Chapter 77. Land Use: Managing the impacts of agriculture on catchment water quality

(Original title 'Land Use: Management as watersheds')

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1 Synopsis

Poor water quality in lakes, rivers, estuaries and groundwater can be attributed to indirect effects of agricultural land use. Land management practices that lead to excess nutrients, sediment, pathogens and agrochemicals reaching waterbodies, or that negatively affect the hydrology, habitat and structure of waterbodies, can degrade water quality. This chapter describes farm and landscape management practices that can potentially prevent or mitigate adverse water quality impacts. Some examples of improved catchment water quality in response to changes in agricultural management are reported. Challenges associated with collecting evidence of the effects of agricultural management on water quality at the catchment scale, the potential for pollution-swapping, the cost-effectiveness of mitigation practices and approaches to governance of water quality management are also discussed.

2 Keywords

catchment, watershed, agriculture, water quality, mitigation measures, Best Management Practices (BMP), conservation, nitrogen, phosphorus, erosion, suspended sediment, lag time

3 Agricultural catchment water quality

3.1 Surface water quality problems

Fresh water is a scarce and valuable resource (FAO, 2011). Conservation and equitable distribution of freshwater is therefore critical to sustaining ecosystem services and global food production (Rockstrom *et al.*, 2009) (see Chapter 83).

Agriculture uses 70% of global freshwater and so optimising water use through advanced irrigation, farm and food processing systems is paramount to meeting the globally increasing demands for food, particularly in the face of a changing climate (see Chapters 86, 87 and 241). As well as the quantity of available freshwater, all human activities affect the quality of freshwater resources, and degradation of water quality in turn increases the scarcity of freshwater (Peters and Meybeck, 2000). Impaired water quality also limit ecosystem services, and human welfare and livelihood (Ongley, 1996; FAO, 2011). Poor water quality can cause loss of aquatic and riparian biodiversity, ecosystem stability and recreation value, poor human health (e.g. due to unsanitary drinking water and toxins from harmful algal blooms), physical disruption to water supply systems, shellfish contamination, fish kills and reduced aquaculture production (Carpenter *et al.*, 1998; Cheng and Kimble, 2001; Schindler, 2006; Withers and Haygarth, 2007; Kay *et al.*, 2009).

Eutrophication and sedimentation are two common processes leading to water quality degradation. Eutrophication is an increased rate of organic matter supply to a waterbody which can lead to excessive algal growth, species composition changes, taste and odour problems, changes in aesthetics, and oxygen depletion when algal blooms decompose. Both natural and human-induced processes lead to hypoxic (reduced oxygen) and anoxic (no oxygen) conditions in waterbodies (Rabalais *et al.*, 2010). The main cause of eutrophication in water-bodies is the over-supply of nutrients. Nitrogen supply commonly limits eutrophication in marine waters and phosphorus supply commonly limits eutrophication in freshwaters. However co-limitation or the reverse scenarios also occur (Boesch *et al.*, 2001). Sedimentation is the deposition of suspended soil and other particulate matter on river, lake and estuary beds. Accelerated levels of suspended and deposited sediment can disturb habitats for macroinvertebrates, aquatic flora and fish spawning (Donohue and Molinos, 2009). Other significant causes of water quality degradation include accumulation of pesticides, other persistent organic pollutants and heavy metals, and changes in salinity, pH, thermal regime and hydromorphology (i.e. water body structure, habitat and hydrological processes) (Tognetti and Lawrence, 2002).

3.2 Impacts of agriculture

Detrimental water quality impacts in rivers, lakes, groundwater and coastal waters have been attributed to impacts from agriculture, forestry, urban and industrial land use (Carpenter *et al.*, 1998). During the 1980s and 1990s there was considerable success in reducing nutrient inputs to waterbodies from human wastewater and industrial discharge such that agricultural inputs have become a higher total proportion of overall inputs in many cases (Schindler, 2006; Kronvang *et al.*, 2008). This is the case in Lough Neagh in Northern Ireland where the dominant source of phosphorus gradually switched from urban discharge to agricultural inputs after controls on urban discharge were introduced in the 1980s (Foy *et al.*, 2003). Globally, agricultural ‘grey’ water (polluted freshwater) volumes are estimated to be large (53%) compared with industrial (26%) and domestic (20%) users of ‘blue’ (surface and groundwater) and ‘green’ (consumed rainwater) water (Mekonnen and Hoekstra, 2011). Current nitrogen use is estimated to far exceed the planet’s boundaries of sustainability and phosphorus transfer from land to water has almost reached the planet’s functionally sustainable threshold (Rockstrom *et al.*, 2009).

Pressures (i.e. stressors on the environment) on waterbodies in agricultural catchments include enhanced losses of nutrients and eroded soil, changes to the natural hydrologic regime, increased losses of agrochemicals, pathogens and organic compounds, and acidification (Carpenter *et al.*, 1998; Blann *et al.*, 2009; Quinton *et al.*, 2010; Carpenter *et al.*, 2011; Foley *et al.*, 2011).

In Europe, pressures from agriculture are significant in 40 % of rivers and coastal waters, and in one third of lakes and transitional waters (EEA, 2012). Brown tides in China have been linked to turbidity, dissolved organic carbon and metals exports which have been partly linked to intensification of agriculture (Gobler *et al.*, 2011; Qiu, 2012; Zhang *et al.*, 2012a). In the Aral Sea basin in Central Asia, over-extraction of water for irrigation caused a 75% decline in the lake volume between 1960 and 1995, and an increase in salinity of the lake and land leading to large-scale abandonment of agricultural land use and poor life expectancy, health and drinking water supply (www.fao.org/nr/water/aquastat, Accessed 18/11/2012). Other examples of where impaired water quality has been linked to agricultural land use include the Baltic Sea in Europe (Gustafsson *et al.*, 2012), Chesapeake Bay in the USA (Simpson, 2010), and the Great Barrier Reef in Australia (Waterhouse *et al.*, 2010).

4 Technical options for water quality mitigation

A range of stewardship approaches, including engineering solutions that treat the symptoms as well as management changes that minimise the pressures, are required to mitigate against anthropogenic pressures on the planet's resources (Steffen *et al.*, 2011). In agriculture, practices aimed at mitigating environmental degradation are referred to interchangeably as 'mitigation measures', 'stewardship approaches' and 'best, recommended, conservation or sustainable' management practices and can be classified as cultural or structural measures. Cultural measures (e.g. fertiliser application rate, form, placement and timing) are land management practices which modify the spatial and temporal availability of nutrients and pollutants for mobilisation and transport to waterways. Structural measures (e.g. slurry storages, riparian fences and vegetation) are those which modify the pathway of nutrients, pollutants and water to a receiving water body. A range of structural and cultural mitigation measures can be applied at different stages of the link between land management and water quality impact. These stages can be conceptualised as a transfer continuum for materials that are mobilised from sources via pathways of transport and later delivered to a waterbody where they may cause an ecological impact (Haygarth *et al.*, 2005). Figure 1 highlights key components of the nutrient transfer continuum for nutrients. Many mitigation measures have benefits for both the off-farm environment and the farm system itself by sustaining the soil, animal, landscape and water resource through, for example, increased output from similar or lower nutrient use, reduced soil compaction and erosion and enhanced on farm water quality (Ridley, 2005; Gourley *et al.*, 2007; Zeckoski *et al.*, 2007; Simpson *et al.*, 2011; Soane *et al.*, 2012). Mitigation measures that target nutrient, agrochemical, sediment and pathogen losses, and greenhouse gas emissions at plot, field and farm scales have been identified for a range of agricultural industries and climates (Ongley, 1996; United Nations Environment Program, 1998; Sharpley *et al.*, 2000b; Campbell *et al.*, 2004; McKergow *et al.*, 2008; Kay *et al.*, 2009; Merriman *et al.*, 2009; Monaghan, 2009; Sharpley *et al.*, 2010; Newell Price *et al.*, 2011). Some key measures are described in this section. The effectiveness of each mitigation measure is often highly site specific and needs to be targeted to each agricultural landscape to avoid neutral or negative impacts on the environment.

