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Monitoring and modelling diffuse pollution from agriculture for policy support: UK and European experience

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ABSTRACT

The need to understand and mitigate diffuse water pollution from agriculture (DWPA) using a range of monitoring or modelling techniques and abatement methods has never been greater. In response to the widely reported detrimental environmental impacts of such pollution and the desire to safeguard water resources, a number of important legislative drivers have been introduced, including the Water Framework Directive (WFD) for EU member states. Efforts to commission research and introduce policy options that address the key requirements of over-arching legislation, increasingly point to a number of common and important issues for policy makers. Whereas our understanding of, and ability to predict, pollutant loadings is reasonably well developed, coupling such pressures to ecological impacts remains a difficult task due to the limited functionality of available toolkits. It is important for mitigation programmes to consider multiple pollutants especially given the risks of pollution swapping and to support the uptake of abatement options that are economically and socially acceptable to the stakeholders involved. Appropriate spatial targeting of mitigation methods will continue to come under scrutiny, especially in the context of additional environmental pressures like climate change. Given its key role in governing the transfer and fate of priority nutrients and contaminants and its well-documented negative habitat impacts, sediment must be given a higher profile in diffuse pollution policy. The latter does, however, require further investigation of background sediment loads necessary for healthy habitats and associated sediment standards or thresholds, in order that catchment compliance can be more reliably assessed. Delayed water quality response to the mitigation of DWPA must be assessed and understood, as a means of informing stakeholders and policy options. A further challenge is posed by the need to place DWPA in the context of pollution from alternative sectors so that a more holistic approach to understanding and managing pressures and impacts and engaging stakeholders can be encouraged.

As a consequence of the widely reported environmental impacts of diffuse pollution from agriculture, various national and international obligations have been set to drive the establishment and implementation of management strategies to help safeguard waterbodies. Such legislation includes the EU Water Framework Directive (WFD) (2000/60/EC) as well as longer-standing policy drivers represented by, amongst others, the Nitrates (91/676/EC), Fish (78/659/EC) and Shellfish (79/923/EC) Directives. Water policy in the EU will increasingly be coordinated under the over-arching remit of the WFD as a

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means of providing an integrated management structure for meeting key environmental objectives. In short, the WFD seeks to; prevent further deterioration of water resources, promote sustainable water use, and enhance protection of aquatic environments using Programmes of Measures (PoMs). The WFD is especially novel and demanding, because it also comprises economic analyses and full public participation in establishing River Basin Management Plans (RBMPs). Given the substantial body of work being commissioned and delivered to provide policy makers with the evidence base for addressing diffuse pollution from agriculture, it was considered timely to synthesise experience from across the EU in this special issue.

Three modes of monitoring are specified by the WFD to inform management decisions; surveillance monitoring to assess longer-term water quality; operational monitoring to characterise waterbodies in danger of failing targets, and; investigative monitoring to understand the primary causes of non-compliance. A number of emerging tools are available to underpin these requirements, including biological early warning systems and whole-organism bioassays, as well as on-line, in situ and passive monitoring systems for chemical water quality (Allan et al., 2006). Given the problems of reliability and representativeness encountered with traditional monitoring approaches and the failure to identify standardised methodologies (Dworak et al., 2005), opportunities certainly exist for exploring the scope for the wider adoption of new monitoring tools. Due to the lack of consistent approaches to surveillance and the demanding timeframe of the WFD, increasing emphasis has been directed towards using modelling and Decision Support Tools (DSTs) to inform agricultural diffuse pollution policy. Examples of modelling studies are provided, including the contribution by Silgram et al. (this issue).

An important shortcoming of the WFD is its failure to take explicit account of the risks posed by climate change (Wilby et al., 2006). In relation to primary WFD objectives, climate change could result in variable 'good status' at reference sites, disproportionate costs and the failure, or at least reduction, in the efficacy of PoMs (Limbrick et al., 2000; Wilby et al., 2006). Recent climate change scenarios developed for the UK Climate Change Impacts Programme (UKCIP02) (Hulme et al., 2002) suggest wetter winters with increasing likelihood of heavy storm events. Such scenarios could have important implications for diffuse pollution management strategies. Equally, it is instructive to note that projected changes in climate are characterised by numerous uncertainties related to climate variability, gaseous emissions and the robustness of available modelling frameworks. It is clearly meaningful to weight the multiple uncertainties associated with climate change scenarios (Wilby and Harris, 2006). The relationships between climate change and diffuse pollution from agriculture are likely to be complex. Increased flooding, for instance, could mobilise enhanced sediment loads and associated contaminants, potentially exacerbating impacts upon aquatic ecosystems (Wilby et al., 1997). Alternatively, more severe droughts could reduce pollutant dilution, thereby increasing toxicity problems (Landrum et al., 1984).

