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# Policy Implications of Pollution Swapping

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# Policy implications of Pollution swapping

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## Abstract

Pollution swapping can be defined as the increase in one pollutant as a result of a measure introduced to reduce a different pollutant. Although pollution swapping is widely understood it has received relatively little research attention and receives little consideration in agri-environmental policy. Evidence of pollution swapping in constructed wetlands, riparian buffer zones, cover crops, crop residue retention and no-tillage is examined in this paper. These widely used mitigation options are all successful at reducing diffuse pollutants but literature review shows that there is potential for them to increase levels of one or more other pollutants. There is potential for the widespread adoption of mitigation options to result in unexpected increases in some pollutants.

There are a number of barriers to the recognition of pollution swapping in agri-environmental legislation including a lack of tools to evaluate the relative impacts of different pollutants, gaps in our knowledge of the impacts of mitigation measures on non-target pollutants and institutional barriers.

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**Introduction**

Pollution swapping occurs when a mitigation measure introduced to reduce levels of one pollutant results in increased levels of another pollutant; one pollutant is 'swapped' for another. For example, riparian buffer zones are used worldwide to reduce levels of sediment, phosphorus and nitrogen entering waterways by overland flow. However, they have the potential to result in waterlogged soils which leads to an increase in emissions of greenhouse gasses such as nitrous oxide and methane. Although the concept of pollution swapping and the mechanisms by which pollutants are produced are widely understood, there have been very few attempts to summarise the available information for different mitigation options. Stevens and Quinton (in press) reviewed a range of mitigation options available for combinable cropping systems and found that pollution swapping was surprisingly common. All of the mitigation options investigated were likely to result in some form of pollution swapping. They also identified large gaps in our knowledge of the impact on pollutants that were not directly targeted by the mitigation option implemented.

Partly as a result of this lack of synthesis, there has been virtually no consideration of pollution swapping in policy making. This has potentially serious consequences if policy and funding available to farmers results in large uptake of mitigation options targeting a single pollutant. This paper will summarise the evidence for pollution swapping and build upon Stevens and Quinton (in press) to discuss how current agri-environmental policy is impacting upon it. We will also reflect on how future policy would need to be adapted to incorporate consideration of pollution swapping.

At a European level there are several key pieces of legislation to which pollution swapping is directly relevant; the 2000 EU water framework directive (WFD) (European Commission, 2001), the forthcoming Thematic Strategy on Air Pollution and the Directive on Ambient Air Quality and Cleaner Air for Europe (Commission of the European Communities, 2005). Member states of the European Union will need to ensure that they meet the requirements of these directives. In the case of the WFD member states will need to ensure that inland waters meet good ecological status. This is not closely defined in the directive and is open to interpretation. However, it is clear, perhaps for the first time, that member states will be required to control diffuse pollution as well as point sources of pollution: water managers will need to look beyond the water course and take account of land use within the catchment (Moss, 2004). Optimising land use to prevent ecological deterioration will require consideration of how pollutants will be swapped rather than considering pollutants in isolation. As discussed later in this paper, pollutants can also swap between phases, and while the pollution of water bodies by diffuse agricultural sources in included within the WFD, environmental managers will also need to consider air quality. In an assessment of the impact of the Thematic Strategy on Air Pollution and the Directive on Ambient Air Quality and Cleaner Air for Europe it is recognised that agriculture will have a role to play in improving air quality, although this is mostly associated with livestock and ammonia emissions (Commission of the European Communities, 2005). The reform of the Common Agricultural Policy (CAP) is seen as a potential mechanism for reducing agricultural emissions (Commission of the European Communities, 2005) and as member states develop their own policy in which to deliver CAP reform, there is at least the potential for pollution swapping to be taken into account.

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This paper will focus on pollution swapping for diffuse pollutants, the pollutants that will be discussed are: suspended solids, nitrate ($NO_3$), ammonia ($NH_3$), nitrous oxide ($N_2O$), dissolved and particulate phosphorus (DP and PP), dissolved organic carbon (DOC), carbon dioxide ($CO_2$), methane ($CH_4$), hydrogen sulphide ($H_2S$), biocides (including pesticides, fungicides and herbicides) and pathogens.

The above pollutants are transported to their sink via a number of different pathways and have a number of potential impacts. In many regions dissolved pollutants are moved predominantly by subsurface pathways while particulate pollutants move by overland flow. Suspended solids are predominantly delivered to receiving waters by overland flow. They increase the turbidity of receiving waters and are frequently associated with the transport of other pollutants. PP is one of these pollutants. PP contributes to eutrophication of waters and soils providing a long term store of P that becomes available to plants over time. Pathogens are also transported with suspended solids. They originate from manures and slurries applied to crops as fertilisers. Pathogens pose health threats to wildlife, bathers and water supplies.

