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Diffuse Pollution Swapping in Arable Agricultural Systems

C. J. Stevens and J. N. Quinton

Department of Environmental Sciences, Lancaster University, Lancaster, LA1 4AU, UK

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Abstract

Pollution swapping occurs when a mitigation option introduced to reduce one pollutant results in an increase in a different pollutant. Although the concept of pollution swapping is widely understood it has received little attention in research and policy design. This study investigated diffuse pollution mitigation options applied in combinable crop systems. They are: cover crops, residue management, no-tillage, riparian buffer zones, contour grass strips and constructed wetlands. A wide range of water and atmospheric pollutants were considered, including nitrogen, phosphorus, carbon and sulphur. It is clear from this investigation that there is no single mitigation option that will reduce all pollutants.

Keywords: Pollution swapping, cover crops, crop residues, buffer zones, no-tillage, constructed wetlands.

Corresponding Author:

Carly Stevens

Department of Environmental Sciences, Lancaster University, Lancaster, LA1 4AU, UK

Tel: 01524 383500

Email: C.J.Stevens@open.ac.uk

I. Introduction

Arable agriculture is considered to be a major contributor to diffuse nitrogen, phosphorus and carbon pollution. Mitigation of diffuse pollution is an important part of the Water Framework Directive but there is a conflict between pollutants. Pollution swapping occurs when a mitigation option or best management practice (BMP) is introduced to reduce loss of one pollutant, but in doing so inadvertently leads to an increase in another pollutant; in effect, one pollutant is 'swapped' for another. Although pollution swapping has been recognised for a number of years, there have been very few attempts to draw together information on the potential for pollution swapping across a range of diffuse pollution mitigation options in agricultural systems.

This study focuses on agricultural systems, specifically on combinable crops. A wide range of BMPs are available to farmers to control diffuse pollution from such crops. Indeed, incentive scheme legislation, such as the Entry Level Stewardship scheme (ELS), encourages or requires farmers to adopt BMPs in order to reduce losses of one or more pollutants. However, little thought is given to the impact of BMPs on losses of other pollutants. This study presents the results of an extensive literature review on a variety of BMPs and pollutants. The review is not intended to be exhaustive, but to highlight trends in published data and to demonstrate the effect of mitigation options on different pollutants.

The mitigation options that have been investigated in this study are: cover crops, residue management, no-tillage, riparian buffer zones, contour grass strips and constructed wetlands. These were chosen because they are widely promoted as being useful for controlling diffuse pollution (e.g. 26, 38, 91, 107, 171, 125). A variety of water and atmospheric pollutants with a wide range of environmental, economic and health effects were considered (table 1).

II. Cover crops

Cover crops (also called catch crops) are used in agricultural systems throughout the world to reduce losses of nutrients through leaching and to protect the soil surface from erosion. Other potential benefits of cover crops have been suggested, including weed suppression, carbon sequestration, integrated pest management (38), the provision of a source of forage in integrated farming systems (143), fixation of nitrogen, improvement of soil structure and reduced surface sealing of the soil (154). There is also evidence that cover crops have an effect on evaporation and infiltration processes because of different patterns of surface cover and crop growth through the season, compared to a conventional crop system (167).

The use of cover crops dates back to ancient agriculture. However, it was not until between 1930 and 1945 that the role of cover crops in reducing nitrate leaching was first studied (104). This has since become the primary function of cover crops in modern European agriculture. In 1991 the EC Nitrates Directive was passed, which required all member states to establish nitrate vulnerable zones (NVZs). Within these areas there is a requirement to establish codes of good agricultural practice that aim to minimise nitrate (NO_3) leaching in order to safeguard drinking water supplies and prevent eutrophication. The Directive suggests that cover crops should be considered as a good agricultural practice where land would otherwise be left bare. In addition, local measures have been taken to encourage the use of cover crops in some countries: for example, the Swedish Parliament has passed a resolution requiring at least 50% of arable land in the south of Sweden to have a winter cover crop (187).

Cover crops are generally sown either in autumn, immediately following harvest, or in the spring, when they are under-sown below the crop. Cover crops can then be incorporated into the soil ready for the next season's crop to be sown. A wide variety of species, including both legumes and non-legumes can be sown as cover crops. The major non-leguminous species used in temperate climates are grasses, particularly rye grass (*Lolium perenne*). Among leguminous species, clover (*Trifolium* sp.) and hairy vetch (*Vicia villosa*) are commonly used. Leguminous species offer the advantage of fixing atmospheric nitrogen in the soil and potentially providing additional nitrogen for the following crop.

Most of the recent research into the effectiveness of cover crops for reducing pollution has focussed on NO_3 leaching. Cover crops reduce NO_3 leaching by intercepting nitrogen that would otherwise be lost from the plant–soil system. During the winter, when the ground is bare and evaporation is low, there is a greater potential for water to move through the soil profile. With no crop to take it up the mineralized nitrogen can be leached from the soil. Cover crops return this mineralized nitrogen to the organic pool, thus reducing leaching losses.

Reductions in NO_3 leaching with cover crops compared to bare fallow have been widely reported in the literature (e.g.193, 184, 156), although results are very mixed. A review of the literature shows reductions in NO_3 leaching ranging from 0 to 98%, with an average annual reduction of 48% (figure 1). This variation in study results is due to differences in soil textures, crop rotation, cover crop species, rainfall during the sampling season, cover crop success rate, fertiliser rate, planting date and whether the cover crop was incorporated into the soil.

Ritter et al. (143) were not able to identify any significant reduction in NO_3 leaching from cover crops. Their study was conducted on loamy sand soils in

Delaware, USA with a rye cover crop and irrigated corn. Ritter et al. (143) identify the importance of planting date and weather conditions in ensuring a good cover crop growth and optimum nitrogen uptake. They point out that cover crops do not fit easily into some crop rotations and it may not be possible to establish good crop cover early enough in the year to prevent NO_3 leaching. Measurements taken over eight years at Gleadthorpe Research Station, Nottinghamshire, UK, Shepherd (156) reported variable success of cover crops. Cover crops were found least effective when drainage started early, allowing NO_3 to leach before the cover crop was established.

Long-term frequent use of cover crops can lead to an increase in mineralizable nitrogen in the soil. Aronson and Torstensson (7) report the results of a seven-year study conducted on sandy loam soils in the south of Sweden. During most of the experimental period significantly less NO_3 was leached from catch crop treatment than from bare fallow. However, towards the end of the experiment a poorly developed catch crop showed significantly higher NO_3 leaching. This demonstrates that if catch crops are no longer used or fail, there is potential for enhanced NO_3 leaching.

Leaching is not the only route by which NO_3 can potentially be lost from cover crop systems. Overland flow provides a second route which is strongly influenced by crop cover. Increased infiltration and evapotranspiration have the potential to reduce the volume of overland flow, but despite this the NO_3 concentration in the overland flow may increase. In a review of the impacts of cover crops on surface water runoff, Sharpley and Smith (152) identify both increases and decreases in the NO_3 concentration of runoff, which they attribute to differing climatic, soil and crop factors. NO_3 concentration in runoff ranged from an increase of 31% with cover crops as opposed to bare fallow, to a reduction of 87% with cover crops. Sharpley and Smith

(152) emphasise the need for flexible management solutions that can account for site-specific factors.

Nitrogen can also be lost from soils as the greenhouse gas nitrous oxide (N_2O). The effect of cover crops on N_2O emissions has received little research attention. Vinther et al. (193) were not able to identify any significant differences in N_2O emissions between four different crop rotations (some including cover crops) on clay soils in Denmark. The N_2O emission was spatially variable and depended on current and previous land use. Some authors have suggested it is possible that cover crops could reduce emissions by reducing the mineral nitrogen accumulated in the soil (40).

Cover crops have traditionally been grown to protect the soil from erosion, and they continue to be used to reduce soil and sediment losses. They reduce runoff volumes by encouraging infiltration, and minimise the area of soil surface exposed to raindrop impact, thereby reducing splash erosion and detachment. Cover crops can also reduce the velocity of overland flow, with the result that it can detach and transport less sediment (38).

Klienman et al. (81) investigated the use of a simultaneous corn and cover crop system on loamy soils in New York. They found significantly less runoff and reduced suspended solids with cover crops than with the control treatment of corn alone. Following the application of dairy manure (50 kg total P ha^{-1}), suspended solids in runoff were measured. After one day the amount of suspended solids in runoff was reduced by an average of 65% with a rye cover crop and 76% with a clover cover crop, compared with the corn only control plots. When manure was applied at a higher rate (100 kg total P ha^{-1}) the reductions were not as great, but they were still apparent: suspended solids were reduced by 33% for rye and 7.3% for clover. These reductions were hypothesised to be due to less ground cover in the control plots.

Langdale et al. (88) reviewed the use of cover crops for reducing soil losses and found significant reductions in erosion. In field trials, cover crops have reduced soil loss by 7–87%, with an average reduction of 52% (figure 1). As for NO_3, the range of values presented here are affected by soil textures, crop rotation, cover crop species, rainfall during the sampling season and cover crop success rate.

As a considerable fraction of phosphorus (P) lost from combinable crop fields is lost as particulate P, there is scope for cover crops to reduce P pollution. In a laboratory experiment on loamy soils, Bechmann et al. (13) found that total P (TP) concentrations in runoff from soils planted with a rye grass cover crop were reduced by 75% compared with bare and manured soil. TP was strongly correlated with the suspended solids in the bare and manured soils, indicating that the majority of the P lost was bound to sediments. P is also lost dissolved in surface waters or in leachate, which is not as effectively reduced (and may even increase) with cover crops.

Sharpley and Smith (154) reviewed studies investigating a range of cover crops and cropping systems. They found that a majority of studies showed reductions in the TP concentration of runoff. Working on a wheat crop system on clay loam soils with ryegrass as a catch crop, Ulén (187) also reported large reductions in TP in runoff (up to 94% in some plots) with a cover crop, compared to bare fallow, although on average particulate P was not reduced by cover crops, but showed similar concentrations to control plots. Staver and Brinsfield (167) reported a reduction in total P concentration in runoff by 44% in a conventionally tilled corn system on silty soils in Maryland. In a no-till system this was increased to 63%. They found that only a small percentage of the P lost from crops was lost in overland flow; this, however, would appear to be in contrast to many cover crop P studies. PO_4-P was significantly higher in runoff from experimental plots with a cover crop than from those with bare ground (187, 154).

