6. Accelerated Loss of Nutrients: Summary of Issues

Problem

Increased availability and subsequent transfer of various forms of nitrogen (N) and phosphorus (P) from land to water. Evidence of increased concentration of nutrients, well above the expected 'background' conditions in all types of waters. Chronic and acute contamination has a direct implication for human health but also wider concerns for ecosystem structure and functioning.

Impact

Drinking water quality (recent evidence of link to childhood diabetes), localised eutrophication (freshwaters, estuaries and coastal) causing reduced bio-diversity and in some cases potentially toxic 'blue green' algal blooms, long-range and cumulative transport to the marine environment (e.g. North Sea).

Areas at Risk

The loss of nutrients may occur from throughout an agricultural landscape. However there are particular combinations of the physical environment with management aspects which exacerbate potential loss. A defined transport pathway is important. Therefore proximity of a 'nutrient source' to a watercourse and particularly the presence of drainage systems increases the risk.

- **Loss of nitrate:** freely draining soils that have the ability to nitrify especially those that are under regular cultivation.
- **Loss of soluble forms of phosphorus:** particularly occurs from coarse textured soils often having a limited capacity to retain (adsorb) P and aggravated by farming systems with a large annual P surplus so that soils become 'saturated'.
- **Loss of particulate associated N and P:**(see 'erosion risk') and also proximity driven factors such as river bank collapse (natural and enhanced by livestock) and runoff from areas of hard standing.
- **Areas at risk:** shallow groundwater is especially at risk from nitrate contamination. Standing waters from cumulative P loading. Some rivers and coastal waters although the impact is site dependant, due to non-linear relationships that exist between solute concentration and algal growth (impact). It is possible that naturally oligotrophic systems are at a greater risk per unit increase in the loss of N or P.

Practical Actions

Transport mechanisms and loss pathways for N and P differ. Nitrate (often the predominant form of N lost) is leached and therefore linked to drainage conditions. It can be transported long distances laterally. In contrast, the strong affinity of P for soil means that its loss is often (although not exclusively) associated with soil erosion and fine sediment loss. These significant differences in properties are reflected in the emphasis and approach that specific legislation and remedial measures adopt. For example, while action programmes for Nitrate Vulnerable Zones are inclusive of the total catchment area, those for sediment loss are targeted at specific management actions of localised features (eg. river banks).

Practical actions for reducing nutrient enrichment include:

- reducing the N/P capital of the system by removing agricultural land and/or restricting fertiliser/ manure use.(e.g. increasing the proportion of spring sown crops would reduce N applied).
- improving the efficiency of N/P use through farm nutrient budgeting (codes of good agricultural practice), more precise applications and timing of fertilisers/manures (see NVZ guidelines).
- minimising the production (nitrification) of the mobile nitrate anion in soil (e.g by reducing the extent of cultivation).
- intercepting nutrients somewhere along their transport pathway (e.g. riparian buffer strips/wetlands or for sediment, physical boundaries such as hedges). Management of riparian zones has had mixed results; it could be argued that these 'end of the line approaches' can only offer short-term improvements. The continued management of riparian zone is critical because of their potentially highly sensitive position adjacent to the watercourse.
- managing sediment 'hot spots', achieved through stabilisation of stream bank, fencing of stock, providing drinking troughs, moving feeding rings regularly, separation of clean and dirty water in farm yards, maintenance of drains.
- adopting good soil conservation techniques (e.g. avoid cultivation right up to the stream bank reduces potential for sediment loss and/or direct inputs of fertilisers and pesticides to surface waters; establishing good ground cover in autumn-sown crops on sensitive soils; reducing seedbed cultivation to keep a coarser tilth.

Linkages

Soil erosion risk reduction through soil conservation.
Biodiversity enhanced through appropriately designed riparian woodlands.
Streamwater ecology potentially improved through enhanced water quality (reduced sediment and nutrient inputs). Note possible negative effect in terms of shading and inappropriate leaf litter.
Nutrient surpluses likely to be reduced when land removed from agriculture/ nutrient budgeting.