4.1 Mitigating sources

Structural and cultural practices that optimise the magnitude, timing and spatial distribution of nutrient, contaminant and sediment sources can mitigate losses to catchment waterways and in many cases also increase farm profitability, particularly when efficiency of the farming system is also increased (Monaghan *et al.*, 2008). Source management is often more practical and feasible than water management in rain-fed farming systems. In the Republic of Ireland, for example, over half of the mandatory agricultural mitigation measures are focussed on managing point and diffuse nutrient sources (Statutory Instrument 610, 2010).

A European Union-wide upper limit on the rate of organic nitrogen in manures that can be applied through spreading or animal grazing has been set at 170 kg ha^{-1} (with some exceptions allowed) in an effort to minimise nitrogen (and phosphorus) losses to receiving waterbodies. Practices that maintain agricultural production whilst decreasing the use of fertilisers are also encouraged. For example, containing manures and slurries from stock that are housed in farm yards enables reuse and redistribution of the captured nutrients to fields. Investing in the infrastructure and spreading equipment and developing the skills required to utilise manure nutrients is an on-going challenge (Kleinman *et al.*, 2012). To maximise the value of nutrient use, and minimise losses to the environment, manure and fertiliser should be applied at appropriate rates, timings and locations to match crop requirement, maximise crop uptake and minimise leaching of nitrate and volatilisation of ammonium compounds (Di and Cameron, 2002; Lalor *et al.*, 2011). Soil management practices that minimise nitrogen leaching include use of cover crops, minimal ploughing of pasture (particularly in early autumn), improved stock management and precision farming (Di and Cameron, 2002). Certain fertiliser formulations can also be used to reduce nitrogen losses from cropping systems to the environment (Chen *et al.*, 2008).

Comprehensive reviews of measures suitable for mitigating phosphorus sources in a range of farming systems are provided by Kronvang *et al*. (2005), Nash and Halliwell (1999) and Sharpley *et al*. (2000a). Availability of phosphorus sources for loss to waterways can be minimised by matching fertiliser and manure application rates to match crop needs, and not applying fertilisers to soils that have stores of plant-available phosphorus higher than crop requirements (Richards, 2006). Timing phosphorus applications to fields in ways that avoid forecast heavy rain, choosing appropriate fertiliser formulations and placing fertiliser away from the main water flow pathway (e.g. subsurface placement of phosphorus fertilisers and manures) can reduce the risk of losses in runoff and drainage (Hodgkin and Hamilton, 1993; Nash *et al.*, 2004). Avoiding stocking and/or fertilising areas within fields that seasonally saturate can reduce losses because the wettest parts of the landscape contribute a disproportionately large amount of runoff and nutrients to downstream waterbodies (Melland *et al.*, 2008; Sharpley *et al.*, 2011).

A range of soil amendments (e.g. gypsum, lime, bauxite mining residues siderite, refuse ash, dredged river sediments, alum hydrosolids, ferrous sulfate and cement kiln dust) increase the soil's capacity to retain phosphorus against leaching mostly due to their iron, aluminium or calcium content (Summers *et al.*, 1996; Callahan *et al.*, 2002;

Murphy and Sims, 2012); although this is not currently widely practiced. Soil management practices that maintain a threshold level of groundcover and minimise compaction will minimise erosion and runoff of nutrients. Practices include rotating stock between fields and optimising stock densities to efficiently utilise but overgraze pasture, controlling farm machinery traffic and minimising tillage operations (McCaskill *et al.*, 2003; Agouridis *et al.*, 2005; Deasy *et al.*, 2010; Godwin, 2012).

4.2 Mitigating pathways

A key mitigation strategy for reducing nutrient and contaminant loads in surface and subsurface pathways is to reduce the interaction between water, as a transporting medium, and the source of the nutrient or contaminant. Interaction can be reduced by reducing the volume and/or energy of water flow, by redirecting the pathway of water flow or by removing the source from the water flow pathway, either in space or time. For example, nitrate nitrogen that drains from the rootzone via old root channels or subsurface drains (i.e. high energy flow) may rapidly reach a stream without attenuation (Figure 2). In contrast, nitrate that flows through lower energy water pathways such as soil drainage and riparian zones may be retained and naturally attenuated (depleted) by plant uptake or by biological transformation (e.g. via denitrification) into gaseous nitrogen forms (see 4.5 ‘Pollution Swapping’). For phosphorus, the optimum spatial arrangement of mitigation practices on a farm can be guided by identifying areas that transfer disproportionately high amounts of phosphorus. These areas usually have both a high source of phosphorus and a high potential for surface runoff and are termed critical source areas (Sharpley *et al.*, 2011). A range of phosphorus and nitrogen loss risk assessment indices (e.g. http://wpindex.soils.wisc.edu/, Accessed 15th December 2013) and models have been developed to identify critical source areas in farmed landscapes (Buczko and Kuchenbuch, 2007; Buczko and Kuchenbuch, 2010) and are the subject of continuing research (e.g (Shore *et al.*, 2013)).

The volume of soil drainage can be decreased through increasing plant water uptake by planting deep rooted perennial forage crops instead of shallow-rooted annual crops (White *et al.*, 2003). Efficient irrigation management can also decrease volumes of surface runoff and subsurface drainage (Wilcock *et al.*, 2011). Slowing water in drainage ditches using vegetation and/or low grade weirs allows attenuation of nitrogen and phosphorus (Ensign *et al.*, 2006; Kröger *et al.*, 2011). Retention of runoff water in constructed wetlands can also enable denitrification of nitrate and uptake and sediment adsorption of phosphorus, sedimentation, degradation and decay of pathogenic bacteria (Scholz *et al.*, 2010; Wilcock *et al.*, 2012) and pesticides (Moore *et al.*, 2007; Gregoire *et al.*, 2009). Sediment traps and flow diversion terraces in fields facilitate deposition of sediments entrained in field runoff (Yang *et al.*, 2009; Ockenden *et al.*, 2012). Vegetated riparian buffers have some potential to mitigate inputs of nitrogen, phosphorus, sediment and faecal inputs (Lovell and Sullivan, 2006; Collins *et al.*, 2007; Kay *et al.*, 2009). Buffers are most effective for reducing sediment and sediment-associated nutrients such as phosphorus because surface runoff is slowed allowing for enhanced deposition of sediment. Infiltration is also enhanced so some attenuation of dissolved nutrients and contaminants also occurs.

