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A review of the efficacy of contemporary agricultural stewardship measures for ameliorating water pollution problems of key concern to the UK water industry

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Abstract

The UK water industry faces a number of water quality issues which mean that capital must be spent on treating raw water in order to meet regulatory standards. Moreover, other policies exist that require improved water quality (e.g. the Water Framework Directive) and contemporary regulation is encouraging water companies to deal with the problem at source, rather than relying exclusively on ‘end-of-pipe’ treatment solutions. Given that much of this pollution results from agricultural practices, agricultural stewardship measures could offer a means of source control. Although numerous schemes are available that encourage farmers to adopt environmentally friendly farming practices, uncertainty exists as to the specific impacts of these measures on water quality. The current study has, therefore, reviewed the scientific literature to establish those agricultural stewardship measures that have been proven to impact water quality for three pollutant groups of key concern to the UK water industry, namely dissolved organic carbon, nutrients and pesticides. It has been found that, whilst for many measures there is little or no evidence for impacts on water quality, a range of stewardship practices are available that have been proven to improve water quality. Their effectiveness is subject to a number of factors though (e.g. soil type and pollutant chemistry) and so they should be implemented on a case-by-case basis. Further research is needed to ascertain more fully how contemporary agricultural stewardship measures really do impact water quality.

Keywords: Agriculture; stewardship; water quality; dissolved organic carbon; nutrients; pesticides.

1. Introduction

Water may become polluted with a range of contaminants due to the use of land for agriculture (e.g. Hooda et al., 2000; Lovell and Sullivan, 2006). Of these pollutants, dissolved organic carbon (DOC) (Freeman et al., 2001; Holden, 2005; Wallage et al., 2006), nutrients (nitrogen (N) and phosphorus (P)) (Heathwaite et al., 1996; Haygarth and Jarvis, 2002; Dorioz et al., 2006) and pesticides (Environment Agency, 1999; Blanchoud et al., 2007; Garrod et al., 2007) represent the most significant issues for some land-owning UK water utilities due to the need to remove them from raw waters to meet regulatory standards. Whilst nutrients (Brett and Benjamin, 2008) and pesticides (Brack et al., 2007) also represent a direct ecological risk, DOC is problematic due to the formation of carcinogenic trihalomethane compounds during the chlorination process (Nieuwenhuijsen et al., 2008). Although a range of potential pollutant sources exist in addition to agriculture, including rural sewage treatment works, septic tanks (Ahmed et al., 2005; Gaddis et al., 2007) and amenity usage of pesticides (Knapp, 2005; Lapworth and Gooddy, 2006), agriculture is regarded as the key reason for their presence in UK waters (Defra, 2004).

The costs of treating these pollutants to meet drinking water standards is highly significant to water companies and ultimately paid for by the consumer. Pretty et al. (2000) estimated the costs of treating pesticides, nitrate, phosphorus (and sediment), and organic carbon (and sediment) in water for drinking in the UK to be £120 M, £16 M, £55 M and £106 M respectively. Monitoring and advice on pesticides and nutrients is estimated to cost a further £11 M per annum. In addition to drinking water standards, environmental standards are also imposed by the Water Framework Directive (WFD) (EC, 2000), which specifies that all waterbodies must be of good chemical and ecological status (or potential) by 2015 and that the costs of any clean-up should be charged to the polluter. Whilst the ecological impacts of chemicals in water (Ashauer et al., 2007; Brack et al., 2007; Gilliom, 2007) are known to result in additional economic losses, these cannot be calculated at present due to a lack of information (Pretty et al., 2000).

