control, flood protection and enhancement of aquatic habitat quality (Kondolf 1994; Sear 1994). As restoration projects vary greatly in goals, scale and budget, techniques used are diverse and range from adding boulders or cobbles to the stream bed, removing dams, constructing deflectors and riffles to remeandring channel.

Stream restoration programs may affect water quality since (i) they encourage vegetation on the banks and in the channel, (ii) river flow is slowed down and flush events are buffered and (iii) the stream is reconnected with the water table.

(i) *Vegetation*

- Plants assimilate nutrients from the water, which may lead to 15% of NO_3^- loss from the river water according to Fennessy and Cronk (1997).
- Plant development enables accumulation of organic matter that is favorable to the establishment of denitrifying bacteria (Gift et al. 2010).
- Finally, trees adjacent to streams supply coarse woody debris that lead to a more diverse stream channel morphology (Harmon 1986).

(ii) *Hydrological buffering*

- Firstly, the creation of very slow to stagnant zones allows sedimentation. Moreover, such zones may foster denitrification at the interface between low oxygen sediments and high NO_3^- loads from the streams.
- Most restoration projects aim at buffering high flow events. As elevated level of nutrients are often observed in streams with flash flood hydrology (Norton and Fisher 2000; Pionke et al. 2000), positive effects on water quality can be expected from increased water retention during flow events.

(iii) *Groundwater connectivity*

- The reconnection between the stream and its water table enhances transfers of dissolved solutes between surface and subsurface water, which can lead to an increasing in denitrification rates (Kasahara et al. 2006).

Performance

There is a need for research regarding the quantification of the actual impact of stream restoration on denitrification rates and NO_3^- uptakes (Bernhardt et al. 2005; Klocker et al.

2009). Only few studies aim at effects of stream restoration projects on water quality, probably because of the following difficulties in evaluating their performance:

- Comparison between restored and non-restored streams is difficult since each stream has a individual characteristics; comparison of situation before and after restoration in one stream is complex since hydrology and nutrient loads vary greatly between years and other changes in the catchment may occur during observation period.
- Although many projects are undertaken, few involve adequate monitoring before and after the restoration works.
- Subsurface hydrologic flow paths, which may greatly influence denitrification processes (Böhlke et al. 2004; Kaushal et al. 2008; Mulholland et al. 2008), are difficult to evaluate.
- Moreover, comparison of denitrification rates under different restoration designs and hydrological conditions is likely to reveal a high variability of effectiveness (Klocker et al. 2009).

Total NO_3^- retention values have been calculated from isotopic measure in small to medium stream reaches in the USA: they range from 1.1 mg-N.m^{-2} h^{-1} to 3.1 mg-N $m^{-2}h^{-1}$. Kaushal et al.(2008) indicate that the denitrification potential is higher in a restored reach of a river than in a non-restored reach, with 77.4 ±12.6 µg N (kg sediments)$^{-1}$ d^{-1} and 34.8 ± 8.0 µg N kg^{-1} d^{-1}, respectively. In contrast, Velinsky et al. (2006) observed very few differences in stream water chemistry after a small dam was removed.

Other benefits of stream restoration projects, such as habitat quality, sediment trapping and flow management, are more widely studied.

Boundary conditions

As stream restoration works lead to re-suspension of sediments, they can lower water quality temporarily. Moreover, sediment deposits could in turn change hydrology of downstream channel.

Stream restoration can lead to creation of very fast and aerated paths, for example little cascades between ponds. Other projects that consist in addition of cobbles and boulders, also aim at enhancing the streambed habitat by creating oxygen rich zones. However, higher oxygen may lead to a lowering of NO_3^- retention, because denitrification occurs only under low oxygen conditions. However, in some experiments, fast clogging was observed in sediment and nutrient rich streams, which led to recreation of anaerobic zones (Kasahara et al. 2006).

High flow events greatly reduce N retention in streams. Therefore stormwater management strategies that increase hydraulic residence time in watersheds should be a focal point in stream restoration to improve conditions for N reduction (Booth 2005; Kaushal et al. 2008; Walsh et al. 2005).