Separation of clean and dirty water by using appropriate guttering and drainage in farm yards reduces the potential for nutrients and sediment to become mobilised and diverting runoff from farm lanes can prevent nutrients directly entering streams (Wilcock *et al.*, 2007).

A range of technologies have the potential to remove nutrients from water. Phosphorus is adsorbed and precipitated by aluminium, iron or calcium compounds in natural, industrial by-product or artificial media such as iron oxide, limestone, steel slag, melter slag and bauxite mine red mud residue (Buda *et al.*, 2012; Klimeski *et al.*, 2012). Nitrate is removed by denitrification using permeable carbon reactive media (Fenton, 2008), and concentrations of suspended solids, chemical oxygen demand and total nitrogen are decreased through physical filtration, as well as absorption and biological uptake in aerobic woodchip filters (Ruane *et al.*, 2011).

4.3 Mitigating direct delivery

Limiting direct contact between nutrients and other contaminants and a waterbody itself provides a direct means for mitigating losses. Faecal contamination of waterways and erosion from trampled stream banks can be reduced if stock are excluded or discouraged from entering a waterbody. Partial or total stock exclusion can be achieved by riparian vegetation, fencing, providing bridges for crossing and providing alternative shade and water sources and by managing grazing rotations (Agouridis *et al.*, 2005; Collins *et al.*, 2007; Kay *et al.*, 2008). Minimum distances to watercourses are routinely included in fertiliser spreading and pesticide spraying codes of practices.

4.4 Mitigating impacts

In some cases, the impact of eutrophication can be mitigated through engineering, chemical treatment and hydromorphological modification, however, these approaches are often prohibitively expensive and the mitigation effect is often temporary. The Mondego estuary in Portugal provides an example of where reducing the water residence time and redirecting inflows and associated nutrient loads to enter a deeper section led to improved water quality status according to some biological and physico-chemical indicators (Lillebo *et al.*, 2007). Elsewhere, recovery of eutrophic lakes, dams and estuaries has been accelerated by (usually costly) technologies that remove nutrients including chemical amendment (with lime, ferric aluminium sulphate and other phosphorus binding products), aeration, dredging, harvesting macrophytes, flushing to reduce water residence times and manipulation of the food web (Humphries and Robinson, 1995; Schindler, 2006; Gafsi *et al.*, 2009). Phosphorus binding and filtering products have also had some success situated near to and within streams (McDowell and Nash, 2012) and manipulating light and temperature through shading can also minimise eutrophic impacts (Bowes *et al.*, 2012).

4.5 Pollution swapping

Pollution swapping is a term used to describe the outcome of mitigation practices that is positive for one natural resource but negative for another. Stevens and Quinton (2009) reviewed the pollution swapping potential of mitigation methods (cover crops, residue management, no-tillage, riparian buffer zones, contour grass strips, and constructed wetlands) in combinable cropping enterprises. They found potential for pollution swapping via increased greenhouse gas emission after retaining crop residues, establishing riparian buffer strips or constructing wetlands for sediment loss control, via increased nutrient leaching due to crop residue retention, and via delayed runoff of nutrients, particularly soluble phosphorus and dissolved organic carbon due to the reduced efficacy of riparian buffers and constructed wetlands over time. In another case, vegetated riparian buffers reduce eroded sediment inputs to streams; however, less suspended sediment is then available to adsorb phosphorus from the water so concentrations of the most algal-available phosphorus form can increase as a result (McKergow *et al.*, 2006). The suitability of zero tillage cropping varies depending on soil type and climate and can lead to increased greenhouse gas emissions from wet heavy clay soils and increased runoff of dissolved phosphorus from accumulation of phosphorus near the soil surface in some circumstances (Soane *et al.*, 2012). Table 1 summarises some of the advantages and disadvantages of zero till cropping in Europe. Mitigation measures, therefore, need to be targeted to the desired outcome and to specific soil types and agricultural systems in order to optimise both mitigation and production.

5 Governance, costs and benefits of water quality mitigation

5.1 Costs and benefits

The costs of implementation, the technical feasibility, and the adoptability of practices can all constrain the effectiveness of farm practice change measures related to water quality (Buckley *et al.*, 2012). In the European Union, the Water Framework Directive requires member states to calculate the cost-effectiveness of policies that are implemented. Derogations from Water Framework Directive objectives are allowable where it can be demonstrated that achieving such objectives would involve disproportionate costs (Kallis and Butler, 2001). Calculating cost-effectiveness requires both a measure of the costs of implementation and a measure of the effect of the policies on water quality (Balana *et al.*, 2011). Because of the complexities of estimating the costs of a suite of mitigation measures in a diverse agricultural socio-economic and physical landscape, costs are often modelled for a range of actual or theoretical farming systems scenarios (Fezzi *et al.*, 2008). Similarly physical effects of measures are difficult to measure with certainty (see Section 6) and are therefore frequently modelled (for example as marginal abatement costs) rather than measured directly (Fezzi *et al.*, 2010). Importantly, the impacts of mitigation measures vary between agricultural landscapes and over time, so cost-effectiveness needs to be measured or modelled for each landscape and timeframe of interest.

Cost-effectiveness of mitigation measures has been modelled for a range of farming systems and landscapes. Source mitigation approaches such as altering the way nutrients are managed at farm level are often cost neutral or cost beneficial (Ribaudo *et al.*, 2001; Zhang *et al.*, 2012b). For example reducing the risk of nutrient transfer to the aquatic environment through more efficient use of chemical fertiliser potentially has a double dividend effect of increased returns to agricultural production (Barnes *et al.*, 2009; Huhtanen *et al.*, 2011; Buckley and Carney, 2013). In the United Kingdom, optimal cost-effective measures for reducing nitrate leaching were modelled as reducing stocking rates and annual grazing duration, and substitution of cropping area to grassland (Fezzi *et al.*, 2008; Cardenas *et al.*, 2011). For mitigation of phosphorus in runoff, measures that targeted pathways of loss were modelled as more effective than source management, and amongst the most cost-effective of the pathways measures were sediment traps and riparian buffers (GBP 4-8 (USD 6.5-13, conversion rate as of 15th December 2013) kg^{-1} phosphorus conserved) (Haygarth *et al.*, 2009). In contrast, to mitigate phosphorus exports from dairy farms in Australia and New Zealand, source management strategies were calculated to be more cost-effective (NZD 0-200 (USD 0-165) kg^{-1} phosphorus conserved) than using amendments or edge-of-field methods to capture phosphorus (McDowell and Nash, 2012). Similarly, in Ireland, source management to deplete surplus soil P was identified as the most cost-effective long term strategy for water quality improvement in the Lough Melvin catchment (Schulte *et al.*, 2009).

At the river basin scale, Roberts et al. (2012) identified, through modelling, that costs to implement agricultural, forestry and river management changes sufficient to meet aspired nutrient pollution reduction targets for the impaired Gippsland Lakes in eastern Victoria, Australia (AUD 1 billion (USD 0.9 billion) over 25 years) exceeded the available environmental budget. They proposed that discussion around environmentally acceptable mitigation measures should centre on cost-effective, politically realistic, and technically feasible options. Optimising the spatial arrangement of measures also increases the cost-effectiveness (Qi and Altinakar, 2011; Doole *et al.*, 2013). Cost benefit analysis, which places monetary values on both costs and the effects of measures, identified that farmyard improvement and establishing vegetative buffers in critical source areas was effective in reducing phosphorus loss but only became cost beneficial 15 years after implementation (Rao *et al.*, 2012).