Research Gaps

Establish if a direct linear linkage does exist between changes in soil nutrient status and N and P loss. This is essential for the most effective targeting of management options and would question the likely benefits from a nutrient budgeting approach.
Quantifying temporal and spatial lags in the land/water nutrient transfer sequence.
Linking a nutrient concentration with an actual impact.
Integrated nutrient, pesticide and microbiological management at field, farm and catchment scale (e.g. introducing minimum tillage systems may have benefits for nitrate and sediment loss but there may be a need for greater use of pesticides).
Rigorous testing of buffer strips under Scottish conditions.

6. Accelerated Loss of Nutrients: Critical Commentary

6.1. Background

Intensification and specialisation have been central to the modernisation of farming practices and together have led to greatly increased productivity. One resulting feature has been a geographical separation of livestock and arable production systems occurring at both a regional and national scale. For the reasons discussed below this has lead to a fundamental breakdown of farm-level nutrient cycles. Importation of feed is often a necessary feature of intensive livestock enterprises resulting in a flow of associated nutrients from arable to livestock systems.

Within the UK (as many countries) the complementary transport of livestock wastes in the opposite direction is uncommon, leading to the storage of this material and application to a very much reduced land base. Smith *et al*., (2000; 2001a and b) have reviewed manure production figures for England and Wales. Uncertainties associated with predicting the short-term availability of nutrients from livestock waste has exacerbated the situation. It has meant that, until recently, little if any compensation in the amounts of artificial fertilisers was made in advisory recommendations (e.g. Ythan, Domburg *et al*., 2000b). The legacy has been a substantial positive nutrient balance when calculated at the national scale (Sibbesen and Runge-Metzger 1995; and for P in the UK see Withers *et al*., 2001). There is a great deal more variation in the magnitude of the surplus at the regional or individual farm scale (Domburg *et al*., 2000a) for a range of farm types. It is evident from farm budgeting that, while intensive arable systems can use nutrients comparatively efficiently, they tend to rely upon artificial fertilisers. Livestock enterprises on the other hand, while also being productive, tend to be associated with substantial nutrient surpluses. The implementation of farm level nutrient/waste management plans (such as those carried out by SAC) can go a long way to reduce these surpluses and maximise nutrient use efficiency.

Using the example of P it can be demonstrated that we have approximately doubled the P content of some soils (Withers *et al*., 2001 and Soil Use and Management, special supplement 1998). When combined with soils having low capacities to retain P (e.g. certain coarse textured soils) this creates a particular set of problems (e.g. large areas of The Netherlands and Belgium). Thus, nutrient balances differ spatially producing a heterogeneous distribution within a catchment and they also vary temporally. The latter is associated with individual crops and rotations e.g. potatoes currently are associated with a large P surplus which is 'distributed' over the remaining rotation period.

6.2. Environmental consequences of nutrient surpluses

The environmental consequences of nutrient surpluses are various but can be considered within the context of world resources and long-term sustainability (see summary of recent conference workshop proceedings of Food chain 2001, Uppsala, Sweden http://www.foodchain2001.org/media/pressrelease/2001/FC2001final.html with workshop to be published in European J. Agronomy).

Other potential environmental impacts can arise from either a build up *in situ* of potentially toxic elements (e.g. cadmium impurities in soil phosphate rock sources, pesticide residues) or the offsite transport of various substances. The extent of any ‘impact’ will depend not only upon the quantity and availability of nutrients that are lost but also the sensitivity of the receiving (or any downstream) ecosystems. This has implications for establishing target values or threshold concentrations: do you base them on the source (e.g. soil) or the point of impact (e.g. sensitive water). The first assumes some direct link exists between a soil target value and an impact, which requires a direct relationship between a change in soil status, a loss and an impact (recent emphasis by the US EPA for example is moving towards establishing maximum daily loads for various substances, which we consider offer exciting possibilities (Parry, 1998; Whittemore, and Beebe, 2000)).

While there is considerable interest (both advisory and political) in developing nutrient budgeting approaches to better manage the flows of nutrients at farm and landscape scale (sustainability), it is important that there is a demonstrable effect upon loss. While some studies suggest this is the case for particular conditions (Pote *et al*., 1999; Heckrath *et al*., 1996), this might not be the situation for P where physical aspects such as susceptibility to erosion are equally important (Edwards *et al*., 2000).