In a wheat cropping system in Texas, Sharpley et al. (151) found that a sorghum cover crop reduced soil loss and associated particulate phosphorus, but dissolved and bioavailable phosphorus were greater with a cover crop. They speculated that this might be due to the contribution of P from vegetative material, which can potentially be an important source in runoff. Sharpley (153) found that in the absence of fertiliser application, up to 90% of soluble P found in runoff can be from cover crops. This can be especially important when cells are lysed by freezing and thawing, or by senescence. Furthermore, by reducing erosion there may be an increase in the surface P status of the soil (151), particularly in no-till systems (section IV).

Other pollutants have received little or no attention with regard to cover crops. Vinter et al. (193) found no difference in CO_2 emissions between cover cropped systems and bare fallow. Other gaseous pollutants such as methane and hydrogen sulphide have not been investigated; however, as cover crops do not cause waterlogging of the soil there is unlikely to be any effect. It has been suggested that cover crops provide an opportunity to sequester carbon in the soil (38), as they increase the soil carbon content through the incorporation of the cover crop. There is also the potential for cover crops to increase the dissolved carbon organic content of runoff owing to greater microbial activity, and this area deserves some research effort. Similarly, the impact of cover crops on pesticide pollution has received very little attention, although the potential for reduction in overland flow with cover crops could reduce the mass of pesticide transported.

Cover crops – Summary (figure 1)

- Cover crops generally reduce NO_3 leaching when they are successfully established, although long-term use may lead to a flush of NO_3 when cover cropping is ceased. NO_3 in overland flow may be increased or decreased. There are no changes apparent in N_2O emissions.

- Cover crops generally reduce soil losses, as compared to bare fallow.
- Losses of particulate P are generally reduced in runoff, but losses of dissolved reactive P may be increased by cover crops. This form of P is biologically available.
- There is no difference in CO_2 emissions between cover crops and bare fallow.

III. Crop residues

Crop residues are either removed or left on the soil surface. Retained residues can be used as mulch and to control wind (e.g. 109) and water erosion. Residues which are incorporated into the soil improve soil condition and infiltration, but they are less effective at reducing erosion than residues left on the soil surface (107). Residues left on the soil surface protect the surface from sealing and crusting, so increasing the potential for infiltration; they also increase surface roughness and create small diversions and retention reservoirs, slowing runoff velocity. Working on a clay loam soil, Myers and Wagger (120) did not find any increase in infiltration with residues but they did find significant reductions in the amount of sediment in overland flow. This could be due to reduced splash erosion (32, 107, 150) or a reduction in the velocity of overland flow reducing entrainment.

A review of the literature (figure 2) gives an average reduction in soil loss of 78%, with a range of 40–100% from the use of crop residues. A number of studies have found that greater reductions in sediment loss occur at higher levels (>40% cover) of residue surface cover (e.g. 56, 103, 108, 118). However, even small amounts of residue (12% cover (108)) have the potential to greatly reduce soil losses, and mulching rates that are sufficiently low to not have an adverse effect on early growth and crop yields will reduce erosion (107,108, 111).

Residues from different crops vary in their effectiveness at reducing erosion and overland flow: some will decompose before high-risk periods for soil erosion are over (107), with incorporated residues generally decomposing more quickly (146). Brown *at al.* (31) found that the size of the residue was not important in relation to how much it reduced erosion.

As crop residues decay they release nutrients which may be lost in surface runoff or by leaching (152). Manipulating nitrogen usage with crop residues is a complex process determined by the relative timings of N immobilisation, mineralization and plant uptake. Perhaps as a result of this, the use of crop residues has only been moderately successful for reducing N leaching. Thomas and Christensen (181) conducted a lysimeter study on sandy loam soils. They found NO_3 losses were not significantly different when rye and barley residues were left on the soil surface, although the results indicated a slight reduction in leaching. Short-term reductions in NO_3 losses were identified but these were balanced by later increases. Rainfall simulator studies have also shown an increase in NO_3 and ammonia (NH_3) in leachate with an increasing residue application rate. Leachate (from simulated rainfall) was collected from corn residues with field loading rates of 5, 7, 10 and 15 tons ha^{-1}. Volume-weighted nutrient concentrations of NO_3 and ammonia increased by 16% (NO_3) and 41% (ammonia) between the lowest and highest residue application rates (147). Similar results were identified by Stenberg et al. (168). Deeper incorporation and the use of finely ground residue leads to greater N immobilisation and a reduced risk of leaching (4).

Despite increased N leaching, crop residues have generally been successful in reducing N in overland flow. Mostaghimi et al. (118) found that 750 kg ha^{-1} of rye residue left on the surface (in no-tilled areas) resulted in a reduction of 86% in NO_3, 97% in NH_3, 98% in Kjeldahl N, 97% in sediment total N and 99% in total N when compared to the control. However, higher residue rates of 1500 kg ha^{-1} resulted in a smaller reduction in the total N load in no-till areas compared to the control, and increased loads of NO_3 in conventionally tilled areas. Residues were less successful in conventionally tilled areas, with a 64% reduction in total N compared to the control. In a no-till corn cropping system N losses in overland flow were reduced by 76% on land with 100% residue cover, compared to a control with no residue cover. This reduction can be attributed to smaller volumes of overland flow (182). When rainfall simulations were carried out on a dry soil, Torbert et al. (183) found that initiation of surface runoff was delayed and loss of nutrients was reduced by surface-spread corn residue. There was a 97% reduction in N content of surface runoff on dry soil and 95% reduction on a wet soil.

Soil moisture is higher in areas where surface residues have been applied than in areas left bare, and this creates conditions that are more conducive to N_2O production. This is as a result of reduced evaporation, an increase in the carbon content of the soil and a supply of easily mineralizable N (71, 190). At a residue application rate of 8 tons ha^{-1} of wheat (left on the surface), N_2O emissions increased from 1.6 kg N_2O ha^{-1} (in the control) to 2.8 kg N_2O ha^{-1}. At the very high residue application rate of 16 tons ha^{-1}, N_2O emissions were to 3.5 kg N_2O ha^{-1} (71). The amount of N_2O

emission depends on the quantity and quality (C:N ratio) of the residue; highest emissions are found after the incorporation of residues with a low C:N ratio (9, 191).

There has been relatively little research into P in relation to crop residues. As with N, there is potential for an increase in P losses as P in the residues is mineralized. In an incubation experiment, Sharley and Smith (154) found that significantly greater amounts of P were leached from surface-applied residues than from incorporated residues. Greater leaching losses of P have also been found in field trials using a rainfall simulator. Working on a clay soil with corn residue applications of 5, 7, 10 and 15 tons ha^{-1}, Schreiber (147) found that the concentration of P in leachate increased with increasing residue cover. This was explained by a greater contact time between the rainfall and the crop residue as flow rates of overland flow are reduced, allowing more time for P to leach from residues.

Crop residues have been much more successful in reducing P losses in overland flow than in leachate. Both Torbert et al. (183) and Andraski et al. (5) found that PO_4 losses in surface runoff were reduced with corn residue. Torbert et al. (183) found a seven-fold reduction. These reductions have been attributed to increased infiltration and reduced sediment losses (5).

Crop residues increase the C content of soils. This has a number of important advantages in improving soil structure and moisture retention, and increasing C sequestration. It also causes some potential concerns regarding the loss of C in leachate and overland flow, or by gaseous emissions. Losses of total organic carbon in leachate were found to increase with the amount of corn residue applied (147). However, Tiscareno-Lopez et al. (182) found that total organic matter in runoff was reduced by 85% as corn residue cover was increased from zero to 100%. The reduction in carbon lost was primarily due to a reduction in the volume of runoff produced.

Emissions of CO_2 increase significantly with residue application, although the amount of CO_2 emitted can depend on the type of residue and the soil. Velthof et al. (191) investigated CO_2 emissions from a range of residues on both sandy and clay soils. The combinable crops investigated (barley, wheat and maize, both fertilised and unfertilised) all gave very similar results, with an average increase of 60% on sandy soils and an average increase of 47% on clay soils. Govaets et al. (57) found very different results with wheat and maize residues on a coarse sandy clay soil in Mexico. In this no-till system, CO_2 production was lower when residue was incorporated than when it was removed.

Soils frequently have both positive (emission) and negative (consumption) fluxes of CH_4; with crop residues negative fluxes are more frequent and there is generally a larger CH_4 consumption where residues are retained rather than being removed (70). Jacinthe and Lal (70) identified a weak (not significant) trend for increasing CH_4 emissions with increasing residue cover, suggesting that this is an area requiring further investigation.

As crop residues reduce runoff and sediment losses they also have a great potential to reduce losses of pesticides. Concentrations of pesticides in overland flow are only reduced by a small amount in runoff. However, the reductions in runoff quantity results in reductions in pesticide losses. In a laboratory rainfall simulation of overland flow concentrations of atrazine and metolachlor, Smith et al. (164) found a significant reduction with 30% residue cover one and eight days after application. Myers et al. (121) conducted a field trial with mowed corn stover left on the surface, and found an 11% reduction in pesticide loss.

The other pollutants being considered in this study have not been investigated in relation to crop residues, although, as with N_2O, there is a possibility of increased H_2S

release due to wetter soils. Pathogen movement is also likely to be reduced due to increased infiltration and reduced overland flow volumes. Both are areas requiring further investigation.

Crop residues – Summary (figure 2)

- Crop residues are very successful in reducing sediment losses, even at low cover.
- N, P and C in runoff are also reduced. However, losses in leaching may increase.
- Gaseous emissions of N_2O and CO_2 can increase with crop residues; the pattern for CH_4 is less clear.
- Losses of the pesticides atrazine and metolachlor can be reduced using crop residues.

IV. No-tillage

No-tillage (NT) is often promoted as a way of reducing diffuse pollution, particularly soil erosion, and as a means of sequestering carbon. In this review we define NT as a system where the soil surface has not been disturbed prior to seeding and where crop residues are left on the soil surface. Conventional tillage (CT) is defined here as a system which inverts the soil using a mouldboard plough. There are a number of other forms of tillage which lie between these two extremes, but for the purpose of this review we will contrast NT with CT. NT is generally accepted as being beneficial to the physical condition of the soil. Soils under NT generally have higher organic matter, more stable aggregates (199, 185), lower susceptibility to soil crusting (185), and more soil faunal and microbial activity (58, 185) leading to increased infiltration. As NT does not mix the soil, nutrients and agrochemicals accumulate at the soil surface and concentrations are generally higher in this region than in CT soils (80, 53).