Interactions between water table and stream bed are favorable for nutrient retention (see above). Therefore, restoration projects should consider riparian water table (Gift et al. 2010). Given the connections between riparian water table and stream water quality, an intact riparian zone is an essential condition for good nutrient retention in the stream (Fennessy and Cronk 1997).

Cost

Goals and techniques used for restoration vary widely between projects; this is also reflected in the costs. Figure 4.1 is based on the experiences of 37 099 projects in the USA, including some that do not enter our “stream restoration” category. As most of the projects in this database do not include monitoring or assessment (Bernhardt et al. 2005), outcomes of the projects are unknown.

MEDIAN COSTS FOR GOAL CATEGORIES

NRRSS goal category	Median cost	Examples of common restoration activities
Aesthetics/recreation/education (A/R/E)	$63,000	Cleaning (e.g., trash removal)
Bank stabilization (BS)	$42,000	Revegetation, bank grading
Channel reconfiguration (CR)	$120,000	Bank or channel reshaping
Dam removal/retrofit (DR/R)	$98,000	Revegetation
Fish passage (FP)	$30,000	Fish ladders installed
Floodplain reconnection (FR)	$207,000	Bank or channel reshaping
Flow modification (FM)	$198,000	Flow regime enhancement
Instream habitat improvement (IHI)	$20,000	Boulders/woody debris added
Instream species management (ISM)	$77,000	Native species reintroduction
Land acquisition (LA)	$812,000	
Riparian management (RM)	$15,000	Livestock exclusion
Stormwater management (SM)	$180,000	Wetland construction
Water quality management (WQM)	$19,000	Riparian buffer creation/maintenance

Figure 4.1: Costs for goal categories, median from river restoration projects in the US, (Bernhardt et al. 2005)

4.2 River wetlands

4.2.1 In-stream wetlands

Approach of the method

Once nutrient loads reach streams, they can directly affect the aquatic environment. Compared to wetlands, conditions for removal of NO_3^- of phosphorus, either by sediment deposition or denitrification, are less favorable in streams, where flow speed and oxygen concentration are high (compare section 3.2). To increase stream “self-purification”, utilization of in-stream wetlands (ISW) has been proposed as mitigation measure against diffuse pollution by some authors (Gilliam et al. 1997; Stone et al. 2003).

ISW can be created simply by impounding or adding a control structure to the stream (Gilliam et al. 1997; Stone et al. 2003), for example by simply reinforcing a beaver dam (Hunt et al. 1999; Stone et al. 2003). The wetland will then occupy the valley bottom (Figure 4.2).

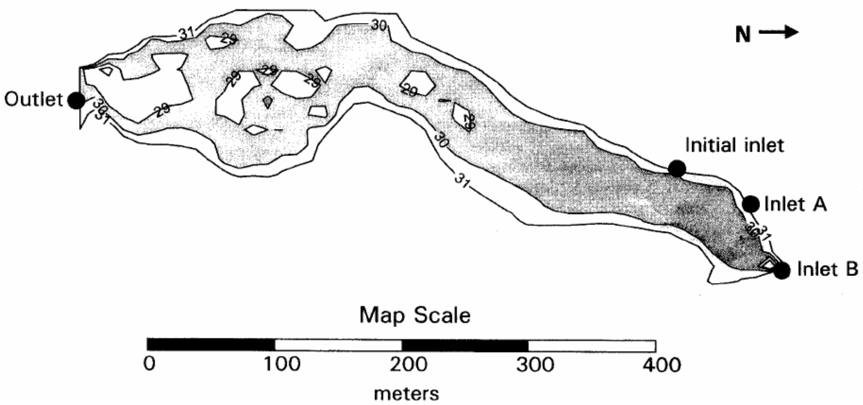


Figure 4.2: Contours of an in-stream wetland in North Carolina (Hunt et al. 1999; Stone et al. 2003)

Performance

Whereas reduction of nitrogen in a free-flowing stream is very low (Haggard et al. 2001; Jansson et al. 1994), it has been shown that this reduction could rise to approximately 50% after the construction of an in-stream retention pond or wetland (Hunt et al. 1999; Jansson et al. 1994). Reduction in TN concentration through an ISW in North Carolina (Figure 4.2) was 54%, whereas 71% of the NO_3^- was removed and NH_4^+ was slightly released (Hunt et al. 1999; Stone et al. 2003).