Agricultural pollutants can be treated to meet drinking water standards using engineered solutions, although as the costs can be significant, in both economic and environmental terms, control of these pollutants at source is desirable and a range of management techniques are available that aim to achieve this. These include measures that seek to reduce inputs of pollutants to catchment systems (e.g. reduced usage of chemicals), those that reduce the transport of pollutants from agricultural land (e.g. improved soil management) and others that aim to capture and degrade pollutants that have been transported towards waterbodies (e.g. buffer zones and wetlands). For a number of years, agri-environment schemes have been available to land managers in order that these measures can, theoretically, be implemented without compromising the financial viability of farm businesses. Recently (since 2005), agricultural stewardship has been pursued with renewed vigour due to the importance of controlling agricultural pollution and a number of highly significant policy developments have taken place, particularly Common Agricultural Policy (CAP) reform (Defra, 2005a) and the development of new agricultural stewardship schemes; Entry Level Stewardship (ELS) (Defra, 2005b) and Higher Level Stewardship (HLS) (Defra 2005c). These new policies that aim to control agricultural pollution offer opportunities for water companies to encourage implementation of measures on the ground that could reduce water pollution and, thus, result in capital and operational expenditure savings. At present, however, understanding of the impacts of these land management measures on water quality is uncertain. Whilst some recent work has been undertaken (Parry et al., 2006; Cuttle et al., 2007) this has not covered DOC and has only discussed pesticide pollution to a limited extent. Moreover, empirical evidence has not been thoroughly reviewed and modelling has been relied upon to determine some likely impacts on water quality. If water companies are to build these land management measures into their business plans then a sound knowledge of their impacts is urgently needed. The current review summarises peer-reviewed literature in order to develop a state-of-the-art understanding of the effects of contemporary agricultural stewardship measures on water pollution by DOC, nutrients and pesticides. This information could be used in the business planning of water companies and by other interested parties, such as Government and its agencies, as well as to guide future research in this area.

2. Dissolved organic carbon/water colour

Only catchments dominated by organic soils will generate DOC levels significant to the water industry (Holden et al., 2007a) and so it is only stewardship measures for moorlands that offer water companies an option for reducing DOC. Limited moorland options actually exist in current stewardship schemes and even less data are available to indicate their efficacy for improving water quality.

Some work has shown grip blocking to significantly (by up to 70 %) reduce DOC concentrations in some cases (Wallage et al., 2006; Armstrong et al., 2008) (Table 1). This could therefore offer water companies that take raw water from the uplands a means of controlling this significant problem. Many moorland areas in the UK have been drained (gripped), particularly during the 1960's and 70's, to increase agricultural productivity (Robinson and Armstrong, 1988). Damming these drains raises the water table, slows peat degradation and reduces the transport of DOC (and therefore water colour) off-site (Holden et al., 2007a; Worrall et al., 2007). Effects on the composition of the DOC are uncertain with Wallage et al. (2006) reporting more colour per unit carbon, indicating an increase in humic substances, but Armstrong et al. (2008) showing more easily treated colour. Grip blocking may not always result in decreased DOC/colour contamination however. In some cases DOC may increase after blocking (Worrall et al., 2007) and in others the peat may not necessarily recover its original physical and chemical properties (Freeman et al., 2001; Holden et al., 2006; Wallage et al, 2006; Holden et al., 2007b).

Further research is needed if water companies are to be able to pursue other catchment management measures available in stewardship schemes with the expectation of reducing DOC contamination of streams. Holden et al. (2007a) comment that virtually nothing is known about the impacts of moorland burning on water quality and soil hydrology, although a number of papers have eluded to the fact that increased burning will lead to higher levels of water colour (Mitchell and McDonald, 1995; Garnett et al., 2000). A study at Moorhouse in the northern Pennines showed that severe burning reduced the water holding capacity of the soil and created a more flashy hydrograph (Robinson, 1985), factors that could increase the generation and delivery of DOC to surface waters. Burning also leads to increases in the amount of heather that is present and this has subsequently been shown to increase the density of soil pipes, which move runoff from soils to streams, lower the water table and increase the generation and flux of colour to surface waters (Holden, 2005). Data describing the impacts of livestock grazing on water colour are almost entirely lacking from the literature, although one study found there to be no significant difference between soil water colour in grazed and ungrazed plots (Worrall et al., 2007).

3. Nutrients

In comparison to DOC/water colour, water companies may select from a much wider range of agricultural stewardship options which may reduce pollution of waterbodies by nutrients. A number of these would require that utilities work with farmers to reduce inputs of fertilisers into catchment systems. Limiting nitrogen additions to crop requirements (Lord and Mitchell, 1998; Coelho et al., 2006, 2007) or quantities specified in Nitrate Vulnerable Zone (NVZ) regulations (Vertés et al., 1997; Lord et al., 1999; Hanegraaf and den Boer, 2003) have been found to reduce water pollution substantially (Table 1). Nitrate losses have been reduced to 10 kg ha^{-1} (Goulding et al., 2000) and leaching to groundwater (1 m depth) by 57 % using this mechanism (Lord and Mitchell, 1998). Whilst impacts on nitrogen compounds have been desirable, phosphorus concentrations in runoff will be affected to a much lesser extent due to their build-up in soils however (Stålnacke et al., 2003, 2004). It has, therefore, been suggested that 10 years would be needed to see a reduction in dissolved phosphorus whilst a number of decades would be required in order to observe a decline in particulate-associated phosphorus concentrations reaching waters (Withers et al., 2001; Haygarth et al., 2002). In some case, reductions in nutrient losses to water have been negligible, however, due to soil type, crop and prevailing hydrological conditions (Dukes and Evans, 2006; Harmel et al., 2006; de Ruijter et al., 2007), leading some workers (Macgregor and Warren, 2006; Schröder et al., 2007) to be sceptical of the benefits of these measures as many farmers claim already to be applying nitrogen below specified limits and yet water pollution is still occurring.