As with PP, DP contributes to eutrophication, although in this case it is rapidly available to algae. $NO_3$ and $NH_4$ are also mainly transported in the dissolved fraction and again contribute to eutrophication. Additionally $NO_3$ is implicated in methemoglobinemia (blue baby syndrome) and $NH_4$ is toxic. Biocides and DOC are also transported via subsurface pathways. Biocides are potentially harmful to a wide range of biota and can bioaccumulate higher up the food chain. DOC is associated with water colour and although not directly harmful it is associated with increased water treatment costs.

Waters and soils are not the only sink for pollutants; the atmosphere is also an important destination. Gaseous emissions occur naturally from soils but under certain conditions, such as waterlogging, which increases anaerobic processes, they are enhanced. $CO_2$, $CH_4$ and $N_2O$ are all greenhouse gasses emitted from soils. $H_2S$ can also be emitted from waterlogged soils, this is a toxic gas which contributes to acid rain.

In their review Stevens and Quinton (in press) examined different mitigation options for reducing diffuse pollutant losses from combinable crops. These were: constructed wetlands, riparian buffer zones, cover crops, crop residues and minimum or no-tillage. These mitigation options will also be the focus of this discussion.

**Evidence for pollution Swapping**

**Constructed wetlands**

Wetlands have been investigated extensively for the treatment of point source pollution (e.g. mine waste (Demin et al., 2002) and urban storm water (Lee et al., 2006)) but there is an increased interest in the construction of wetlands for the management of diffuse pollution. There have been comparatively few studies investigating the use of constructed wetlands to treat diffuse pollution, the majority of information relates to semi-natural wetlands or those constructed for other purposes.

Design and environmental conditions in constructed wetlands vary considerably but they generally seem to be effective at sediment removal, retaining between 43 and 88% (Stevens and Quinton, in press).

Wetlands also reduce phosphorus concentrations by sedimentation of soil-bound nutrients, sorbing nutrients onto sediments and vegetation assimilation. Removal due to vegetation may be seasonal (Picard et al., 2005) and the lowest removal rates can occur in winter and spring when most of the P enters the wetlands (Kovacic et al., 2000). As with sediments, retention of phosphorus in constructed wetlands draining catchments containing combinable crops is very variable, ranging from 1 to 91% with an average of 35% (Stevens and Quinton, in press). For nitrogen denitrification is an important removal process. Nitrogen removal is between 11 and 42% with an average of 29% (Stevens and Quinton, in press). $NO_3$ has an average removal of 26% but $NH_4$ removal is generally low and some experiments report $NH_4$ production (Braskerud and Haarstadt 2003, Kovacic et al., 2000).

N removal by denitrification can lead to $N_2O$ production if denitrification is not complete. Emissions are exacerbated by high water $NO_3$ content (Stadmark and Loenardson, 2005) therefore wetlands receiving large amounts of $NO_3$ and those with fluctuating water levels (Mitsch et al., 2005) are most likely to have high $N_2O$ emissions. $CH_4$ and $CO_2$ are also emitted from waterlogged areas. Constructed wetlands 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 (Johansson et al., 2005). Altor and Mitsch (2006) report that seasonally wet areas will have lower methane emissions than permanently wet ones.

Braskerud and Haarstad (2003) investigated the retention of 13 pesticides in a large wetland within a 22ha catchment. They found that retention rates varied between pesticides, with a range of between -2 and 40% retention over two years. Retention was higher in the first year than the second year. In their review, Oliver et al (2007) report very high retention of fecal organisms in constructed wetlands but they draw attention to the need for long term recording. The ability of a constructed wetland to continue to retain pollutants over time is a potential cause for concern and effectiveness may be reduced with age (Mitsch et al., 2005, Fink and Mitsch, 2004, Braskerud, 2002, Braskerud and Haarstadt 2003).

**Riparian buffer zones**

Riparian buffer zones are strips of vegetation along waterways that aim to prevent pollutants entering the waterway by a number of mechanisms including retardation of flow and consequent deposition of sediment and sediment-bound contaminants, interception by vegetation, plant uptake and infiltration. They have been widely used to reduce the impact of soil erosion 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 (Syversen, 2002) although are likely to become less successful at removing sediment over time as it builds up within the buffer (Borin et al., 2005, Magette et al., 1989).