5.2 Water quality governance

Rogers and Hall (2003) in (Hoekstra, 2006) define water governance as ‘the range of political, social, economic and administrative systems that are in place to develop and manage water resources, and the delivery of water services, at different levels of society’. Recognition of the hydrological connection between upland and lowland parts of catchments has resulted in watersheds (river basin boundaries) frequently being used as administrative boundaries for water management (Ferrier and Jenkins, 2010). For example, natural resource management planning within river basin boundaries is an integral component of the Water Framework Directive legislated in Europe since 2000 (Official Journal of the European Community, 2000). Many countries also share river basin resources (e.g. Mekong, Nile, Danube) which leads to a high dependency on water quantity and quality governance upstream (Hoekstra, 2006; Murphy and Glasgow, 2009).

A range of policy instruments and mechanisms are used in water quality management, specifically to link upstream and downstream water users (Tognetti and Lawrence, 2002). These mechanisms include regulatory instruments (Statutory Instrument 610, 2010; Daroub *et al.*, 2011), economic instruments and mechanisms to increase market access (Dabrowski *et al.*, 2009; BenDor and Riggsbee, 2011); modifying organisational structures (Ridley, 2005); education, awareness-building and participatory approaches (Ridley, 2005; Ulén and Kalisky, 2005; Bergfur *et al.*, 2012; Doody *et al.*, 2012). Table 2 shows an example of the suite of agricultural policies imposed since 1985 to reduce nutrient transfers to waterbodies in Denmark (Kronvang *et al.*, 2008). The choice of mechanism(s) can be usefully informed by an account of the likely ratio and type of public and private benefits that will occur as an outcome of the desired structural or cultural practice change (Pannell, 2008). Adaptive planning and implementation approaches and fit-for-purpose governance structures are key features of what is known as integrated catchment management

(Hammer *et al.*, 2011; Pahl-Wostl *et al.*, 2012). The potential beneficiaries of mitigation practices such as land and water user groups and commodity boards are being increasingly involved in catchment and water governance (Millenium Ecosystems Assessment, 2005). Payments for ecosystems services (FAO, 2004), and farmer-led movements such as Landcare in Australia (Youl *et al.*, 2006), exemplify recognition of the multiple benefits (i.e. environmental, food and fibre, employment, community) provided by farmers as stewards of the land.

6 Monitoring and evaluation of mitigation effectiveness

6.1 Accounting for investments in environmental management

Despite the large number of plot to farm scale studies on technical mitigation options for water quality, there is a relative paucity of evidence that these measures improve water quality at larger spatial scales and also over long temporal scales. Evidence for the effects of mitigation measures on water quality is required to account for public and private funds spent implementing those measures, to inform the scientific foundation for implementation of a measure and to help inform expectations about the potential in time and space for those measures to achieve anticipated water quality targets. In some cases expectations of natural resource condition improvement are not realised despite expenditure on research, development, extension, incentives/subsidies and penalties, or insufficient monitoring exists to demonstrate whether change has occurred. Since 2002, a nationwide evaluation of over 50 years of conservation practices and 38 catchment assessment studies (the Conservation Effects Assessment Project) was initiated in the USA to account for USD 6 billion in expenditure on these practices (Weltz *et al.*, 2005). In Australia, AUD 1.4 billion (USD 1.25 billion) was spent over 7 years to remediate salinity of soils and groundwater; however, there was little evidence of mitigation as a result of this expenditure (Pannell and Roberts, 2010). In the European Union, monitoring the impact that policies affecting agricultural practice have on water quality is mandatory in zones declared as Nitrate Vulnerable Zones (e.g. in the England, France, Sweden, Czech Republic and the Walloon region of Belgium) and is a pre-requisite for stocking rates above a European Union cap on manure-nitrogen loading to be permitted. Some member states have identified the whole country as a Nitrate Vulnerable Zone (e.g. the Republic of Ireland, Austria, Luxembourg, Germany, Denmark) and have accordingly established national and agriculture-specific water quality monitoring programs to compliment country-specific national regulations that include many of the measures highlighted in section 4 (Fraters *et al.*, 2011).

6.2 Water quality targets and standards

To measure the effectiveness of mitigation measures, the target condition relating to the water resource needs to be defined. Targets may be defined as chemical load, concentration and/or exposure, degree of sedimentation, biological quality, hydromorphology or a combination of indicators (Ongley, 1996). In the USA, total maximum daily loads are used as targets and are defined as the maximum amount of a pollutant that a waterbody can receive and still meet water quality standards (http://water.epa.gov/lawsregs/lawsguidance/cwa/tmdl/, accessed 07/11/2012). Legislation requires that total maximum daily loads are set for impaired waters; however, implementation of measures to achieve the load reductions is largely voluntary or incentivised (Helmers *et al.*, 2007). The European Union Water

Framework Directive (Official Journal of the European Community, 2000) set a target for all water bodies to attain at least 'good' water quality status by 2015 (with six year review cycles if the first target could not be met). In order to monitor progress towards this target, an inter-calibration process was conducted (e.g. McGarrigle and Lucey (2009)) so that water quality status can be compared across the wide range of bioregions and water body types across Europe or to account for shared bioregions between jurisdictions. Individual member states have subsequently set their own chemical, biological and hydromorphological water quality standards. For example in The Republic of Ireland, standards for drinking water and standards designed to protect ecological status have been legislated, with ecological status being constrained by the most limiting of a range of chemical, biological and hydromorphological indicators (Bowman, 2009). The Australian and New Zealand guidelines for fresh and marine waters require threshold (or 'trigger value') chemical concentrations and biophysical status to be established based on conditions in reference water body types (Anon, 2000; McDowell *et al.*, 2013). The choice of indicator of system quality or change can influence assessments of whether mitigation measures have been successful or otherwise. For example, Lillebo *et al.* (2007) found that estuary quality either did not change, improved or worsened in response to mitigation measures depending on which quality status indicator was used in the assessment. Some indicators may not be able to pick up changes due to specific mitigation measures. For example, if loads of nutrients from diffuse sources of episodic overland flow are reduced by reducing agricultural soil phosphorus levels, but in-stream biological quality responds mainly to low-flow nutrient concentrations, then the benefits of the overland flow mitigation measures may not be reflected by biological indicators.

6.3 Mitigation effect monitoring methods

Many of the mitigation methods implemented on farms have been recommended to or by policy makers a result of process studies that relate action to response for individual or small groups of measures (Kronvang *et al.*, 2005; Carton *et al.*, 2008; McDowell *et al.*, 2009). Effectiveness of implementation of measures can also be informed by scenario analysis and modelling of the complex interactions of land management with environmental variables (Silgram *et al.*, 2008; Oenema *et al.*, 2009; Vigiak *et al.*, 2011; van Grinsven *et al.*, 2012). A growing number of studies have also directly measured impacts of mitigation measures at the catchment scale.