Other measures aim to reduce nutrient concentrations in water not by reducing inputs to catchments but by changing the way in which they are applied. The injection of slurry, rather than broadcast spreading, has resulted in reductions of 93, 82 and 94 % of dissolved reactive P (DRP), total P (TP) and algal-available P (AAP) in runoff (Daverede et al., 2004). Moreover, nutrient losses from poultry litter were reduced by 80-95 % (Pote et al., 2003) whilst incorporation of inorganic fertilizers has been found to reduce nutrient losses to the water environment to background levels (Pote et al., 2006). Where tile drains are present losses may be greater though (Coelho et al., 2007), highlighting that implementation of stewardship measures needs to be carried out on a site-specific basis. Other fertiliser-specific measures are available for implementation (i.e. not allowing runoff from in-field manure heaps, not applying organic fertilisers when the soil is saturated and not applying manure within 10 m of a surface water and within 50 m of a borehole) although these demonstrate the dearth of scientific evidence for the impacts of many measures on water quality.

Some specific soil management measures have also been proven to be effective at reducing nutrient pollution. Planting a green cover crop is one of the single most effective ways of decreasing the risk of nitrate leaching (Shepherd et al., 1996) and, in general, cover crops lead to a 50 % reduction compared to a winter-sown cereal (Goss et al., 1988; Shepherd et al., 1993; Lord et al., 1999). Good establishment before the start of drainage is key to getting the most from a cover crop and uptake of N can actually range between 10-150 kg ha^{-1} (Fielder and Peel, 1992; Shepherd, 1999).

Ensuring a rough soil surface by ploughing or discing is another soil management measure which can have a useful, but variable, impact on nutrient transport (Angle et al., 1993; Rasmussen, 1999; Benham et al., 2007). The transport of soluble P in surface runoff may be reduced by a factor of 2-3 compared to an untilled surface (Zeimen et al., 2006) although some workers have found that nitrate leaching is unaffected (Stoddard et al., 2005) due to site-specific factors (Rasmussen, 1999). Farmers may also be able to help water companies by working fields along the contour and Withers et al. (2006) found no significant differences in runoff quantity, sediment and total P concentrations where tramlines ran across-slope compared to areas without tramlines. Schonning et al. (1995) also compared the effects of the direction of drilling (winter wheat) on runoff, soil loss and total P for two sandy Danish soils. Reductions of 9 %, 13 %, and 12 % (Site 1) and 19 %, 58 %, and 57 % (Site 2) were reported for runoff volume, suspended solids and total P losses respectively. Even if the direction of traffic is unaltered, conservation tillage techniques can have significant impacts on nutrient losses to water. Mean losses in surface runoff were reduced by 63, 67, 46 and 49 % for total nitrogen, total Kjeldahl nitrogen, ammonia and nitrate respectively whilst reductions for total phosphorus and orthophosphate were 73 and 17 % (Benham et al., 2007). Winter N losses from drained plots at Brimstone Farm averaged 24 % less from land that had been direct drilled instead of ploughed (Goss et al., 1988). A comparison of concentrations of sediment and P in runoff from the Greensand and Chalk soils showed them to be consistently lower when the soil was minimally tilled rather than ploughed (Withers et al., 2007), with the benefits of reduced cultivation being attributed to better surface cover and a firmer surface for tractor wheelings. Impacts of reduced tillage on soil macroporosity (which has significant implications for nutrient transport) have been noted, with Schjonning and Rasmussen (2000) demonstrating a smaller volume of macropores in the top 20 cm of soil compared to a ploughed treatment. Johnson and Smith (1996) also found that shallow cultivation, rather than ploughing, decreased N leaching by 44 kg N ha^{-1} over a five-year period but that the difference between cultivation types diminished over time. Conversely, some research has shown that minimum tillage can actually increase nutrient pollution. Carter (1998) reviewed a large number of studies carried out on a range of soil types and found that, whilst the technique was effective in reducing particulate associated P in 31 % of studies, no effect occurred in 8 % and increased P loss actually resulted in 23 % of cases. The same study also showed that conservation tillage increased leaching volumes and nitrate loss to groundwater. Whilst some work has shown that direct drilling decreases soil macroporosity, other studies (Shipitalo et al., 2000; Petersen et al., 2001) reported that the most effective way of reducing macroporosity was intensive cultivation (i.e. ploughing) and that conservation tillage increases transport through macropores, partially attributable to the increased activity of earthworms (Edwards and Lofty, 1982). The build up of nutrients as a consequence of surface applications and limited mixing associated with reduced cultivation has been reported (Rasmussen, 1999), particularly in grassland soils (Haygarth and Jarvis, 1999)