Phosphorus removal is very closely related to sediment removal when the surface runoff has a high particulate concentration but, a large fraction of the sediment-bound phosphorus is associated with fine clay which is not easily deposited (Syversen and Borch, 2005). Dissolved and reactive phosphorus loads have been found to increase by as much as 50% (Uusi-Kamppa, 2005, Daniels and Gillman, 1996).

As with wetlands, denitrification is an important method of removing nitrogen but this is both spatially and temporally variable. Stevens and Quinton (in press) report results ranging from an increase of almost 20% (Magette et al., 1989) in $NO_3$ exiting the buffer zone compared to that entering it, to a decrease in nitrogen load of up to 99% (Patty et al., 1997), with a mean reduction of 35%. There is also potential for buffer zones to reduce nitrate levels in sub-surface waters, especially for forested buffers. In a review of six studies, Osborne and Kovacic (1993) found $NO_3$ removal from subsurface waters varied between 40 and 100%.

Buffer zones have also been reported to significantly reduce losses of both pesticides (Lacas et al., 2005, Arora, 2003) and fecal bacteria (Coyne et al, 1995, Young et al., 1980).

Despite the potential for buffer zones to remove pollutants, there have also been increases reported. In a sub-watershed scale study in Maryland, Peterjohn and Correll (1984) found a 2.9-fold increase in DOC. As with constructed wetlands there is also potential for an increase in greenhouse gas emissions. There are much higher levels of $N_2O$ produced in riparian buffer zones than field margins, with forested buffers producing seven times more $N_2O$ than grassed ones (Hefting et al., 2003).

**Cover crops**

Cover crops are grown in the period when the ground would normally be in fallow. They provide surface cover thus reducing loss of sediment, increasing nitrogen uptake and increasing infiltration. Cover crops are generally sown either in autumn, immediately following harvest, or in the spring, when they are under-sown below the crop. They 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.

Studies have generally shown that cover crops can successfully reduce nitrate leaching (e.g. Vinther et al, 2004, Torstensson and Aronsson, 2000, Shepherd, 1999). Stevens and Quinton (in press) identify an average reduction in leaching of 48% and a range of 0 to 98%. It has also been suggested that there is potential for cover crops to reduce nitrous oxide emissions from soils by reducing the amount of mineral nitrogen stored in the soil (Dalal et al., 2003). However, there have been both increased and decreases reported in losses of nitrate in overland flow. Sharpley and Smith (1989) report nitrate concentrations in overland flow ranging from an increase of 31% with cover crops as opposed to bare fallow, to a reduction of 87% with cover crops.

Cover crops have been very successful for reducing sediment losses (Langdale et al., 1991) and Stevens and Quinton report reductions of between 7 and 87%, with an average reduction of 52%. As a considerable fraction of phosphorus (P) lost from combinable crop fields is lost as particulate P, cover crops can also reduce P losses. In their review, Sharpley and Smith (1989) report that the majority of studies show reductions in the total phosphorus concentration of runoff. In a wheat cropping system in Texas, Sharpley et al. (1995) found that a sorghum cover crop reduced soil loss and associated particulate phosphorus, but dissolved and bioavailable phosphorus were greater with a cover crop. As this phosphorus is rapidly available to plants it could be important in eutrophication. Other pollutants have received little or no attention in relation to cover crops.

**Crop residues**

Crop residues can have benefits both when they are left on the surface and incorporated into the soil. Residues which are incorporated into the soil improve soil condition and infiltration. Left on the soil surface they 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.

Crop residues have been very successful at reducing losses of sediment. Stevens and Quinton (in press) give an average reduction in soil loss of 78%, with a range of 40–100% from the use of crop residues. Even small amounts of residue (12% cover (Meyer et al., 1970)) have the potential to greatly reduce soil losses although many studies have found greater reductions with increasing cover (e.g. McGregor et al., 1990, Meyer et al., 1970, Mostaghimi et al., 1992).

As crop residues decay they release nutrients, potentially leading to and increase in nitrate leaching. Thomsen and Christensen (1998) conducted a lysimeter study on sandy loam soils. They found volume-weighted nutrient concentrations of $NO_3$ and $NH_4$ increased by 16% and 41% respectively with corn residues applied at a rate of rates of 5 15 tons $ha^{-1}$compare to 5tons $ha^{-1}$. Increased phosphorus (Sharpley and Smith, 1991) and carbon (Schreiber, 1999) leaching losses have also been identified.

Nutrient losses in overland flow are reduced by crop residues. In a rainfall simulation study Torbert et al. (1999) found that initiation of surface runoff was delayed and loss of nutrients was reduced by up to 97% with surface-spread corn residue. Pesticide loss is overland flow is also reduced (Smith et al., 2002, Myers et al., 1995).

