Final Report **Project No. WFD13**

APPRAISAL OF RURAL BMP'S FOR CONTROLLING DIFFUSE POLLUTION AND ENHANCING BIODIVERSITY

December 2004

SNIFFER
SCOTLAND & NORTHERN IRELAND FORUM FOR ENVIRONMENTAL RESEARCH

DISSEMINATION STATUS

Unrestricted

RESEARCH CONTRACTOR

This document was produced by:

SAC

West Mains Road,

Edinburgh

EH9 3JG

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Appraisal of Rural BMP's for controlling diffuse pollution and enhancing biodiversity

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EXECUTIVE SUMMARY

a) Best Management Practices (BMPs) are available for control of farm derived diffuse pollution (sediment, sediment bound P, soluble nutrients, faecal indicator organisms and pesticides). BMPs have five main points of application: planning/general farm measures, in-field, field margin (or riparian), in-stream and steading measures. BMP efficacy is enhanced if a “treatment train” approach is adopted: firstly reduce inputs, then detain potential runoff water, then prevent soil dislodgement and runoff downslope, then intercept sediments before reaching watercourses.
b) Among the most effective BMPs are nutrient budgeting, leaving winter stubble after cereal crops, cultivation after row crops, headland subsoiling, use of buffer strips, restricting water access by livestock, and constructed wetlands. Field margin and watercourse BMPs are not recommended as alternatives to other BMPs, but may be important as additional options within the treatment train.
c) A theoretical analysis of buffer strip design for sediment removal suggested that slopes >7° have low efficiency of sediment removal, especially where significant flow concentration occurs. Therefore steeply sloping zones should not be included in the specification of the overall width required for pollution control. Sediment removal was sensitive to flow velocity, but not very sensitive to buffer width.
d) The efficacy of the Green’s Burn Buffer strip (Loch Leven catchment) during storm events was assessed. Before buffer strip installation the mean event total P concentration (TP) (for 3 significant events) was 1.1 ±0.6 mg TP/L and after installation (3 significant events) it was 0.5±0.16 mg TP/L. For suspended solids (SS) the values were 233 mg/L ±115 and 94±62 respectively. More data need to be collected before firm conclusions can be drawn.
e) Sediment collected in the Green’s Burn buffer strip following a major storm event contained lower P and organic matter content than the field soil, which suggests that the finer colloidal material (which contains a large proportion of the readily available P) is not being collected effectively by the buffer. Most sediment deposition occurred before or within the first few metres of buffer and this suggests that a buffer width of >10m may not be warranted for sediment control alone. Larger buffer strip widths may be warranted for promotion of diverse vegetation and food resources for birds, but this depends on active and appropriate management, to ensure a diverse habitat emerges.
f) Invertebrate diversity in the Green’s Burn buffer strip was poor. A greater amount of heterogeneity in vegetation type and structure would be needed to improve

biodiversity. To this end, active management, either in the buffer (more frequent cutting, leaving cut vegetation in swaths, some open space), or in field headlands (leaving crops unharvested, sowing of small areas of winter seed bearing crops such as rape or kale, to promote bird activity) could be encouraged.

g) The efficacy of fencing to reduce dairy stock access to water was studied in the Cessnock catchment, Ayrshire by event sampling up and downstream of fenced and unfenced reaches of subcatchments. There was a clear impact of the farming on pollutant concentrations in both the fenced and unfenced reaches studied. For all pollutants studied, the dominant factor controlling the pollutant concentration was the scaled discharge, but data were inconclusive in evaluating the effect of fencing on faecal coliform pollution, because of the impact of other sources of pollution such as steading sump overflow.

h) Major structural bypass routes (steading sump drains, surface runoff drains from fields) occurred within both arable and dairy farm systems studied, and these should be carefully identified and managed along with diffuse pollution BMP installation. Farm auditing is necessary to identify diffuse pollution sources and is a prerequisite for assessing whether BMPs are being correctly and effectively used by farmers.

i) Field plot studies to investigate the relative efficacy of buffer strips for faecal coliforms and inorganic sediments showed that faecal coliform capture is less efficient than inorganic sediment capture. This suggests that fencing improves water quality mainly through reduced bankside access, not through improved filtration of runoff. This means that provision of drinking water supply off stream may be a more cost-effective pollution control measure than fencing. However, as it is not known whether this will significantly reduce stock movement on the banks and into watercourses, further studies are recommended.

j) Monitoring of BMPs needs to be event based, and both inlet/upstream and outlet/downstream need to be sampled. We recommend a fixed installation of flow proportional sampling, with routine servicing, discarding of waters when no events occur. For routine analysis we recommend focus on robust determinands (TP, SS, DOC (dissolved organic carbon), turbidity, total N or NH_4 (with a nitrification inhibitor, where feasible)), rather than highly labile/degradable determinands (SRP (soluble reactive P), FIOs, nitrates), in order to make assessment cost-effective, timeous and feasible. FIOs and SRP will be useful for more detailed studies.

k) Conversion from winter to spring cereals is one potential BMP which has been investigated on a spatial basis across Scotland. For each parish in which arable or mixed cropping is dominant, a probability factor for winter cereals occurring on soils with a moderate or high risk of erosion has been determined. Nearly 70% of land classified as 'spring and winter sowing dominant' occurs on land with a medium or high risk of soil erosion. The areas where this factor is highest are along the east side of Scotland, from the Laurencekirk area southwards with the highest probability occurring in a number of parishes in East Lothian.

l) Soils within 10 metre buffer strips of watercourses have been aggregated into 'wet' and 'dry' categories based on their HOST classification. This enabled us to identify the proportion of soils within the 10 metre buffers that have a drainage impedance and therefore likely to have field drains that will reduce their effectiveness in preventing a proportion of leachable pollutants (e.g. nitrates and FIOs) from reaching water courses.

m) Areas with predominantly grassland farming have the highest proportion of 'wet' potential buffer strips - much of Ayrshire and Central Scotland - and areas with predominantly winter cereals areas also have relatively high proportions, for example the Merse of Berwickshire, the Lothians, Fife, parts of Strathmore and the Peterhead area.

n) If all the watercourses in Scotland had 5 or 10 metre buffer strips applied then the ‘loss’ of land from agricultural production would be approximately 1.3 and 2.7 %. Note that this does not equate with loss from agriculture as many of these areas will be in scrub or woodland already. The actual loss of land based on actual implementation is likely to be much less than this.

1. INTRODUCTION

PRINCIPLES OF CONTROL OF FIELD DERIVED DIFFUSE POLLUTION USING BEST MANAGEMENT PRACTICES (BMP'S)

Diffuse pollutants from farmland include nitrate, ammonium, oxidisable organic carbon, soluble and particulate phosphorus, pesticides and faecal indicator organisms (FIOs), as well as sediment in its own right. Diffuse pollutants can be transported by surface runoff, by subsurface "interflow", via field drainage systems, and through deep percolation to groundwater which later contributes to stream baseflow. Pollutants such as phosphorus may be associated with particulate (>2μm) or colloidal materials (<2μm), whereas, for nitrate, solution phase transport is more important. A large proportion of the total annual diffuse pollution loss, especially for those pollutants that are physically associated with particulate material, occurs during episodic high intensity storm events. Although the main transfer of particulates and colloids will occur via surface runoff, transport of particulate-sorbed pesticides, colloid and bacteria can also occur through vertical movement and through field drains. Agriculture is a major source of all of these pollutants, and they cause downgrading of both surface water and groundwater quality. In Scotland, highest priority is attached to:

- The mitigation of phosphorus pollution because of the numerous lowland lochs, such as Loch Leven, Kinross and the Lunan Lochs, Dunkeld, which have a high biodiversity interest and are subject to P limited eutrophication;
- The maintenance of microbiological standards for bathing water, which have been shown to be strongly affected by livestock in catchments dominated by dairy farming;
- The prevention of groundwater pollution by nitrates in designated Nitrate Vulnerable Zones;
- The upgrading of surface water quality with respect to oxidisable organic matter and ammonium, associated with polluting discharges from both point and diffuse sources.
- The reduction of the levels of pesticides residues to be found in surface waters draining arable areas and areas where sheep dipping is practiced.

Best Management Practices (BMPs) are measures that can be applied to control farm pollution that results from small distributed sources (diffuse pollution), rather than discharge from a single pipe or point source. Four main points of application across the farm have been identified (D'Arcy, pers. comm): planning/general farm measures, in-field, riparian (or field margin) and steading measures. We would include a fifth category, namely in-stream measures, such as ponds and wetlands, which may be dealing with both field and steading sources of pollution. Planning measures include development of farm and field based nutrient budgets, waste management plans and pesticide spraying plans. In-field measures include conservation tillage, headland subsoiling, choice of crop and contour strips. Riparian and field margin measures include buffer strips and livestock exclusion by fencing. In-stream measures involve some modification or construction of surface water bodies and watercourses (such as constructed wetlands and settling ponds). Steading measures include slurry storage, cattle access routes and separation of clean and dirty water.

Hilton (2005, in prep.) summarises the range of BMPs available in these different components of the farm landscape. He recognises that BMPs should be applied in a hierarchy or “treatment train”. These are firstly to reduce inputs, secondly to detain water to reduce peak flows, thirdly to prevent dislodgement of bare soil (and soil-bound pollutants), fourthly to prevent runoff downslope, and finally to intercept sediment before it reaches watercourses, and this hierarchy should be borne in mind in planning an approach to minimising diffuse pollution through BMPs.

There are many examples of successfully implemented BMPs. For example Dampney et al (2002) state that riparian buffer strips are effective for reducing pollution by nutrients, sediments, FIOs, BOD, veterinary medicines and pesticides. It is to be expected that they will also function to promote farmland biodiversity, as the field margin will have different vegetation from field and watercourse, and will act as a reservoir for invertebrates, and a corridor for flora and fauna. In addition they have the potential to provide increased access for the public. In Scotland, important issues regarding their likely effectiveness need to be addressed. For example where runoff is concentrated (eg from ridged crops such as potatoes) and where glaciated landscapes make techniques such as contour cultivations very difficult, large flows may be generated at specific points at the field edge, making buffer strips ineffectual (see Fig 1). Moreover the long-term removal of particulates depends on efficient filtration, adsorption and sedimentation of suspended material in the buffer strip. The effectiveness could be expected to vary due to local site specific factors such as width and gradient of the buffer strip, position in the landscape, soil texture and slope, with sedimentation rate of entrained particles and with the seasonal growth and type of vegetation cover.

The specific objectives of this project as set out by SNIFFER in their tender document were as follows:

A. To review relevant data relating to the effectiveness of BMPs;
B. To establish pilot sites representative of land use, for the monitoring of effectiveness of buffer strips;
C. To undertake monitoring at a range of scales in order to assess the effectiveness of buffer strips;
D. To specify the effectiveness of buffer strips for a range of diffuse pollutants;
E. To compare the effectiveness of buffer strips with in-field measures such as hedgerows;
F. To recommend a monitoring protocol which SNIFFER members can use to assess the effectiveness of BMPs and agri-environment schemes and scope a national study.

These objectives have been met in this report as follows:

Chapter 2

a. A review of the literature relating to the effectiveness of BMPs for pollution control, considering in field, field margin and in-stream methods[1].
b. Development of a theoretical approach for assessment of site suitability for buffer strips.

Chapter 3

a. Establishment of pilot sites and field plots in arable and dairying areas of Scotland, for the monitoring of the effectiveness of buffer strips and stock exclusion for pollution control and biodiversity enhancement .
b. Monitoring of these sites at a range of scales in order to assess the effectiveness of buffer strips in terms of water quality and biodiversity enhancement.
c. Conclusions drawn from field data on the effectiveness of buffer strips for a range of diffuse pollutants and guidance on optimal width and vegetation management.

Chapter 4

a. Assessment of the appropriateness of the reviewed (Ch 2) and monitored (Ch 3) BMPs to Scottish conditions and suggestions for criteria for assessment of site suitability;
b. Recommendations for a monitoring protocol which SNIFFER members can use to assess site suitability and efficacy of BMPs ;
c. Scoping of the development and application of a rule-based GIS approach to determine the potential extent and location of the deployment of selected BMPs in Scotland.

Chapter 5

Overall conclusions

REFERENCES

Dampney P.M.R., Mason P., Goodlass G. & Hillman J. 2002. Methods and measures to minimise the diffuse pollution of water from agriculture, – a critical appraisal. Report to Defra for contract No. NT2507. 131pp.

Hilton J. 2005 (in press). BMP Handbook. Information for advisers on BMPs for minimising diffuse agricultural pollution.

[1] We have not considered in depth the BMP groupings mentioned by Hilton (2005) of steading pollution control, as we understood steading pollution to be a point source, as discussed in the review of Dampney et al. (2002).

2. REVIEW OF BEST MANAGEMENT PRACTICE IN COMBATING AGRICULTURAL DIFFUSE WATER POLLUTION FROM FIELD SOURCES

Diffuse pollution from agricultural sources can be categorised into four broad classes, dealing with:

- Pollution from excess nutrients causing eutrophication problems.
- Siltation problems caused by the erosion and removal of soil mineral particles.
- Potential human health problems arising out of the spreading of faecal indicator organisms (FIOs) associated with farm livestock.
- Pesticides and other chemicals

Nutrient pollution is chiefly caused by nitrogen and phosphorus from excess or mis-timed fertiliser applications (in both inorganic and manure forms), and the severity can also partly depend upon individual weather events. Nitrogen pollution is mainly caused by the removal of soluble nitrate nitrogen from soil and manure by leaching to drain-flow in addition to surface run-off, whereas although some phosphorus pollution is of soluble material, the main transport pathway is of sorbed phosphorus being transported on eroded mineral particles. Therefore the factors controlling phosphorus pollution are often synonymous with those that govern the erosion of soil mineral particles and the ensuing siltation problems in waterways[2]. Likewise, the prevention methods will often be similar. FIOs (total coliforms (TC), faecal coliforms (FC) and faecal enterococci (FE)) can be transported either in suspension in leached and run-off water, or attached to particles in run-off water.

In the following section we shall examine the methods available at each stage of the "treatment train", and assess their effectiveness in controlling each of the categories of pollution identified at the start, and their appropriateness within each of the regions and farming systems mentioned above.

IN-FIELD METHODS (INCLUDING PLANNING TOOLS)

A comprehensive list of management tools by which eleven separate pollution indicators could possibly be controlled in the agricultural environment were identified in a major report to Defra on methods of control for diffuse pollution and measures taken to implement control (Dampney *et al.,* 2002). Those methods judged to be most relevant and most effective have been selected and reproduced in Table 1 below, for those methods which exercise control at the first stage of within-field defence against pollution. They have been set out in approximate "treatment train" order.

[2] However, where organic and soluble forms of P are prevalent, this may not be the case.

Table 1: Possible control methods to reduce the release of pollutants in-field, including planning tools. ++ = strong positive effect; + = positive effect; - = negative effect. Farming systems are arable (A) or livestock (L)(grazing) with either mixed farms or manure used on arable farms (L/A)

Control Method	Position in "treatment train"	Farming System	Pollutant category			
			Soluble	**Sediment**	**Pesticides**	**FIO**
Planning tools:						
Crop nutrient management planning	1	A/L	++			+
Crop pesticide management planning	1	A & L			++	
Restricted spatial and temporal application of crop inputs	1	A & L	++		++	+
In field tools:						
Over-winter ground cover	3	A	++	++	+	+
Spring sowing and early seedbeds	2,3	A	+	++		
Avoid the use of tramlines in winter	2,3,4	A	+	++		
Strip cropping	2,3,4	A	+	++	+	+
Stocking rates	1,3	L	++	++		+
Cultivation for soil stability (inc. Sub-soiling)	2,3,4	A	-	++	+	+
Minimal cultivation	2,3	A	+	+	+	+
Contour management	2,3,4	A		+	+	+
Feed composition	1	L	+			
Manure application rate	1	L/A	+			+
Manure treatment	1	L	+			+
Closed spreading period	4	L/A	++			+
Soil incorporation of manure	4	L/A	-	-		+/-

Treatment train order: 1=reduce inputs,2=detain water,3=prevent soil dislodgement,4=prevent runoff

Crop nutrient management planning

Much of the pollution by soluble plant nutrients is because they have been supplied in excess, either prophylactic applications of extra amounts of inorganic nitrogen and phosphorus, or the nutrient content of applied livestock manure not being taken into account when applying fertiliser. A greater attention to existing levels of nutrients in soil and records of all applied fertiliser to land would help a great deal in planning only the necessary amount for the next crop, and current recommendation systems such as SAC fertiliser recommendations (SAC, 2003), RB209 (Anon, 2000), or models such as MANNER (Chambers et al., 1999) help in doing this a great deal.