A number of livestock management techniques have been proven to reduce nutrient pollution. A significant relationship has been reported between grazing intensity and N losses to water (Huging et al., 1995) and, under extensively managed pasture, N leaching losses were reduced by 69 %. Limiting overgrazing through careful management can, therefore, have significant benefits for the water environment. More heavily grazed fields usually receive higher levels of fertiliser, however, and it can be hard to separate these two factors (Cuttle et al., 2004). It is also possible that nutrient losses could still be significant from pasture where overgrazing is not occurring but where stocking densities remain high. Similarly, limiting soil poaching by grazing of saturated soils and not locating supplementary feeding sites on poorly drained areas can significantly improve runoff quality. Using exclusion cages, Kurz et al. (2006) demonstrated the effect of cattle on soil physical properties and nutrient losses in overland flow. Grazed areas were characterised by 57–83 % lower macroporosity, 8–17 % higher bulk density and 27–50 % higher resistance to penetration than areas from which the cattle were excluded. Increased concentrations of total N, organic P and potassium (K) were measured in surface runoff from the grazed areas. Other workers have reported high P losses in land drainage that could only be attributed to heavy winter sheep grazing, with concentrations in drain waters reaching up to 20 mg P l^{-1} and nearly a third of the total annual P loss occurring during one month immediately after the sheep had been grazing the study site (Jordan and Smith, 1985). In another study, the effect of different grazing pressures on P export in surface runoff generated after artificial rainfall events resulted in 2, 7.6 and 291 mg total P m^{-2} loss for ungrazed, lightly grazed (4 stock ha^{-1}) and heavily grazed land (>15 stock ha^{-1}), respectively (Heathwaite and Johnes, 1996).

In some instances water companies may be able to encourage farmers to take certain actions through the provision of capital grants. Unpublished research by Kay et al. in the Ingbirchworth catchment in South Yorkshire (one of Defra’s Associate Catchment Senstive Farming pilot projects) has indicated that farmers would be much more likely to install fencing to exclude livestock from watercourses if supported financially. Parkyn et al. (2003) reported that streams in New Zealand within fenced-off areas showed rapid improvements in visual water clarity and channel stability, although nutrient and faecal contamination responses were actually variable and significant changes in macroinvertebrate populations were not apparent. Soluble reactive phosphorus decreased by up to 33 % in some streams but was found to increase by up to 20 % in others. Similarly, total N decreased by up to 40 % in some fenced-off streams but increased by up to 31 % in others. More positively, when a fenced-off area of 335 m length and 10-16 m width was created to stop dairy cattle entering a North Carolina stream, total organic nitrogen, Kjeldahl nitrogen and total phosphorus were reduced by 33, 78 and 76 % respectively (Line, 2003). Further encouragement can be provided, particularly on tenanted land, to provide water troughs so that cattle do not have to drink from streams (Sheffield et al., 1997). In this study total phosphorus concentrations were reduced by 54 %, whilst total nitrogen concentrations fell by 81 %.