A large number of studies have compared soil erosion rates from NT and CT soils. In a review of 28 studies with plot sizes from 0.13 m² to 750 m² (171, 172), soil loss was changed by between 100 and -100% of that found in the CT treatment, with a mean reduction of 69% (figure 3). This is attributed to the more stable soil structure under NT (66). Typically, the reduction in soil loss is greater than the reduction in overland flow (figure 3); however, there is considerable variation in the results of the studies reviewed. This is in part due to the different scales and measurement techniques employed by the investigators, but it is also to the result of variability in soil response to tillage. This variability is due to inherent soil properties and the antecedent conditions when the tillage takes place. Many of the studies reported were only of short duration, which may make it difficult to realise the benefits of NT, which can take up to four years to become apparent (16).

As much of the phosphorus found in the soil is associated with particles in the silt and clay size fraction (173), the lower sediment losses associated with NT give rise to a lower TP loss than CT. This is despite higher concentrations of P at the soil surface in NT systems (53), leading to higher TP concentrations in overland flow. Dissolved P losses in overland flow are less commonly quoted in the literature, but the studies reviewed (16, 60, 42, 197) all found higher concentrations of dissolved P from NT areas than from CT areas (figure 3). This is because while overland flow volumes may be reduced by NT, the concentrations of dissolved P in the runoff are higher from treatments with less soil disturbance (16, 129, 142). Higher concentrations of dissolved P may lead to higher dissolved P losses from sites with lower runoff (197, 142, 148). In a Brazilian study, dissolved P concentrations in the runoff were five times greater than those from the CT plots on a Hapludox soil subjected to rainfall simulation four years after imposing the treatment. This was a result of P concentrations in the upper 0–0.025cm of the soil being 5.3 times those in the CT treatment (16).

NT leads to increases in the concentration of N in the surface of the soil associated with residue and fertiliser additions (80). NO_3 losses in runoff tend to be small relative to the loss by leaching. However, studies have shown that the proportion of dissolved N and P relative to the total N and P lost is higher in NT systems. A three-year study of nutrient losses in overland flow from CT and NT maize plots on a silt loam soil in Missippi, USA found that solution losses for N and P ranged from 0.6 to 9% from CT and 39.1 to 53.9% from NT plots (102).

It is not clear whether NT encourages leaching losses. Better soil structure encourages infiltration – the converse of reducing surface runoff (figure 3); however, this is not always translated into greater N leaching losses. Concern has been raised about the need to use more N fertiliser in NT systems because of the build-up of organic matter under NT, which leads to increased N immobilisation at the soil surface (100). Randall and Iragavarapu (134), Malhi et al. (98) and McConkey et al. (100) found lower residual NO_3-N content within the profile of NT soils compared to CT soils. Work on an 11-year study on a poorly drained soil in Minnesota, showed that even when drain flows were higher due to NT, NO_3 fluxes through them were 5% lower (134) due to the lower NO_3 concentration from NT (12 mg l^{-1} N-NO_3) compared to CT (13.4 mg l^{-1} N-NO_3).

There is some evidence to suggest that herbicide concentrations in surface runoff are greater from no-till than from more intensive tillage operations. This is due to the accumulation of pesticides at the soil surface and the lack of soil mixing. For herbicide losses to be lower from the NT, surface runoff needs to be reduced to an amount which compensates for the higher concentrations; therefore the literature contains conflicting results. This is reflected in figure 3 which uses results from 65 comparative studies of NT and CT plots under natural rainfall in nine separate studies for five pesticides. It shows that, although the mean is a reduction of 68% in pesticide load, in some cases pesticide losses increased by up to twice those of the CT plot. Other recent literature contains similar contradictions: a nine-year study of pesticide losses from seven small (<1ha) watersheds Shipitalo and Owens (160) found that average herbicide losses from NT watersheds were 1.4 to 3.3 times those from disked watersheds, despite the fact that the NT watersheds generated 1.4 times less runoff. This contrasts with work in Germany (180) which found that NT reduced surface runoff losses of soproturon, metolachlor, and terbuthylazine from large (2.4ha) plots by 30%.

Tillage also has a variable effect on leaching losses of pesticides. Studies conducted at Coschoton, Ohio conclude that there is only likely to be a few percent difference between herbicide leaching losses from CT and NT, even in extreme circumstances, such as heavy rainfall following a herbicide application, although non-adsorbed chemicals are expected to move deeper into the soil due to the better macropore network in NT soils (160). Work at Beltsville MD showed consistently greater concentrations of atrazine in shallow (4m) groundwater under NT plots compared to CT plots. However, these differences were not significant, due to considerable inter-well variability (69).

Few studies have been carried out on the effects of tillage on pathogen transport. Most studies of vertical pathogen losses have been carried out in the laboratory using soil cores. Using this method, Gagliardi and Karns (54) found no significant difference between a no-till and a disturbed (ploughed) treatment. Tyrrel and Quinton (186) have suggested that the transport of microorganisms in overland flow will be closely linked to sediment, indicating that reduced tillage is likely to reduce their transport. The incorporation of manures into the soil reduces losses of presumptive faecal coliforms compared with surface applications (133, 130).

Soil organic matter is generally considered to increase under NT, as crop residues are normally retained. In a literature-based meta-analysis of 56 paired comparisons of organic C stocks under ploughed and NT systems, Puget and Lal (128) found NT had a positive effect on C stocks in 42 of the comparisons and a negative effect in 11 of them. Of these, significant differences were found in 10 of the comparisons where there was a positive effect. Mean sequestration rate was 330 kg C ha^{-1} y^{-1} (95% confidence interval 47–620 kg C ha^{-1} y^{-1}). The increased sequestration of C is likely to be due to increased residue additions with NT, and perhaps lower C oxidisation. No significant difference was found in CO_2 emissions (42.1–81.7 mg C m^{-2} h^{-1} for all treatments) measured by Liu et al. (92) for tillage and nitrogen placement combinations in a long-term continuous corn experiment in Colorado. We could find few studies which compared losses of TOC and DOC in overland flow or drainage from NT and CT soils. Work over a 15-year period in Ohio (123) on six <0.8 ha watersheds found that total C content of sediments passing over a flume from NT (26.1 g C kg^{-1}) and chisel-ploughed (20.7 g C kg^{-1}) watersheds were not significantly different. Mean leaching losses of DOC from 7 x 7 m plots over a seven-year period in Wisconsin (33) were lower (435 kg C ha^{-1}) from NT than from the chisel-ploughed treatment (502 kg C ha^{-1}), but the differences were not significant.

It should also be noted that NT requires lower energy inputs. Lal (87) calculates that CT operations produce 35.3 kg carbon equivalents (CE) ha^{-1} compared to 5.8 kg

CE ha^{-1} for no-tillage systems. Higher emissions of 23 kg C ha^{-1} for NT and 67–72 kg C ha^{-1} depending on crop type for CT are suggested by West and Marland (196). Values for the energy inputs associated with fertilisers, seeds and pesticides are somewhat higher than those from machinery use, ranging between 48 and 202 kg C ha^{-1} for NT and 40 and 156 kg^{-1} for CT (196). In each of the systems considered, total C emissions associated with NT cultivations (71–225 kg C ha^{-1}) are lower than from CT (107-228 kg C ha^{-1}).

Much of the work on the effect of tillage on N_20 emissions is contradictory. Emissions are highly dependent on soil, climate and fertilisation history, as well as on tillage. Arah et al. (6) found that the date of sampling followed by soil and tillage type had a significant effect on N_2O concentrations within the soil. However, they also found that the differences between sites were greater than those between treatments. Work in Argentina found that N_20 losses measured in chambers over a 90-day period were 0.190 kg N ha^{-1} for conventional tillage and 0.350 kg N ha^{-1} for no tillage (124). In Scotland, Ball et al. (11) also found higher emissions of N_2O from NT soils compared with CT soils. These contradictions may be due to variability in soil properties, particularly moisture, or how long the NT treatment has been established. Using 44 data points from studies around the world, Six et al. (162) modelled N_2O emissions after changing from CT to NT and concluded that after 20 years N_2O fluxes would decrease. No information on the influence of no-tillage on H_2S emissions could be found.

There are few studies comparing CH_4 fluxes in CT and NT. In field studies (11, 92) and soil cores (68), NT soils oxidised more CH_4 than CT soils, and the modelling study of Six et al. (162) concludes that there would be a significant enhancement of CH_4 uptake (0.6 kg ha^{-1} y^{-1}) with NT. Field studies in Canada (59) contradict these findings and suggest that, as with N_2O emissions, there is likely to be considerable variation in this response.

No-tillage – Summary (figure 3)

- NT reduces soil erosion and overland flow.
- Overland flow losses of agrochemicals are reduced under NT.
- Carbon sequestration is enhanced by NT.
- There are no clear differences between leaching losses of agrochemicals from NT and CT.
- Gaseous losses of CH_4 and N_2O do not differ between NT and CT.

V. Riparian Buffer Zones

Riparian buffer zones (RBZs) are bands of vegetation located on land down-slope of agricultural fields, bordering surface waters. They are also known as riparian or vegetative filter strips. RBZs aim to provide erosion control and remove nutrients and pesticides from water entering a river or stream (from surface runoff and groundwater) via retardation of flow and consequent deposition of sediment and sediment-bound contaminants, interception by vegetation, adsorption onto plant and soil surfaces, plant uptake, infiltration, dilution with rainfall and microbial processes. RBZs vary in length (distance from edge of buffer to river) and vegetation composition; grasses are commonly used, but buffer zones can consist of other vegetation types, including trees. Although a number of papers have focused on the optimal design for RBZs (e.g. 2, 174), there has been no consensus on this. However, it is clear that buffer zones must be suitably located to be effective and should be designed for the type and quantity of pollution at each location (63). Furthermore, farmers will frequently want to put the minimum area of land necessary out of production in order to protect water quality, so the efficiency of buffer zones should be maximised (97, 46).

The effectiveness of the RBZ depends on many factors, including species of vegetation, soil type, soil texture, subsurface drainage characteristics, temperature slope, barrier length, relative sizes of the filter strips and runoff areas, soil moisture, topography, activities on the cropped land, volume of runoff and the nutrient loading rates (122, 35, 110). Some of the key processes for pollutant removal with the RBZ are bacterially mediated, making them highly dependant on the hydrology of the buffer zone (63).