Where N inputs are balanced with crop requirements, nitrate leaching following crops appears to be determined mainly by post-senescence mineralisation of soil nitrogen, rather than by recent fertiliser inputs. The observation that nitrate leaching risk following arable crops is relatively insensitive to fertiliser input up to a 'break point' at the economic optimum, and thereafter increases more steeply has been reported by many workers including Shepherd & Sylvester-Bradley (1996) and MacDonald et al., (1989). These results suggest that the steep increase in nitrate leaching risk above a break point can be explained in terms of a surplus of nitrogen supply (from all sources) over total crop nitrogen use. The most cost-effective action to reduce nitrate leaching is to ensure that N inputs do not exceed the economic optimum.

Crop pesticide management planning

Pesticides applied to crops, whether herbicides, fungicides or insecticides, can find their way into local watercourses either as diffuse pollution from field application or from point sources of pollution at farm steadings. The entry routes from diffuse sources, in order of priority (highest first) are; spray drift, drain-flow, leaching, surface run-off, through-flow and base flow seepage (Carter, 2000). Those from point sources arise from; spillages, washings and disposal, faulty equipment, tank filling, sumps and soak-aways, over-spray & consented discharges (Carter, 2000). Many of these potential pollution routes can be obviated by careful management and planning. Simply ensuring adequate training of operators, inspection and repair of equipment, adequate storage tanks and drainage systems, choice of product and use according to proper protocols for environmental safety, will go a long way to reducing both forms of pollution (Carter, 2000). More specifically for diffuse pollution sources, the maintenance of some of the other mechanisms listed here, such as buffer zones and vegetative barriers will provide good reduction mechanisms.

Restricted spatial and temporal application of crop inputs

The restriction of when and where nutrients and pesticides may or may not be applied should be part of the above two management plans. These may restrict the application of nutrients and manure to times when land will not be damaged or excessive rainfall is unlikely, and similarly for pesticides when wind speed and direction are not such that will cause problems.

Over-winter ground cover

The simple expedient of ensuring a good vegetative ground cover during the winter period will operate at two levels to control pollution by both leaching and surface runoff transport routes. Ground cover can simply be the stubble and litter of last season's crop (and this is also beneficial to wildlife). Chopped straw from the combine, which is a feature of modern "stale seedbeds" in minimum tillage regimes (SMI, 2003), has been shown effective in partially controlling erosion (Chambers *et al.*, 2000a). If a spring crop is to follow then a specifically sown cover crop (such as winter rye) is often considered a better option for erosion control in England, but this is often difficult to achieve in Scotland, because of unsuitability of timing for cover crop sowing because of later harvests, and slow, anaerobic decomposition of cover crop residues in the spring, following ploughing. Nonetheless, well established cover crops may provide a more complete barrier to the erosive effects of rain during the drainage period, than the rather sparser cover that winter cereals provide. In addition they tend to maintain an adequate infiltration capacity to the soil, and by these two mechanisms will combat sediment loss (and associated phosphorus losses). Nitrate loss by surface run-off will be controlled by the same mechanism, but losses from leaching will not necessarily be abated.

The use of winter cover crops is designed to reduce leaching by causing labile nitrate nitrogen (which is over 90 % of leached nitrogen) in the topsoil to be taken up into the crop cover rather than be leached. A specific crop need not be sown, as re-growth of weeds and/or volunteers from the previous crop will also take up N. Sown winter crops, with the exception of early sown oilseed rape do not tend to take up sufficient (<10 kg ha^{-1} N) (Cuttle *et al.,* 2004). However, care has to be taken to establish a crop cover early, so that soil organic nitrogen mineralised from the cultivation event is taken up by the crop before significant winter drainflow commences. Nitrogen levels may be particularly high following certain high value, and well fertilized crops, such as potatoes, brassicas and peas, and set-aside land where N has accumulated over a season (Goulding, 2000). A well sown cover crop, such as winter rye or forage rape, can reduce winter leaching losses by 10 – 15 % in drainage water compared with fallow land (Shepherd & Lord, 1996) and if established early, August or early September, a cover crop can reduce N leaching by 50 % taking up 40 – 50 kg ha^{-1} N (Lord *et al.*, 1999), and Johnson *et al.* (2002) found them particularly effective after wheat and barley reducing average losses from 57 to 12 kg ha^{-1} N over 10 years. Similarly Shepherd (1999) reported an average uptake of 25 kg ha^{-1} N on sandy soils. Cover crops require careful management however, having to be ploughed in before they develop further in the spring and possibly becoming a N immobilising residue in the seedbed. In addition their use can lead to drier seedbeds in the spring (Davies *et al.,* 1996), and a reduction in hydraulic re-charge in the autumn and over-winter drainage. Ease of cultivation in Jan/Feb and sufficient time to allow initial rapid decomposition of cover crops to be completed before sowing are also important considerations (Ball, pers.comm).

Crop cover during autumn/winter is an important factor determining the fate of the nitrate present in soil in autumn. An early-sown and well-established autumn crop can take up a substantial amount of N (e.g. 20-80 kg/ha) during autumn and early winter and thus decrease the amount left in soil and exposed to leaching. In MAFF trials, where nitrate leaching is measured directly, there was at least 20 kg N/ha less nitrate leaching in a winter cereal sown in mid-September compared to bare fallow. However, autumn sowing is not effective in decreasing leaching if emergence or sowing is late (e.g. in October) due to limited crop cover and growth. Leaching losses following slurry applications to grass in September are also reported to be slightly lower than in October, reflecting greater slurry N uptake by the grass crop at the earlier application.

The actual nitrogen uptake of winter-sown crops or grass during the autumn to winter period will vary greatly depending on crop type, weather, soil conditions, sowing date and other agronomic factors. Typical N uptake rates during the autumn/winter for early sown and well growing crops were reported by Aitken (2002) to be:

Winter oilseed rape	50-100 kg N/ha
Winter barley	20-30 kg N/ha
Winter wheat	5-10 kg N/ha
Grass	20-30 kg N/ha

Spring sowing and early seedbeds

Speirs and Frost (1985) documented the incidence of soil water erosion on arable land in eastern Scotland and concluded that a disproportionate number of erosion observations (65%) were associated with winter cereals crops (19% of tillage area), whereas spring cereal crops (60% of tillage area) were associated with only 5% of erosion events observed. This strongly suggests that reducing the area of winter cereals grown, while leaving the stubble over winter to protect the soil surface would be a particularly appropriate measure for reducing pollution from sediment bound pollutants. A move to spring sowing could also, by delaying cultivation until spring, avoid the mineralisation of large amounts of nitrate in the autumn just before the hydraulic recharge and its subsequent leaching during the over-winter drainage period. Vinten et al. (1991) found an indication of more nitrate leaching (30±10kg N/ha, n=6) from fallow plots of clay loam soil that had been cultivated in autumn (chisel ploughing and subsoiling) compared to those left in stubble (21±2 kg N/ha, n=2). Dunn et al. (2004) reported that the difference between N fertiliser input and N offtake, based on national data, was 29 , -8 and 27 kg N/ha for winter wheat, spring barley and winter barley respectively.

Avoid the use of tramlines in winter

Tramlines represent extreme forms of wheelings and are compacted areas within the crop which will be prone to surface run-off during periods of excess rainfall. Because they tend to be carried out in the same place year after year, compaction can become well established at the base of the cultivation layer, reformed annually at the surface and if they run up and down slope can be the site of water shedding and even rill erosion in places. They have been demonstrated as sites associated with accelerated erosion of particulate P from arable land (Chambers *et al.*, 2000a), and at the least their formation should be delayed until there is some degree of protective crop canopy cover over the soil surface.

Strip cropping

Strip cropping is another method aimed primarily at reducing the surface erosion by run-off within fields. It operates by sowing the crop in strips (20 m wide or more) along contours of slopes whilst leaving narrow (4 m or at least a drill width) strips of fallow or drilled grass between them. These grass strips thereby act as “filters”, although it has been shown that sediment deposition does not take place within the strip, but in areas of ponding upslope of the strip (Ghadiri *et al.*, 2000). When studied under controlled conditions, Ghadiri *et al.* (2000) found that narrow strips of only 20 or 40 cm were only effective for larger sediment particles, and that fine fractions passed through the strip unaffected, and were even augmented by erosion from within the strip. In conclusion they stated that such narrow strips did not reduce downslope transport of pollutants by surface run-off (Ghadiri *et al.*, 2000). Melville & Morgan (2001) however, worked with 1 m wide strips of either *Festuca ovina* or *Poa pratensis* grasses or bare soil, in a field setting on a 5^0 slope. Both grass species significantly reduced run-off losses compared to the bare surface (less than 10% of the soil loss), but there was no difference between species despite different sward densities and canopy structures. It was again shown that deposition occurred upslope of the barrier, not within it (Melville & Morgan, 2001). Although, shown to be less effective above, bare soil strips are in effect one system used in the USA, where strips are left un-tilled between tilled areas for maize (Morrison & Sanabria, 2002). However, the no-tilled strips in this case do contain stubble and harvest residues left on the surface, which act as some sort of barrier to surface water movement. The mechanism described above where ponding occurs upslope of the barrier, and so is prevented from leaving the field, is essentially the same as that discussed in the section on vegetated buffer zones. Strip cropping can be considered a form of contour management.

Cultivation for soil stability (including sub-soiling)

Many problems of sediment transport and surface run-off can be avoided if good management practice is more strictly observed with respect to the timing and soil conditions that apply when cultivation is carried out. Problems arise when soil is worked when it is too wet to be friable and in a plastic condition. Compression causes compaction at the base of the topsoil, and shear damage occurs at the surface under wheels, or at plough depth where a "pan" can be created. Both lead to impeded rainfall infiltration and eventually exacerbate run-off and erosion at the surface (NSRI, 2001).

Other cultivation practices such as sub-soiling (to alleviate compaction and break plough pans), have a positive effect, if carried out correctly, in that the connectivity of fissures in the topsoil with a permanent under-drainage system is improved. This will widen the window for soil trafficability and lessens he likelihood of the structural damage mentioned above. Other practices also aid the infiltration of rainfall, such as leaving a rougher seedbed, with not too fine a crumb structure nor rolled to a high bulk density.

Minimal cultivation

It could be considered that reduced cultivation methods are a mechanism to control diffuse pollution, simply by being within the range of options available for cultivation for soil stability. Minimal (also often termed reduced) cultivation methods rely on shallow tine or disc cultivators being used instead of a plough and require fewer passes with subsequent harrows to generate an adequate seedbed. They are best used when allied to the retention of straw and harvest residues on the surface and the practice of controlling weeds in a "stale" seedbed (SMI, 2003). Surface erosion is reduced by the residue cover and rough surface, and the risk of topsoil damage and panning are avoided by not using a plough. Seedbeds produced by minimal tillage methods generally retain more moisture and nitrogen than those produced by conventional means, due partly to a reduction in the amount of nitrogen mineralised in the soil caused by soil disturbance (Dampney *et al.,* 2002).

Direct benefits of reduced nitrate leaching have also been demonstrated from both reduced and zero tillage. Compared with direct drilling of autumn cereals, ploughing led to 20 % more nitrate leaching (Goss *et al.,* 1993), though conversely direct drilling led to greater losses in spring cereals. In Lincolnshire Johnson & Smith (1996) found reduced cultivation systems led to reduced nitrate leaching of 44 kg ha^{-1} N over five years, though the effect diminished over time. Reduced tillage systems also tend to increase the organic matter content of surface soil, which itself leads to a more stable structure and better infiltration rates of incident rainfall, though may lead to higher nitrogen mineralisation rates eventually when ploughed again. Pierce *et al.* (1994) found rates were almost doubled to 18.4 g m^{-3} N when long-term no-till soils were ploughed.

Contour management

Contour management is simply the carrying out of cultivation, seed-bed preparation and drilling along the contours of the land rather than up and down slope. This counteracts the tendency for rill erosion to begin along the length of channels created by the soil operations. The degree of slope, along with the volume of flow discharge is one of the most influential factors governing the detachment rate of soil particles (Zhang *et al*., 2002), and a study of erosion in England and Wales found that rill erosion in winter wheat fields was mainly associated with valley features that concentrated flow, and wheelings and tramlines as well as poor crop cover (Chambers *et al.,* 2000a). Losses of P at the edge of fields from this form of erosion was estimated at 18 % of the total loss from agricultural land in England and Wales (Chambers *et al.,* 2000a), though effective control measured included those that help increase rainfall infiltration and reduce surface run-off. Contour management helps achieve these features by maintaining surface roughness, in the direction of water flow, and has been shown to effectively reduce sediment losses by 25-50 % on uniform slopes of up to 8^0 (Riding & Rast, 1989). It was cited as being one of the least effective mechanisms at controlling losses by Withers & Jarvis (1998) however, where losses of 6.3 kg ha^{-1} a^{-1} P were nearly twice those of the next least efficient method (conservation tillage) though much lower than no control at all (10 kg ha^{-1} a^{-1} P). It was however, one of the cheaper methods to apply and for that reason could be widely practised. Another reason would also be that it is considered an effective method of attenuating pesticide losses associated with surface flows, as well as excess nutrients (Carter, 2000).

It is however, only recommended for gentle slopes of less than about 8^0 gradient and is difficult to achieve on varied slopes without introducing channelling effects (Frost *et al*., 1990). At higher gradients safety concerns and discomfort dissuade operators from using this method of cultivation. In situations where a large bare surface and obvious channelling routes exist, such as potato ridges running up and downslope after cereals, then a secondary cultivation like sub-soiling across the downslope end of ridges may be effective. This practice increases vertical soil pore continuity to intercept waterflow and create penetration rather than run-off, impeding sediment removal before the field edge.

Stocking rates

Although there is relatively little information on the effects of stocking density on P and sediment loss (Withers *et al*., 1997), in predominantly grazing areas one cause of pollution problems can come from over-stocking. Schepers and Francis (1982) measured 40 % less P loss from pastures when cattle were not grazing , compared with when they were stocked. Heavily grazed pasture produced twice as much run-off as lightly grazed in a Devon catchment, and P losses were predominantly in particulate form from areas where the stocking density has been high enough to cause compaction and poaching damage to the soil surface (Heathwaite *et al.,* 1990). Intensively managed permanent pasture may also accumulate considerable amounts of available P in the topsoil leading to higher losses in run-off (Haygarth *et al.,* 1998).

Grazing also increases N loss by leaching from pasture, when compared to areas of cutting management (Scholefield *et al*., 1991; Cuttle *et al*., 2004). This has been shown to be considerably worse on sandy soils (Scholefield *et al.,* 1991), and so the siting of permanent pasture on heavier land if possible, would be the first management option to minimise diffuse N pollution, although this may enhance faecal indicator pollution losses. A lower stocking rate on pasture will reduce the area affected by urine and dung patches (hotspots for nitrate leaching), and also allow fertiliser rates to be reduced, thereby lowering the overall potential for N leaching (Cuttle *et al.*, 2004). This reduction has been demonstrated in Germany (Huging *et al.*, 1995) where 27 kg ha^{-1} N was lost from extensively grazed systems compared with 87 kg ha^{-1} N from intensive systems. In addition to reducing the stocking rate, limiting the time that grazing is allowed will also reduce N leaching. Removing cattle from pastures in early autumn has a very marked effect on lowering the availability of N in the sward for leaching. Grass at this time of year is less able to utilise N in urine and dung and so this becomes available for leaching (Kessler *et al.,* 2000), and Sherwood (1986) has shown that this available N increased from 3 % of that in urine during the spring, to 30 –50 % during September to November. Cuttle *et al.* (2004) detail the results from many other studies that also show the benefits of restricted grazing season on nitrate losses from pasture, and also benefits from more efficient grazing and stock management. Vinten et al (2002) also showed that restricted grazing before ploughing out of grass leys is an effective way of reducing nitrate leaching.