The installation of ‘edge of field’ measures (i.e. buffer zones and wetlands) could potentially offer significant water quality gains to water companies. A number of management issues need to be considered for buffer zones as Table 2 shows that their effectiveness for reducing concentrations of nutrients in surface waters is very variable and actual operational efficiency will be highly season and location specific. Important factors include soil properties, climate, vegetation cover, physical dimensions, sediment characteristics and the presence of underdrainage (Barling and Moore, 1994; Tate and Nader, 2000). Unfortunately, the maximum delivery period of nutrients (i.e. winter) (Uusi-Kämppä et al., 2000) overlaps with the least efficient period for many buffer zones due to a combination of high local water tables, reduced infiltration capacities and poor plant growth/cover. The highest rates of suspended solids deposition (and therefore particulate associated phosphorus) occur in the upper part of the buffer strip, and retention rates decline with increasing width when expressed as an amount per unit area (i.e. g m^{-2} y^{-1}). Poor filtering efficiency of the finest material may be an issue however (Le Bissonnais et al., 2004; Owens et al., 2007), especially because this represents the most reactive and preferentially enriched soil fraction (Syversen and Borch, 2005).

Recommended widths range from 3-200 m (Castelle et al., 1994) although 5-15 m is most common and Haycock and Burt (1993) reported that the majority of nitrogen capture occurred in the first 5-8 m. Long-term management is a key issue - Dorioz et al. (2006) state that the retention of phosphorus is unlikely to be sustained and that dissolved phosphorus release from the buffer zone will increase. Lovell and Sullivan (2006) note a host of more wide-ranging limitations of buffer zones for treating nutrients in runoff, including a lack of catchment-scale research, a need for more clearly defined and targeted goals, a lack of cooperation between scientific disciplines and agencies, an absence of accountability from landowners for investment in buffers, as well as limited attention to the aesthetic quality of buffers. It is perhaps somewhat surprising that such a recent review is still raising what are rather basic issues.

Wetlands have often been shown to be very effective at removing nutrients from runoff (Table 2), although operational efficiencies again vary seasonally and with time. For example, seasonal removal percentages of nitrate by a wetland were 100, 35, 55 and 96 % of the autumn, winter, spring and summer loads respectively, with a total removal of 55 % (Larson et al., 2000). Generally, the efficiency of wetland systems is reduced during high flow periods when retention times are shorter Koskiaho et al. (2003). Whilst there are other examples which appear to operate well (Jansson et al., 1998; Koskiaho et al., 2003), there are also others which do not (Wedding, 2000; Braskerud, 2002). The ratio wetland:catchment area is often used as an indicator of retention capacity and whilst wetland size is recommended to be 1-5 % of the contributing catchment (Kadlec et al., 2000), many ponds and constructed wetlands are often <0.3 % (Braskerud, 2002). These authors argue that, unlike buffer strips, wetlands are more effective at retaining the finer clay-sized material with the mean annual retention of suspended solids being 57-71 %. Despite the fact that much information is available on the impacts of some stewardship measures for nutrients, none is available for many.

4. Pesticides

A wide range of measures exists within contemporary agricultural stewardship schemes that seek to reduce pesticide pollution by limiting their input into catchment systems. Some of these have been proven to have very significant impacts (50-100 % reduction in concentrations in runoff and surface waters) (Table 1), including not spraying when surface runoff is likely to be generated or enter land drains (Barnes and Kalita, 2001; CPA and AIC, 2004). Measures to reduce spray drift can also be highly effective at reducing pesticide pollution of water bodies and it has been shown that drift can be reduced by between 20 and 50 % using core-tipped rather than flat nozzles (de Snoo and de Wit, 1998) whilst band spraying may reduce drift by 90 % (van der Zande et al., 2001). Windbreaks (e.g. miscanthus) can also reduce drift significantly; a wind-break that was 0.5 m above the crop (sugar beet) reduced drift by 80 % and when this height was raised to 1 m then drift was further reduced to 90 %. Moreover, biobeds offer a very effective means of combating pesticide pollution by degrading residues in waste and washings by over 98 % in some instances (Fogg et al., 2004; Spliid et al., 2006). In contrast, taking measures to reduce reliance on pesticides would seem to have a negligible effect on water pollution. Of the limited evidence that is available (Pacini et al., 2003; Hole et al., 2005) losses from farms with reduced inputs appear to be similar to those from conventional farms. Sheep dip pollution may be combated by disposing of spent sheep dip to land or farming organically. The effectiveness of the first measure will depend on the physico-chemical properties of the compounds used and the characteristics of the land disposed to (Grant et al., 2002; Cooke et al., 2004; Levot, 2007). Appropriate siting of dip disposal areas is, therefore, critical and detections of sheep dips in watercourses have previously been attributed to poor citing (Virtue and Clayton, 1997). No studies have quantified the impacts of organic sheep farming on pesticide pollution of the water environment to date. A further input reduction measure available to water companies is reversion of arable land to grassland which has been shown to reduce pesticide application to land generally (Herzog et al., 2006).