Some of the complications that can arise from nutrient budgeting have been highlighted recently for the Ythan NVZ (Edwards *et al*., 2002) where the benefits likely to arise from a reduction in N surplus at the catchment scale will probably be evident over longer time periods. The significance of budgeting for reducing nitrate leaching and any associated improvements to the ‘local’ estuarine ecology are less clear. Even the longer-term benefits of recovery programmes may be difficult to predict if the results of Grimvall *et al*. (2000) are generally applicable. These authors recently reported the slow and smaller progress in the plans of Nordic countries to reduce N loading to the Baltic Sea and suggest that the inertia of the systems that control the loss of nutrients were seriously underestimated in the initial forecasts.

Research in Scotland has focused on the Ythan catchment. Three aspects of the catchments N cycle were considered where there was an attempt to place the period of fertiliser input, river loads to the estuary and algal uptake within a temporal context. A greater proportion (70%) of the average 1700 tonnes of nitrate-N leaves the catchment and passes through the estuary during the period September to March. Once in the river it is likely that most of the nitrate passes rapidly into the estuary which has a variable but comparatively short residence time (days). Fertiliser-N application peaked in April, with none apparently being applied during November to January. Previous work has demonstrated that the direct leaching of nitrate is not a significant source in this catchment. The algal growing season is typically between April to August which coincides with the period of smallest nitrate loads and concentrations. Comparison of the actual quantities of N involved indicates that algal uptake was predicted to be <0.2 % of the annual N losses. Significant temporal dislocation exists between the period of fertiliser application, nitrate loss and algal uptake. The conclusions of Edwards *et al*. are important for future cost-benefit analyses and highlight the need for a greater consideration of system functioning: -

'The designation of areas under the Nitrate Directive requires only nitrate from agricultural sources to be considered and while for many situations (e.g. drinking water standards) this may be sufficient, this may not be enough when impacts are the result of eutrophication. It is important to gain an overview of the temporal and spatial aspects of the various components of the N cycle so that a particular change in management can be matched with a change in the nitrate signal. A greater understanding of the linkages between the nutrient cycles of component parts of the system is required before the likely success of specific remedial actions can be evaluated. While it is likely that the proposed action plan could reduce nitrate leaching in the long-term, the extent to which these reductions are temporally distributed and relate to periods of active algal growth are still not clear.'

6.3. Practical remedies for reducing losses

Numerous practical remedies for reducing nutrient losses are identified in the PEPFA codes, and guidelines set out by Scottish Agricultural Pollution Group some of which are highlighted in the accompanying fact sheet. The costs to individual farmers of many of these practical measures are generally assumed to be low, although estimates of expected costs are rarely published alongside the guidelines. An exception to this is the recently published draft action programme measures to be applied in the Ythan Nitrate Vulnerable Zone (NVZ).

There is also considerable debate regarding the effectiveness of different measures to achieve the desired objective of reducing nutrient losses. A major part of the debate is focussed on the assumption of fundamental links between individual system components that are implicit to these measures. Some of these issues in relation to specific measures are considered in more detail below.

6.3.1. Nutrient Budgeting

Nutrient budgets are being increasingly proposed as a means through which nutrient losses can be reduced. There is a range of different budgeting approaches in terms of system complexity and accuracy. Whilst accuracy can be shown to increase with complexity, so do the information requirements, and therefore, the costs. A comprehensive assessment of the costs to farmers of nutrient budgeting relative the effectiveness of different types of budget has yet to be undertaken. Furthermore, given that the assumption of a direct link between farm level nutrient surpluses and potential losses of N and P has yet to be adequately demonstrated, their effectiveness as tools for reducing nutrient losses remains highly questionable. The work of Hooda *et al*. (1998) for example showed the tremendous difference in nitrate leaching that could be expected from pasture located in SW Scotland compared with NE Scotland, due very much to basic differences in edaphic and climatic properties. It is these factors that vary regionally that have not been adequately considered in current recommendations/legislation.