Maintaining a lower stocking density may also reduce the risk of FIOs from livestock dung reaching waterways and hence ultimately contact with human populations. However, Vinten *et al.,* (2004) showed than reduction in stocking rate alone is likely to be less cost effective than improved mitigation.

Feed composition

The risk of nitrate and phosphorus pollution from livestock areas can be reduced by lowering the amount of these nutrients in the system, and hence the excess that may be open to transport off site. Currently, dairy cows recover about 20 % of the nitrogen they ingest, though Bussink & Oenema (1998) contend that 43 % recovery is theoretically possible, and therefore the scope for reducing N in excreta is high. A review by Moorby & McConchie (2002) reported various measures that could be adopted to reduce N ingestion, and their modifications of sward composition, silage management and concentrated feed formulation are briefly discussed in Cuttle *et al*. (2004). A review by Castillo *et al.* (2000) concluded that the most important change however, would be to lower the overall N intake. When intake is above 400 g day^{-1} N (150 g crude protein kg^{-1} dry matter), most excretion is in urine, and this could be reduced by 40 % (and in dung by 19 %) by lowering the crude protein intake from 200 g kg^{-1} dry matter to 150. A study by Metcalf *et al.* (1996) indicated the likely benefits in practice, by feeding a specially formulated low-N diet to dairy cows, which reduced urinary –N loss from 169 to 122 g day^{-1} and faecal N from 178 to 157 g day^{-1}, with no milk yield penalty. Many other studies have been reviewed by Cuttle *et al.* (2004), mainly of dairy cattle, with the consensus that reducing the nitrogen in the currently high-N diet, could lead to a 30 % reduction in excreted nitrogen and thereby that available for leaching. Jarvis (1993) estimated that 21 % of leached N from a dairy farm was from manure, and so a 30 % reduction in excreted N could lead to a 6% reduction in total leaching losses.

Withers *et al*. (1997) also considered dietary manipulation of benefit in reducing the recycling of P in animal manure. These authors also point out that up to 80 % of dietary P intake in different classes of livestock is excreted, which suggests a considerable scope for the reduction in either total P intake or composition of feeds. One demonstrated modification is that pig feeds containing microbial or cereal phytase instead of calcium phosphate to improve digestibility, led to 30-60 % less P being excreted (Han *et al*., 1997). Budget assessments at the farm scale have shown that an excess of 20 % which can be found in dietary P, can lead to soil accumulation (Haygarth *et al.,* quoted in Cuttle *et al.,* 2004), which may be open to erosion and transport.

Secondary pollution issues may also be addressed by modification of imported animal feed, as there is the potential that methane emissions could be lowered in future by this means (Shepherd *et al*., 2003). Also, the inclusion of the heavy metals copper and zinc in pig and poultry and cattle feeds, can cause soil pollution problems over many years of affected manure application (Nicholson *et al*., 1999). If this becomes high enough, then translocation of sediment could lead to a diffuse pollution contribution from heavy metals as well as eutrophicating agents.

Manure application rate

It is axiomatic that a reduction in manure application rate will reduce nitrate (and phosphorus), and this has led to the introduction of the limit of 250 kg N /ha/year equivalent in Nitrogen Vulnerable Zones, which now cover approximately 12 % of agricultural land in Scotland. For intensive livestock units, this may mean transport of the manure off the farm (Smith *et al.,* 2000). Even when applied at rates well within the recommendations of the Code of Good Agricultural Practice, annual applications can lead to a situation of increasing N leaching (Shepherd 2001). This came from an increasing pool of labile organic nitrogen in the topsoil, which mineralises an increased amount of N each autumn. The largely unknown long-term dynamics of manure applications has contributed to the view that these should be according to a well designed manure management plan, which took into account the fertilizer value of manure (Smith *et al.,* 2000).

Similarly, manure application rates will have an influence on P loss in run-off. This relationship has been shown to be exponential rather than linear (Sherwood & Fanning, 1981), and suggests that inorganic P dressings should be adjusted to accommodate manure P additions. Other work by Withers & Bailey (2003) suggests that splitting the manure application will help keep phosphorus losses at a minimum, and this practice was also found to reduce the risk of nitrogen pollution by sub-surface flow when slurry was applied during late winter and early spring months (Parkes *et al.,* 1997). They recommended that application of several doses of slurry of less than 35 m^3 ha^{-1} was preferable to fewer higher dose rates.

Manure treatment

The treatment of livestock manure by composting or thermal means has the aim of reducing pathogen survival rates due primarily to temperature elevation, but also by producing sub-optimal pH and moisture regimes (Dampney *et al.,* 2002). There is the additional effect of accelerating the degradation of pesticides and veterinary pharmaceuticals. However in many of the areas of Scotland with the biggest manure management problems, waste management is slurry based, as straw is unavailable. This technology is therefore more likely to be effective in the mixed farming areas (NE and SE Scotland).

Closed spreading period

By avoiding fixed periods when heavy rainfall is likely and soil moisture at field capacity, then the risk of soil damage is less probable, and the immediate leaching and run-off of soluble and readily available nitrogen from the manure, is also reduced (Dampney *et al.,* 2002). This does however, necessitate sufficient manure storage capacity being available, and increases the possibility of odour nuisance during open periods when spreading does occur. Aitken et al 2001 reported that the available waste storage capacity is inadequate on a high proportion of Scottish livestock farms. 21% of farms had less storage than 1 month of the farm's waste production. The equivalent figures for 1-2 months was 18%; 2-3 months, 10%; 3-4 months, 12% and 4-5 months, 9%. 29% of farms had more than 5 months storage available, while 71% had less than 5 months storage. The risk of enhanced nitrogen leaching from applied manure is a problem during the autumn and winter months when utilisation of available nitrogen in the manure is poor (Smith & Chambers , 1993), but will also depend upon manure type, as the plant available N content varies considerably. Beckwith *et al.* (1998) found that manures with a high plant available N content (cattle and pig slurry and broiler litter) led to high leaching risk in September to November, but FYM with little available N resulted in much less leaching. The risk of N leaching from different manure types during autumn and winter months has been summarised by Cuttle *et al.* (2004)(adapted from Chambers *et al.,* 2000b) in Table 2.

Table 2: Estimated leaching loss (% of applied total-N) as affected by manure type, application time, and land use (reproduced from Cuttle *et al.* (2004) and adapted from Chambers *et al.,* 2000b).

	Proportion of total-N leached (%)		
	Arable		**Grassland**
Application date	**FYM**	**Slurry/poultry**	**Slurry**
September	7	19	12
October	5	16	17
November	3	10	10
December	2	4	1
January	1	2	0

Recommendations are that manure is applied in the spring when available N will be used most efficiently (Anon, 2000), but despite this Chalmers (2001) found that about 25 % of applications to grassland were still in the autumn. This is largely because of a perception that applying manure during the growing season will affect herbage quality and palatability (Laws & Pain, 2002). Aitken et al. (2001) found that about 50% of Scottish livestock farms applied manure and slurry in the spring.

Several literature reviews have demonstrated that the greatest nitrate losses from autumn/winter applied organic manures occurred from manures with large amounts of readily available N (e.g. Smith and Chambers, 1993; Beckwith et al 1998 etc).

Unwin et al., (1991) reported N leaching losses of 123 and 139 kg N/ha equivalent to 39% and 49% of the total N from poultry manure applied to a loamy sand in October and November. Smith and Chambers (1993) reported that nitrate losses following cattle slurry were significantly greater from the September to November applications compared to the control and December and January applications. Nitrate losses were greatest from the November timing (64 kg N/ha lost).

Beckwith et al (1998) found that manures with a high proportion of available N (i.e. cattle/pig slurry and broiler litter) applied to a sandy soil in September, October and November consistently increased N leaching losses over the four years at two sites. N leaching rates were significantly lower in December and January while FYM caused consistently less N leaching.

The ammonium (readily available) N content of manures is rapidly converted to nitrate-N and can then be used by plants or lost by leaching. Slurries and poultry manures are 'high' in readily available N (40-60% of total N) compared with FYM which is 'low' in readily available N (10-25% of total N). The rest of the N that manures contain is organic N which is released (mineralised) slowly over a period of months to years.

Soil incorporation of manure

When applied to arable land the rapid incorporation of manure and slurry is often advised to avoid the odour problems and gaseous pollution associated with it, and this is also the thinking behind slurry injection in grasslands. However, this could lead to cases of "pollution swapping", where gaseous pollution is exchanged for an increased risk of water pollution by excess nitrogen in solution (Williams *et al.,* 2000). It is one of only two cases in Table 1, where a negative impact is considered possible for soluble nutrients and sediment problems. Its impact on pathogens could be both positive in that it would restrict their transport, but also negative in that it may increase their survival in the soil (Dampney *et al.,* 2002).

Economic analysis of options for in field BMPs

Frost and Ramsay (1996) carried out a costed reconnaisance survey of erosion control options within the Greens Burn catchment, Perth and Kinross. The estimated reduction in erosion losses obtained for a range of potential erosion control measures is given in table 3.

Table 3. Estimated reduction in erosion losses in Greens Burn catchment obtained for a range of potential erosion control measures.

Option	Description	Erosion Reduction Expected	Annual net fall in profit (%)
1(a)	Cultivation Change[3] + 20% Grass	50%	11
1(b)	Cultivation Change + 40% Grass	75%	40
2(a)	Spring sown cereals & OSR	50%	49
2(b)	Spring sown cereals & OSR + 20% Grass	75%	55
3(a)	Increase Grass to 40% of rotation	50%	34
3(b)	Increase Grass to 60% of rotation	75%	58
4(a)	Diversion Terracing (wider spacings)	50%	23
4(b)	Diversion Terracing (narrower spacings)	75%	54

Annual net fall in profits associated these measures, for a typical farm for the catchment, (205 ha, 34% spring cereals, 21% winter cereals, 12% potatoes, 12% cabbage, 11% rotational grass for silage) is also given. This analysis strongly suggested that the suite of cultivation measures recommended (all stubbles left unploughed until shortly before following crop, no rolling of winter seedbed, ploughing of land after potatoes or brassicas immiediately after harvest, contour cultivation where feasible) was the most cost effective way of reducing erosion losses. Conversion from winter to spring cereals is expensive because yields are typically lower, and increasing the grass % in the rotation (as a grass margin, this would also serve to provide wide buffer strips), while more effective in terms of erosion control (75% control) would also be very expensive. However, with decoupling of the CAP from production to environmental payments, the relative costs of these measures may change. In addition, farmers will have to comply with a range of conditions under the Good Agricultural and Environmental Condition rules (SEERAD, 2004) which should help to mitigate erosion problems.

From our discussions with the farmer at Wester Gospetry (see Chapter 3, Box 1), it would seem there would be a lot of resistance to using looser seedbeds on winter cereals because of problems with weed control and non-uniform germinationHowever, he clearly recognises the value of overwinter cover, and subsoiling after harvest to promote infiltration. In field with late harvested potatoes, a grubbing tool is used to break up the beds, to reduce soil erosion. Strip cropping is not considered feasible because of the extra man-power resources required. .

FIELD MARGIN METHODS

Within the list of management tools identified in Table 2 of the aforementioned report to Defra on methods and measures of diffuse pollution control (Dampney *et al.,* 2002), were several aimed further down the treatment train, at the field margin, and the more widely accepted of these have been selected and reproduced in Table 4 below. These measures should be considered once those further up the treatment train (planning,

[3] All stubbles left unploughed until shortly before following crop, no rolling of winter seedbed, ploughing of land after potatoes or brassicas immiediately after harvest, contour cultivation where feasible.

input reduction and in-field measures to reduce runoff of sediment and pollutants) have been assessed and implemented, where appropriate.

Table 4: Possible methods to control the transport of pollutants to waterways (at the field margin). Extracted from Dampney *et al* (2002)(except those in italics). ++ = strong positive effect; + = positive effect; - = negative effect. Farming systems are arable (A) or livestock (L)(grazing) with either mixed farms or manure used on arable farms (L/A). Tie up with position in treatment train of Hilton (2004) also given.

Control Method	Position in treatment train	Farming System	Pollutant category			
			Soluble	Sediment	Pesticides	FIOs
Barrier ditches	5	A	+	+	+	+
Vegetative barrier strips and fences	5	A & L	+	+	+	+
Reed-beds	6,7	A & L	+	+	+	+
Constructed wetlands	6,7	A & L	++	+	+	+
Riparian buffer zones	6	A & L	+	+	++	+
Restricted access of livestock to waterway	6	L	+	+	+	++

5=field margin, 6=watercourse margin, 7=in-stream

Barrier ditches

A ditch that runs parallel to watercourses or transverse perpendicular to the flow of surface run-off, will intercept particulate and soluble pollutants before the reach free water, and has been found particularly effective for dilute effluents and dirty water from dairy steadings. By trapping sediment the ditch eventually fills up and needs regular maintenance (clearing) if overloading and failure are to be avoided. Maintenance measures should also follow guidance re maintenance of biodiversity.

Vegetative barrier strips and fences

Vegetative barrier strips can range from simply tall grass allowed to grow along fence-lines and watercourses, which provides a simple barrier and against surface run-off and windblown particles, to more densely planted strips at the edges of fields and along watercourses. These planted strips can be considered “buffer strips” (discussed in the section below on buffer zones), but could include taller species grown as interception plants to provide a barrier against spray drift of pesticides, so that aerosols do not reach free water (Carter, 2000).

Reed-beds

These can be considered an extreme form of vegetative barrier strip for surface run-off and sub-surface flow. Sited at outflows and points of confluence for water shed from fields, planted beds of *Phragmites australis*, provide both a physical barrier to water movement which causes sediment to settle, but also consists of a site of biological activity where the inactivation of FIOs and degradation of chemical pollutants will be enhanced.

Constructed wetlands

Similarly to reed-beds outflow areas of fields can be engineered to develop into artificial wetland ecosystems, where nitrogen and phosphorus nutrients will be captured and taken up by vegetation, before they can enter free-water and possibly exert a eutrophicating influence. The anaerobic conditions they will be developed in still water areas, at least in parts, will also encourage the full de-nitrification pathway for nitrates through to gaseous di-nitrogen. They will be discussed in more detail in the next section for "Watercourse methods".

Riparian buffer strips

Buffer strips are uncultivated zones at the edges of fields, adjacent to water courses and ditches. They function as pollutant control agents by a number of mechanisms, including sedimentation, true filtration, infiltration of water, adsorption and pollutant uptake by vegetation (see below). They also serve as a potential source of biodiversity in the landscape, and as corridors for wildlife. Often operated in conjunction with other in-field measures, and work by creating distance and a physical barrier between the source of pollution and watercourse. The maintenance of a buffer zone will allow sediment to settle, nutrients to be absorbed by bank-side vegetation and aerosols to be intercepted in the air by trees and tall vegetation. Their development can be traced to studies of the natural ecology of riparian zones (Naiman & Decamps, 1989), and they are adopted in afforested areas (Olson *et al*., 2002; Weston 1995) as well as agricultural areas(Ducros & Joyce, 2003). They can range from a few metres of grass or natural vegetation up to complex strips > 50 m in width running the length of waterways. One such complex system in the USA, is described by Hubbard & Lowrance (1994) as comprising three sections; 1) a narrow band (5-10 m)of permanent indigenous trees immediately adjacent to the stream; 2) a forest management zone where biomass production is maximised; and 3) a grass buffer strip < 10 m wide located between agricultural areas and the forested zone. However, any one of these zones, or similar combinations, can be considered and maintained as buffer zones and will be effective to some degree at reducing diffuse pollution. In different regions of Scotland simple grassed buffer strips of 3- 10 m wide, strips of natural bankside vegetation (<10 m) or afforested strips can all be envisaged, as appropriate depending on whether the area is predominantly grassed upland with afforested areas, intensive dairy grasslands, mixed agricultural land or arable lowland. Which is more appropriate will also depend upon the predominant form of pollution to be dealt with as they are more effective at different functions, and their application to UK conditions has been reviewed by Muscutt *et al.* (1993).