Mitsch and Cronk measured a removal of 63 to 96% of phosphorus concentration in an ISW (Mitsch and Cronk 1992)

Finally, similar to wetlands in section 3.2, ISW provide habitats for aquatic and non-aquatic species, and can be considered as landscape improvement. However, ISW can also block passage for migratory fish species.

Boundary conditions

According to (Mitsch and Cronk 1992), ISW creation is a reasonable alternative only in low-order streams. Gilliam et al. (1997) suggest locating them in the upper reaches of the watershed, along first- and second-order streams, to guarantee their efficiency.

Construction or rehabilitation of ISW is subject to the local framework on water policy. A particular issue may be the fish continuity of streams, which is blocked if streams are dammed over the entire cross section without measures for fish passage.

Prior to construction, topographic survey should be done to assess the future wetland extension.

As for upland wetlands (section 3.2), wetland performance in removing diffuse pollution is expected to vary greatly. During large flow events, ISW can release accumulated pollutants (Gilliam et al. 1997). Particularly for phosphorus, sediments would have to be removed on regular intervals.

Cost

The construction needs for ISW creation or restoration are very low: in the case study of Hunt, Stone et al.(Hunt et al. 1999) it only consisted in repairing a (beaver) dam.

Furthermore planting costs may be reduced to null, if vegetation is left to natural succession. Thus the main cost for this mitigation mean would be land acquisition, which depends highly on local circumstances.

Potential monetary benefits are expected to be similar as in section 3.2.

4.2.2 River diversion wetlands

Approach of the method

While wetlands can be placed directly in-river in low-order streams (section 4.2.1), river diversion wetlands are also applicable to high order streams.

According to Mitsch et al.(2005), “a river diversion wetland is a wetland on the adjacent floodplain or behind artificial levees that receive water by pumping or gravity flow from the main channel of a river and includes such floodplain features as oxbow lakes, backwater swamps and other riparian wetlands.” Example of experimental case studies are represented in Figure 4.3.