Nitrogen removal in RBZs can be by denitrification, retention by vegetation, or transformation followed by immobilisation in the soils (35, 61). Of these, denitrification is the most important mechanism, although this is both spatially and temporally variable (35). Partially as a result of this, the effectiveness of RBZs for removing nitrogen from surface runoff shows great variation. Review of the literature shows results ranging from an increase of almost 20% (97) in NO_3 exiting the buffer zone compared to that entering it, to a decrease in nitrogen load of up to 99% (125), with a mean reduction of 35% (figure 4). Making comparisons between different studies is very problematic because of variation in the buffer width, species composition, buffer area to field area ratio, soil type and runoff conditions influencing the ability of the RBZ to remove pollutants. However, the general trend is that in the absence of field drains even narrow buffer zones reduce NO_3 losses.

A number of experiments have demonstrated how successful RBZs can be for NO_3 removal. In Marano, Italy, Borin and Bigon (19) reported a 90% reduction in NO_3, leaving a 5m grass buffer with an additional line of trees. They also found that the zone of influence of the buffer extended beyond its margins, as a result of extensive plant roots systems. Peterjohn and Correll (126) found similarly high rates of nitrogen removal in Maryland, with large reductions in the nitrogen content of overland flow: a 79% reduction in NO_3, 73% reduction in ammonium and 62% reduction in organic nitrogen. Combining results for both surface runoff and groundwater, the RBZ retained 89% of the nitrogen entering the system – much higher than the 8% retained by the same area of cropland. Borin et al. (20) found a 78% reduction in the mass of total nitrogen lost from experimental plots with a 35m buffer zone, compared to those without one. Their results indicated that the amount of nitrogen lost is a factor of the quantity of water leaving the field.

Buffers are not always successful in the removal of nitrogen and can even cause increases in the nitrogen loading. Individual catchment hydrology is critical to the success of RBZs (89), but there also appears to be a relationship with buffer length. A number of studies have considered the effect of RBZ length on nitrogen removal (e.g. 97,106, 174) and it is generally true that a longer buffer zone will be more effective in removing nutrients. Magette et al. (97) found that 9.2m buffer zones were more effective at removing nitrogen than 4.6m buffers on a sandy loam soil. Plots were treated with either 30% urea ammonium NO_3 solution at a rate of 112 kg N ha^{-1}, or broiler litter with a nitrogen content approximately equal to 353 kg N ha^{-1}. For the ammonium nitrate solution, the 9.2m buffer gave an average reduction (compared to the control) of 51%, whereas the 4.6m buffer gave an average increase of 15%. Where broiler litter had been applied the 9.2m plot resulted in a 28% average reduction, whereas the 4.6m plot gave an increase of 20%.

Buffer zones can act on shallow groundwaters through vegetative uptake and by providing carbon for denitrification (165). Haycock and Burt (63) estimated that uptake by microbial biomass or denitrification accounted for 60–70% of the NO_3 reduction in groundwaters in an RBZ. In a review of 10 experimental plots in six studies, Osborne and Kovacic (122) found NO_3 removal from subsurface waters varied between 40 and 100%. Groundwater mediation by RBZs is primarily associated with trees (46), but grasslands also have the potential to remove NO_3 from groundwaters. Osborne and Kovacic (122) found forest buffers were significantly more efficient than grasslands at removal of NO_3 from groundwater. Haycock and Burt (63) found an 82% reduction in NO_3 concentration in waters passing under a floodplain. In a survey of NO_3 losses from sites with and without hardwood buffer zones, the highest NO_3 concentration occurred in areas without RBZs (165).

Despite this success, the removal of NO_3 from groundwater and overland flow by denitrification presents a potential problem. N_2O is an intermediate of denitrification and is an important greenhouse gas. N_2O is an important product of denitrification when NO_3 loading in the buffer zone is high (65). Production is variable in the environment with hotspots of production. Soil type and moisture content are the major control; secondary controls include fertiliser use, carbon source and soil temperature (61, 96). There are much higher levels of N_2O produced in RBZs than field margins, with forested buffers producing seven times more N_2O than grassed ones (65).

RBZs have been widely used to reduce the impact of soil erosion. They decrease the amount of soil entering waterways by reducing flow velocity of overland flow and consequently increasing the deposition of sediment. Buffer zones also increase the surface roughness, further reducing the runoff velocity (174). Review of the literature (figure 4) suggests that RBZs reduce the sediment load in surface runoff between 0 and 99%. The average reduction is 75%, suggesting that RBZs are highly effective in removing sediment from surface runoff, although it should be noted that many of the studies were carried out at the plot scale and RBZs may be less effective in landscapes that encourage flow accumulation. For sediment deposition to occur it is essential that runoff passes slowly through the buffer (44). The area upslope of the buffer is the most important area for deposition, as this is where flow is initially slowed (145). The majority of deposition within the buffer occurs in the upper area (110, 178). Over time the sediment will build up, initially filling depressions and eventually burying vegetation (106). Larger particles are more easily trapped within a RBZ than fine particles (93, 174). The slower settling velocities of fine clay particles mean that they require a greater distance to settle from the flow. Loch et al. (93) compare the settling velocities of 0.02mm diameter particles with 0.002 diameter particles. The former would settle out in 48 seconds in a 20mm deep flow – this is achievable within a 10m buffer on a shallow slope. For the smaller particle size it would require 90 minutes for the particles to be deposited – this is not a feasible retention time in an RBZ. Silt and sand are deposited in RBZs, although fine and medium clay particles are too small and are only deposited when aggregates are formed (177, 178).

Longer buffer zones clearly have the potential to provide greater deposition opportunities for sediment, even under concentrated flow conditions (18). Abu-Zreig et al. (1) found that filter length rapidly increased the proportion of sediment trapped up to a length of 10m; however, after 10m this increase tailed off, giving very little change in the quantities of sediment trapped between 10 and 15m. Vegetation type also has the potential to alter the RBZ's trapping efficiency. Syversen (178) found a forested buffer zone trapped significantly more particles than a grassed one. There have been fewer catchment-based studies, but it is known that buffers do not perform well at trapping sediment in converging landscapes. Here water and sediment is concentrated in valley bottoms before passing across the buffer and into the stream.

P removal is very closely related to sediment removal when the surface runoff has a high particulate concentration (2). Despite this, RBZs are generally less effective at removing P than sediment, potentially because a large fraction of the P is associated with fine clay, resulting in an increase in concentration of P in the sediment that passes across the RBZ (177). Abu-Zreig et al. (2) found that although short buffer zones were good for removing sediment they were less effective for P removal. Review of the literature shows that between 7 and 85% of TP is removed (figure 4).

The removal of particulate P in RBZs occurs by deposition of sediments. Dissolved P is mainly removed by sorption by soil and uptake by vegetation. Infiltration and filtration are also important (190). Sediment removal is not the only process that is dependant on a reduced flow rate and consequently longer buffers are more effective than short buffers for P removal. Syversen (175) found that a 10m buffer was significantly more effective in removing P than a 5m buffer. However, Abu-Zreig et al. (2) reported a steady increase in P trapping efficiency up to 10m, but this increase declined after 10m. Working at a watershed scale, Reed and Carpenter (137) found that the shape and continuity of the buffer was more closely related to the P retention than the length of the buffer.

RBZs are more effective at removing some forms of P than others. A number of studies have identified increases in reactive or dissolved forms of P as runoff waters pass through RBZs (188, 189, 41, 126): several of these studies found increases of over 50% in the dissolved or reactive P load (188,41). Spruill (165) also found increases in shallow groundwater concentrations of P associated with RBZs, which is in agreement with Osborn and Kovacic (122), who suggest that forested buffers may also leak P to shallow groundwater.

Increases in dissolved or reactive P may be associated with vegetation type or management. In comparisons between forested and grassed buffers and grass and mixed vegetation buffers, the cutting and removal of vegetation appeared to be the key difference between a reduction and increase in reactive P. The source of this P is most likely to be leaching from decaying vegetation (188, 189).

Carbon has received relatively little attention with regard to RBZs, although there is some evidence for increases in dissolved organic carbon (DOC) reaching waterways where forested buffers are present. In a sub-watershed scale study in Maryland, Peterjohn and Correll (126) found a 2.9-fold increase in DOC and an increase in the proportion of organic carbon per unit of sediment from 1.5 to 8.2%. A second study, in North Carolina, compared buffer and non-buffer areas in a watershed. This study found an increase in DOC in shallow and deep groundwater under forested RBZs. Increased levels of DOC in groundwaters have a number of potential impacts. Carbon is important for denitrification and so can lead to increased N_2O losses. It also influences the water pH and is related to CO_2 losses (165). If the buffer zone is saturated there is also an increased chance of methane production.

Sulphur has received even less attention than carbon with regard to RBZs. However, there is potential for hydrogen sulphide production in saturated buffers.

RBZs have also been used to reduce faecal bacteria losses from manure amended soils. There is some similarity between manure-borne TP concentration and faecal coliform concentration (170). This is because faecal bacteria are very small and would behave much like clay particles (36) which P binds to. The small size of the faecal bacteria means that RBZs are limited in their potential to reduce losses. Coyne et al. (36) reported a 59% average reduction in faecal bacteria leaving an RBZ than entering, and Young et al. (198) reported a 70% reduction. Despite these successes, Coyne et al. (37) warn that there is potential for RBZs to become a reservoir for sediment-bound bacteria: they found that by the end of a one-hour simulated rainfall event (intensity 64mm hr^{-1}) the flow-weighted mean concentrations of faecal bacteria leaving the buffer exceeded those entering.

RBZs are also quite effective for the removal of pesticides, with reductions of between 0 and 100% and a mean of 78% reported in the literature (figure 4). Lacas et al. (86) presented a thorough review of the effectiveness of RBZs for trapping pesticide runoff. They found RBZs intercepted between 13 and 100% of pesticide runoff. Arora et al. (8) also presented a review of current literature showing very similar results of between 11 and 98%. Examination of the literature shows that, despite this wide range, in a majority of studies pesticide retention in RBZs is high (see figure 4), although this may not be sufficient to meet EU limits for environmental and drinking water (20).