A key emerging issue for the management of agricultural diffuse pollution at catchment scale is the engagement of stakeholders in decision making and mitigation strategies. It is evident that mitigation options must be sustainable whilst addressing the wider needs of society (Gerrits and Edelenbos, 2004). One important challenge comprises the need to improve interaction and linkages between scientific experts and stakeholders in order that the capacity to mitigate diffuse pollution is maximised (Gerrits and Edelenbos, 2004; Allan et al., 2006). A good example of such interaction is provided by the England Catchment Sensitive Farming Delivery Initiative (ECSFDI) which has been focusing upon farmer engagement and the nurturing of working partnerships between farmers, agricultural advisors, water companies, competent authorities and conservationists in 40 priority catchments. But, the development of the evidence base and tools to support such interaction is dependent upon multidisciplinary scientific collaboration (Blum and Eswaran, 2004).

RBMPs under the WFD need to be founded on appropriate spatial targeting of control options as a means of helping to address cost-effectiveness (Collins et al., 2007). On account of the focus at catchment scale, it is therefore necessary to consider a range of landscape and management factors that potentially combine to enhance the risk of pollutant loss to watercourses. The blanket application of control options is neither effective in reducing loss (Granlund et al., 2005), nor cost-effective (Schleich et al., 1996). Landscape risk is difficult to manipulate, but land management involving factors such as nutrient applications, cultivation or cropping methods and the maintenance of drains can be influenced by policy drivers. There is substantial scope for developing high-resolution GIS toolkits to support the optimal placement of mitigation features such as buffer strips and retention ponds in the landscape. Increased uptake of mitigation methods does not necessarily mean that options are being targeted most appropriately for protecting water quality status. Catchment officers and stakeholders would clearly benefit from understanding the potential for further reductions in pollutant losses consequent upon altering the siting of mitigation options. This is particularly the case in the context of the placement of options available through agri-environment schemes such as Entry Level Environmental Stewardship in the UK.

Various measures have been developed to reduce losses of diffuse water pollutants including nitrate, phosphate, sediment, faecal indicator organisms, etc. (e.g. Cuttle et al., 2007). The development of options that have multiple benefits to the farmer (e.g. measures that reduce water wastage, improve animal health or improve crop growth as well as reducing diffuse pollution) are particularly valuable as they can be implemented with little or no overall cost to the farmer. These "win-wins" are being promoted, for example, through the England Catchment Sensitive Farming Delivery Initiative. Several mitigation options for reducing diffuse pollution from agriculture are also beneficial in reducing the volume of runoff and therefore have a bearing on reducing the risk of flooding. This is particularly true of those options that address soil compaction and the hydrological capacity of soil.

Pollution swapping, whereby mitigation of one pollutant has a negative effect on another, is a particular issue when implementing policy options for tackling diffuse pollution. The impact of interventions on other pollutants and other factors should be considered when instigating mitigation strategies for individual pollutants. An example of a potential pollution swapping effect has been demonstrated by Laws et al. (2007). Changing the timing of slurry application from winter to spring to reduce the risk of nitrate leaching was shown to potentially increase ammonia emissions on short grass swards. There is therefore a need to take a holistic approach to mitigation. Future modelling work needs to predict the interaction between multiple pollutants and to assess the best strategies for deploying policy measures for the mitigation of diffuse water pollution from agriculture.

The emphasis of the WFD requires the integration of modelling experience and tools in order to couple physico-chemical variables with ecological quality (Horn et al., 2004). Various catchment scale models have been coupled to in-channel water quality routines, thereby linking terrestrial loads and in-channel transport, e.g. SWAT and QUAL2E (Santhi et al., 2001) and MIKESHE and MIKEII (Hafno et al., 1995). The greater challenge is, however, to provide sufficient functionality for linking water quality and ecological indicators of habitat health. Synthesised approaches will rest upon the assumption that it is possible to derive meaningful relationships between physico-chemical pressures and the response of indicator species used to reflect the status of aquatic ecosystems. For example, the British River Invertebrate Prediction and Classification System (RIVPACS) (Wright et al., 1984) and the Australian River Assessment System (AUSRIVAS) (Simpson and Norris, 2000) are based on the assumption that the physico-chemical environment at any given site has the capacity to influence the structure of macroinvertebrate communities (Clarke et al., 2003; Hargett et al., 2007). Similarly, the Sediment Intrusion and Dissolved Oxygen (SIDO) transport model (Alonso et al., 1996) simulates the relationship between salmon survival and spawning habitat quality in gravel-bed rivers. Diffuse pollution pressure models need to be coupled with ecological tools like RIVPACS and SIDO so that scope for achieving good status now and in the future, taking into account projected land use and climate change and uptake of mitigation options can be explored. In doing so, careful consideration will need to be given to the time and cost demands of the sampling protocols underpinning ecological tools (Haase et al., 2004; Hering et al., 2004) and the increased computational complexity. Linking the impact of diffuse pollution control options or combined strategies on habitat condition will help to satisfy the true focus of the WFD and will provide a basis for target setting. Coupled tools will therefore provide policy makers with modelling systems that are truly fit for purpose and which pinpoint specific ecological windows of interest (e.g. the salmonid spawning season) as opposed to annual loadings and their abatement. To date, many modelling studies have simulated the impact of mitigation strategies on annual pollutant loss as opposed to pressures during ecological windows.