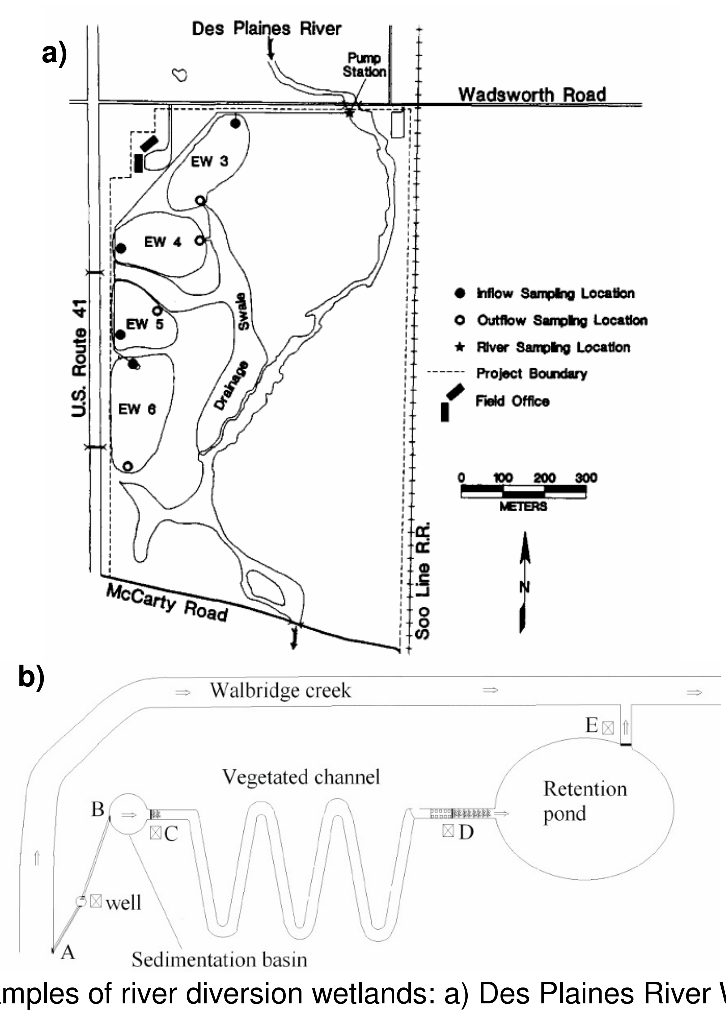
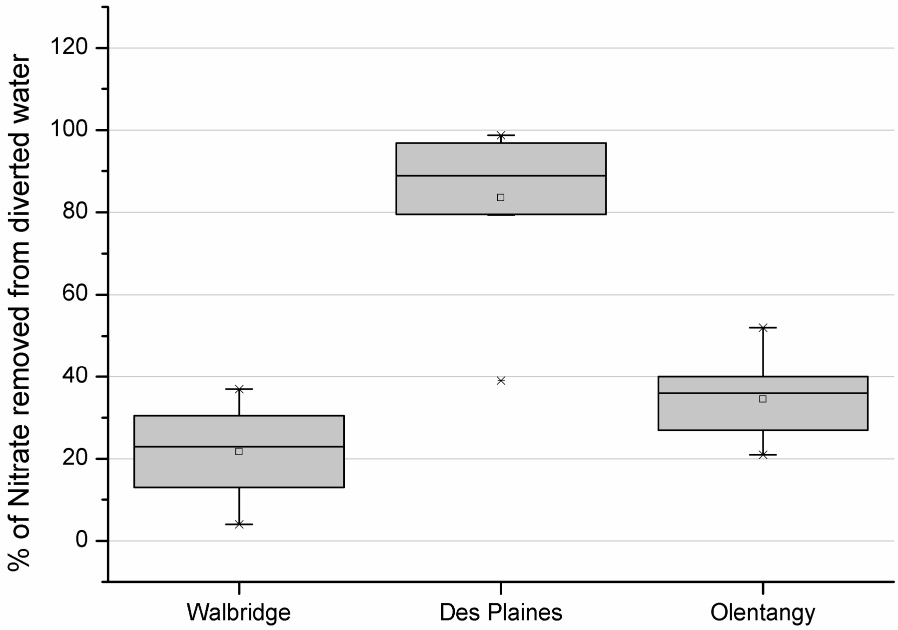


Figure 4.3: Examples of river diversion wetlands: a) Des Plaines River Wetland Demonstration Project in the USA (source: Hey et al. 1994b) and b) Walbridge Creek wetland in Canada (source: Kröger et al. 2007).

Performance

River diversion wetlands have been used for urban river treatment (Jing et al. 2001). However, studies on high order streams that drain mainly agricultural watersheds are rare; therefore we will focus on three available studies. Observed NO_3^- retention fractions of these three case studies are presented in Figure 4.4.

Total P reduction fractions vary from 25% (Kroeger et al. 2007) to 99% (Hey et al. 1994b) and TSS percent removal range from 76 to 99% in Des Plaines experiment (Hey et al. 1994b).



Case study	Walbridge	Des Plaines	Olentangy
Inflow pattern	5% of river flow	Constant flow	Alternating hydrologic pulses, constant flow and flooding
Total Wetland area [m²]	1215	90 000 (sum of the 4 wetlands)	20 000 (2 wetlands)
Mean water loading [m yr^{-1}]	109	2 to 19 (depends on the wetland)	20 to 30
Range of river NO_3^- concentration [mg-N L^{-1}]	3-5	1.2-1.9	1-8

Figure 4.4: NO_3^- retention rates and main characteristics from three case studies (left to right): Walbridge river wetlands, in Québec (Kröger et al. 2007), Des Plaines River Wetlands Demonstration Project, Illinois (Hey et al. 1994b) and Olentangy River, Ohio (Mitsch et al. 2005)

A dependence of removal rates on river concentration has been observed in some studies (Mitsch et al. 2005; Phipps and Crumpton 1994). All the case studies are situated in the northern hemisphere with continental climate. They show high NO_3^- loads during early spring and late fall precipitation events. Whereas the wetlands were nitrogen sinks during those high NO_3^- loading events, they became nitrogen sources during periods of low nitrogen loading (Phipps and Crumpton 1994).