Removal of pesticides in RBZs is mainly due to infiltration of soluble components. Sedimentation, dilution with rainwater and adsorption to plants and soils are also important (86, 110, 125). Krutz et al. (85) identify the latter as especially important under saturated conditions. The relative importance of infiltration and sedimentation will depend on the chemistry of the pesticide. Different pesticides vary in their solubility and the strength with which they are adsorbed to soil particles. This can have a considerable influence on their removal from overland flow by RBZs. Some pesticides such as the herbicides atrazine and metolachor are relatively water-soluble and are moderately absorbed onto the soil, making infiltration more important. Others, such as diflufenican and lindane, have lower water solubility, but are more strongly absorbed to the soil, making sedimentation more important for their removal.

There have been a number of potential problems identified with the use of RBZs to reduce pesticide runoff. Degradation decomposes the pesticide into by-products that can have a higher reactivity in the soil than the parent molecule (86). These may be trapped in the buffer and then released once degraded (192). There is also some evidence of potential for leaching of pesticides through the soil profile in buffers (140) although levels of leaching are less than in cropped areas (15, 140).

Despite the very positive reports of the effectiveness of RBZs for removing pollutants, there are some questions regarding their effectiveness over time. Several studies have reported a reduced effectiveness over a number of years or after repeated simulations (21, 97). This is especially true of sediment and sediment-bound pollutants. The depth of sediment in the buffer increases over time, altering its geometry. This has the potential to lead to overtopping (145) or concentrated flow (44, 75). Another bypass mechanism is artificial field drainage, which under some conditions means a considerable quantity of water leaves the system without passing through the RBZ (89). There is also potential for buffers to turn into a source of sediment and nutrients as soils that have previously been trapped, are released (190). Buffers are not effective when overwhelmed by concentrated flows (44).

Riparian buffer zones – Summary (Figure 4)

- RBZs are an effective method of removing NO_3 from overland flow and groundwater in hydrologically suitable situations. There are likely to be high N_2O emissions from some RBZs.
- Sediment trapping by RBZs is also very effective where flows are not concentrated, but some management may be needed to prevent sediment build-up.

- RBZs reduce P loads in overland flow, but are potentially a source of dissolved reactive P.
- Increased levels of DOC have been associated with RBZs.
- RBZs are thought to be quite effective for the removal of faecal bacteria and pesticides, although there is potential for re-release of both.
- There is potential for RBZs to collect pollutants and release them at a later date.

VI. Contour grass strips

Contour grass strips (CGSs) or vegetative barriers work on the same principals as RBZs, but are ribbon-like bands of grass, typically 2 to 4m wide (91) located within fields rather than at the field edge. They act to reduce slope length, which in turn reduces runoff velocity, allowing time for sediment to settle (39), and act as barriers to overland flow causing ponding and the deposition of sediments in front of the barrier. They have received considerably less research attention than RBZs.

CGSs have predominantly been used to reduce sediment losses. In laboratory experiments Ligdi and Morgan (91) found CGSs were effective at removing sediment on 5% and 10% slopes. Only dense vegetation was effective on a 20% slope. At 20% and above the CGSs were sources of sediment. In a flume experiment with different grasses and flow rates, Dabney et al. (39) found the CGSs to range from 15 to 79% in their effectiveness.

Although CGSs are regularly referred to as filter strips, the main mechanism for sediment removal is settling (22, 39, 55, 91). CGSs are only able to filter large particles, due to the large flow spaces in the vegetation (39).

Sediment trapping mainly occurs in the backwater that forms upslope of the CGS. The reduction in flow velocity in the backwater causes coarse sediment to settle out. Finer sediment settles out in fans below the strips. The length of this backwater is determined by the slope, vegetation density in the strip and the flow rate; the strip width is not important in determining the efficiency of the sediment trapping (55). Debris and plant residues can become trapped in the strip, and this increases hydraulic resistance, causing deeper backwaters and increased trapping (39, 55). Jin et al. (72) found a 10% increase in sediment trapping efficiency when mulch was introduced to a barrier in a flume experiment. In high flows, the strips can become overloaded and the barrier can be submerged. Flume experiments have shown that once submerged the whole structure can be undermined, washing away soil from around the plant roots (22).

Grass type is very important. Grasses that form dense uniform barriers and have dense root mats will be most effective in reducing sediment losses. Grasses that are not sufficiently rigid or have a low stem density have the potential to increase sediment losses as the barriers are overwhelmed (22).

There has been very limited work into the effectiveness of CGSs in reducing pollutant losses, although there is potential for CGSs to reduce the same pollutants as RBZs. Eghball et al. (48) showed that narrow (0.75m) grass hedges established approximately on the contour, were effective at reducing both P and N losses in runoff. Dissolved P, bioavailable P, particulate P, NO_3 and NH_3 loads in runoff were all significantly reduced compared with plots without a CGS.

In a flume experiment Krutz et al. (85) investigated the effectiveness of Buffalo grass filter strips for trapping atrazine and its metabolites. They found that in a 60-minute simulation 22% of the atrazine was retained in the CGS and 19% of the atrazine metabolite.

Contour grass strips – Summary (figure 5)

- There is good potential for CGSs to reduce sediment losses.
- CGSs can reduce nitrogen and phosphorus in runoff.
- A reduction in pesticide losses is possible, although results show that the reductions are not large.
- More research is needed into the effectiveness of CGSs for trapping pollutants.

VII. Constructed wetlands

Wetlands are created for a number of reasons: their value as high-diversity habitats, to mitigate against habitat loss, and for the treatment of wastewater (113). Initially wetlands were used predominantly for the treatment of point source pollution, but there has been an increased interest in the use of wetlands for the treatment of diffuse urban and agricultural pollution (149). The term 'wetland' covers a wide range of habitats and in the context of this paper will be used to encompass all wetland types used to treat wastewaters, including ponds, marshes and reed beds. These may be situated on low order streams, receive pumped water or receive flows from other sources such as overland flow. The design of wetlands varies considerably between studies; however, there are a number of factors which have been identified as important in determining how effective wetlands are at removing pollutants. These include biological, physical and chemical factors on both short and long time scales (136), including hydraulic loading, retention time, depth of water column, pollutant concentration in inflow, soil type, presence or absence of vegetation, water chemistry, shoreline development, wind effects and temperature (50, 169).

Constructed wetlands (CWs) have been used extensively for sediment removal from a range of wastewaters. Runoff from combinable crops has received surprisingly little attention, but the results from other systems can provide a considerable amount of relevant information.

Sediment entering a CW is removed primarily by settling. This means that a number of hydrologic factors are important in determining the retention of sediment. Using a laboratory experiment, Stephan et al. (169) suggest that an increase in the flow velocity causes a reduction in settling – this could be due to reduced residence times. Braskerud (26) suggests that as larger soil particles and aggregates are transported with higher velocity flow, retention may increase with velocity; this is because larger particles which settle more readily, are transported in greater quantities and then deposited in the CW. However, this is not in agreement with Kadlec and Hey (76) who suggest that sediment load is unimportant, as wetlands trap sediments at their inlet. In order to maximise settling of suspended sediment, uncontaminated water should be directed away from the CW (26). Vegetation can also have considerable influence over sediment removal, as plants change the flow through the wetland. Depending on the flow rate and sediment input, plants may increase or decrease deposition by changing flocculation rates, creating local turbulence, reducing velocity and providing local deposition surfaces (169, 189). Quantity of vegetation is also important in a CW, with possible seasonal differences. At low (20%) vegetation cover in a small CW 40% of sediment was re-suspended, but at 50% vegetation cover re-suspension was insignificant (26).

Due to differences in design and environmental conditions, sediment retention in CWs from agricultural catchments varies between 43 and 88%, with a mean of 69% (figure 6). Kadlec and Hey (76) report retention of 88% in a series of six wetlands covering approximately 12ha at the Des Plains river wetlands demonstration project in the USA, 6.6% of the agricultural and urban catchment. Braskerud and Haarstad (29) reported a much lower sediment retention of 43% in a sedimentation pond with vegetated filters draining a 22ha catchment of agricultural crops. However, the catchment area to wetland area ratio in this study was lower at 0.003. Examining a range of CWs in Southern Norway, Braskerud (26) observed sediment retention of 45 – 75% of sediments. Clay retention was high in this investigation (57%), suggesting that aggregates form allowing fine particles to be removed.

Wetlands reduce phosphorus concentrations by sedimentation of soil-bound nutrients, sorbing nutrients onto sediments and vegetation assimilation (short- or long-term storage depending on biomass turnover and the life time of the vegetation (136)). Removal due to vegetation may be seasonal (127) and the lowest removal rates can occur in winter and spring when most of the P enters the wetlands (83). In addition to the factors described at the start of this section, the ratio of CW area to catchment area, CW area and oxygen concentration in sediments are all important factors controlling P retention (189, 27, 51). The oxygen concentration of the water and redox potential of the sediments can be affected by flow rate (52). An experiment using wetland soils has also shown that P concentration of the water has the potential to change the retention capacity of the CW. When P concentrations in water are low, P may be released from soil pore water into water column; however, as P concentration in overlying waters increased, retention by the soils also increased (47).

As with sediments, retention of TP in CW draining catchments containing combinable crops is very variable, ranging from 1 to 91% with an average of 35% (figure 6). This value is similar to that found by Uusi- Kamppa et al. (189), who investigated CW draining catchments with various vegetation types. They found an average of 17% retention in free water surface (FWS) wetlands. The majority of the CWs identified in this review were FWS wetlands, which Uusi-Kamppa et al. (189) suggest have lower retention than other wetlands for which they found 41% retention. Fisher and Acreman (51) reviewed 57 natural wetlands and also found that swamps and marshes are most likely to retain P.

Some CWs have been considerably more successful at retaining P. The Des Plains river wetlands demonstration project (described above) uses continuously pumped water; here P retention was 81, 91, 67 and 79% in the four wetlands (76, 112). In a second wetland with continuously pumped flow, removal was 70% (112).

As in buffer zones, in CWs N is primarily retained by microbial processes (26, 127, 139). Reinhardt et al. (139) found 96% of the N removed was accounted for by denitrification. The remaining 6% was accumulated in sediment. Plants can also provide supplemental N removal (127).

Examination of the literature shows an average TN removal (in CW draining catchments containing combinable crops) of 29% with a range of 11–42%. NO_3 has an average removal of 26%. NH_4 removal is generally low and some experiments report NH_3 production (28, 83). Organic N is also retained by CWs (figure 6): Braskerud (28) report retention of 17%, which they attribute primarily to sedimentation.