The efficiency of diffuse pollution control depends on a number of factors, which link with the mechanisms. The most important factors affecting performance are:

a. type of pollutant.

Vegetated buffer strips can be over 90 % effective at removing sediment bound P (Withers *et al*., 1997) and it is grass or dense herbaceous vegetation that is most effective for this. However, some studies have found them to be only moderately effective in the longer term (Dillaha et al., 1989). Buffer strips can prove equally effective for sediment bound pesticides, such as chlorpyrifos, 80 % of which was retained in grassed buffer strips in studies by Arora *et al.* (2003) compared with 49.7 and 51.2 % of the less strongly bound pesticides atrazine and metolachlor respectively. Their experiment also showed that a field to buffer strip area ratio of 30:1 was all that was necessary to achieve this level of removal. There was not a significant increase in removal when the ratio was 15:1. Boyd *et al.* (2003) showed that herbicides such as acetochlor and atrazine, which are mainly transported in solution in surface and sub-surface flow, are controlled more by infiltration rate than deposition, and so it is advisable that buffer strips are not located over permanent under-drainage systems which may collect such infiltration (Haycock & Burt, 1991).

Soluble phosphorus and nitrate pollution are also not contained very well by narrow vegetated buffer strips (Withers *et al*., 1997; Cuttle *et al*., 2004). A paired study of headwater catchments in the UK showed grassed buffer strips to be ineffective at reducing nitrate outflow to streams (Leeds-Harrison *et al*., 1999), though they can be effective at removing nitrogen in storm run-off from grassland (Heathwaite et al., 1998) where more of the N is in organic and particulate form, and because buffer strips will impede surfae runoff more than throughflow..

Schmitt et al. (1999) also found that removal of particulate materials by grass buffers was much better than removal of soluble pollutants. Little extra pollutant removal was obtained by extending buffer strip width from 7.5 to 15m. Young trees and shrubs planted in the lower half of the 15m buffer had no effect on buffer performance.

Bingham et al. (1980) found considerable variation in the efficiency of removal of the various pollutants, which can be attributed mainly to their degree of association with sediments. Fig 2 shows a summary of results for 7 rainfall events of 21-140mm. Syversen (2001) found that in simulated runoff experiments at 3 sites, with 5m and 10m buffers, grassed or forested, the retention of total P, total N and suspended solids was measured, during applications of 5L/sec over a 5m width of buffer for 2000L or 20,000L. SS retention (particle size not given) was>80% in most cases, intermediate for total P 40-60% in most cases and low for total N(,40% in most cases). However, the total N and total P were as soluble forms in this experiment, and flows are quite low (shallow sheet flow, not concentrated runoff). Retention efficiencies for total N and P were better in natural runoff conditions, and retention of SS was similar. Retention efficiency of the finest clay fraction (0.06-0.2um) was greater than larger fractions.Syverson (2001) interprets this as transport as aggregates. If this is the case, the extent of P transport will depend on whether the soil aggregates deflocculate during transport but it could also be because the finest clay particles will be more readily filtered, as their diffusion coefficient is significant (Ives, 1975).

Coyne et al. (1995) compared the transport and deposition of Faecal coliforms and soil from 22m long poultry manure treated plots through grass filter strips 9m wide, during simulated rain of high intensity (64 mm/h for 90 minutes – an approximately 10 year return period event). Whereas sediment trapping accounted for 99% of input on both plots, 24% and 57% of faecal coliform inputs were transported across the two plots into runoff. Note that there were no concentrating effects in these field plots, so where runoff becomes concentrated, or catchment areas relative to buffers are larger, this event would be a lot more frequent than a 10 year return.

Goel et al. (2004) found that the effectiveness of buffer strips receiving soil suspensions at 1 L/m buffer/sec varied with the pollutant in the order SS≈Total N≈Total P > soluble P≈faecal coliforms > nitrate. The main other factor affecting performance was buffer strip width, but the conclusion for flows of this magnitude or higher, is that soluble nutrients and FIOs are not very efficiently removed.

b. Buffer zone width.

Buffer strip establishment leads to reallocation of agricultural land to non-productive land. Recommendations vary from 10-60m for sediment removal and 5-90m for nutrient removal depending on slope, vegetation, soil etc(Castelle et al., 1994). Bingham et al. (1980) investigated the effect of grass buffer zone length in reducing the pollution from land application areas in grassed clay loam soils, with ratio of length of buffer zone to waste application zone varying from 0 to 2.6. Syversen (2001) found that increase in width led to higher retention but only by a small amount (7.4, 1.9 and 4.8% average increase across the 3 sites for total P, SS and total N 10m instead of 5m. The larger volumes led to poorer retention. Forested buffer strip showed higher retention of SS but not nutrients. Syversen (2001) studied the retention efficiency of buffer strips as a function of (i) Buffer zone width, (ii) amount of surface runoff water, (iii) seasonal variation and (iv) vegetation type at 4 sites in SE Norway. Relative retention increased with width, but specific retention decreased, because retention efficiency was highest in the upper part of the buffer. Winter retention of sediment was 15-35 times higher than summer retention because more sediment transport occurred in winter.

c. Slope.

Buffer strips are less effective on steeper gradients, or where outflow is concentrated into channels which may carry run-off across narrow (< 10 m) strips (Withers *et al.*, 1997). Rose et al. (2003) showed that the deceleration of flow velocity which occurs upslope of a buffer strip at modest land slopes plays a crucial role in the net deposition of sediment laden water. These authors described how re-entrainment of particles, and the carriage of larger particles further into the vegetated zone, increase progressively with gradient and sediment flow rate. Slopes >10% may not be effective when establishing buffer strips (Dillaha et al 1997) as these will tend to erode, rather than acting as sinks for sediment, especially if the slope of the adjacent farming area is less severe. Decreased efficiency with slope increase from 11 to 16% (Dillaha et al (1989) and 7 to 12% (Robinson et al. 1996) have been reported. This effect may however be mitigated to some extend by vegetation. Sediment deposition in buffer strips occurs at the leading edge of the vegetated area where a "hydraulic jump" occurs (Rose *et al.*, 2003). This deposition on the leading, field-side, edge of vegetated buffer strips leads to a wedge shaped barrier or "berm" forming which may subsequently cause ponding at the field edge. This may at first be beneficial by increasing infiltration rates over run-off, but will eventually cause transverse flow along the edge of the strip and lead to channelling and concentrated flow outbreaks across the strip. For that reason buffer strips require periodic maintenance to remove these "berms" (Withers *et al.*, 1997). Slopes of <1% may also be unsuitable because of limited hydraulic gradients for infiltration and lateral movement of water (Hayes and Dillaha, 1992).

d. Vegetation.

Permanent pollutant removal in forest is generally rather higher than in grass. However tree density is important as this will affect ground vegetation. If it is too dense then ground vegetation may be shaded out. Rapid vegetation growth leads to high permanent removal. Amongst tree species, aspen, poplar, and willow all have the potential for have high nutrient uptake. It should, however be remembered that uptake will only occur in the growing season. Density, height and stiffness of vegetation also affect efficiency of retention. Uusi-Kamppa (2002) found negative retention efficiency of dissolved P during snowmelt from a forest buffer strip compared with a reference plot. There was a positive retention from a grass buffer strip (cut, yield removed in summer). Leakage of dissolved P from frozen vegetation occurred in winter. Dilaha and Inamdar (1997), particle retention efficiency decreases with time, due to sediment accumulation. However Syversen showed no significant reduction in retention efficiency with time over 1992-1999. Uusi-Kamppa (2002) also showed increased efficiency with time in Finland (1992-2000).

The field margin may be different for grass fields compared with arable. Water may run along ploughed margins in the last furrow in arable fields, but connection in grass fields is direct therefore there is less likelihood of flow concentration in grass. On arable fields, a “jagged edge” to the ploughed field margin may be beneficial in providing extra sediment storage and less flow concentration.

Grassed buffers are best harvested to remove nutrients and to ensure vegetation does not get too long, when particularly for concentrated flows, the vegetation will collapse and submerged flow (less effective sediment removal) will occur. Short vegetation is often denser than high vegetation, and tall vegetation will tend to lie flat during high runoff periods. Stiffer vegetation is more likely to provide protection than vegetation which falls over during flows over the buffer. For example, false oat grass () or tussock grass (Deschampsia cespitosa) are better than cocksfoot (Dactylis glomerata), which is better than ryegrass (Lolium spp.) or meadow grass (Poa spp.).

e. Contributing area.

Variable source area control is common in temperate catchments, where the close proximity of the water table to the land surface leads to overland saturated flow. Nearly all biologically active P export in the Brown Catchment, Pennsylvania, was attributed to less than 10% of the area (Pionke et al., 1997). The source area relative to buffer area is an important design feature and Haynes and Dillaha (1992) recommend a ratio of <50:1.

f. Subsurface soil processes – plant uptake and denitrification.

The interception of diffuse nutrient pollution by riparian zones is a recognition of the natural function of nutrient uptake and removal by riparian forest and vegetation along waterway corridors and river terraces (Naiman & Decamps, 1989). Soluble nitrate and phosphorus enters these zones by both surface run-off and sub-surface flow from higher ground, and they also moves parallel to the watercourse within the riparian zone (Naiman & Decamps, 1989). Removal before it has a chance to enter the waterway itself occurs by plant uptake into the bankside and terrace vegetation, and also by microbial denitrificaton in the case of nitrates. The review by Cuttle et al. (2004) cites many cases where nitrate concentrations of 5-20 mg L^{-1} in the water entering a buffer strip are reduced to < 2 mg L^{-1} in that leaving the zone, and the optimum width of the zone seems to be governed mainly by the soil’s water holding characteristics. Although Jacobs and Gilliam (1985) state that buffer strips are most effective when the form of nitrate was primarily as run-off, Heftig & Klein (1998) also found loadings could be reduced in sub-surface groundwater flow by 95 % (40 mg L^{-1} reducing to <2 mg L^{-1}).

Riparian zones need not be particularly wide for subsurface nitrate removal to occur. Haycock & Burt (1991) found that most of the nitrate removal occurred within the first 5 – 8 m of a buffer strip. Groundwater nitrate levels fell by over 60 % within the first 3.3 m of entering an afforested buffer strip in the USA (Schoonover & Williard, 2003), which was attributed to both uptake and denitrification during the spring, but probably only de-nitrification during the summer months. Nutrient retention of 67 to 94 % for N and 81 – 97 % for P, was achieved by wider buffer strips of 31 and 51 m respectively in Estonia where the vegetation was a natural complex of wet meadow and grey alder species (Mander *et al*., 1999). This nutrient retention was always the case when incident concentrations were above 5 mg L^{-1} N (0.15 mg L^{-1} P), but could be zero or negative when low levels of 0.3 mg L^{-1} were in the incoming water (Kuusemets *et al.,* 2001).

Removal of nitrogen from the riparian zone by the microbial process of denitrification can be a very effective means, but depends upon the maintenance of anaerobicity in the soil. For this reason it was found to be almost 100 % effective over time for nitrate entering the “near-stream zone” of a riparian buffer strip in the USA, but hardly at all for an “upslope area” (Ettema *et al*., 1999). The upslope area did not maintain anaerobic conditions, nor contain high labile carbon levels in the soil, which have also been found to govern the location of de-nitrificaton hot-spots (Addy *et al*., 1999). It is thought that afforestation of buffer strips is one means of providing high concentrations of labile carbon in the soil, but care must be taken to maintain an understorey vegetation, that does not lead to an eroding surface (Cuttle *et al*., 2004). Promoting denitrification however, is not necessarily beneficial. If conditions are only partially anaerobic, then increased nitrous oxide (N_2O) emissions can result, and indeed a study of N_2O emissions from a riparian zone in the Netherlands, found significantly higher emission rates (20 kg ha-1 a^{-1} N) from afforested areas, than from grassland (2 – 4kg ha^{-1} a^{-1} N) (Heftig *et al*., 2003). The main process for N removal is subsurface denitrification, not nutrient uptake (Vought et al., 1994).

Overall, the available information suggests buffer strips of 5-20m will be effective, for dealing with a significant proportion of sediment based inorganic pollution, but effectivness is limited for colloidal size particles, infrequent large storm events and steep slopes. In addition, their effectivness for some pollutants is indirect: prevention of access to water reduces trampling of banksides by livestock and direct input of faecal coliforms to water; uncropped buffers prevent direct input and spray drift of pesticides; biodiversity value may be high, but this depends on active vegetation management. For soluble pollutants, buffer strips need to be modified to slow down the movement of subsurface water.

Restricted access of livestock to waterways

Kunkle (1970) found that grazing near a watercourse significantly impacted stream bacterial densities, while livestock grazing more distant land had little impact. Several factors determine the potential of livestock grazing to serve as a nonpoint source of pollution and Sweeten and Reddell (1978), list the following 1) stocking density, 2) length of grazing period 3) average manure loading rate 4) manure spreading uniformity by grazing livestock 5) disappearance of manure with time. Under low summer flows it is likely that this material is retained being only re-suspended and mobilised during subsequent rainfall events and a rising storm limb (Crowther et al., 2002, Stephenson and Rychert, 1982). The significance of direct deposition of faecal matter by cattle in mixed beef and dairy farming areas of New Zealand has been recently demonstrated by Nagels et al., (2002). Using an artificially generated flood event, they eliminated wash-in of FIO from terrestrial catchment sources but still observed increased FC counts of greater than two orders of magnitude.

In livestock areas, a full buffer strip cannot be maintained, as drinking points are often required. However, these can become point sources of sediment transport, from both the bank-side and stream bottom themselves being eroded by the action of livestock, but also enhanced surface run-off from poached and trampled areas in the immediate hinterland of the drinking point. By restricting livestock access to stream sides at points where soil will not be prone to damage and at times when poaching will not occur, this source can be considerably reduced, though other drinking points in fields may have to be installed.

Livestock and livestock waste management

Inamdar *et al*. (2002) evaluated the effectiveness of BMPs in the U.S. for reducing bacterial pollution at the watershed scale and over the long term. Major BMPs implemented were manure storage facilities, stream fencing, water troughs, and nutrient management. Inamdar *et al*. (2002) concluded that although BMP implementation can be expected to accomplish some improvement in water quality, BMP implementation alone may not ensure compliance with current water quality standards.

Aitken (2002) summarised the main operational techniques which farmers can use to mitigate against the risk of FIO contamination of watercourses as follows:

- Prevent contamination of clean water from roof and yard systems, with dirty water, to allow recycling or direct discharge of clean water to watercourses.
- Reduce volumes of dirty water by increased use of roofed areas for waste storage, feeding and standing stock.
- Minimise volumes of contaminated water by. using low volume wash equipment.
- Regularly inspect, repair and maintain storage structures and associated equipment and ensure contingency plans are available to all staff in the event of system failure.
- Prepare and use a Farm Waste Management Plan and always apply manure or contaminated water in accordance with the Code of Good Agricultural Practice.
- Site water troughs and feeding areas well away from watercourses and avoid regular stock movement on tracks where run-off to watercourses is likely and, where possible, prevent stock from standing in watercourses while drinking.
- Ensure good waste management operations by training staff to appropriate competence standards in the use of equipment and in the landspreading of wastes.