Boundary conditions

Wetland flow pattern can be managed in different ways: i) reach regular flow pattern in wetland independent of river flow (the case of Des Plaines River wetland), ii) divert a percentage of the river flow (Walbridge experiments) or iii) provide flood pulses (part of the Olentangy study).

River diversion wetlands are situated in river floodplain; therefore their hydrologic functioning may be disrupted during flooding, as has been observed in the Walbridge experiment Kröger et al.(2007). Wetland design and monitoring must be adapted to possible flooding events.

River proximity has another consequence in the wetland functioning: as groundwater table is near, seepage outflow from the wetland can lead to a reduction of surface outflow from the wetlands, as has been observed in one of the Des Plaines wetlands (Hey et al. 1994a; Phipps and Crumpton 1994). It can be expected that denitrification could be enhanced by sub-surface passage, but no monitoring of groundwater quality enabled to show this effect.

It has to be noted that two of the examples in Figure 4.4 are very large (> 20 000 m^2), beyond the wetland size, which could typically be implemented in an agricultural landscape.

Boundary conditions described for upstream wetlands in section 3.2 are also applicable to river diversion wetlands.

Cost

To the costs indicated in section 4.2 the implementation and maintenance of a pump or other flow regulating structure that controls water diversion from the river may have to be added (depending on topography), as well as channels between the river and the wetland.

Monetary benefits can be expected to be the same for other types of wetlands.

Chapter 5
Conclusions

The overview of existing approaches in chapters 2 to 4 shows that there are feasible options with the potential to mitigate NO_3^- pollution from drained agricultural land. In general, approaches with high removal potential are :

- controlled drainage (section 2.2),
- bioreactors at the tile level (section 2.3.2),
- reactive swales (section 2.4.2),
- constructed wetlands (section 3.2) and
- river-diversion wetlands (section 4.2.2).

Most practical experience exists for constructed wetlands (globally) and for controlled drainage (mainly in the USA), whereas the other systems are currently at an experimental state.

The other presented approaches are not suggested for high flow events from drained surfaces, based on this literature review for different reasons:

- Removal of artificial drainage (section 2.1) is not seen as a feasible option in agriculturally used catchments for economic reasons.
- Systems that focus on the treatment of groundwater, denitrification walls (section 2.3.1), hedgerows (section 2.4.1) and riparian buffers (section 3.1), cannot tackle high flow events, since they focus on slow, regular flow conditions. Moreover, they are susceptible to preferential flow paths, which are difficult to control and may reduce the catchment-based NO_3^- removal to zero. However, if groundwater flow conditions are regular, denitrification walls or riparian buffers may be efficient measures. For hedgerows there is too little information to judge its potential for reduction of diffuse pollution.
- In-stream wetlands (section 4.2.1) will be difficult to implement from a legal point of view, since they disrupt fish continuity of streams. However, if applicable they can be efficient measures to remove nitrate and phosphorus from streams, supporting similar removal processes as river-diversion wetlands discussed below.
- Finally, stream restoration (section 4.1) seems promising on a theoretical level; however its effect on river water quality cannot be quantified based on existing case studies.

The comparison of the systems with high NO_3^- removal potential is difficult, since (i) they focus on different points on the water pathway between agricultural land and surface water, (ii) units of measured NO_3^- removal are different depending on type of system and finally (iii) each approach is only applicable for specific pre-conditions. Nevertheless we tried to put the different approaches into perspective by applying them to two model plots, one with tile drainage (Figure 5.1a) and one with surface drainage (Figure 5.1b) using the boundary conditions in Table 5.1. The values in Table 5.1 are put together using typical values found in literature on tile-drained agriculture, as well as some local aspects (e.g., precipitation) from a case study in Brittany.