A range of measures are available that may reduce pesticide transport to watercourses through improved soil management and it is well documented that higher levels of organic matter encourage sorption of certain pesticides and reduce their mobility (Ding et al., 2002; Hernandez-Soriano et al., 2007). Other factors are also important though, including the properties of a substance, the clay content of the soil, the pH of the soil solution, and the coverage of ion exchange sites (Delle Site, 2001; Beulke and Brown, 2006). Facilitated transport due to an increase in the DOC and colloidal content of soil water may actually lead to the increased mobility of pesticides however (Worrall et al., 1995; Li et al., 2005). Organic amendments may also alter the pH of the soil solution and, therefore, the degradation rate of pesticide residues, the degradation rate of carbofuran being reduced for example (Worrall et al., 2001). Whilst previous studies have shown that conservation tillage reduces runoff generation and soil erosion, the fate of pesticides is less certain (Uri, 1998; Rose and Carter, 2003; Ghidey et al., 2005). Although overall delivery to waterbodies will be reduced by at least an order of magnitude due to runoff production and sediment transport being lower than in conventional production systems, pesticide concentrations will be higher in both the aqueous and particulate phases under minimum-tillage due to the smaller quantities of runoff in which residues will be present. This may not be a significant issue at the catchment scale, however, as pesticides will be diluted in streams and if mass losses from land are actually lower under minimum-tillage then stream concentrations may be lower (Kenimer et al., 1987; Tebrügge and Düring, 1999; Shipitalo and Owens, 2006). The build up of soil macropores in no-till systems may be problematic though and increase pesticide losses (Smith and Chambers, 1993; Tebrügge and Düring, 1999; Holland, 2004). Ensuring the presence of a rough soil surface will limit the mobility of pesticides in the environment as a finer soil tilth increases a soil's water holding capacity and, thus, reduces runoff production and pesticide movement (Brown et al., 1999; Hyer et al., 2001). Tillage of the soil surface by discing or ploughing will also disrupt macropores in the soil and so reduce pesticide transport by encouraging the transfer of solutes from macropores to micropores (Jarvis et al., 1994) and reducing the connectivity of desiccation cracks with land drains (Kay et al., 2004). Current agricultural stewardship schemes are likely to do little to reduce pesticide transport to waterbodies via this mechanism, however, as tillage is only encouraged following harvest. Whilst this practice may be useful for reducing soil erosion and transport of nutrients in the post-harvest period when soils are relatively bare, pesticide application will take place at different times prior to this cultivation. It is well known that the most significant pesticide transport usually occurs in the first period of runoff generation after application, before much time has elapsed for degradation to take place and sites available for chemical sorption in the soil may be saturated (Ng and Clegg, 1997; Kamra et al., 1999; Zehe and Flühler, 2001). In order to have a significant impact on pesticide transport, tillage would have to be carried out repeatedly whilst the crop was growing and pesticides were being applied.

As for nutrients, buffer zones and wetlands can have a significant impact on the environmental fate of pesticides although it is generally accepted that only a limited amount of empirical research has been carried out (Harris and Forster, 1997; Andreoli and Tellarini, 2000; Kleijn et al., 2001). In the context of many of the measures advised under agricultural stewardship schemes, however, a considerable body of research is actually available and a number of studies have highlighted the importance of buffer strips as a management technique for limiting surface water pollution by pesticides (Klöppel et al., 1997; Patty et al., 1997; Dabrowski et al., 2002). Specific changes in pesticide mass losses and concentrations due to the creation of buffer zones are shown in Tables 3 and 4 respectively. Strongly sorbed compounds have been found to require a buffer zone of only several metres to be trapped, with greater width having little additional effect. For hydrophilic compounds a more linear relationship has been reported, where greater width increases the chances of the pesticide being retained and degraded (Krutz et al., 2005). Those studies reported in Tables 3 and 4 have generally employed buffer zones of 5-20 m. Other work has addressed the issue of buffer zone size by comparing this to catchment area and Arora et al. (2003) found that small buffer zones (30:1 ratio between drainage area and buffer strip) were just as effective as larger ones (15:1 ratio). Of key importance to the water industry is the fact that research that has been carried out to-date is of limited use in determining the effectiveness of buffer zones from improving water quality at the catchment scale (and therefore treatment works). Although some studies have investigated the fate of pesticides in wetland systems this subject area is not understood as well as for nutrients and sediment (Schulz and Peall, 2001). Some studies have shown that wetlands reduce mass losses of pesticides by 25-100 % (Table 5). The size of a wetland relative to the catchment from which it is receiving runoff is a key issue when considering the use of wetlands for treatment of pesticide residues in runoff. Constructed wetlands on farms covering