Costs for these operational techniques are given in Aitken et al (2004).

Potential for managed headlands to improve the biodiversity value of grasslands

As a result of agricultural intensification, the majority of in-bye agricultural grasslands in the UK now lack botanical and structural complexity. Plants within intensively managed swards are allowed little capacity to set seed while the majority of invertebrates within these grasslands are either too small to be utilised by birds or inaccessible because of dense vegetation or impenetrable soils. However, the plant and invertebrate biodiversity value of such swards can generally be enhanced by ensuring that the sward contains a mixture of grasses and broad-leaved plants together with a range of vegetation heights and structures (from short open swards with patches of bare soil to tussocks of tall vegetation). Such a varied vegetation structure is also essential to allow farmland birds access to the potential food items present within the sward. However, although such a diversity of sward composition and structure can be achieved by reducing the intensity of grazing or mowing management that the field is subjected to, trying to extensify or alter the timing of the management practised over the whole of a field is not always practical or economic for the farmers concerned. To this end, the possibility of improving biodiversity value by only altering the sward conditions at the edge of a field has been investigated in southern Scotland through the establishment of grassland conservation headlands (e.g. Haysom *et al*. 2004).

These consisted of 6-10 metre wide strips established along the edge of intensively managed grassland fields in which the grassland vegetation was not cut or grazed during spring and summer. The use of herbicide in a limited number of narrow strips through the headland was also considered necessary at establishment in order to create strips of bare ground and thereby allow access by birds (and their chicks) into the headland and to encourage natural colonisation and regeneration of other plant species. This was particularly necessary in situations of high soil nutrient status where otherwise the associated perennial rye-grass would simply grow tall and then collapse - thereby preventing effective access of birds (and their chicks) to the invertebrates within the headlands. By the same token, grazing or cutting of the headlands during the autumn and winter was necessary to remove the vegetation and thereby keep the sward within the headlands open; removal of any cuttings was also essential in order to assist the lowering of soil nutrient levels. The choice of boundary on which to site each headland was also made with care in order to reduce the chances of noxious weeds encroaching into the headland from neighbouring fields and habitats. Where livestock were present in the remainder of the field between April and August, exclusion of the livestock from the headland required the use of permanent fencing (since temporary fencing proved ineffective at preventing access by determined livestock into the headland).

Both small-plot and field-scale studies have shown that these headlands enhance invertebrate prey abundance, but despite this few birds were observed foraging in these headlands. This appears to be related to the fact that it is very unlikely that any such headland established on its own within a sea of intensively managed grasslands will actually provide sufficient resource to sustain any increase (from a very low base) of the birds using those grasslands for any significant period of time. Although these headlands can clearly benefit vegetation and invertebrates, the high costs of fencing and lack of 'return' in terms of farmland birds means that such a headland approach is difficult to justify purely from a biodiversity perspective.

Conversely, concern for water quality and human health mean that riparian buffer strips will increasingly be used to reduce the amount of livestock faecal material (i.e. diffuse pollution) entering watercourses from intensive grasslands. Current management of these areas is driven solely by the need to exclude grazing livestock from watercourses and prevent contamination from livestock-related practices (such as slurry-spreading). However, the establishment of riparian buffer strips provides an opportunity to try to offset the biodiversity losses due to intensive grassland production by ensuring that species diversity is maximised.

To explore this, a three-year SEERAD project has been started with the aim of increasing understanding of the key factors affecting biodiversity within the riparian buffer strips and the potential role such strips play in the movement of surface-active invertebrates between other farmland habitats (e.g. species-rich grassland, hedgerows, woodlands) and intensive grassland. Biodiversity value will be assessed in terms of vegetation composition and structure, invertebrate abundance and species composition and the occurrence of water voles along watercourses where riparian buffer strips have and have not been established. The research will focus on the Cessnock Water (Ayrshire) where work is ongoing to install and monitor mitigation measures for diffuse pollution. The results will help inform the development of recommendations of ways of enhancing the biodiversity value of riparian buffer strips while ensuring that their key function of reducing diffuse pollution is maintained. In this way it is hope that it will be possible to provide understanding to help accrue multiple environmental benefits from the establishment of buffer strips in the future.

WATERCOURSE METHODS

Within the list of management tools identified in Table 2 of the aforementioned report to Defra on methods and measures of diffuse pollution control (Dampney *et al.,* 2002), were several aimed at the upper reaches of waterways outside of field margins, and the more effective of these have been selected and reproduced in Table 5 below. These can be considered the last stage of defence against pollution in the major watercourse downstream, and are relevant to both field and steading sources of pollution. They are not dealt with separately by Hilton (2005), but are subsumed in steading measures. However we think they have a role also in control of diffuse field sources of pollution, so include them here as a separate category.

Table 5: Possible control methods to control the transport of pollutants in waterways after they have left fields and their margins. Extracted from Dampney *et al* (2002)(except those in italics). ++ = strong positive effect; + = positive effect; - = negative effect. Farming systems are arable (A) or livestock (L)(grazing) with either mixed farms or manure used on arable farms (L/A)

Control Method	Farming System	Pollutant category			
		Soluble	Sediment	Pesticides	FIOs
Reduced watercourse maintenance	A & L	+	++	+	+
Constructed wetlands	A & L	++	+	+	+
Settling ponds	A & L	+	++	+	+

Reduced watercourse maintenance

Watercourse maintenance itself can be the source of sediment problems down stream, so this should be kept to a minimum and carried out when flow is low. The damage to stream and bank-side macrophyta should be kept to a minimum, though their complete removal off-site may be necessary from eutrophicating situations. Inadequate maintenance however, may reduce the effectiveness of field drainage and other pollution control methods, and even increase local flooding potential.

Constructed wetlands

Wetlands are ecozones that buffer the interactions of terrestrial and aquatic systems and as such have a pivotal role in the landscape (Reddy and Gale, 1994). Wetland BMPs designs that range from simple settlement ponds to sequenced filter bed systems with a considerable engineering design requirement. Other examples include swales although Mazer et al., (2001) suggested that flow velocity and hydraulic loading during storm events would appear too large to permit sedimentation of silt and clay sized material. There is also a distinction between surface flow constructed wetlands and those that involve subsurface flow, with the latter being reported as being more efficient (Hill and Sobsey, 2001). Such constructed wetlands have been introduced to treat a wide range of wastes. They are especially suited to treating effluents from septic systems and are being successfully used to treat a variety of farm wastewater in Ireland (Harrington et al., 2004).

A recent survey in the USA (Knight et al., 2000) demonstrated the range of farm wastes treated and these included dairy manure and effluent/wash water, runoff from concentrated cattle feeding operations, pig and poultry manure. This review also indicated the possible need for pre-treatment stages in certain cases in order to maintain the long-term health of the wetland system. Dairy farm effluent (characteristically has a high COD and FC content) passing through constructed wetlands receiving a daily hydraulic loading of 0.01 m^3/m^2 reported 99.3% removal rate for FC in the summer and 95.8% in the winter. Adsorption onto soil was suggested to be a key mechanism for retention (Kern et al., 2000). Other mechanisms thought to be involved in the removal of FC include, sterilisation through exposure to UV radiation (Sinton et al., 2002) and retention and subsequent grazing.

Retention occurs through a combination of sorption process involving the solid phase and predation, so the contact time (or residence time) of effluent within the system is a critical factor influencing the degree of treatment and contaminant removal. Decamp and Warren, (2000) demonstrated that very different removal kinetics could be expected between planted gravel beds and unplanted soil beds with a much faster rate of removal in the former situation. Most of the *E. coli* removal occurred in the first third of the bed. These systems appeared to be more effective when there is a combination of soil and gravel beds and where the inflowing water is made to percolate/infiltrate through substrate rather than flow over the surface. Bacterial removal efficiency was shown to be inversely related to the flow rate of effluent through a surface flow reed bed system treating sewage effluent in Yorkshire (Perkins and Hunter, 2000). Williams et al., (1995) reported gravel bed wetland systems operated optimally with a residence time of about 6 hours, although it was easy to overload the system in which case the efficiency is reduced dramatically. It is probably the case that wetland systems operate under optimum conditions when they receive a regular flow of effluent. High efficiencies in wetland with a retention time of 4 days were presented by Thurston et al., (2001).

A range of substrates have been compared for filter systems and have included gravel, soil, river sand or mature sewage sludge, with or without macrophytes. Sand filled filter systems have been reported to be more effective than soil systems for septic effluent (Harrison et al., 2000). Manios et al., (2002) reported high rates of FC removal from primary treated waste water for all treatments and especially gravel filled beds but no differences between planted and unplanted reed beds. Coleman et al., (2001) reported enhanced treatment efficiencies in vegetated wetlands compared to gravel only. Gerba et al., (1999) compared three wetland systems, a duckweed covered pond, a multi-species surface flow and multi-species sub-surface flow wetland. The most efficient was the sub-surface flow system while the pond was efficient at removing larger organisms (Gardia and Cryptosporidium) probably through sedimentation. The performance efficiency of a constructed wetland and water pollution control pond were compared for stormwater events by Davies et al., (2001). The less effective removal of FC and nutrients by the control pond was attributed to its inability to retain fine, clay sized particles to which the pollutants were predominantly associated. It was suggested that a setting pond and SSF wetland would be a successful combination. Especially as Tanner et al., (1998) report the possibility of clogging in their gravel substratum.

Constructed wetlands can also be located in the upper reaches of water-courses themselves, where the level of pollution risk does not warrant their installation at field margins. This may be prove particularly effective at or after the confluence of several small streams that may collect low levels of pollution together until it poses a threat at medium scale stream size. Wetlands can be adapted to include natural features at stream headwaters (Hill, 1990) or artificial constructions at drain outflows (Petersen *et al*., 1992) and operate by creating shallow (< 0.5 m according to Uusi-Kämppä *et al*., 2000) vegetated areas with slow moving water, which impeded sediment flow and provide a sink for soluble nutrients. They can be effective in nitrate pollution (Kovacic *et al*., 2000), removing 28 % of incoming N (when incoming levels were of the order 5 – 15 mg L^{-1} N), and also P (Withers *et al*., 1997) though as phosphorus can collect in bottom sediments they can act as a store of P that can be flushed out during storm events causing greater short-term pollution.

They are best used in conjunction with other methods , such as buffer strips (Kovacic et al., 2000) and/or settling ponds (Uusi-Kämppä *et al.,* 2000), and Fig.3 shows how a constructed wetland with these various features may look (from Uusi-Kämppä *et al.,* 2000). When compared with settling ponds on their own constructed wetlands retained 41 % of the incident total P compared to 17 % in the ponds, though these values were often much lower than buffer strip mitigation (27 – 97 % reduction depending on width)(Uusi-Kämppä *et al.*, 2000). Another drawback in comparison with buffer strips, is that accumulated nutrients in the vegetative biomass can be harvested and removed from buffer strips but not effectively so from wetlands. Kuusemets & Mander (2002) considered constructed wetlands less effective than buffer strips, and calculated that an interception of 2.2 – 2.6 t a^{-1} N and 12 – 15 kg a^{-1} P would be made from a 460 m buffer strip along a stream in acatchment, but only 1.66 – 2.76 t a-1 N and 3 – 4.5 kg a-1 P from a wetland on the stream.

Constructed wetlands are often the last barrier for sediment run-off mitigation, but fortunately Braskerud *et al*. (2000) found that their performance can increase with hydraulic load and the sediment found within them was clearly indicative of the sediment from the watershed. The area of the wetland need only 0.03 to 0.07 % of the watershed they collect from (Braskerud *et al*., 2000) and so are more area efficient than buffer strips. In studies over 3 to 7 years Braskerud (2001) found that wetlands effectively retained 45-75 % of soil particles, 21-44 % of phosphorus and 3 –15 % of nitrogen in the incoming water-flow. Higher reductions of 78 % nitrate N and 80 % P were found in a constructed wetland down-slope from dairy pastures in New Zealand by Tanner *et al*. (2003), though only over the short-term. However, they are only effective for pesticide transport mitigation when incorporated into buffer strips over considerable distances (100 – 400 m)(Moore *et al.,* 2002).

Settling ponds

One effective mechanism for progressively reducing sedimentation risk, often as part of a constructed wetland system (Figure 3), is a series of small ponds along the flow path, such that particles in the in-flowing water can settle out in the still water. The depth of these ponds should be > 1 m compared to the shallower vegetated areas (Uusi-Kämppä et al., 2000). The outflow from each pond is thereby progressively clearer, and vegetation in the ponds assists the removal of nutrients and creation of still areas by providing a physical barrier. Tanner & Sukias (2003) in New Zealand, also looked at linking ponds and wetlands to treat the effluent from sewage dairy and piggery waste waters, finding that bacterial indicators were regularly reduced by one log-unit, but consistent achievement of cfu counts below 500 $(100ml)^{-1}$ was difficult.

DEVELOPMENT OF A MECHANISTIC APPROACH TO ASSESSING BUFFER STRIP EFFICACY

There are a number of mechanisms which contribute to pollutant removal by buffer strips. A proper assessment of the likelihood of efficacy of buffer strips requires a physical and quantitative appreciation of these mechanisms. These include:

Sedimentation

Small particles have a higher surface area, therefore they retain more nutrients by adsorption. P occurs as adsorbed and solution P. Both influence algal growth (as adsorbed P will desorb, in the higher solution: solid ratio of freshwater, compared with soil). However smaller particles also sediment less readily. Aggregation affects sedimentation, as although aggregates have a lower density than primary particles, they are much larger. Factors that reduce velocity of flow increase sedimentation. Reduction in slope or increase in surface roughness may decrease velocity. Flow in the buffer can be submerged (below height of vegetation) or non-submerged. Higher trapping efficiencies occur if the vegetation is not submerged. Time to inundation depends on incoming sediment load. Short vegetation is often denser than high vegetation, and tall vegetation will tend to lie flat during high runoff periods.

Filtration

Sedimentation as a removal mechanism for smaller, colloidal particles is likely to be poor, although in many cases clay particles will be transported in aggregates that act as silt or fine sand sized particles (Braskerud, 2001). Finer particles can be removed by buffer strips, by a process of filtration, due to interception and diffusion of colloids to filtering surfaces.

Ives (1975) gives equations which predict the efficiency of these processes. For cylindrical collector geometry, the capture efficiencies for diffusion and interception are:

$$\eta_D = 3.64A_f^{\,1/3}\,(D/2a_fU)^{2/3} \qquad (1)$$
$$\eta_I = 2A_f(a_p/a_f)^2 \qquad (2)$$

η_D, η_I = filter capture efficiency for individual filter element by diffusion, interception, respectively A_f = packing function, D = diffusion coefficient, a_f = radius of cylinder. U=fluid viscosity, a_p = particle radius.

From these expressions, the efficiency of filtration for particles of different sizes can be estimated. Fig 4 gives an example of the effect of particle size for a slow flow rate case (5×10^{-4} m^3/m buffer width/sec). It can be seen that at these low flow rates, significant diffusion of colloids towards buffer vegetation surfaces improves capture of sub-micron particles. Fig 4 also shows that it is the particles in the 1-10μm size range (also not removed very well by sedimentation, see Table 6) that are least efficiently removed by the diffusion and interception mechanisms.

Infiltration

This reduces surface runoff and removes fines which are not removed by sedimentation. Permanent vegetation leads to better soil structure and therefore better infiltration than in tilled soil. Moreover, the permanent vegetation means that soils under buffer strips are often drier than in adjacent arable fields, promoting extra infiltration. Deep root systems in buffer strip plants favour infiltration. Delgado et al., (1995) suggested that design of buffer strips for receiving input from treatment plants should have the objective of complete infiltration of water, but for buffer strips designed to control diffuse pollution from rainfall runoff, this is rarely feasible. Forthe large events and concentrated runoff where buffer strip function is seriously tested, infiltration effects are likely to be relatively small.