The dependence on microbial processes leads to a seasonality in N removal, which has been identified in a number of investigations (e.g. 28, 131). Both nitrification and denitrification are inhibited at low temperatures and water needs to be retained in the CW for a longer time period for N removal to occur (26, 131). It is also possible that flood events in winter remove carbon needed for denitrification (28). Fisher and Acreman (51) found N removal from natural wetlands was most closely related to oxygen content of sediment, degree of waterlogging and redox potential, all of which are important factors controlling denitrification. As discussed in relation to buffers, N removal by denitrification can lead to N_2O production if denitrification is not complete. This means wetlands should be located in areas with high NO_3 concentrations in water for optimal denitrification. Emissions are exacerbated by high water NO_3 content (166) therefore wetlands receiving large amounts of NO_3 and those with fluctuating water levels (113) are most likely to have high N_2O emissions.

CH_4 and CO_2 are also emitted from waterlogged areas. CWs emit methane at similar rates to natural wetlands with similar vegetation. This means that areas previously under agriculture will have greatly increased emissions by converting them to wetlands (73). Several studies have reported methane and CO_2 emissions from CWs (113, 166), but there has been relatively little attention given to CWs in comparison to natural wetlands.

In contrast to carbon losses to the atmosphere through CH_4 emissions, the picture is mixed for organic carbon in waters. Jordan et al. (75) report an average of 36% TOC retention over two years in a FWS wetland. Over three years Kovacic et al. (83) report TOC retention of 7, 6 and -11% in three CWs. DOC was exported from these wetlands in over half of the wetland years, giving no significant change in carbon.

As with other wetland habitats there is potential for a CW to emit H_2S, although there has been no research on this to date. The removal of pathogens has also received very little attention in relation to CWs in agricultural catchments. CWs are used for wastewater treatment and are effective at removing pathogens by sedimentation (79).

A number of experiments have been conducted to investigate pesticide removal in CWs, with removals of between 36 and 100% with a mean of 79% (figure 6). Several mesocosm studies have shown very high pesticide removal (chlorothalonil – 94% removal after 24 hours (158) and chloropyrifos – 83% removal after 84 days (116)); however, retention times are longer than found in many CWs. With very low pesticide inputs, Schulz and Pearll (149) found up to 93% retention of azinphos-methyl and 100% retention of chloropyrifos and endosulfan after a single storm event. Using simulated runoff, Moore et al. (2000) found retention rates of 68 and 36% for 73µg l^{-1} and 147µg l^{-1} atrazine, respectively.

Braskerud and Haarstad (29) investigated the retention of 13 pesticides in an 840m^2 FWS wetland within a 22ha catchment. They found that retention rates varied between pesticides, with a range of between -2 and 40% retention. For all of the seven pesticides tested over a two-year period, retention was much lower in the second year of the experiment; for example, propachlor had a retention of 67% in the first year but dropped to 14% in the second year.

The ability of a CW to continue to retain pollutants over time is a potential cause for concern. Sediments, total phosphorus, pesticides and organic N retention have all been found to decrease with CW age (113, 50, 29, 28). Braskerud (26) found wetlands filled with sediment in 8–20 years, although accumulated sediment can be dug out and the wetland should regain its functionality (2005). A 10-year experiment conducted by Mitsch et al. (113) confirmed this when they found their experimental wetland became a sediment source after nine years. Vegetation may also contribute to aging effects as plants may take up less nutrients once they are well established (95).

Constructed wetlands – Summary (figure 6)

- CWs are effective in removing sediments by sedimentation.
- P is generally retained in CWs, although their effectiveness is variable.
- N is removed by microbial processes in a CW, but retention rates are not generally high.
- CWs constructed for pollutant retention emit greenhouse gasses.
- CWs have the potential to remove pesticides, although they may not be effective over a long time period.
- Efficiency of a CW for sediment and nutrient trapping may also decrease with time.

VIII. Conclusions

Figures 1 to 6 summarise how each of the mitigation options impacts on the various pollutants investigated in this study. It is clear from these graphs that there is no single mitigation option that will reduce all pollutants. It is also a very challenging task to compare the relative impacts of the different pollutants, as their effects are apparent over differing temporal and spatial scales. For example, eutrophication may be an issue of local concern as phosphorus-rich water enters a lake, with impacts over very short timescales, whereas N_2O oxide has no short-term impacts, but is a powerful greenhouse gas contributing to a global problem with long-term impacts. Because of the opposing impacts that different mitigation options have on pollutants, it is not possible to recommend a single strategy for reducing diffuse pollution. Instead we must make some recommendations with regard to how to select the most appropriate mitigation option. Pollution swapping should be considered when selecting a mitigation option and the most appropriate option should be selected on a site-by-site basis. Introducing schemes nationwide and encouraging farmers to install a single mitigation option will result in unnecessary increases in some pollutants, even though it may reduce the impact of the target pollutant. When considering the most appropriate mitigation option to use, the first consideration should be which pollutant(s) is the target of concern: some may be more pressing than others and mitigation options should be applied to tackle this. However, longer term implications should be considered as well as short-term ones. Maintenance costs and lifespan are also an important consideration, as poorly maintained mitigation options can become a source of pollutants rather than a sink. A mitigation option should be selected that is appropriate to the location, including soil type, climate, location in the catchment, landscape features and hydrology. It is beyond the scope of this paper to make recommendations for each of the mitigation options; however, these issues have been addressed for many of the options available.

This paper has identified some considerable gaps in our knowledge of the impact of mitigation options that been applied throughout the world on different pollutants. Vegetative barriers and cover crops are the two mitigation options with particular need for further research. Pollutants that are in particular need of further investigation include total organic carbon, methane and hydrogen sulphide. Research is also needed into quantifying the relative importance of different pollutants in the environment.

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Table 1. The range of pollutants discussed in this study and the environmental, economic and health effects.

Pollutant		Effect
Suspended sediments		Increases turbidity and transports other pollutants
Nitrogen (N)	Nitrate (NO_3)	Contributes to eutrophication Implicated in methemoglobinemia (blue baby syndrome)
	Ammonia (NH_3)	Contributes to eutrophication Toxic
	Nitrous oxide (N_2O)	Powerful greenhouse gas
Phosphorus (P)	Dissolved phosphorus (DP)	Contributes to eutrophication – rapidly available to algae
	Particulate phosphorus (PP)	Contributes to eutrophication – available to plants over time
Carbon (C)	Dissolved organic carbon (DOC)	Associated with water colour increasing water treatment costs
	Carbon dioxide (CO_2)	Greenhouse gas
	Methane (CH_4)	Greenhouse gas
Sulphur (S)	Hydrogen sulphide (H_2S)	Toxic gas, contributes to acid rain
Pesticides		Potentially harmful to biota, can bioaccumulate
Pathogens		Pose health threats to wildlife, bathers and water supplies

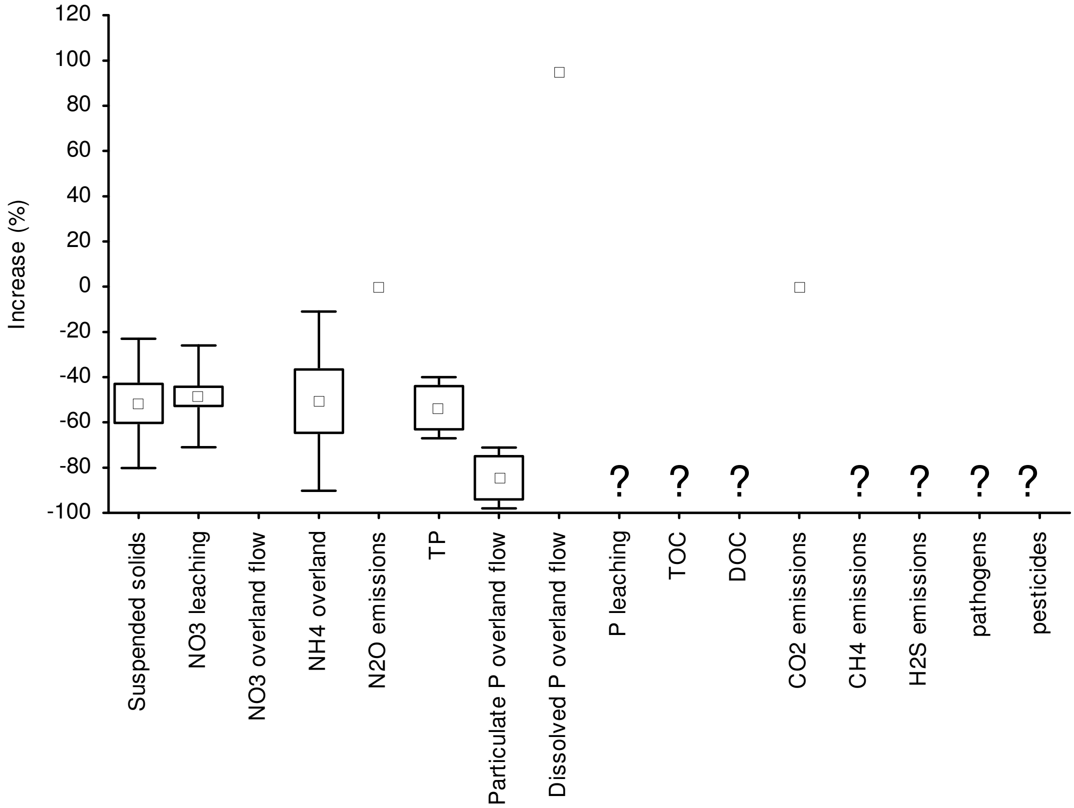

Figure 1. Percent reduction (-) or increase (+) from cover crops when compared to control plots in: suspended solids, NO_3 leaching losses, NO_3 losses in overland flow, NH_4 losses in overland flow, N_2O emissions, total P losses on overland flow, dissolved P losses in overland flow, P leaching, total organic carbon losses in overland flow, dissolved organic carbon losses in overland flow, CO_2 emissions, CH_4 emissions, H_2S emissions, pathogens in overland flow and overland flow pesticide losses. Data for suspended solids was taken from 81 and 88 (n=12). NO_3 leaching losses were taken from 7, 12, 14, 34, 43, 99, 104, 105, 114, 117, 119, 156, 157, 181, 184 (n=38), NH_4 in loads in overland flow from 152 (n=8), N_2O emissions from 193 (n=1), total P losses in overland flow data were from 167, 187(n=2), particulate P losses from 187, 167 (n=2) and dissolved P losses from 187(n=1). CO_2 emissions were taken from 193 (n=1). ▫ Mean, ▯ Mean ± Standard error, ├┤ Mean ± standard deviation, ↑ indicates trend reported in literature, ? indicates no information. No error bars indicates insufficient data.