6.3.2. Low-input and organic farming systems

There is considerable debate surrounding the effectiveness of extensification and/or organically managed systems to reduce nutrient losses. Reduced productivities (extensification) do not automatically mean improvements in nutrient use efficiencies (which can be high for intensive pig/poultry units for example) or reduced losses. The ploughing up of pasture whether 'conventional' or 'organic' is likely to cause the stimulation of nitrification and increased nitrate leaching (Davies *et al*., 2001). Similarly, ensuring a balanced nutrient supply is likely to prove more challenging for organic systems restricted to using low solubility fertilisers (such as rock phosphates). A recent review of organic farming has been published (Stockdale *et al*., 2001) and many of these issues raised are complex and need to be more fully explored. There is also considerable debate as to whether the costs of moving to low-input and organic systems, along with impacts on output prices of large shifts in production systems, make low-input and organic systems a cost-effective option for all farmers.

Whilst the cost-effectiveness of adopting alternative production systems to achieve the single objective of nutrient reduction is questionable, it is likely that adoption of such systems could lead to improved biodiversity (see chapter 9) and reductions in pesticide residues (see chapter 7). This highlights the need for evaluations of cost-effectiveness to take into account multiple objectives.

6.3.3. Buffer strips

The effectiveness of riparian buffer strips for nutrient retention can be questioned for a number of reasons; for example, in many situations these areas tend to be by-passed by drains and ditches. It is also apparent that their effectiveness may well be enhanced by alternative locations within catchments away from riparian areas (Blackwell *et al*., 1999). Furthermore, there is still no strong evidence for fixing a minimum width of buffer zones, which will presumably depend upon the combination of many local site factors.

Whilst the cost-effectiveness of riparian buffer strips to achieve the single objective of nutrient reduction is questionable, they are likely to have other positive benefits, including enhancement of 'habitats' and stabilisation of stream banks. As with the adoption of low-input and organic systems, the cost-effectiveness of buffer strips should be examined in the context of multiple outputs.

6.3.4. Demonstration Projects

There are examples of 'demonstration' schemes that have shown the benefits of actively involving farmers (e.g. Loch Leven). These initiatives play an important role in examining the cost-effectiveness of different measures in a 'local' context. However, the transferability of the measures adopted, in terms of both cost and effectiveness has yet to be fully explored.

In summary, it is increasingly evident that many practical measures have environmental effects and impacts that are inter-linked and often additive (see for example Skinner *et al*., 1997) suggesting that cost-effectiveness and cost-benefits assessments of different measures must become more inclusive.

6.4. Policy issues

This chapter identifies a number of practical measures aimed at reducing nutrient losses, many of which are well documented in the codes of best and good agricultural practice. There is considerable evidence to suggest that at present a very significant proportion of farmers fail to adopt even the most basic of these practices. This suggests that policy intervention may be required just to bring all farmers up to a minimum set of management standards, let alone encourage widespread implementation of more costly management practices.

There is a range of policy mechanisms available to government for encouraging change in agricultural management practices. These are generally classified as regulation, economic instrument and voluntary approaches. Regulatory, or "command and control" approaches include legislation covering systems of licensing, regulatory controls on certain activities, for example through the designation of Nitrate Vulnerable Zones (NVS) and prosecution of offences. Economic instruments provide an alternative to the traditional regulatory approach and in the context of pollution control include instruments such as emission charges and tradable permits. Voluntary approaches include the encouragement of compliance with codes of best or good agricultural practice through education and information, or voluntary entry into a government scheme such as the Nitrate Sensitive Area (NSA) scheme.

The adoption of many of the remedial measure and practical actions set out above can be encouraged through any one of the policy options outlined above. However, the full costs, the distribution of the costs and the effectiveness of the different policy interventions vary considerably. Whilst an examination of the relative advantages and disadvantages of alternative policy mechanisms is beyond the scope of this commentary, it is an important consideration in any discussion of achieving practical action at the farm level.

6.5. References

Blackwell, M.S.A.., Hogan, D.V. and Maltby, E. (1999). The use of conventially located buffer zones for the removal of nitrate from diffuse agricultural run-off. *War. Sci. Technol*., **39**, 157-164.

Davies, M.G., Smith, K.A., Vinten, A.J.A. (2001). The mineralisation and fate of nitrogen following ploughing of grass and grass-clover swards. *Biol. Fert.Soils*, **33** (5): 423-434.