Theory of sedimentation in buffer strips

From a sensitivity analysis Munoz-Carpena et al. (1999) suggested that the most important parameters for the hydrology component of a transport model in VFS are initial soil water content and vertical saturated hydraulic conductivity, and sediment characteristics (size, deposition velocity and sediment density). Abu-Zreig (2001) using a simulation model showed that the size range of the inflowing sediment material was extremely important for determining the sediment trapping in buffer strips. The trapping efficiency of clay sediment in a 15 m length buffer strip was predicted to be 47% compared with 92% for silt in incoming sediment. This tends to support the commonly reported observation that retention efficiency is non-linear with increasing buffer or filter strip width.

Barfield et al. (1979) developed a set of equations to predict the deposition of sediment during concentrated discharges into a buffer strip with rigid vegetation, as a function of vegetation spacing and height, inflow rate, slope, sediment concentration, particle size and buffer width. This enables outflow sediment concentration to be calculated for varying intensity of flow. Hayes et al. (1979) also illustrate the use of a non-steady state version of this model to predict deposition in a buffer strip of rigid vegetation (once vegetation bends over, incoming sediment deposition will be greatly reduced, because the flow velocity will not be impaired, so this case is not considered).

The deposition of sediment in filter strips during high intensity, long return period events, with concentrated flows will depend primarily on enhanced settling promoted by retardation of flow within the buffer. The difference in slope between field and buffer, and the retarding effect of the vegetation are responsible for this reduction in flow velocity. During such events the buffer strip will be divided into four zones (Figure 5). In zone A, sediment transport is along the top of the inundated vegetation, and essentially all the incoming sediment is transported. Zone B is an advancing, wedge shaped deposition front, which will develop an equilibrium slope characteristic of flow, vegetation and sediment conditions. In Zone C, it can be assumed that sufficient sediment has been deposited on the flow path of the water that surface irregularities are filled up, allowing all sediment entering this zone to be transported as bedload. In zone D, sediment transported as bedload is trapped in surface irregularities. In this zone, infiltration of water will also occur, leading to a change in flow velocity and consequent enhanced sedimentation.

For zone D, empirical studies under controlled conditions by Tollner et al. (1976) showed the trapping efficiency could be given by:

$$T = \exp\{-0.00105\ (VR_s/\nu)^{0.82}\ (L(t)\ V_s/Vd_f)^{-0.91}\} \qquad (3)$$

Where: V= mean flow velocity, R_s=hydraulic radius (See below), ν = kinematic viscosity, L(t) is effective buffer width, V_s = sedimentation velocity of particles, d_f= depth of flow. The value of R_s is given by:

$$R_s = S_s d_f/(2d_f+S_s), \qquad (4)$$

where S_s= spacing between vertical elements of buffer vegetation. The value of V is given by :

$$V=(1/n)\, R_s^{\,2/3}\, S_c^{\,½} \quad (5)$$

Where S_c is channel slope and n is Manning's roughness coefficient (taken as 0.0072).

The continuity equation in the absence of infiltration is given by:

$$Q_w = Vd_f \quad (6)$$

Where Q_w = volumetric flow rate per unit width of buffer.

In zone C, the sediment load transported for a given depth of flow (calculated for zone D) and channel slope is determined from the expression:

$$Q_{sd} = \phi\, \gamma_s \sqrt{[\, ((\gamma_s-\gamma)/\gamma)\ g\ d_p\,]} \quad (7)$$

Where :
γ_s=sediment density
γ=water density
g = gravitational acceleration
d_p=particle diameter

ϕ is given by an empirical relationship found from sediment transport studies in controlled flow conditions:

$$\varphi = 1.08\ \phi^{-0.28} \quad (8)$$

Where $\varphi = ((\gamma_s-\gamma)/\gamma)\ d_p / s_c r_s$
S_c= channel slope

Table 6 gives calculations of the equilibrium sediment concentration and flow depth in zone C, and the trapping efficiency and sediment concentration in the outlet from zone D, under conditions of steady state flow, for a range of discharge, filter length, particle size and slopes. Note that the value of 5x 10^{-2} m^3/m/sec corresponds to the estimate mean annual flood for West Hillfoot field, Greens Burn (see below), concentrated into a 1m width of buffer (Hayward, personal communication). The model makes it clear that zone D removal efficiency in this case is low. Moreover, the equilibrium sediment concentration in zone C is quite high, leading to sediment concentrations in the discharge from the buffer in the order of 3000 g/L. If flow from this field runoff is spread over a greater width of buffer (eg 100m, ie the margin of a square 10 ha field) efficiency of sediment removal is very much higher.

Table 6. Sensitivity analysis for filter strip efficiency in Zone D as a function of flow rate, filter length and particle diameter.

Volumetric flow rate per unit width	**Effective buffer width**	**Particle diameter**	**channel slope**	**Flow depth**	**Equilibrium concentration Zone C**	**Efficiency zone D**	**Outlet concentration**
q_w	**L**	d_p	S_c	d_f	C_{sd}	**Efficiency zone D**	C_o
m^3/m/sec	**m**	**M**	**%**	**mm**	**mg/L**	**%**	**mg/L**
5.E-02	20	5.E-05	0.05	58.3	4901	39%	3012
5.00E-03	20	5.E-05	0.05	6.6	8772	95%	434
5.E-04	20	5.E-05	0.05	3.0	20271	99%	224
5.E-03	**5**	5.E-05	0.05	6.6	8772	83%	1481
5.00E-03	**20**	5.E-05	0.05	6.6	8772	95%	434
5.E-03	**50**	5.E-05	0.05	6.6	8772	98%	197
5.E-03	20	**2.E-05**	0.05	6.6	72263	71%	21025
5.00E-03	20	**5.E-05**	0.05	6.6	8772	95%	434
5.E-03	20	**2.E-04**	0.05	6.6	613	99%	3
5.E-03	20	5.E-05	**0.01**	13.0	66	96%	2
5.00E-03	20	5.E-05	**0.05**	6.6	8772	95%	434
5.E-03	20	5.E-05	**0.1**	6.0	89746	92%	6733

The table gives an indication of conditions where sedimentation in buffer strips are likely to be less effective ie, for particles <50um, slopes >5%, discharges > 5 x 10^{-3} m^3/m buffer/sec, the potential sediment concentration discharging the buffer will exceed 400 mg/L, under conditions where significant soil erosive losses occurring. Moreover over time the sediment will accumulate in the buffer, so the effectiveness of the buffer vegetation will decrease (L will decrease with time), as it fills with sediment in Zones A and B. In addition the sedimentation described above in zone D only occurs so long as there is capacity for deposition of bedload in zone D. This capacity is not specified explicitly in the model, but it is expected to be small compared with sedimentation occurring in zone B, for relatively long events. Barfield et al (1979) provide a method to estimate the slope of the deposition wedge and its rate of advance (ie zone B behaviour). The extent of advance of the deposition wedge will depend on the length of time the water is flowing, and for this we need a method to estimate return periods and sediment content of runoff water.

Estimation of input discharges and pollutant loads for design purposes

In order to obtain a full assessment of the impact of buffer strips on pollutant loads, based on transport mechanisms within the buffer strip, it is necessary to have information on the flow of water and pollutants into buffer strips, as a function of weather conditions, soil types, topography, land use etc and the design criteria to be followed Prediction of hydrological processes depends on whether runoff is controlled by the development of saturation excess in variable source areas (the approach used in TOPMODEL (Beven and Kirkby, 1979)) or by infiltration excess, where rainfall is of sufficient intensity to cause surface runoff without saturated conditions occurring in the soil profile. The latter approach is used for the prediction of infrequent flood events (eg Flood Studies Report, 1975). We believe that the latter approach will be more appropriate for predicting surface runoff events that cause the majority of soil erosion, on a regional basis. The mean annual maximum instantaneous flood from a field or small catchment can be estimated using the Poots and Cochrane formula (Wilson, 1990) as follows:

$$Q_{m1} = 0.0136\mathrm{AREA}^{0.866}\ \mathrm{RSMD}^{1.413}\ \mathrm{SOIL}^{1.521} \qquad (9)$$

Area = catchment area in km^2; RSMD= net 1-day rainfall of 5 years return period less soil moisture deficit; SOIL=soil index.

The instantaneous flood for a different return period (T) is given by:

$$Q(T)/Q_{m1} = u + [\alpha(1-e^{-ky})/k] \qquad (10)$$

Where for Scotland, u=0.82 to 0.84, α=0.18 to 0.22, k=-0.2 to –0.3 (Flood Studies report, 1975), y = Gumbel number given by:

$$y = -\ln[-\ln(1-(1/T)] \qquad (11)$$

And the instantaneous flood for a different duration (d, in days) is:

$$Q(d)/\ Q(T) = 1/(1+Bd)^N \qquad (12)$$

Where N=0.5, B = 2.16 (mean of 64 stations, Flood studies report, 1975).

If we assume that a watercourse with potential for buffering is lined by square fields of area A, which act as individual hydrological units, as far as runoff is concerned, and that a width of field W contributes runoff, then the contributing area of field is given by:

$$\mathrm{AREA} = W\sqrt{A} \qquad (13)$$

This area can be used to calculate the mean annual flood discharge, and the design discharge (Q(d)) from each field.

The specific discharge across the buffer strip, per unit width will be given by Q(d)/W.

Table 7 gives values of the specific discharge, as a function of return period, field area and contributing width.

Table 7. Specific discharge, as a function of return period, field area and contributing width.

Field area ha	contributory width m	RSMD mm	duration hours	return years	Discharge across buffer m^3/m/sec
1	50	40	24	10	**2.96E-05**
10	50	40	24	10	**2.85E-05**
100	50	40	24	10	**2.74E-05**
10	**10**	40	24	10	**6.01E-06**
10	**50**	40	24	10	**2.85E-05**
10	**100**	40	24	10	**5.56E-05**
10	50	**20**	24	10	**1.07E-05**
10	50	**40**	24	10	**2.85E-05**
10	50	**80**	24	10	**7.58E-05**
10	50	40	**1**	10	**4.85E-05**
10	50	40	**24**	10	**2.85E-05**
10	50	40	**48**	10	**2.19E-05**
10	50	40	24	**1**	**1.24E-05**
10	50	40	24	**10**	**2.85E-05**
10	50	40	24	**30**	**3.80E-05**

It is clear that these discharges should be dealt with comfortably by zone D of a buffer strip, if sediment loads are small, so the advance of zone A and B is slow. However, if flows are concentrated into specific channels across the field boundary (say 1% or 10% of field boundary), then these discharges become large enough to cause problems for zone D of even quite wide filters.

Regional analysis of buffer strip efficacy

We have developed a spreadsheet version of the model of Barfield et al. (1979), linked to the above analysis of flood frequency, enabling us to produce predictions of likely overall removal efficiency of eroded particles, for storm events of a given frequency and duration. We have calculated the effect of buffer strip slope on the equilibrium sediment concentration in zone C. We have done this for the sediment transport occurring from a field during a design erosion event (see below for calculation method) for Eastern Scotland: ie 1 year return period, 24 hours flow, 50m wide contributory area from a 10 ha field, flow concentrated into 10% of the riparian margin, 1 tonne/ha soil erosion loss, particle diameter 20μm. With a P content of 0.2%, this would correspond to about 2 kg P/ha loss from the contributory area. This translates into a flow of 1.25 x 10^{-4} m^3/m buffer/second, with a sediment concentration of 30 g/l. Figure 6a shows some results of these calculations. It can be seen that the equilibrium sediment concentration that flows in zone C approaches the input concentration to the buffer rapidly when the slope exceeds about 7%.

We also calculated the effect of sediment concentration input to the buffer on the rate of advance of the sediment wedge (zone B). This is shown in fig 6b for a 5% slope. These two figures provide a basis for potential design criteria for an effective buffer:

1. Zone B should not extend beyond 50% of the buffer width, as once Zone B approaches the end of the buffer, the sediment concentration leaving the buffer would begin rapidly to approach the input sediment concentration and no sediment removal would occur.

2. Equilibrium sediment concentration in zone C is 50% or less of the concentration of sediment in the design event. This means that 50% particulate pollutant removal would be occurring. Removal of particulates during small events would be much more effective than this, but a target of 50% removal for large events, would give a tangible effect of buffer strips in control of P pollution to surface waters.

For the design conditions given, this means that the buffer strip slope should not exceed about 7%.

The required width of buffer would depend on the eroded soil present in the design event, and this is also shown in Fig 6b. Thus for a 1 tonne/ha erosion event, contained in a 1 year/24 hour return period storm, it would require a buffer of 7m width, to achieve 50% removal. A 20 m buffer would provide 50% removal of a 3t/ha erosion event.

The outcome of this theoretical approach is that we should not expect buffers with a slope of >7% to be effective if flows are concentrated, and that it is reasonable to expect 50% removal of sediments from relatively infrequent events on moderate slopes, with moderate (10m) buffer widths, so long as flow concentration does not exceed a factor of 10. This brings into question the value of steep sided banks as buffer areas for control of high sediment discharges. It also highlights the need for targeted measures, rather than installing very wide riparian zones for erosion control, where flows are likely to be concentrated, as occurs in the undulating landscape associated with much of Scottish arable farming.

These models could be further developed to obtain spatially distributed, rational guidelines about the likely efficacy of rigid vegetation buffer strips as a function of height, width, flow velocity, slope and sediment runoff concentrations.

Conclusions of theoretical assessment of buffer strip function

1. Buffer strips potentially (site specific) show effective retention of suspended solids and associated sorbed P or pesticides, but are much less efficient removal of soluble nutrients, even where sheet flow transport off adjacent fields is occurring.
2. The value of increasing from 5m to 20m buffer strip for sediment removal is often relatively small, from a pollution control point of view.
   Slopes > 7% are not recommended for buffers, as sediment deposition, especially for the finer, pollutant rich sediments, will be poor. Hence when identifying riparian zone widths required for buffer strip installation, we provisionally recommend that steeply sloping areas be discounted, and measurement of widths begin from the point in the landscape where slopes are <7%.

3. Relatively short, but stiff vegetation is likely to be more effective in removing sediment than tall vegetation, if prone to collapse.

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3. FIELD STUDIES

The nature of diffuse pollution is episodic with the majority of pollutants being transported from the land surface to watercourses during storm events. Therefore in order to assess the effectiveness of BMPs it is necessary to monitor during high flows. Moreover, because diffuse pollutants can reach water from a range of sources, good local knowledge of the likely routes of pollutant transport is vital. For these reasons, pilot sampling sites on small streams were established, in collaboration with farmers owning the riparian land. Both arable and dairying areas of Scotland were included, to monitor the effectiveness of both buffer strips and stock exclusion for pollution control and biodiversity enhancement. The sites were monitored at a range of scales from field to catchment, and for a range of pollutants. The sites were in an arable area (Greens Burn, near Kinross, where a buffer strip was already installed) and in an intensive grassland area (Cessnock Water, Ayrshire where BMPs were in existence or about to be implemented).

The effectiveness of buffer strips also requires to be tested under controlled experimental conditions. This can present logistical problems at a catchment scale and experiments have been carried out at a field scale on small plots where surface runoff can more readily be followed and flow conditions controlled. In association with theoretical work in Chapter 2, this aids extrapolation of results to other situations.

GREENS BURN

Study area

The Greens Burn, to the north of Loch Leven, drains only 10% of the Loch Leven catchment but contributes an estimated 30% of the river-borne total phosphorus load to Loch Leven. This is due to a number of factors including climate, soil erodibility, topography, crop management, and soil erosion control practice. Compared to most other tributaries draining to the loch it has a high loss coefficient of Total Phosphorus at 0.46 kg ha^{-1} yr^{-1}. For the 14.4km^2 catchment this is a load to the Loch of 0.667 tonnes yr^{-1} (MacEachern, 2001).