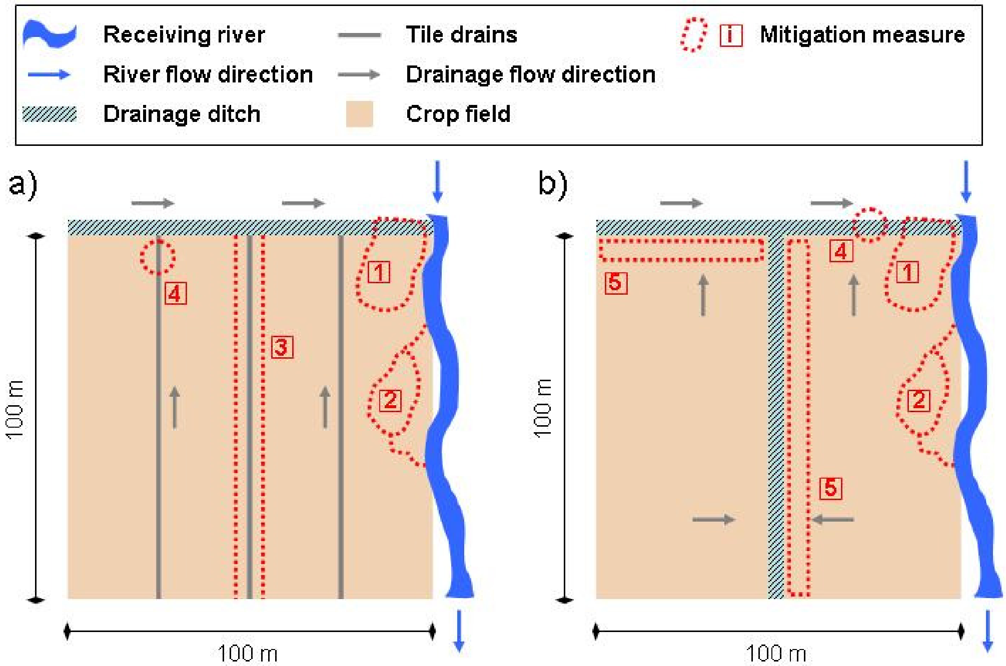


Figure 5.1: Model plots with (a) tile drainage and (b) surface drainage. Mitigation measures are: (1) constructed wetland, (2) river-diversion wetland, (3) woodchip reactor parallel to tiles, (4) controlled drainage and (5) riparian buffer

The simple estimation results in potential NO_3^- removal between 14 and 54 % for the tile drained system and between 28 and 82 % for system drained by surface ditches (Table 5.2). Based on the simple calculation assuming constant denitrification rates, constructed wetlands (# 1a and 1b) show an increase in retention if flow period is longer. As a result, a river diversion wetland (# 2) shows a better performance than a constructed surface flow wetland on a drainage ditch (# 1a), which carries water only during the rain season. First experiments by De Haan et al. (2010) indicate that straw filled constructed wetlands (# 1b) may be more efficient, however since the estimation is based on a single study results have to be treated with care. The cost efficiency for wetlands is clearly highest, if construction cost are low, i.e., if only a dam has to be constructed. In turn, absolute NO_3^- retention is highest in a more complex system with carbon addition.

Table 5.1: Boundary conditions of model plots in Figure 5.1

Parameter	Plot a	Plot b
Annual precipitation [mm]	700	
N applied [kg-N ha^{-1}]	180	
River flow [L s^{-1}]	500	
River NO_3^- concentration [mg-NO_3^- L^{-1}]	50	
Annual water transfer to drainage ditch [mm]	233 (33 %)	140 (20 %)
Annual N loss to drainage [kg-N ha^{-1}]	36 (20 %)	18 (10 %)
NO_3^- concentration in drainage water [mg-NO_3^- L^{-1}]	68	57
Period of drainage flow [months yr^{-1}]	3	4

Mitigation on drainage systems (# 3 and # 4) leads to higher NO_3^- retention in a tile drained system and still to good removal for surface drainage. The installation of denitrification walls parallel to tiles (# 3) is the system with highest NO_3^- retention; despite this performance it is similarly cost efficient as a classical wetland, given the high effort of installing the walls. Controlled drainage (# 4) in turn, is the most cost efficient approach of all the systems in Table 5.2 and thus a very interesting approach for flat drained agricultural areas.