A water catchment, often also referred to as a watershed, is the area of land from which rainfall eventually drains into a surface waterbody. Whilst surface water catchments are defined by topography and are separated by watersheds, or catchment divides, groundwater zones of contribution to surface waters do not always follow the same topographic boundary as a surface water catchment. Water catchments provide discrete biophysical spatial units that account for all the anthropogenic and natural chemical, biological and physical processes that influence the mobilisation, transfer, attenuation and delivery of materials from their sources to receiving surface water bodies. Headwater catchments which supply water to perennial streams have a significant influence on downstream water volumes and quality despite often being a large distance away (Alexander *et al.*, 2007). An understanding of the degree to which land use and management in these headwater catchments influences inherent hydrological and biogeochemical cycling is therefore important in terms of managing lake, groundwater and coastal water quality.

Medium size headwater catchments, or meso-catchments (also referred to as micro-basins (Yates and Bailey, 2006) or small watersheds (Meals *et al.*, 2010)), are commonly 1-100 km^2 and incorporate 1st – 3rd order streams. Measuring the effects of agricultural mitigation practices on water quality in meso-catchments offers advantages over smaller and larger scales in that the size can minimise inputs from non-agricultural pressures and enable adequate stakeholder involvement in implementing and/or monitoring agricultural practices, and the integrated and 'net' impacts of attenuating and mobilising processes and farm types and practices are accounted for (Iital *et al.*, 2008; Fealy *et al.*, 2010). Larger catchments usually include other significant influences such as forestry, industrial and municipal land use and as the size and scale of the catchment increases, the effects on water quality of individual mitigation practices become more difficult to discern ((Kiersch, 2002) (Table 3)).

Approaches for measuring water quality impacts of agricultural mitigation practices in meso-catchments range from measuring water quality over a time series, such as before and after a land management change (e.g. (Jaynes *et al.*, 2004)), and/or over a spatial series such as in paired catchments with and without agricultural practice change (e.g. (Schilling and Spooner, 2006)) or over a gradient of practices or catchment types, and by cause and effect studies that measure sources, pathways and impacts of practices (e.g. (Wall *et al.*, 2011), Figure 3).

A dose-response relationship is also sometimes used to describe the increasing impacts on water quality where the agricultural pressure increases. The European Environment Agency uses the DPSIR framework to assess change in the state (S) of natural resources due to changes in specific drivers (D) and pressures (P) that can have an impact (I) and are the focus of policy responses (R) (EEA, 2012). These conceptualisations highlight the biophysical links in space and time between agricultural pressures and water quality in receiving water bodies. Nutrients, for example, can be transferred from both point and diffuse sources. Point sources include discharges from intermittent or persistent discrete sources (such as a farmyard) to a water flow pathway or water body. Diffuse transfers, are derived from non-point sources, which are spatially widespread, such as nutrients in soil. The connectivity of these sources with receptor waterbodies depends on the activity of, and attenuation potential along, the water flow pathway to the receptor. For example, overland flow transport of soil and nutrients tends to occur episodically during storms and from parts of a catchment that are prone to generate runoff (e.g. saturated areas and hard surfaces).

Effects of agricultural practice on water quality are challenging to measure at the meso-catchment scales because of i) resource constraints associated with establishing sufficient monitoring infrastructure and collecting land management information (Cherry *et al.*, 2008), ii) uncertainty in cause-effect relationships due to the complexity of hydrological, climatic, biogeochemical and anthropogenic processes occurring in time and space, and because iii) long time scales are normally needed to identify trends in data due to variable time scales and time lags between implementation of mitigation measures and responses in water quality (Spooner *et al.*, 1987; Meals *et al.*, 2010). These challenges are discussed below.

6.3.1 Resource constraints

Due to the sometimes 100-fold variation in phosphorus concentrations between baseflow and stormflow in streams, and the likelihood that phosphorus measures are aimed specifically to mitigate surface sources of phosphorus, sufficient sampling intensity of stormflow phosphorus concentrations is required to accurately measure flow-weighted concentrations and loads (Jordan and Cassidy, 2011). Stormflow samples are costly to collect and analyse on a continuous and highly spatially distributed basis. The cost of sampling one site every seven hours for one year and analysing for nutrients was estimated at EUR 5000 (USD 6900) once-off for equipment and EUR 30000 (USD 41200) for sampling at 2012 Euro values (Melland *et al.*, 2012). Technologies such as continuous bankside analysis provide opportunities for continuous analysis that importantly capture high flows but are still costly (EUR 50000 (USD 68700) once-off for equipment and EUR 30000 (USD 41200) for sampling) and are often impractical to deploy at multiple locations (Jordan *et al.*, 2007). Passive sampling may offer a compromise between increasing the spatial coverage of sampling and collecting continuous (integrated) nutrient concentrations over storm and base flows but the technology is still at the research and development stage (Jordan *et al.*, 2013).

6.3.2 Uncertainty in cause-effect relationships

Effects of measures are difficult to dissociate from potentially overriding/swamping attenuation effects such as groundwater denitrification and stream nutrient uptake and from counteracting effects such as increased intensification of agricultural production (Sutton *et al.*, 2009). The effect of mitigation needs to be large, compared with background processes, so that impacts can be measured and measures can be effectual (Tomer and Locke, 2011). Further to these factors, rates of implementation of mitigation measures are often difficult to control, particularly when there is reliance on voluntary adoption (Yates *et al.*, 2007). Uncertainty in measurement can affect estimates of cost effectiveness and equity of cost sharing of mitigation (Khadam and Kaluarachchi, 2006). Gren and Destouni (2012) suggest that calculation and presentation of a range of estimates of nutrient loads based on different models is likely to reduce barriers to implementation of measures by identifying commonality in model outcomes (such as consistent attribution of nutrient source or apportionment of mitigation costs) and thus reducing arguments about model outcome uncertainty.

6.3.3 Time lags between implementation of mitigation measures and responses

The time between when a land management activity occurs, and when either a positive or negative response in water quality occurs, depends on the type of pollutant and the potential for that pollutant to be mobilised or attenuated along the path it travels to reach a waterbody. Lag times for water quality improvement in groundwater and groundwater fed waterbodies after nitrate mitigation can be as long as decades (Fenton *et al.*, 2011), particularly where travel rates through aquifers are slow and there is little opportunity for denitrification (for example, oxic chalk aquifers in Denmark (Collins and McGonigle, 2008; Fenton *et al.*, 2011; Windolf *et al.*, 2012). To identify trends in acid sensitive lake chemistry, 10 years of sulphate chemistry records were predicted to be sufficient for trend analysis but trends in nitrate were not be detectable over this time frame due to biological processes of nitrogen transformations (Skjelkvale *et al.*, 2005). For phosphorus and sediment movement via surface flow pathways, chemical fixation of phosphorus to soil particles and retention and re-mobilisation of particulate and soluble phosphorus forms throughout landscape and channel beds can cause years to pass before water quality improvements from phosphorus mitigation practices occur (Walling, 1999; McDowell *et al.*, 2003; Stutter *et al.*, 2007; Schulte *et al.*, 2010).

Contaminant flow times can impact not only the timing of the response, but the degree of response. Generally, the longer the contact time between a contaminant and its surrounding flow path media (in this case soils or geological strata) the longer time available for attenuating reactions (such as sedimentation, denitrification or phosphorus adsorption) or mobilising reactions (such as phosphorus desorption and nitrification) to occur (Barrow and Shaw, 1975; Haag and Kaupenjohann, 2001). Even after water quality improvement, however, recovery of aquatic biological structure and function is not guaranteed and can be limited by a complexity of factors including for example, extinction of endemic species during the eutrophic phase (Schindler, 2006; Carpenter *et al.*, 2011). Using modelling, no improvement in Baltic Sea eutrophication was predicted in response to decreased nutrient loads since the 1980s due to at least a 20 year time lag in water quality response (Gustafsson *et al.*, 2012). Further to time lags that occur in the biophysical response of water bodies, implementation of cultural and structural changes to farms and farm practices takes time, even where measures are made mandatory by governments or industry (Kronvang *et al.*, 2008).