The Loch Leven Area Management Advisory Group, formed in 1992, produced the Catchment Management Plan for the Loch Leven Catchment, Perth and Kinross (LLCMP). One of the LLCMP objectives was to "produce, and initiate implementation of a practical land use strategy for the catchment", and this included the requirement for five farms (Wester Gospetry, Wester Balgedie, Newlands, Channel and Mawcarse) to establish and manage buffer strips in the Pow/Greens Burn sub-catchment". The 20m wide buffer strips were established in 1997 (MacEachern, 2001).

Previous study on pollutant transport

MacEachern (2001) carried out an investigation into pollutant transfer into the Greens Burn. Samples were taken during storm events using a remotely triggered autosampler at SEPA's permanent hydrometry station at Damley's cottage (SEPA reference code: 14912), capturing the first flush of pollutants by anticipating storm events and sampling half hourly over a 12 hour period. These samples were analysed for total phosphorus, SRP, and suspended solids. The samples were taken over a 2 year period before installation or whilst the buffer strips were stabilising (28th February 1995 to 11 December 1997). They therefore represent a benchmark, against which data collected in the current project (during 2002-4) can be compared. The annual loads in the 1995-97 study show that up to 65% of the annual total phosphorus load and up to 90% of the suspended solids load was transported during storm events (table 8).

Table 8. Estimated loadings of suspended solids and Total P for six events sampled before the Greens Burn buffer strip was effective (MacEachern, 2001).

Date	m^3/event	kg TP/event	kg SS/event
28/02/95	7045	3.74	837
30/05/96	16	0.45	70
03/07/96	1.4	0.00	9
05/11/96	34165	36.93	7907
11/12/97	17730	73.82	21706
19/02/97	23839	40.27	8299

It was found that good relationships existed between suspended solids, total phosphorus and storm flow volume:

Total Phosphorus (kg) = total volume (m^3) * 0.0013 - 0.2266

Total Suspended Solids (kg) = total volume (m^3) * 0.2675 - 57.119

Sites for assessment of buffer strip function in the current study.

An afternoon was spent surveying possible sites with Brian Darcy/Janette MacDonald of SEPA-DPI (Diffuse Pollution Initiative) on 14th February 2003. No suitable control catchment was identified, against which to compare the buffered Greens Burn, but it was agreed to monitor up and downstream of potato fields in both a buffered section of Greens Burn and a weakly buffered section of the upper reaches of River Eden (upstream of NO1603707320). Figure 7 shows the study area concerned, the upper reaches of the Greens Burn and the river Eden. A v-notch weir (at NO1644906019) and diver (pressure transducer to measure water head continuously), plus 2 Epic autosamplers (at NO1644906019 and NO1674205932) were installed on the buffered reach (see diagram). Two samplers were held at Bush for deployment when appropriate on the unbuffered reach. Flow data were to be extrapolated using the relative area approach for the unbuffered reach.

However, during a further meeting with the farmer at Wester Gospetry on September 11th 2003 it became clear that the upper reach of the Eden would also be affected by sediment and water transported from the farm steading and from unbuffered fields not adjacent to the stream, via a steading sump and drain, bypassing field margins. This meant the effects of the weakly buffered potato field originally identified in the Eden catchment would be difficult to identify. We also established that the lower half of this field was in set-aside, reducing its suitability further. During this visit, we also established that a similar bypass drainage system would affect the Greens Burn by discharging into the Burn across the field above Butterwell House, transporting runoff water and sediment from a highly fertilised field in winter barley (field 12, see Figure 7). We therefore changed our sampling positions to three positions on the Greens Burn, one of which would catch the background flow and sediment from the upper part of the catchment (fields in grass or burnt stubble), one below a well buffered field in potatoes then winter wheat (in 2003), and one below this (at Butterwell House, NO1588106237), which would include the runoff from the unbuffered winter barley field, via a surface drain.

Survey of surface runoff and sediment transport evidence.

Sedimentation in buffer

Prior to the start of the project, in October 2002, the upper buffer strip was visited to observe runoff and sediment trapping by the buffer strip. This showed that considerable quantities of water were passing straight over the buffer in concentrated channels through the grass, without leaving much sediment (figure 1). During the visit on 14th February 2003, a lot more sedimentation was evident, both at the field margin (present since October) and in the buffer strip (Figure 8), and the amounts collected in the buffer strip adjacent to a particularly badly eroded field (field 27) were estimated by volumetric coring. A grid of samples was taken for analysis of total P, loss on ignition, Olsen P and ammonium acetate extractable P for comparison with the field from which the sediment came (table 9).

The efficiency of the buffer strip however can be estimated as follows:

A.

Soil loss from field x soil P % = (buffer deposit x buffer P%) + (transported sediment x sediment P%)

B.

Soil loss = buffer deposit + transported sediment

These two equations have two unknowns, if we assume (table 9):

Field soil P % = 0.078%

buffer sediment deposit = 42 tonnes

buffer sediment P%= 0.031%

transported sediment P% (Damley's cottage) = 0.4% (see Table 12)

solving we obtain the soil loss = 48.1 tonnes, and soil P loss = 3.75 kg . The P trapped in the buffer is 1.3 kg, so efficiency is only 35% .

Although these calculations are approximate, they do demonstrate the problem with the buffer strip only trapping the coarser sediment (soil aggregates and mineral particles of sand and coarse silt size range): the finer sediment and colloidal material is not trapped, and this contains most of the adsorbed P.

Table 9. Analysis of samples from sediment in Greens Burn buffer (see Fig.8 and 9).

	Total P		**Loss on ignition**		**Olsen P**		**Ammonium acetate/acetic acid extractable P**	
	Mg/kg	Sd	(%)	sd	mg/L	sd	mg/L	sd
Field	778.0	57.6	5.2	0.4	9.7	0.2	41.0	0.9
Sediment in buffer	307.0	35.6	1.4	0.1	8.6	0.3	21.9	2.3

Tied ridging trial

A field scale runoff trial was installed in spring 2003 in one of the potato fields at Wester Gospetry (field 16). This consisted of hand built tied ridges along one furrow, with sediment traps installed at the end of this furrow and on a control furrow. No sediment transfer was observed during the summer, and the traps were removed prior to potato harvest. On the visit made on September 11th 2003, I proposed to the farmer to omit final rolling of the soil in one 6m pass after sowing of winter barley in one field, so that erosion losses could be assessed compared with the rest of the field at the end of the winter. However this did not take place.

Event sampling and analysis

Damley's cottage sampler

A SEPA stage recorder has been running on Greens Burn at Damley's cottage for many years. Visual observations made during a storm event in April 2003 made it clear that some road runoff may have the potential to contaminate samples, so the sampler intake position was changed, but not until late September 2003. It was agreed with SEPA that the trigger for sampling events would be 10mm rainfall in the previous 6h, based on SEPA's rainfall gauging, with warnings of potential events based on SEPA's link with the Met. Office. No triggering would be done between midday Friday and Saturday midnight. As well as samplers to be triggered for inorganic and organic analysis, a SONDE was installed for continuous NH_4, DO, conductivity and turbidity measurement. Table 10 summarises the actual occurrences of this trigger from Jan 2003 to May 2004, in order of approximate event discharge volume at Damley's cottage.

Table 10. Occurrences of rainfall trigger for SEPA sampler, from Jan 2003 to May 2004. Data are given in order of approximate event discharge volume at Damley's cottage.

Year	Month	day	Hour	rain in prev 6h	approx. event size[1]	Comments
				mm	m^3	
2003	11	29	18	11.6	32097.6	all samplers triggered
2003	12	20	12	10.6	19353.6	not triggered (Saturday)
2003	1	17	10	10.2	13305.6	Not triggered (before project)
2003	12	31	23	14	3283.2	Not triggered (New Year)
2003	9	22	3	22.8	2764.8	SAC samplers triggered 21:45 on 21/9. Only sampler C worked. Delayed rise and little sediment present anyway
2003	4	30	9	12.2	0.0	SAC samplers triggered (no event due to dry antecedent conditions)
2004	5	4	6	10.6	Not available	Discussed with SEPA, but forecast suggested insufficient for SEPA trigger.
2004	8	9-10	17	not available	Not available	SEPA sampler and SAC samplers triggered. SEPA sampler did not capture the whole event [2]

[1] Based on daily discharge at Damley's cottage less baseline, therefore only approximate
[2] Event collected after the end of the project, but included for information

As can be seen the SEPA sampler was successfully triggered only once during the project (29 November 2003), but this was for a good sized event. However, we also obtained event data from 9 September 2002 from Bill Craig of SEPA Edinburgh. This event shows the influence of road runoff? but the information can be analysed, as the road runoff influence is clearly separated from the main peaks in the chemographs. An event was also captured in August 2004, after the end of the project.

SAC samplers

The shortage of trigger events led us to try to catch smaller events at the SAC samplers, even in the absence of the trigger for the SEPA sampler, and Table 11 summarises the occasions when the SAC samplers were triggered, whether successful or not. In addition to the event of 29 November 2003, 3 further events were captured in at least two of the SAC samplers, but only one other event was large enough to merit detailed analysis (8 January, 2004).

Figure 10 gives a comparison of storm events sampled at Damley's cottage before (1995) and after (2002 and 2003) installation of the buffer strip. Total P, suspended solids and discharge are shown. The catchment area at Damley's cottage is 10.5 km^2, so 1000 L/s corresponds to a scaled discharge of 8.2 mm/day.

Table 11. Summary of sampling events in the Greens Burn catchment, during 2002

Date	Sampling sites	**Flow measured during sampling period(m^3)**		**Analysis SAC**	**Analysis SEPA**
		SAC weir	**SEPA**		
9 September 2002	Damley's cottage Only	Not installed	17012 (11h from 16:000	-	TP,DP,DO,T,EC, NH4,TON,SS,pH
29 November 2003	Damley's cottage SAC samplers A,B,C	782 (12h from 11:30)	21847 (7h from 17:00)		TP,DP,DO,T,EC, NH4,TON,SS,pH
8 January 2004 (24h from 10:30)	SAC samplers A,B,C Only	696 (23h from 10:30)	25714 (24h from 10:30)	SS Selected samples analysed by MLURI for TP,TN only	
24 April 2003 30 April 2003 20 Jun 2003 22 Sept 2003 18 November 2003 30 January 2004 5 May 2004		Events not large enough to merit sample recovery; or SS low; or samplers failed.			

T= turbidity , SS=suspended solids, DOC=dissolved organic carbon, NH_4 = ammonium N, NO_3=nitrate N, OP = orthophosphate, TP = total phosphate, EC = electrical conductivity, DO=dissolved oxygen

Figure 11 shows the SS in water samples on the Wester Gospetry sub-catchment collected during storm events on 28-29 November 2003 and 8 January 2004. The samples taken in the small sub-catchment have much higher [SS] than at the SEPA sampler. All samplers show a rise in [SS] at the very start of stage rise, but sampler B, below a well buffered section falls again quite quickly. At sampler C, below a section of the buffer where bypass of runoff water from a winter barley field can occur (see Figure 7), a continued increase in [SS] occurs. Sampler A, at the top of the catchment, shows a large increase in [SS] as well, but only later in the large event on 28-29 November. This is not translated to sampler B or C. This could suggest that overtopping of a storage reservoir for sediment (eg a field corner, where sediment accumulates) has occurred. The sediment contributing to the high concentration in sampler A is clearly quite coarse (as would be expected in such an event, as it settles before the next sampler). All three SAC samplers show much higher [TP] than the SEPA sampler, which will probably be because of dilution and sedimentation of large particulates, but this needs to be confirmed. Table 12 summarises the estimated loads of suspended solids, total P and N for these two events.

Table 12. Summary of SS, TP and N loads during two events sampled during the project.

	Sampler	SS (kg)	Total P (kg)	total N (kg)	Discharge (m^3)	sampling period (hours)	Peak discharge (L/s)
08-Jan-04							
(maximum 6hr rainfall=5.8mm)	**A**	18.1	*0.06*	*1.24*	656.9	22	
	B	20.2	*0.14*	*4.70*	454.9	14	
	C	75.2	*0.13*	*6.74*	695.7	23	
	DAMLEY'S				25714.8	22	490
29-30 Nov 2003							
(maximum 6hr rainfall=11.6mm)	**A**	*1497.8*	2.22	12.22	781.6	12	
	B	156.7	0.49	12.31	781.6	12	
	C	994.4	1.69	16.45	815.3	12	
	DAMLEY'S	3624.9	14.42	294.89	21847.5	7	1116

Figures in italics estimated from mean nutrient:SS ratios for event samples

Figure 12 gives a comparison of estimated Total P losses from storm events measured at Damley's cottage, Greens Burn before and after buffer strip installation. If we neglect the data point for 11 December 1997, when bankside works may have led to increased sediment loads (MacEachern, 2001), the sediment loads per m^3 of discharge are not significantly lower, although there is a trend in this direction. Before the buffer strip the mean event [TP] (3 significant events) was 1.1 ±0.6 mg TP/L and after installation (2 significant events) it was 0.5±0.16 mg TP/L. For SS the values are 233 mg/L ±115 and 94±62 respectively. The results suggest that the buffer strip is having some positive effect on loading of TP and SS to Loch Leven, but the difference is not statistically significant at 95% confidence. A more detailed analysis of the storm event data from Damley's cottage has been carried out elsewhere (Greig, 2004). A preliminary analysis of correlations (pre and post buffer strip) between flow and pollutant concentrations indicated that a reduction in inputs of suspended solids and total P may have occurred, since the creation of the buffer strips. However, the report recognised the limitations of the dataset, and the need to sample further events.

The value of the buffer strip could be improved if the bypass route, between samplers B and C, were dealt with. The high, and delayed sediment loads at sampler A may be because of delayed release from a reservoir at the corner of field 27 (West Hillfoot), a field that is known to erode sufficiently badly for water and sediment to cut across the buffer strip untreated, in the right conditions. We consider, based on these observations and theory, that the existing buffer strip does not have adequate capacity to deal with the sediment transport events that occur. This situation is one in which the "treatment train" approach has been applied. In other words, firstly the farmer should consider avoiding high erosion risk by growing appropriate crops (spring cereals with stubble left overwinter) rather than eroding crops. This would constitute Good Agricultural and Environmental Compliance (GAEC) in this case. In addition, in field measures such as subsoiling of headlands should be encouraged. If high erosion risk crops are occasionally grown (winter cereals, potatoes or a winter forage crop), inclusion of a water and sediment trapping device in the corner of this field, where flow concentrates, would have significant benefits to water quality, and such a device has been installed, under a SEERAD funded BMP evaluation project (See Fig 13).

Discussions with farmer at Wester Gospetry
Contact was made with the distributor of AQUEEL (a roller which promotes high surface storage of water on vegetable beds) with a view to promoting use on winter cereals. We have tried to establish with cereals advisors whether this is suitable for winter cereals, but the farmer at Wester Gospetry showed no interest in using this, because of shortage of labour for operations in the autumn, so we have not pursued it further.

AV visited W. Gospetry farmer in February 2003 and in September 2003 to discuss erosion control measures and autumn cultivations. Box 1 lists the various topics discussed. We proposed and designed a simple porous wall/weir contruction to reduce soil loss from a field (field 27) that badly eroded last autumn (Figure 13). In September 2003, we also identified two surface drainage systems which bypass the buffer strip, one draining runoff from a field (field 18) which had recently receiving 10t/ha “hen pen” (chicken manure – an estimated 200 kg P/ha) and was sown to winter barley. This provided an opportunity to evaluate buffered and unbuffered reaches , by sampling above and below the surface drainage input from this winter barley field. We also noted a field in kale, to be grazed off in the new year (field 14).