Domburg, P., Edwards, A.C., Sinclair, A.H. and Chalmers, N.A. (2000a). Assessing nitrogen and phosphorus efficiency at farm and catchment scale using nutrient budgets. *J. Sci. Food Agri*., **80**, 1946-1952.

Domburg, P., Edwards, A.C. and Sinclair, A.H. (2000b). A comparison of N and P inputs to the soil from fertilizers and manures summarized at farm and catchment scale. *J. Agri. Sci*., **134**, 147-158.

Edwards, A.C., Twist, H.and Codd, G.A. (2000). Assessing the impact of terrestrially derived phosphorus on flowing water systems. *J. Environ. Qual*., **29**,117-124.

Edwards, A.C., Sinclair, A.H. and Domburg, P. (2002). dentification, designation and formulation of an action plan for a Nitrate Vulnerable Zone: A case study of the Ythan Catchment, NE Scotland. *Europ. J. Agron*. (In Press).

Grimvall, A., Stålnacke, P. and Tonderski, A. (2000). Time scales of nutrient losses from land to sea – a European perspective. *Ecol. Engineer*, **14**, 363-371.

Heckrath, G. *et al*. (1995). Phosphorus leaching from soils containing different phosphorus concentrations in the broadbalk experiment. *J. Environ Qual*., **24** (5), 904-910.

Hooda, P.S. *et al*. (1998). A comparative study of nitrate leaching from intensively managed monoculture grass and grass-clover pastures. *J. Agr. Sci.* **131**, 267-275.

Parry, R. (1998). Agricultural phosphorus and water quality: A US Environmental Protection Agency perspective. *J. Environ. Qual*., **27**, 258-261.

Pote, D.H. *et al.* (1999). Relationship between phosphorus levels in three ultisols and phosphorus concentrations in runoff . *J. Environ Qual*., **28** (1), 170-175.

Sibbesen, E. and Runge-Metzger, A. (1995). Phosphorus balance in European agriculture – status and policy options. In: Tiessen, H (Ed.) *Phosphorus in the global environment: Transfers, cycles and management*. SCOPE 54, John Wiley & Sons, Chichester, pp 43-57.

Skinner, J.A., Lewis, K.A., Bardon, K.S., Tucker, P., Catt, J.A. and Chambers, B.J. (1997). An overview of the environmental impact of agriculture in the UK.. *J. Environ. Manage*, **50**, 111-128.

Smith, K.A.., Brewer, A.J., Crabb, J., Dauven, A. and Wilson, D.W. (2000). A survey of the production and use of animal manures in England and Wales. I. Pig manure. *Soil Use Manage*, **16**(2), 124-132.

Smith, K.A.., Brewer, A.J., Crabb, J. and Dauven, A. (2001a). A survey of the production and use of animal manures in England and Wales. II. Poultry manure. *Soil Use Manage*, **17** (1): 48-56.

Smith, K.A., Brewer, A.J., Crabb, J. and Dauven, A. (2001b). A survey of the production and use of animal manures in England and Wales. III. Cattle manures. *Soil Use Manage*, **17** (2), 77-87.

Stockdale, E.A., Lampkin, N.H., Hovi, M., Keatinge, R., Lennartsson, E.K.M., Macdonald, D.W., Padel, S., Tattersall, F.H., Wolfe, M.S. and Watson, C.A. (2001). *Agronomic and environmental implications of organic farming systems*. Advances in Agronomy, **70**, 261-327.

Whittemore, R.C. and Beebe, J. (2000). EPA's basins model: Good science or serendipitous modeling? J*. Am. Wat. Res. Ass*., **36**, 493-499.

Withers, P.J.A., Edwards, A.C. and Foy, R.H. (2001). Phosphorus cycling in UK agriculture and implications for phosphorus loss from soil. *Soil Use Manage*, **17** (3), 139-149.

6.6. Recent UK reviews

Nitrogen – Soil Use and Management 16: 167-174 Suppl. 1 Jun 2000
Phosphorus - Soil Use and Management 14: 124-130 Suppl. S Jun 1998

Please see Appendix 3 for selected bibliography on nutrients and faecal pathogens