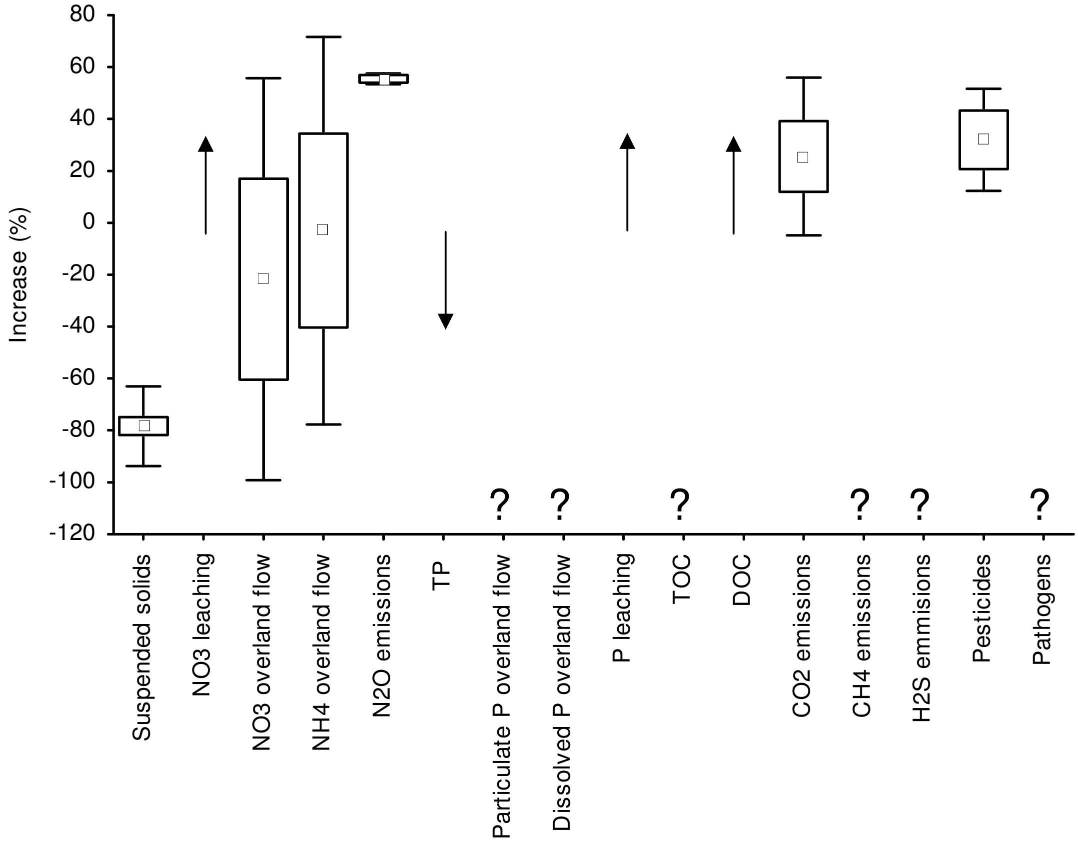

Figure 2. Percent reduction (-) or increase (+) from crop residues when compared to control plots in: suspended solids, NO_3 leaching losses, NO_3 losses in overland flow, NH_4 losses in overland flow, N_2O emissions, total P losses on overland flow, dissolved P losses in overland flow, P leaching, total organic carbon losses in overland flow, dissolved organic carbon losses in overland flow, CO_2 emissions, CH_4 emissions, H_2S emissions, pathogens in overland flow and overland flow pesticide losses. Data for suspended solids was taken from 3, 24, 32, 103, 108, 118, 120 and 150 (n=20). NO_3 and NH_4 losses in overland flow were taken from 118, 183 (n=4), N_2O emissions from 71 (n=2). CO_2 emissions were taken from 191, 57 (n=5) and pesticide losses from 164, 161 (n=3). ▫ Mean, ▯ Mean ± Standard error, ├┤ Mean ± standard deviation, ↑ indicates trend reported in literature, ? indicates no information. No error bars indicates insufficient data.

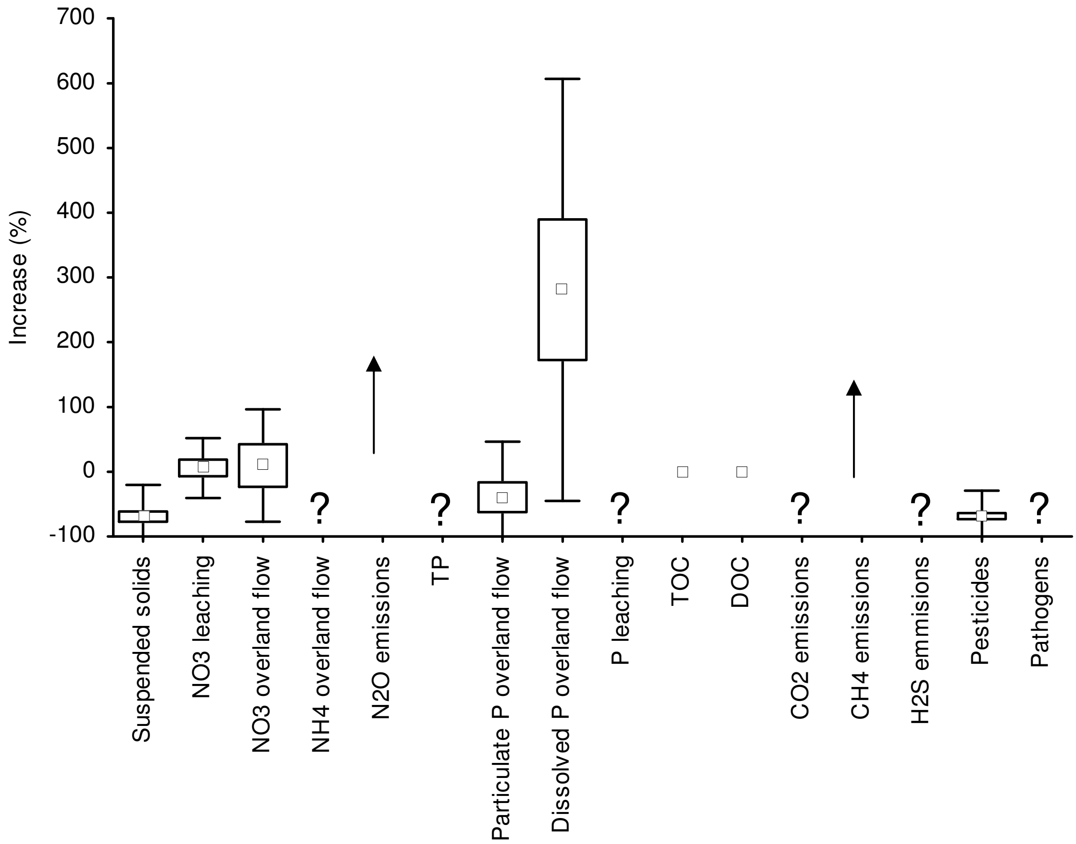

Figure 3. Percent reduction (-) or increase (+) from no-tillage when compared to conventional plough in: suspended solids, NO_3 leaching losses, NO_3 losses in overland flow, NH_4 losses in overland flow, N_2O emissions, total P losses on overland flow, dissolved P losses in overland flow, P leaching, total organic carbon losses in overland flow, dissolved organic carbon losses in overland flow, CO_2 emissions, CH_4 emissions, H_2S emissions, pathogens in overland flow and overland flow pesticide losses. Data for suspended solids was taken from Data for soil loss is taken from 28 plot experiments worldwide reviewed in Strauss et al. (171) and Strauss et al. (172) combined with data from 199, 75, 71, 123 and 141 (n=39). Overland flow NO_3 losses were taken from 60, 102, 155, 62 and 10 (n=13), NO_3 leaching losses from 134, 10, 77, 78, 90 and 179 (n=7), particulate P losses in overland flow data were from 42, 197, 129, 102, 152, 148, 84 and 142 (n=14) and dissolved P concentrations from 60, 42, 197, 102, 155 and 148(n=9). Pesticide losses in overland flow were taken from a review of seven studies on atrazine, cyanazine, simazine and metolachlor by 49 and leaching losses for atrazine, carbofuran, diazinon, metolachlor and terbuthylazine are from 90, 64 and 161 (n=65). ▫ Mean, ▯ Mean ± Standard error, ├┤ Mean ± standard deviation, ↑ indicates trend reported in literature, ? indicates no information. No error bars indicates insufficient data.