6.4 Observed effects of mitigation on water quality

Agricultural mitigation measures have had either no measurable effect, or positive, or negative effects on water quality over periods of 3 to 20 years in meso-catchments in North America, New Zealand, Europe and Brazil (Melland *et al.*, 2013). Beneficial effects occurred over periods of 1.5 to 10 years whereas the time it took for these effects to be measured and identified as significant ranged from 4 to 15 years. These response times tended to increase with increasing catchment size. An increasing number of studies have integrated measurement of environmental impact in terms of biological water quality (Bergfur *et al.*, 2012) along with the more traditional hydrological and chemical indicators. In most catchments where beneficial effects of mitigation measures were successfully measured, combinations of measures that address nutrient or pollutant sources, pathways, delivery and impact have been implemented. Successful farm measures included improved engineering and crop management to reduce runoff and drainage of nutrients and sediment, as well as high rates of implementation of measures across the catchments. In many cases, the potential to measure improvement in one or more water quality indicators was limited by the impact of a few management or weather events. Reasons that water quality did not improve in some studies included the uncertainty inherent in most nutrient flux measurements and a lack of high flow water quality samples that limited the ability of practice impacts to be measured. In other catchments, it was difficult to verify whether a lack of effect was a result of ineffective measures, or because time lags for improvement of water quality were longer than the monitored period. Pollution swapping was identified in some cases. A number of meso-catchment studies attribute water quality improvements largely to changes in land use from a more intensive system such as maize strip cropping to a less intensive system such as alfalfa or due to retirement of land from production altogether (Schilling and Spooner, 2006; Yates *et* *al.*, 2007; Makarewicz *et al.*, 2009). Where land is taken out of production, or where land use intensity is reduced to mitigate water quality impacts, a potential trade-off between water quality and agricultural output may occur. In contrast, and although environmental water quality targets have yet to be met (Windolf *et al.*, 2012), measures including setting minimum plant-available nitrogen percentages for manure nitrogen, and capping livestock density and allowable nitrogen fertiliser rates in Denmark, have led to a significant decrease in nitrogen concentrations in 84% of monitored streams (Kronvang *et al.*, 2008) and at the same time, crop yields have been sustained and livestock production has increased.

7 Sustainable intensification

In order to feed the world's population, which is projected to increase to 9 billion in 2050, policies enabling 'sustainable intensification' are being promoted (Foley *et al.*, 2011). Sustainable intensification has been defined as 'simultaneously raising productivity, increasing resource use efficiency and reducing negative environmental impacts of agriculture' (Bolton and Crute, 2011). For example, de Klein *et al.* (2012) identified, through modelling of a database of farm system information, that by incorporating targeted mitigation strategies into pastoral dairy production systems increases in milk production without a concomitant increase in nitrogen leaching and greenhouse gas emissions is theoretically possible and needs to be tested in the field. Currently there are few catchment-scale examples that demonstrate increased production and economic wealth whilst simultaneously maintaining high quality surface waters in intensive agricultural settings, highlighting that sustainable intensification remains a continuing challenge.

8 Summary

All human activities, including agriculture, have an impact on the quality and quantity of freshwater resources. In turn, maintaining good water quality is imperative for supporting human use of freshwater and sustaining ecosystem function. Agricultural activities can impair water quality through excess movement of soil, nutrients, salt, pathogens and chemicals from land to water. When reaching water bodies in excessive amounts, these often-essential components of agricultural systems can be considered pollutants. Plot and field scale studies have identified a range of technical options that can mitigate off-farm water quality degradation and many options also improve the profitability and sustainability of the farming system. These technical options include limiting the source of pollutant that is available for transport, managing the pathways of travel that a pollutant takes before reaching the waterbody, reducing direct inputs of pollutants into a waterbody, and directly manipulating the waterbody where an impact has occurred.

To maintain or achieve good water quality in agricultural catchments, policies and management recommendations should be developed after consideration of the likely implementation rates (which includes an assessment of cost) and effectiveness of the practices. Consideration of the potential for pollution swapping is also necessary. Headwater river catchments provide useful spatial scales for measuring the effectiveness of practices because these catchments represent the complexity of activities and transport pathways occurring in agricultural landscapes. Catchments are also increasingly used to define administrative boundaries for water resource management. The cost, difficulties in monitoring and collecting information, uncertainty in relating cause to effect and the long time frames needed to measure changes at these scales is challenging. There is, however, a growing body of evidence that water quality can be improved in intensively farmed landscapes. These studies, and other scientifically robust monitoring and modelling endeavours, are important for informing effective, socially acceptable and cost beneficial strategies that can facilitate the sustainable intensification of agriculture into the future.

9 Web links

URL and date accessed	Content summary
http://www.ewater.com.au/ 01/11/2012	eWater Cooperative research centre for water-cycle management and research. Includes software tools and information related to the modelling and management of water resources and water quality.
	water framework directive website (including website of useful graphics)
http://water.epa.gov/type/watersheds/datait/watershedcentral/ 07/11/2012	USEPA website providing information and tools regarding catchment management
http://www.ceep-phosphates.org/	Research into phosphates and the environment and into phosphate recycling
http://www.nine-esf.org/ENA 01/11/2012	Nitrogen in Europe website, which includes the European Nitrogen Assessment published in 2011, information regarding problems and solutions pertaining to nitrogen in the environment and a video link that explains nitrogen issues.
http://water.usgs.gov/wsc 01/11/2012	United States Geological Survey - Science in your watershed - information and tools
http://techalive.mtu.edu/meec/module01/ whatiswatershed.htm 01/11/2012	Visualisation of the definition of a catchment, or watershed.
http://www.iwmi.cgiar.org/index.aspx 01/11/2012	International Water Management Institute publications, tools and resources
http://www.waterfootprint.org/?page=files/home 01/11/2012	The Water Footprint Network publications, tools and resources
http://www.ozcoasts.gov.au/ 13/11/2012	Geoscience Australia – Australian Government Ozcoasts website with information, glossaries and tools regarding coastal water quality
http://www.fao.org/nr/solaw/en/ http://www.fao.org/nr/water/aquastat/ 18/11/2012	FAO State of the World's Land and Water Resources for Food and Agriculture report and background information. Maps. Aquastat – FAO database on water resources and water management in Africa, Asia, Latin America and the Caribbean

10 Glossary

Denitrification	The process of microbially facilitated reduction of nitrate into nitrogen (N_2) gas and other nitrogen oxides.
Ecosystem service	Benefits to humankind from a multitude of resources and processes that are supplied by natural ecosystems.
Grey water	Water output of a food production (or other) system that is degraded in comparison to the fresh (blue and green) water inputs
Hydromorphology	Hydrological, habitat and structural processes and features of rivers, lakes and coastal waterbodies
Nitrate Vulnerable Zone	A term used in Europe for areas of land that drain into nitrate polluted waters, or waters which could become polluted by nitrates.
Pollution swapping	The outcome of mitigation practices is positive for one natural resource but negative for another.