Soil moisture is increased by the retention of crop residues which increases losses of nitrous oxide (Jacinthe et al., 2002). Increased emissions of carbon dioxide (Velthof et al., 2002) and methane (Jacinthe and Lal, 2003) have also been observed.

**No-tillage**

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No-tillage or zero-tillage is where the soil surface is not disturbed prior to seeding and crop residues are left on the soil surface. Soils under no-tillage generally have increased infiltration as a result of higher organic matter, more stable aggregates (Zhang et al., 2007, Tebrugge, 1999), lower susceptibility to soil crusting (Tebrugge, 1999), and more soil faunal and microbial activity (58, Tebrugge, 1999). However, as the soil is not mixed, nutrients and agrochemicals accumulate at the soil surface (Karlen et al., 1998, Franzluebbers and Hons, 1996).

In a review of 28 studies Strauss (Strauss et al., 2002, 2003) found considerable variability in the effectiveness of no-tillage for reducing soil loss. Soil loss was changed by between 100 and -100% of that found in the conventional tillage treatment, with a mean reduction of 69%. The generally lower sediment losses associated with no-tillage result in a lower total phosphorus loss than conventional tillage. It is likely that pathogen load will also be reduced with reduced sediment losses (Tyrrel and Quinton, 2003). Despite these successes, dissolved P concentrations in overland flow are typically higher than in conventionally tilled areas (e.g. Gregory et al., 2005, Withers et al., 2006).

There is potential for no-tillage to encourage leaching losses. This is because better soil structure encourages infiltration – the converse of reducing surface runoff; however, this is not always translated into greater N leaching losses. For both DOC and TOC there appear to be no significant differences between no-tillage and conventional tillage (Owens et al., 2002, Brye et al., 2003).

Stevens and Quinton (in press) review 65 comparisons of no-tillage and conventional tillage in nine separate studies for five pesticides. The studies reviewed demonstrate 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 conventionally tilled plot.

Gas emissions have received relatively little attention and variability on a local scale makes them difficult to measure. Liu et al. (2006) found no significant difference between tillage types in a long-term continuous corn experiment in Colorado. $N_2O$ emissions have been found to increase with no-tillage in studies in both Scotland (Ball et al., 1999) and Argentina (Palma et al., 1997). Results for $CH_4$ emissions are mixed with both increases and decreases in emissions being reported (Ball et al., 1999, Liu et al., 2006, Hutsch, 1998, Gregorich et al., 2006).

Table 1 shows a summary of the impacts that different mitigation options have on different pollutants.

## Implications for policy makers and regulators

The evidence for pollution swapping is compelling, all of the mitigation options examined showed significant reductions in one or more pollutants but also showed increases in another pollutant. Comparing the relative importance of the different pollutants is a very difficult task. There is not currently a tool available which allows the relative impacts of different pollutants to be compared i.e. x% increase in greenhouse gasses to be compared to an increase of y% in nitrate leaching to groundwater. The impacts of these pollutants are on different scales and in different media; nitrate leaching may be important on a catchment scale, but the impact of greenhouse gas emissions is important on a global scale. Some pollutants may be more or less important from one scale to another. Even when contrasting pollutants that are more closely related, for example, those that impact upon water courses such as nitrogen and DOC, assessing their relative impacts is very difficult. In order for pollution swapping to be fully considered in policy making there is a need for a tool that can assess the impacts of different pollutant types on the environment. This tool would need to consider all of the potential impacts and evaluate the severity of each but, most importantly, it would need to provide a means of comparison. Economics provides the best potential for creating this tool through the use of market and non-market evaluation to calculate the cost of the mitigation option and the pollution. This could include market costs such as the cost to the farmer of introducing the mitigation options and account for crop productivity changes as well the costs associated with pollutants such water treatment costs. Non-market costs can also be included by considering the financial value people place upon non-market products such as biodiversity and carbon storage. By applying a monetary value to each mitigation option it would be possible to compare contrasting pollutants.

However, before such a tool can be created scientists need to address some of the considerable gaps in our knowledge. There are a number of areas of considerable uncertainty associated with different pollution mitigation options. The use of contour grass strips is one of those areas. These have received very little attention despite the potential for serving a duel function as beetle banks, which are currently funded under the UK Environmental Stewardship entry level scheme (ELS – EF option 7), and reducing diffuse pollution losses by reducing overland flow. Contour grass strips were excluded from the review of evidence for pollution swapping because there is insufficient research on a large number of the pollutants discussed. Some pollutants have also received little attention, in particular gaseous pollutants ($CH_4$, $CO_2$, $N_2O$ and $H_2S$) and DOC. In general there is a need for further research into the impact of mitigation measures on 'non-target' pollutants.