The issue of sediment targets continues to attract debate and Collins and Anthony (this issue) suggest that an alternative sediment standard, other than the 25 ppm annual mean concentration cited by the EU Freshwater Fish Directive is required by the EU. In the USA, the environmental regulatory framework for water quality is based on the Total Maximum Daily Load (TMDL) programme which strives to attain ambient water quality standards via the control of diffuse and point sources (USEPA, 1991). Cooper et al. (2006) recently applied the approach in the UK. But, it is important to acknowledge that the TMDL approach focuses upon chemical rather than ecological status and so in itself does not totally fulfil the WFD. Although the scope for coupling numeric sediment load targets and ecological status could be explored, a number of uncertainties complicate the setting of critical sediment loads, including the dependency upon catchment and reach type, sediment character, species and critical life-stage requirements. It is important to assess sediment accumulation on, and within, the channel substrate as opposed to fluxes per se and to understand the nature (coarse, fine, organic, inorganic) of that sediment (Greig et al., 2005). A range of alternative sediment targets has been explored, including light penetration, embeddedness, riffle stability and the characteristics of surface or subsurface sediment (Rowe et al., 2003). The utility of these measures is, however, constrained by various issues. In relation to percent embeddedness, for instance, potential problems include the need to calibrate observations by multiple personnel and the fact that this metric does not reflect sediment impact on the egg pocket (Reiser, 1998). Similarly, Rosser and O'Connor (2007) have recently argued that the use of river sediment grain size information to set regulatory targets is hampered by the need to adopt statistically robust sampling strategies in the face of substantial spatial and temporal variations. In view of these issues, the coupling of diffuse pollution pressure and impact models appears to offer an alternative means of investigating targets.

Kay et al. (this issue) discuss microbial pollution from agriculture, highlighting that the evidence base on FIO's requires improvement with some urgency. Current priorities include the need to obtain improved empirical datasets on storm period FIO fluxes to complement low flow datasets and on the capacity to remediate FIO fluxes using a range of control options such as stream bank fencing, wetland construction and dirty water management (Kay et al., 2007). Equally, sediment-associated FIO propagation and storage demands further investigation. Sediment provides a beneficial environment for FIO's in terms of protection from environmental stress and a food source, meaning that sediment delivery provides scope for FIO retention, survival and re-growth (Gerba and Mcleod, 1976; Kantani et al., 2003; Jamieson et al., 2004, 2005). The fact that the propagation of FIOs is strongly influenced by catchment sediment (suspended and bed) dynamics has important implications for safe water policy (Droppo et al., 2006). In addition to FIOs, further work is also needed on the role of water in the transport of other human and animal pathogens.

Bechmann et al. (this issue), Wahlin and Grimvall (this issue), Jackson et al. (this issue), Kronvang et al. (this issue), Iital et al. (this issue) focus upon water quality response to mitigation programmes, highlighting the existence of lags. Stakeholders therefore need to be educated and informed, so as not to be alarmed when water quality problems continue in the face of expenditure on abatement strategies. The time lag issue is especially evident in relation to nitrate pollution from agriculture and projected longer-term trends, given the complex relationship between groundwater response times and nitrate emissions from agricultural sources. Hughes et al. (2007) recently modelled aquifer response times for England and Wales with the results confirming the substantial time-scales (decades) over which nitrate pollution will continue to reach discharge points despite reductions in contemporary surface loadings. Similarly, in the case of phosphorus pollution, existing studies suggest that several years can pass before the effects of best management practices translate into measurable improvements in water quality. Such time lags reflect the accumulation of high levels of P in soils and sediment and the complexity of P redistribution through catchments due to storage and remobilisation at intermediate locations between primary sources and catchment outlets (Boesch et al., 2001; Wang et al., 2002; McDowell et al., 2003). Likewise, sediment control strategies must be underpinned by a sound understanding of sediment sources at catchment scale (Collins et al., 2001). Time lags in downstream sediment water quality can be influenced by inappropriate targeting of mitigation methods and the capacity for non-targeted sources to become more important over time or the remobilisation of sediment from catchment sinks (Ruhlman and Nutter, 1999; Renwick et al., 2005). The sediment budget concept provides a valuable framework for interpreting and predicting catchment response to environmental change (Slaymaker, 2003; Walling and Collins, this issue).

Whilst understanding and mitigating diffuse pollution from agriculture continues to pose demanding challenges, it is equally important to sustain a cross-sector perspective. Cross-sector information helps to engage catchment stakeholders and offers wider scope for targeting mitigation efforts. A delicate balance is therefore needed to ensure that our understanding of, and ability to control, diffuse pollution from agriculture develops in an integrated manner with corresponding efforts targeting alternative, e.g. urban sources. There are clearly many challenges facing pollution scientists and policymakers.

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Dan McGonigle is currently working in Farming and Food Science at Defra with responsibility for research on water quality and integrated farm management. Previously he worked as a farm advisor for Devon Wildlife Trust on the Dart Catchment project working on diffuse pollution issues. He completed a PhD on integrated pest management at Southampton University in 2002.