11 Recommended tertiary courses

1. Bachelor of Science in Natural Resources, Department of Soil Science, College of Agriculture and Life Science, North Carolina State University, NC, USA

2. Bachelor of Science in Land Development, Department of Soil Science, College of Agriculture and Life Science, North Carolina State University, NC, USA

3. Bachelor of Science in Environmental Science and Technology (ENST), Department of Environmental Science and Technology, College of Agriculture and Natural Resources, University of Maryland. MD, USA.

4. Bachelor of Agricultural Science, Agri-Environmental Sciences (DN250 AES), National University of Ireland – Dublin. Dublin, IRL.

5. Bachelor of Agriculture, Melbourne School of Land and Environment, The University of Melbourne, AUS.

6. BASIS Graduate Diploma in Agronomy with Environmental Management, Harper Adams University College, Shropshire, UK.

12 Cross references chapters

11. Environmental service issues: markets and policy *unassigned*
83. Water use: water supply conflicts and challenges for the future
85. Water: water quality challenges from agriculture
86. Water: footprint of food production and processing
87. Water: advanced irrigation technologies
91. Soil: conservation practices
241. Climate change: water and irrigation
243. Ecological infrastructure, natural capital and ecosystem services
242. Pesticide risk reduction

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Table 1. Advantages and disadvantages of ploughing and no-till farming in Europe, although not universally relevant to all regions. Reprinted from Soane *et al.* (2012) with permission from Elsevier.

Mouldboard Ploughing		No-till	
Advantages	Disadvantages	Advantages	Disadvantages
Appropriate loosening of topsoil prior to seedbed preparation	Pan formation below the depth of ploughing (from passage of plough sole and tractor wheels)	Lack of compaction below plough furrow	Crop establishment problems during very wet or very dry spells
Complete burial of weeds, crop residues, lime, other amendments and manure	Excessive looseness to depth of ploughing	High work rates and area capability	Weed control problems
Inversion allows structural development of lower layers in the topsoil	Exposure of bare topsoil to wind and water erosion	Increased bearing capacity and trafficability	Cost of herbicides, herbicide resistance
Exposes soil compacted at harvest to loosening by weather	High susceptibility to re-compaction of topsoil	Reduced erosion, runoff and loss of particulate P	Risks of increased N_2O emissions and increased dissolved reactive P leaching
Increased mixing of nutrients throughout profile	Buried weed seeds brought to the surface	Opportunity to increase area of autumn-sown crops	Reduced reliability of crop yields, especially in wet seasons
Promotes surface drainage leading to warmer, drier seedbed in spring	Reduced trafficability under wet conditions	Stones not brought to the surface	Unsuited to poorly structured sandy soils
Reduced risk of crop diseases	Low work rate and high costs	Drilling phased to take advantage of favourable weather conditions	Unsuited to poorly drained soils
Reliable agronomically in widely differing seasons	Increased CO_2 emissions (fuel and oxidation of SOC)	Increased area capability	Risk of topsoil compaction
Suitable for preparing a seedbed after grass	Greater oxidation of organic matter near surface	Reduced overall costs (fuel and machinery)	Problems with residual plough pans
	Disruption of macrofauna (earthworms, predatory insects)		Increased slug damage
			Unsuited for incorporation of solid animal manures

Table 2. Agri-environmental policies implemented in Denmark since 1985 to reduce nitrogen loss from agriculture. Reprinted from Kronvang *et al.* (2008) with permission from Elsevier.

Table 1 – Summary of the Danish measurers imposed to reduce nitrogen losses from agriculture

Danish policy actions	Policy measures imposed
1985: NPo Action Plan to reduce N- and P-pollution	• Minimum 6 months slurry storage capacity • Ban on slurry spreading between harvest and 15 October on soil destined for spring cropping • Maximum stock density equivalent to 2 LU ha^{-1}. (1 livestock unit = 1 LU corresponds to one large dairy cow) • Various measures to reduce runoff from silage clamps and manure heaps • A floating barrier (natural crust or artificial cover) mandatory on slurry tanks
1987: The First Action Plan for the Aquatic Environment (AP-I), aiming to halve N-losses and reduce P-losses by 80%	• Minimum 9 months slurry storage capacity • Ban on slurry spreading from harvest to 1 November on soil destined for spring crops • Mandatory fertilizer and crop rotation plans • Minimum proportion of area to be planted with winter crops • Mandatory incorporation of manure within 12 h of spreading
1991: Action Plan for a Sustainable Agriculture, aiming to reduce N-losses from agricultural fields by 100×10^6 kg N	• Ban on slurry spreading from harvest until 1 February, except on grass and winter rape • Obligatory fertilizer budgets • Maximum limits on the plant-available N applied to different crops, equal to the economic optimum. The economic optimum is calculated annually, taking into account the mineral N in the soil (from a comprehensive soil sampling system) • Statutory norms for the proportion of manure N assumed to be plant-available (Pig slurry: 60%, cattle slurry: 55%, deep litter: 25%, other types: 50%)
1998: The Second Action Plan for the Aquatic Environment (AP-II)	• Subsidies to establish 16,000 ha wetlands, designed to reduce nitrate leaching through denitrification and reduced demand for fertilizer • Subsidies to enable reduced nutrient inputs to up to 88,000 ha of areas designated as being specially sensitive with regards the environment • An expectation that animal feeding practice would be improved to reduce N excretion • A reduction of the stock density maximum to 1.7 LU ha^{-1} • Subsidies to encourage the conversion of 170,000 ha to organic agriculture • The statutory norms for the proportion of manure N assumed to be plant-available were increased from 1999 (pig slurry: 65%, cattle slurry: 60%, deep litter: 35%, other types: 55%) • Maximum limits on the application of plant-available N to crops reduced to 10% below the economic optimum • Mandatory 6% of the area with cereals, legumes and oil crops to be planted with catch crops • Subsidies to encourage afforestation on up to 20,000 ha
2000: AP-II Midterm Evaluation and Enforcement	• Increased economic incentives to establish wetlands • The N assumed to be retained by catch crops must be included in the fertilizer plans • Further tightening of the statutory norms for the proportion of assumed plant-available N in manure. From 2001; pig slurry: 70%, cattle slurry: 65%, deep litter: 40%, other types: 60%; from 2002 pig slurry: 75%, cattle slurry: 70%, deep litter: 45%, other types: 65% • Reduced fertilization norms to grassland and restrictions on additional N-application to bread wheat
2001: Ammonia Action Plan	• Subsidies to encourage good manure handling in animal housing and improved housing design • Mandatory covering of all dung heaps • Ban on slurry application by broadcast spreader • Slurry spread on bare soil must be incorporated within 6 h • Ban on the treatment of straw with ammonia to improve its quality as an animal feed • Options for planning authorities to restrict agricultural expansion near sensitive ecosystems
2004: The Third Action Plan for the Aquatic Environment (AP-III). AP-III is very closely related to the EU-Water Framework Directive and the EU Habitat Directive. N-leaching must be reduced by further 13% by 2015	• Further tightening of the request for catch crops

Table 3. Measurability of land use effects by basin size (measurable impact, x; no measurable impact, -). Reprinted from Tognetti and Lawrence (2002) with permission from the Food and Agriculture Organisation of the United Nations.