Even where mitigation options are promoted in legislation we do not have sufficient information on their impact on non-target pollutants. Cover crops are recommended as good agricultural practice in the 1991 EC Nitrates Directive but there are considerable gaps in our knowledge of the impact of cover crops on carbon, pesticide and pathogen losses as well as greenhouse gas emissions (Stevens and Quinton, in press). The 1991 European Nitrates Directive requires all member states to establish nitrate vulnerable zones (NVZs) where codes of good agricultural practice 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 (Ulen, 1997). There is a danger that encouraging the widespread use of a single mitigation option could lead to increases in non-target pollutants, in this case there is potential for an increase in losses of nitrate and bioavailable phosphorus

Employing a single "one size fits all" measure is also inadvisable. In different situations and locations some mitigations options may be more suitable than others. For many of the mitigation measures there was considerable variation in the impact of the measure on pollution levels. There is a need for more detailed understanding of the factors that influence the likelihood of success or failure of a mitigation measure and which measure is most likely to succeed in a given situation. Additionally some pollutants may be of greater concern in some areas than others. We need targeted management appropriate to local situation, but in order to achieve this, the people advising farmers need a good understanding of a broad range of issues. Decision support tools are needed to help with this process to a) assess the relative importance of different pollutants in different areas and b) assist in identifying the mitigation option that is most likely to successfully reduce the target pollutant without increasing others (given factors such as local hydrology, soil type, cropping regime). Such a tool would be extremely useful but we do not currently have all the information required to create it.

Although the focus of this paper has been on *pollution* swapping the problem for policy makers is broader than pollution alone. There is also a problem of 'ecosystem service swapping' where an ecosystem service is provided, but in manipulating the ecosystem to do this, a second ecosystem service is damaged. An example of ecosystem service swapping would be the provision of bare fallow for ground nesting farm birds (England and Wales Higher Level Stewardship (HLS) option HF17) which could lead to increased losses of soil and diffuse pollutants, especially if provided on sloping ground or near a water course.

Pollution swapping presents complex problems which present institutional challenges to both policy makers and researchers. For scientists this means taking a broader approach to research by working in interdisciplinary teams to consider the broader implications of mitigation options and ecosystem service provision. Interdisciplinary research presents many difficulties but these can be overcome with increased dialogue, open-mindedness and trust (Stevens et al., 2007). For policy makers it is common for different aspects of environmental pollution to be legislated by different government departments, divisions or organizations. This makes reaching a consensus even more difficult due to the need to balance contrasting objectives.

**Conclusion**

Pollution swapping has received relatively little research attention but has important implications for agricultural policy. There are considerable gaps in our knowledge of the impacts of mitigation options on non-target pollutants which need further research if we are to account for it in policy making. Without this information we cannot even begin to consider pollution swapping in policy making. Once we know the impacts of different mitigation options on both target and non-target pollutants we will be able to begin the process of developing tools to compare pollutants.

Management of diffuse pollution requires a holistic approach with integrated management of not only different diffuse pollutants, but all ecosystem services. As no single mitigation option can reduce all pollutant types this presents a considerable challenge and would require that pollutants were prioritised in given situations or regions. The next step toward this goal is a means to evaluate and compare the impacts of different pollutants.

Integrating pollution swapping, and ecosystem service swapping, into environmental policy is currently a long way off but we should begin the process of working towards it. The interdisciplinary approach and broad knowledge required will present us with considerable challenges but if we are to avoid management solutions that cause problems in the future we will need to look at ways of overcoming these.

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Table 1. Summary of the impact of different mitigation options on pollutants. + indicates an increase in the pollutant with the mitigation option, - indicates a reduction, +/- indicates mixed evidence, ? indicates no evidence and nc indicates no impact. A more detailed analysis of the evidence can be found in Stevens and Quinton (in press).

| | **Cover Crops** | **Crop Residues** | **No-till** | **Buffer zones** | **Constructe d wetlands** |
|---|---|---|---|---|---|
| **Suspended solids** | - | - | - | - | - |
| **N (leaching)** | - | ? | +/- | ? | ? |
| **N (overland flow)** | - | - | +/- | - | +/- |
| **P (leaching)** | + | ? | ? | ? | ? |
| **P (overland flow)** | - | ? | + | +/- | - |
| **Organic Carbon** | ? | ? | + | ? | - |
| **Pesticides** | ? | + | - | - | - |
| **Pathogens** | ? | ? | ? | - | ? |
| **CO2** | nc | + | ? | ? | ? |
| **CH4** | ? | ? | ? | ? | ? |