1 % of the catchment area have reduced pesticide concentrations reaching water bodies to non-toxic levels through sorption and degradation in the wetland (Braskerud and Haarstad, 2003). Some studies have indicated that pesticides are totally degraded in wetland systems rather than simply stored as analyses of sediments have proved to be negative (Chapman, 2003). Despite much positive data, other studies have found that wetlands do not offer an effective way of stripping pesticides from runoff. High concentrations of atrazine, metolachlor and chlorpyriphos (2.5, 0.25, and 1 mg l^{-1} respectively) were not degraded at all in one particular study (Mazanti et al., 2003), although at lower concentrations (2, 0.2, and 0.1 mg l^{-1}) some loss was observed, with detection of degradation products showing that breakdown of the compounds was occurring rather than sorption alone. The structure of a pesticide is important in determining whether it will be effectively removed from water in a wetland system; structures based on nitrogen compounds being degraded most effectively (Fogg et al., undated).

5. Conclusion

The current project has sought to elucidate those agricultural stewardship measures that can be implemented in river catchments with reasonable certainty, based on scientific findings, that improvements in water quality will result, focussing on pollutants of key concern to the UK water industry, namely, dissolved organic carbon, nutrients and pesticides. Whilst those measures detailed in Table 1 have been proven to improve water quality the success of all of these will be site specific due to factors such as soil type, hydrology and pollutant chemistry and so measures should be implemented on a case-by-case basis. Moreover, there is a dearth of information quantifying the impacts of many stewardship measures on water quality, which is perhaps not surprising given that many were developed for terrestrial ecology gain rather than from a water quality perspective. It is highly pertinent to note that no studies have been undertaken to date that have quantified the impact of agricultural stewardship measures at the catchment scale, those that have been carried out have focussed on the plot and individual field scale, and further research in this area is, therefore, urgently needed. It is likely to be important to implement a range of measures throughout an entire catchment (dependant upon farming practices in the catchment) in order that benefits are not negated by areas where new management techniques have not been pursued (Kay et al., 2005). A further pertinent point to be considered when implementing stewardship measures in a catchment is that farmers/land managers have to be given responsibility for implementing certain measures (e.g. controls on N application rates and timing) and it is, therefore, essential that they are adequately trained and can be relied upon to carry out the task effectively. Moreover, research that quantifies the impacts of agricultural stewardship on farm incomes is largely lacking and is urgently needed if farmers/land managers are to be convinced that environmental stewardship represents business sense. Overall, despite significant attention from many stakeholders, there is a striking lack of scientific evidence to underpin the use of agri-environment measures for water quality management. This may limit their usage by businesses, such as the water industry, which are required to make steadfast decisions based on sound economics.

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Table 1. Stewardship measures available in contemporary agri-environment schemes that have been proven to reduce water pollution by dissolved organic carbon, nutrients and pesticides.

Pollutant	Measures scientifically proven to improve water quality
Dissolved organic carbon/water colour	Block grips and gullies
Nutrients	Limit nutrient application to crop requirements
	Limit total N from manures to 170 kg ha^{-1} yr^{-1} (arable) and 250 kg ha^{-1} yr^{-1} (grassland)
	Arable reversion to grassland
	Inject slurry or incorporate soon after application
	Do not apply dirty water to high-risk areas
	Ensure soil is bare for a minimum of time
	Traffic fields across slope
	Use direct drilling
	Avoid poaching
	Limit overgrazing
	Limit livestock access to watercourses
	Buffer zones
	Wetlands
Pesticides	Do not apply when land is frozen, saturated or rain is forecast in next 3 days
	Do not apply when pesticides may enter land drains
	Reduce spray drift
	Use a biobed
	Dispose of spent sheep dip to land

Arable reversion to grassland

Increase and maintain soil organic matter

Ensure soil is bare for a minimum length of time

Use direct drilling

Buffer zones

Wetlands

Table 2. Nutrient removal efficiencies for buffer zones and wetlands.