	Basin size (km^2)						
Impact type	0.1	1	10	10^2	10^3	10^4	10^5
Average flow	x	x	x	x	-	-	-
Peak flow	x	x	x	x	-	-	-
Base flow	x	x	x	x	-	-	-
Groundwater recharge	x	x	x	x	-	-	-
Sediment load	x	x	x	x	-	-	-
Nutrients	x	x	x	x	x	-	-
Organic matter	x	x	x	x	-	-	-
Pathogens	x	x	-	-	-	-	-
Salinity	x	x	x	x	x	x	x
Pesticides	x	x	x	x	x	x	x
Heavy metals	x	x	x	x	x	x	x
Thermal regime	x	x	-	-	-	-	-

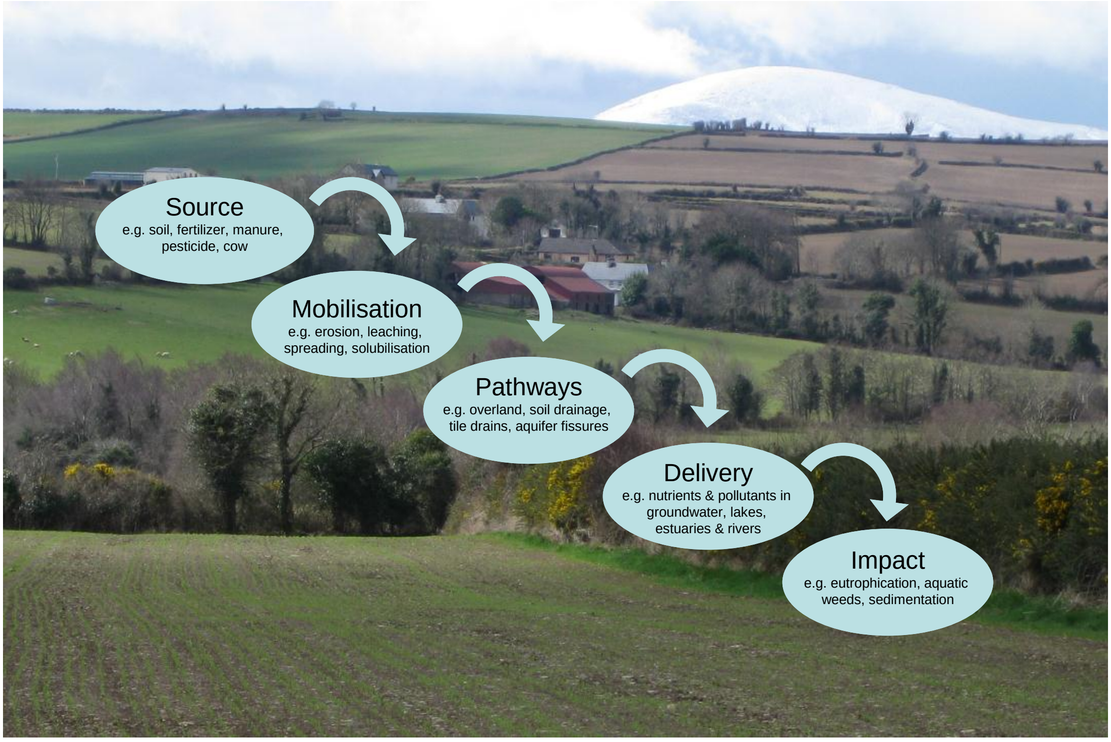

Figure 1. Good agricultural practices and targeted implementation of mitigation measures can reduce the mobilisation, and delivery of nutrient, agro-chemical and sediment sources via pathways to where they can contribute to detrimental impacts in receiving water bodies. Imagery courtesy of Teagasc, Ireland.

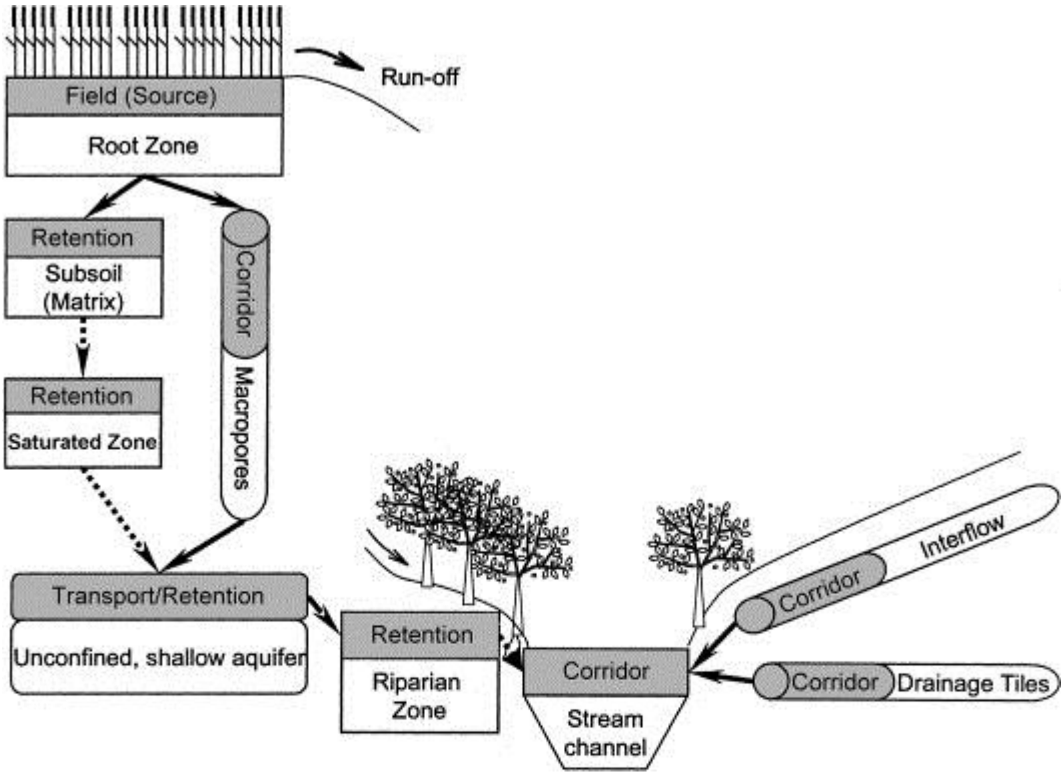

Figure 2. Pathways of nitrate movement from fields to a stream. Nitrate can be transported in water rapidly to streams through natural pipes (corridors) such as tile drains and old root channels and can also flow more slowly through soil and groundwater aquifers and be taken up by plants or biologically transformed into other nitrogen forms (retention). The retention processes attenuate, or deplete, the amount of nitrate reaching the stream. Reprinted Haag and Kaupenjohann (2001) with permission from Elsevier.

Figure 3. A meso-catchment study of the effects of agricultural nutrient management policies on water quality in Ireland (Wall *et al.* 2011) includes measurements of sub-hourly stream discharge and water quality at the catchment outlet, monthly water quality at upstream locations, meteorological parameters, geophysical characteristics and groundwater dynamics and quality in two representative hillslopes, field by field soil nutrient levels and surveys of farm nutrient management practice and financial data. Imagery courtesy of Teagasc, Ireland.