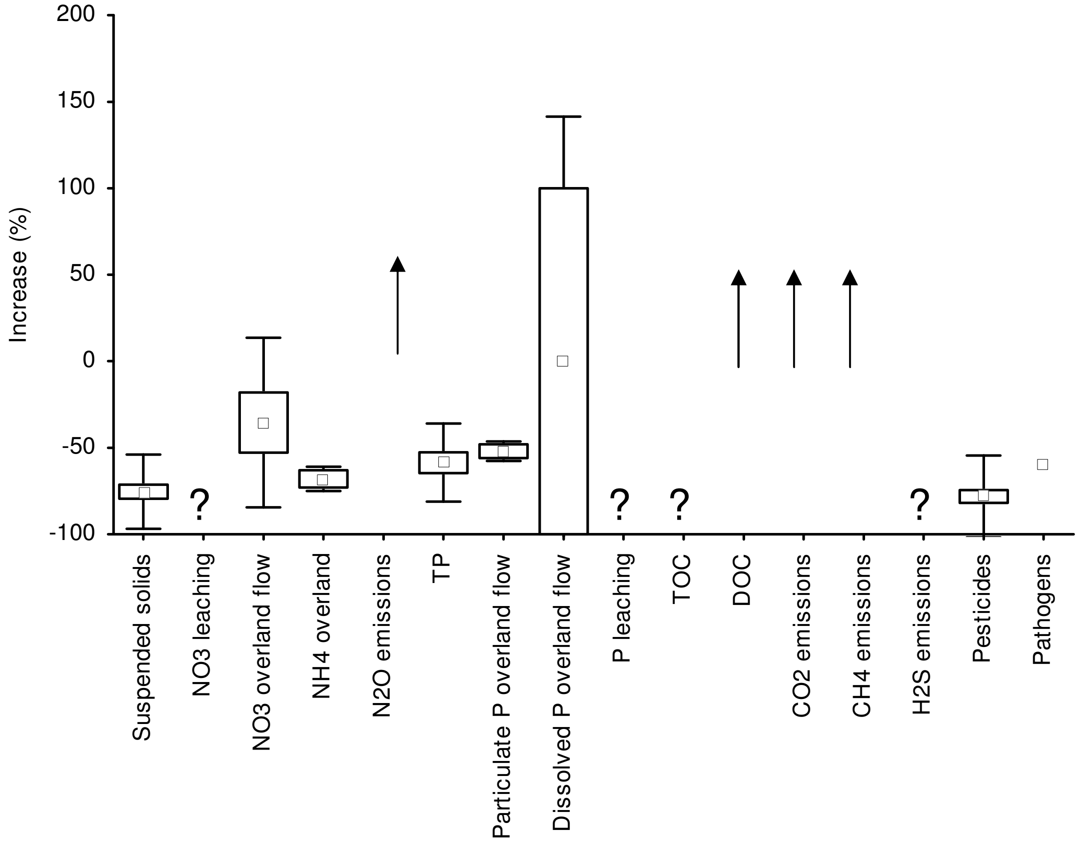

Figure 4. Percent reduction (-) or increase (+) from riparian buffer zones when compared to control plots in: suspended solids, NO_3 leaching losses, NO_3 losses in overland flow, NH_4 losses in overland flow, N_2O emissions, total P losses on overland flow, dissolved P losses in overland flow, P leaching, total organic carbon losses in overland flow, dissolved organic carbon losses in overland flow, CO_2 emissions, CH_4 emissions, H_2S emissions, pathogens in overland flow and overland flow pesticide losses. Data for suspended solids was taken from 1, 25, 36, 41, 44, 97, 106, 110, 125, 144, 176 and 178 (n=27). NO_3 and NH_4 losses in overland flow were taken from 19, 126, 97 (n=8; n=2), total P losses in overland flow were taken from 2, 21, 41, 45, 97, 188 (n=14), particulate P losses were taken from 188 (n=2) and dissolved P losses in overland flow were taken from 20, 188 (n=2). Pesticide losses were taken from 8, 23, 86, 94, 110, 125, 135, 176, 192 (n=42) and pathogen losses from 36 (n=1). ▫ Mean, ▯ Mean ± Standard error, ├┤ Mean ± standard deviation, ↑ indicates trend reported in literature, ? indicates no information. No error bars indicates insufficient data.

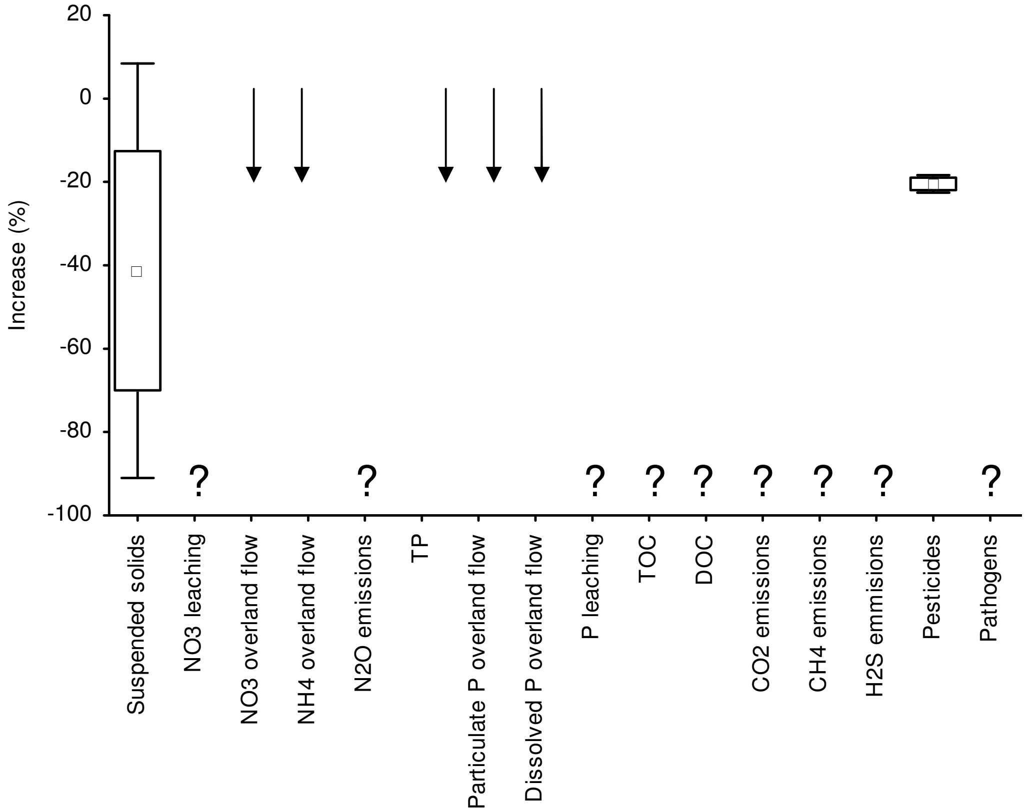

Figure 5. Percent reduction (-) or increase (+) from contour grass strips when compared to control plots in: suspended solids, NO_3 leaching losses, NO_3 losses in overland flow, NH_4 losses in overland flow, N_2O emissions, total P losses on overland flow, dissolved P losses in overland flow, P leaching, total organic carbon losses in overland flow, dissolved organic carbon losses in overland flow, CO_2 emissions, CH_4 emissions, H_2S emissions, pathogens in overland flow and overland flow pesticide losses. Data for suspended solids was taken from 91 and 39 (n=3). Pesticide losses were taken from 85 (n=2). ▫ Mean, ▯ Mean ± Standard error, ├┤ Mean ± standard deviation, ↑ indicates trend reported in literature, ? indicates no information. No error bars indicates insufficient data.

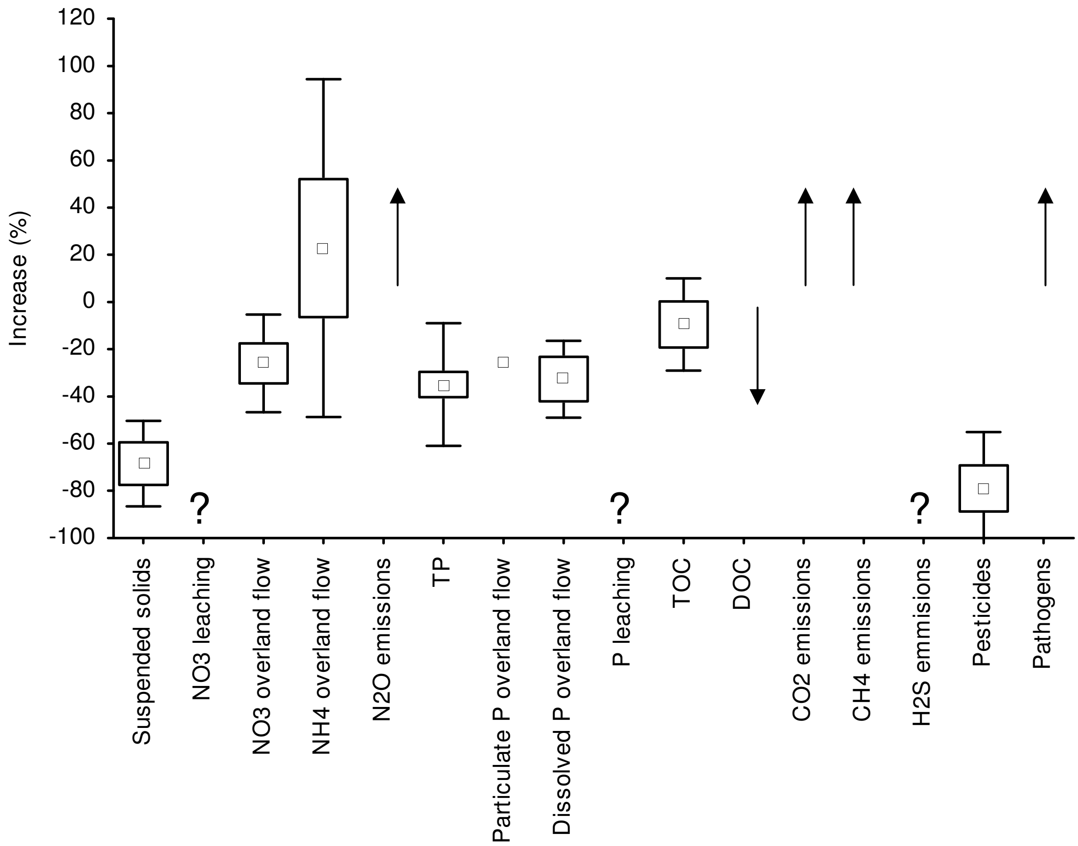

Figure 6. Percent reduction (-) or increase (+) from constructed wetlands when compared to control plots in: suspended solids, NO_3 leaching losses, NO_3 losses in overland flow, NH_4 losses in overland flow, N_2O emissions, total P losses on overland flow, dissolved P losses in overland flow, P leaching, total organic carbon losses in overland flow, dissolved organic carbon losses in overland flow, CO_2 emissions, CH_4 emissions, H_2S emissions, pathogens in overland flow and overland flow pesticide losses. Data for suspended solids was taken from 76, 29 and 26 (n=4). NO_3 losses in overland flow were taken from 28, 50, 83 (n=6), NH_4 losses in overland flow are taken from 28, 83 (n=6), total P losses in overland flow were taken from a review (27) citing 17, 25, 163, 187, 195 plus data from 29, 74, 76, 82, 83, 112, 132, 139 (n=25), particulate P losses were taken from 39 (n=1) and dissolved P losses in overland flow were taken from 82, 83, 139 (n=3). TOC losses were taken from 75, 83 (n=4) and pesticide losses were taken from 29, 116, 149, 158, (n=6). ▫ Mean, ▯ Mean ± Standard error, ├┤ Mean ± standard deviation, ↑ indicates trend reported in literature, ? indicates no information. No error bars indicates insufficient data.