Pollutant	**Effect of buffer zone**	**Reference**	**Effect of wetland**	**Reference**
Total nitrogen	23 % reduction	McKergrow et al., 2003	5-50 % reduction	Alström et al., 2000
	75-94 % reduction	Heathwaite et al., 1998	19-100 % reduction	Jansson et al., 1998
	10 % decrease – 217 % increase	Borin et al., 2005	3-15 % reduction	Braskerud, 2002
	47-100 % reduction	Dorioz et al., 2006	7 % increase – 40 % decrease	Koskiaho et al., 2003
Nitrate	50-100 % reduction	Haycock and Burt, 1993	8 % increase – 38 % decrease	Koskiaho et al., 2003
	No impact (due to macropore flow)	Leeds-Harrison et al., 1999	28 % reduction	Kovacic et al., 2006
	9 % decrease – 232 % increase	Borin et al., 2005	35–100 % reduction	Larson et al., 2000
	95 % reduction	Hefting and De Klein, 1998		
Total phosphorus	6 % reduction	McKergrow et al., 2003	6 % increase – 72 % decrease	Koskiaho et al., 2003
	10-98 % reduction	Heathwaite et al., 1998	53 % reduction	Kovacic et al., 2006

	0-97 % reduction	Uusi-Kämppa et al., 2000		
	31 % reduction	Abu-Zreig et al., 2003		
	60-80 % reduction	Vallières, 2005		
	8-97 % reduction	Dorioz et al., 2006		
	27 % decrease – 41 % increase	Borin et al., 2005		
Soluble phosphorus	16 % reduction	Vaananen et al., 2006	<10 % reduction	Braskerud, 2002
	61 % increase	McKergrow et al., 2003	12-31% reduction	Wedding, 2000

Soluble phosphorus cont.	**Effect of buffer zone**	**Reference**	**Effect of wetland**	**Reference**
	17 % decrease – 475 % increase	Borin et al., 2005	33 % increase – 33 % decrease	Koskiaho et al., 2003
	0-30 % decrease	Dorioz et al., 2006		

Table 3. The effect of buffer zones on mass losses of pesticides to waterbodies.

Pesticide	**Effect of buffer zone**	**Reference**
Atrazine	30 % reduction	Barnes and Kalita, 2001
	83-99 % reduction	Patty et al., 1997
	57-93 % reduction	Popov et al., 2006
Fenpropimorph	71 % reduction	Syversen and Bechmann, 2004
	34 % reduction	Syversen, 2005
Glyphosate	39 % reduction	Syversen and Bechmann, 2004
	48 % reduction	Syversen, 2005
Isoproturon	87 % reduction	Benoit et al., 2000
Lindane	76-100 % reduction	Patty et al., 1997
Metolachlor	40-85 % reduction	Popov et al., 2006
Propiconazole	63 % reduction	Syversen and Bechmann, 2004
	85 % reduction	Syversen, 2005

Table 4. Changes in pesticide concentrations in runoff due to the creation of buffer zones.

Pesticide	**Effect of buffer zone**	**Reference**
Atrazine	53 % reduction	Arora et al., 2003
	25-49 % reduction	Popov et al., 2006
Chlorpyriphos	83 % reduction	Arora et al., 2003
Metolachlor	54 % reduction	Arora et al., 2003
	30-61 % reduction	Popov et al., 2006

Table 5. Changes in mass losses of pesticides to surface waters due to the construction of wetlands.

Pesticide	**Effect of wetland**	**Reference**
Atrazine	25-95 % reduction	Stearman et al., 2003
Azinphosmethyl	77-93 % reduction	Shulz and Peall, 2001
Carbaryl	43 % reduction	Chapman, 2003
Chlorpyriphos	100 % reduction	Shulz and Peall, 2001
	100 % reduction	Chapman, 2003
	47-65 % reduction	Moore et al., 2002
Diazinon	85 % reduction	Chapman, 2003
Dimethoate	100 % reduction	Chapman, 2003
Endosulphan	100 % reduction	Shulz and Peall, 2001
Metolachlor	82 % reduction	Stearman et al., 2003
Simazine	77 % reduction	Stearman et al., 2003

Captions

Table 1

Stewardship measures available in contemporary agri-environment schemes that have been proven to reduce water pollution by dissolved organic carbon, nutrients and pesticides.

Table 2

Nutrient removal efficiencies for buffer zones and wetlands.

Table 3

The effect of buffer zones on mass losses of pesticides to waterbodies.

Table 4

Changes in pesticide concentrations in runoff due to the creation of buffer zones.

Table 5

Changes in mass losses of pesticides to surface waters due to the construction of wetlands.