Finally, groundwater treating riparian buffers were added for the surface drainage plot, since they are a very common measure. They do indeed present a cost efficient system if they work according to plan. However, given realistic doubts in their performance (see section 3.1) they may well turn out have little water quality benefits at a very high loss of agricultural surface.

The efficiency of the compared systems may be different for other substances, such as phosphorus, which is mostly transported attached to particles. Given the collected information in this report phosphorus can be expected to be kept back by systems that increase sedimentation (e.g., constructed wetlands) but not by systems aiming at supporting microbiological processes (e.g., by adding carbon sources). Regarding other contaminants, such as pesticides or veterinary products no recommendation can be made given the low number of existing studies. This is probably due to the large number of molecules and the high cost of analysis.

Table 5.2: Estimated NO_3^- retention of selected systems for model plots described in Figure 5.1 and Table 5.1

Nr.	Mitigation system	Related chapter	Number of studies	Performance		Loss of agricultural land	Initial cost *	Annual cost	Cost efficiency for 5 years
			[-]	[kg-N yr^{-1}]	[%]	[%]	[€]	[€ yr^{-1}]	[€ kg-N^{-1}]
a) Tile drained system									
1a	Constructed/restored surface flow wetland [a]	3.2	21	5	14	1	1600-6200	8	65-246
1b	Constructed infiltration wetland with straw [b]	3.2	1	12	34	1	9400	8	153
2	River diversion wetland [c]	4.3	3	8	21	1	6200	8	164
3	Denitrification walls parallel to tiles [d]	2.3.2	1 (5 yr)	19	54	0	13800	0	142
4	Controlled drainage [e]	2.2	14	16	45	0	1400	90	23
b) Surface drained system									
1a	Constructed/restored surface flow wetland [a]	3.2	25	5	28	1	1600-6200	8	65-246
1b	Constructed infiltration wetland with straw [b]	3.2	1	15	82	1	9400	8	128
2	River diversion wetland [c]	4.3	3	8	42	1	6200	8	164
4	Controlled drainage [f]	2.2	14	8	45	0	1400	90	23
5a	Riparian buffer [g]	3.1		0-6	0-35	24	0	192	30-∞
5b	Riparian buffer with denitrification wall [h]	2.3.1	1	0-18	0-100	24	9200	192	113-∞

* excluding land acquisition

a surface flow wetland; A = 100 m^2, 0.5 m depth; calculation: denitrification at 185 mg-N m^{-2} d^{-1} during flow period plus complete NO_3^- retention of last "fill" of wetland after rain season; cost range for damming and excavation; annual cost for crop loss.

b infiltration wetland, filled with straw based on study by De Haan et al. (2010); A = 100 m^2, 0.5 m depth; calculation as for 1a; ; annual cost for crop loss.

c surface flow wetland; A = 100 m^2, 0.5 m depth; calculation: denitrification at 209 mg-N m^{-2} d^{-1} during entire year; diverted flow should be between 0.1 and 0.5 L s^{-1} to reach residence time between 1 and 7 days; cost estimate for excavation, without pump; ; annual cost for crop loss.

d 6 trenches for 3 tiles, total length 600m, 0.6 m wide, 1.5 m deep; 54 % denitrification for total loads assumed (volume-based rates would yield higher percentage, not clear how calculated)

e 3 control structures at each tile (or alternatively control structure on ditch); values by Evans et al. (1991) are used for calculation, given the large data set. Other authors found both higher and lower values (see section 2.2); annual cost for maintenance

f 1 control structure on collecting ditch; values by Evans et al. (1991) who found the same relative effect as for subsurface drainage; annual cost for maintenance

g 4 buffers for 2 ditches and river, 6m wide; comparably low width of 6m assumed, no planting, no fencing; efficiency highly uncertain

h system 5a extended with wood chip denitrification walls; higher efficiency than 5a expected, but preferential flow paths are still likely

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