Assessment of impact of Greens Burn on invertebrate diversity of Loch Leven

An SAC/Edinburgh University MSc project has been completed by L.Torrance (2003), in collaboration with Laurence Carvalho of Centre for Ecology and Hydrology. The objective was to evaluate the impact of the Greens Burn buffer strip on the invertebrate biodiversity in Loch Leven, in the zone around the mouth of the Greens Burn. In 1994, before the buffer strips were installed, a survey of benthic macro-invertebrates was made across the whole loch (Gunn and Kirika, 1994). Torrance (2003) took ten random core samples of sediment within areas adjacent to the outlet of the Greens Burn into Loch Leven. In addition a transect of samples was taken along lines parallel to (Transect B) and perpendicular to (Transect A) the shore, from the point opposite the middle of the Greens Burn outlet to assess the loch floor for the presence of an alluvial fan. Cores were sieved through 100 μm and 500μm sieves and invertebrates identified to the nearest possible taxon. Loss on ignition of sediment samples was also carried out. Table 13 gives a summary of the findings for the 1994 and 2003 data.

Table 13. Comparison of ecological data in Loch Leven, taken before and after installation of the Greens Burn Buffer strip adjacent to the mouth of the Burn**.**

Sample data	**2003**	**1994**
Total individuals identified	147	514
Total taxa	12	6
BMWP score	38	19
ASPT	3.8	3.8
Shannon index	1.570	1.236
Total Chironomids	29	210
Total Nematodes	215	175
Total Oligochaetes	68	111

The BMWP score for 2003 was twice that of 1994, which is indicative of an improvement in the ecological status of the Loch but the difference was not found to be significant (using a Mann-Whitney U-test, P=0.194) and the ASPT was the same for both years. No significant change in the families present between the two years was found, but the number of chironomids present was significantly lower in 2003 (P=0.030). The organic matter content of the sediment in 2003 declined with distance from the Greens Burn outflow (p=0.019 for transect A and p=0.068 for transect B), but this was not compared with 1994. The number of taxa, ASPT, BWMP and Shannon indices all showed a significant positive correlation with perpendicular distance from the shore, from the outlet of the Greens Burn, but not with distance parallel to the shore. The effect of increasing distance from shore can be interpreted as the effects of increasing wave action with distance from shore, especially as no change was observed parallel to the shore, so this cannot be attributed to the effects of the buffer strip.

Box 1. Summary of discussions about erosion control measures with farmer at Wester Gospetry, 11 September 2003.

1. Winter Seedbeds. Autumn cultivation sequence is ploughing/disc harrowing followed by power harrow/drill followed by rolling. Field 18 was being sown with WB that day (11 September). 10 t/ha hen pen previous day. Agreed that one 6m pass with the
roller would be omitted to see how this did. Rolling would be done later, to ensure good Mn uptake in spring.

2. Earth dam/stone spillway. Discussed building small stone wall using stones retained from separation in spring (stored at quarry).

3. Soil analysis/P fertiliser

No fertiliser used in autumn, but noted that field 18 (see above) had just received 3 ton/acre (10 tonne/ha) of hen pen. This is about 200 kg/ha P_2O_5. Farmer has regular soil analysis done by SOYL precision farming

4. Ploughing of spring sown fields

mainly delayed till spring to leave overwinter stubble.
Not many fields going into winter cereals.

5. Subsoiling. Agreed that field 33 (wee law) would be subsoiled - this field tends to erode into ditch which then discharges to River Eden.It has a set aside strip adjacent to field edge. Evidence of soil building up against wall. Currently in potatoes. Also suggested subsoiling on field 36 (west hillfoot)

6. Nutrient budgeting

Needed for NVZ - discussed levels of manure and fertiliser: 110 units/acre to winter cereals, less to spring cereals. Midden manure spread in singlefield (70-100 overwintered cattle). Farmer did not attend NVZ workshop.

7. Winter livestock access not discussed

Main issue is likely to be runoff from kale field adjacent to steading (grazed from Jan)
This water runs off into farm steading sump and thence along 600m drain to River Eden headwaters. The sump has capacity for about 5m^3 of sediment and was over half full at the time.

8. Cultivations after potatoes

Aims to plough all in autumn. Requested no ploughing on field 16, so that effectiveness of the buffer could be evaluated.

9. Suggested participation in SEERAD BMP Evaluation project - aware of PEPFAA code
though not the acronym - polite interest and subsequently agreed to participation.

10. Other

Farmer pointed out two surface drainage points which are likely to have a major impact on water quality:

a. steading drainage after sump goes to river Eden - in stream wetland for sedimentation a possibility

b. surface drainage from track above Farm - takes runoff from North long - the best soil on the farm

This crosses field 12 and discharges into Greens Burn, bypassing the buffer. As this field is highly fertile, it has been sown early to WB and received 10 t/ha hen pen, this represents probably the biggest P pollution risk in the whole farm - and the traffic leaves the field at the bottom corner exacerbating the risk.

ASSESSMENT OF INVERTEBRATE BIODIVERSITY OF GREENS BURN BUFFER STRIP

The purpose of this survey was to provide some biodiversity context to the wider BMP work by assessing the invertebrate assemblages within the Greens Burn buffer strip established on Wester Gospetry farm. This was achieved by establishing pitfall traps at 6 localities within the Greens Burn buffer strip and comparing the catches of these with pitfall traps established within one arable field acting as a control site.

Methods

Pitfall traps were used to assess the activity density of ground active invertebrates. Traps consisted of plastic beakers (75 mm in diameter and 100 mm deep) partly filled with monopropylene glycol as a killing agent and preservative. A 15 mm mesh grid was secured over the trap mouth with a metal staple to reduce interference by livestock and to prevent small mammals from entering the traps (Downie et al. 2000). At 6 points along the buffer strip (Table 13) rows of 9 pitfall traps (placed at 2 m intervals) were established. One row was situated in the middle (hereafter referred to as Middle) of the buffer strip, while the second was situated 1-2m from the burn fence (hereafter referred to as Edge). In addition, 2 rows of pitfall traps were placed in one spring barley field (Field 17) with one row 4 m from the field boundary the second 30m from the field boundary. Traps were left in situ for a four week period (16 May-16 June 2003) and each row of 9 pitfalls were pooled at collection. At each line of pitfalls the main plant species, height of the grass, the soil penetrability, the percentage of plant trash at ground level, distance from the field fence and the total margin width were measured on 16 June 2003. In addition, 4 random soil cores were take at each of the 7 sampling points on 16 May 2003 and the following analyses conducted: moisture content, percentage organic matter (loss on ignition), pH and extractable P and K.

The following groups of invertebrates were investigated: slugs (Limacidae and Arionidae), earthworms, sawfly larvae, lepidopteran caterpillars, spiders, predatory beetle larvae, leatherjackets, aphids, and plant bugs (homopteran and heteropteran). In addition ground beetles were identified to species level. These groups were selected as they are pest species (e.g. aphids and slugs), beneficial predators (e.g. spiders and ground beetles) or important prey for birds (e.g. leatherjackets and caterpillars).

Table 13: Location of seven sampling points associated with the Green Burn BMP at Wester Gospetry farm. Field numbers are those allocated by the farmer.

Invertebrate sampling point	Located within buffer strip in or adjacent to Field Number
1	Field 12
2	Field 17 (control, within the spring barley field)
3	Field 17
4	Field 27
5	Field 36
6	Field 35 (west)
7	Field 35 (east)

RESULTS

Environmental data

There were physical differences between different areas of the buffer strip, for example the margin was widest at points 6 and 7 (table 14). These sampling points also tended to have shorter vegetation and were dominated by Yorkshire Fog while the other sampling sites tended to be primarily dominated with Cocksfoot (Table 14). The soil at points 6 and 7 was more acidic with a higher moisture and organic matter content (Table 15). Sampling points 1,2 and 3 had the highest available phosphorus in the soil.

Table 14. Physical attributes of the buffer strip looking at mean vegetation height, plant trash at ground level (percentage cover of a 25 x 25 cm quadrat) and penetrability of the soil (where higher readings indicate greater force required to penetrate the soil). Distance of the pitfalls from the field edge and widths of the margin are also provided.

	Distance from field (m)	Vegetation Height (cm)	Margin width (m)	Plant trash	Penetrometer Reading
1 Edge	13.1	93.0	16.0	40.0	82.9
1 Middle	6.1	132.3	16.0	50.0	76.8
2 Edge	4.9	73.7	-	0	69.3
2 Middle	26.5	79.0	-	0	51.5
3 Edge	9.2	140.3	16.0	40.0	69.5
3 Middle	4.8	148.7	16.0	33.3	69.7
4 Edge	6.1	120.3	8.5	10.0	43.1
4 Middle	2.2	146.3	8.5	10.0	43.2
5 Edge	16.4	121.7	19.9	13.3	71.1
5 Middle	10.1	130.7	19.9	16.7	62.0
6 Edge	67.5	42.3	70.0	3.3	66.0
6 Middle	58.0	70.0	70.0	23.3	68.6
7 Edge	68.0	40.3	70.0	10.0	65.8
7 Middle	58.0	30.3	70.0	30.0	62.1

Carabid beetles

A total of 229 ground beetles were identified from the margin consisting of 20 different species. In addition 109 beetles consisting of 14 species were identified from the spring barley field. The average number of ground beetles and ground beetle species richness was higher in the spring barley field than in the buffer strip (Figure 14). The average rarity of ground beetles (site quality score) was not found to differ between the two habitats.

The structure of the ground beetle assemblage was investigated by detrended correspondence analysis (DCA) of the relative abundance data without downweighting rare species (Hill 1979, Oksanen and Minchin 1997). Eigenvalues for axes one to four were 0.4790, 0.185, 0.0455, 0.0264 respectively hence indicating axes one and two account for most of the variation in ground beetle assemblage structure. Figure 15 indicates that the two spring barley sampling sites occurred towards the right hand side of the ordination and Figure 16 indicates that typical arable species such as *Bembidion lampros* and *Agonum dorsale* were more abundant in these sites. There is also a slight separation between the sampling points in the middle of the buffer strip (middle) and those at situated close to the burn (edge) with the former tending to occur towards the top left hand corner of the ordination.

Table 15: Dominant vegetation and other plant species observed at each sampling location.

	Dominant vegetation	*Other vegetation observed*
1 Edge	Cocksfoot Meadow Fescue	Couch Grass, Red Fescue, Yorkshire Fog, Timothy
1 Middle	Timothy, Meadow Fescue	Cocksfoot, Red Fescue, Yorkshire Fog, Bent Grass, Couch Grass
2 Edge	Spring Barley	NA
2 Middle	Spring Barley	NA
3 Edge	Cocksfoot	Red Fescue, Timothy, Meadow Fescue, Rough Meadow Grass, Crested Dogtail
3 Middle	Cocksfoot	Yorkshire Fog, Bent Grass, Rough Meadow Grass
4 Edge	Cocksfoot, Dock	Rough Meadow Grass, *Juncus*
4 Middle	Cocksfoot	Dock
5 Edge	Cocksfoot	Meadow Fescue, Red Fescue, Bent Grass, Creeping Buttercup
5 Middle	Cocksfoot	Red Fescue, Timothy
6 Edge	Yorkshire Fog	Bent Grass, Creeping Buttercup, Creeping Bent, Smooth Meadow Grass
6 Middle	Yorkshire Fog	Meadow Fescue, White Clover, Red Fescue, Sorrel
7 Edge	Yorkshire Fog	Bent Grass, Creeping buttercup, Rough Meadow Grass, Sorrel, Common Mouse Ear.
7 Middle	Yorkshire Fog, Bent Grass	Couch Grass, Creeping Bent, Sorrel, Perennial Ryegrass, moss

The species ordination indicates that species typical of field margins and hedgerows (i.e. *Carbus nemoralis* and *Bradycellus harpalinus*) were more abundant in the edge sites. Paired t-tests indicated that neither carabid species richness (i.e. total number of carabid species) nor site quality score (i.e. average rarity of carabids) differed significantly between pitfalls located at the burn edge and pitfalls located in the middle of the buffer strip (T=0.00, P = NS; T = 0.93, P = NS for species richness and SQS respectively).

Table 16: Soil attributes (based on 4 random soil cores) at each of the sampling points

Sampling Point	*Moisture %*	*% Loss on ignition*	*Organic matter*	*pH*	*P mg/l*	*P*	*K mg/l*	*K*
Point 1	16.4	4.4	Low	5.6	129	High	108	Moderate
Point 2	15.1	4.6	Low	5.6	107	High	104	Moderate
Point 3	19.3	4.7	Low	5.9	109	High	132	Moderate
Point 4	19.8	4.6	Low	5.8	58	Moderate	116	Moderate
Point 5	18.9	4.1	Low	6.0	70	Moderate	88	Moderate
Point 6	26.5	6.4	Low	5.0	32	Moderate	100	Moderate
Point 7	27.1	7.2	Moderate	5.0	15	Low	100	Moderate

The influence of margin attributes (i.e. grass height, soil penetrability, % plant trash, distance from field and the total margin width) on species richness and site quality score were investigated using stepwise multiple regression (Table 17). Where required, data were log transformed to normalise prior to analyses and only variables correlated with species richness or site quality score ($P<0.1$) were included in the models. Site quality score was related to the distance the pitfall traps were set from the field margin with 30.8% of the variation in site quality score being accounted for. Pitfall traps established at a greater distance from the field margin were found to have a higher site quality scores. Species richness was strongly related to grass height and margin width with 63.8% of the variation in species richness being accounted for by these two factors. In particular grass height was negatively correlated with species richness while margin width was positively correlated, indicating that more species were collected in shorter grass and in pitfalls established in areas where the buffer strip was wider.

Ecological Groups
Ground beetles were classified into one of 7 groups on the basis of their ecology (Cole et al. 2002). The relative abundances of these 7 ecological groups were compared between the pitfalls established in the middle and burn edge of the buffer strip and those established in the spring barley field. Data collected from arable (i.e. spring/winter barely, spring oats and winter wheat) and grassland (i.e. semi-natural, intensively grazed and conserved grassland) in the River Earn catchment (1998-2000) was used to compare the ecological makeup of the buffer strip beetles with that typically found in grassland and arable land.

Table 17. Multiple regression indicating influence of environmental attributes on carabid site quality score and species richness. R^2 is the co-efficient of variation.

Environmental factor	Carabid Site Quality Score	Carabid Species Richness
Margin width	-	T=3.85 $P < 0.01$ (+ve correlation)
Grass Height	_	T=2.20 $P = 0.05$ (-ve correlation)
Distance from cereal field	T=2.43 $P < 0.05$ (+ve correlation)	NS
	R^2 (adj) = 30.8%	R^2 (adj) = 63.8%

From Figure 17 it can be seen that the ecological assemblage structure of the middle and edge buffer strip carabids was very similar as was the ecological assemblage structure of the spring barley field at Wester Gospetry when compared to typical arable fields in the River Earn catchment. The ecological assemblage structure of the buffer strip beetles differed considerably when compared to typical grassland (i.e. semi-natural, intensively grazed and conserved grassland) and actually better resembled arable communities than grassland communities. In particular there was a much higher percentage of Group 6 species (small nocturnal and diurnal predators) and a much lower percentage of Group 1 species (medium sized nocturnal predators). In addition Group 7 species (small to medium plant feeding carabids) were more abundant in the buffer strip than the other habitats investigated. It was also found that Group 2 species (large immobile *Carabus* species) which are indicative of semi natural habitats, only occurred in one set of pitfalls at sampling point 7 which was situated in close proximity to a farm woodland. *Other invertebrates.* In addition to carabid beetles, the number of: Arionidae and Limacidae slugs, homopteran and heteropteran bugs, lepidopteran and sawfly caterpillars, earthworms, spiders, leatherjackets, aphids and predatory beetle larvae were counted. The assemblage structure of these invertebrates was investigated by DCA of the abundance data without downweighting rare species (Figures 18 and 19). Eigenvalues for axes one to four were 0.166, 0.080, 0.01 and 0.001, respectively hence indicating axes one and two account for most of the variation in invertebrate assemblage structure.

The spring barley samples were again separate from the buffer strip samples indicating that the assemblage structure differed between the two habitats. The species ordination indicates that the barley field had a higher abundance of spiders than the grass margin. The spatial orientation of the invertebrate assemblage in the ordination space seems to be related more to the location along the margin that pitfalls were established rather than whether they were established in the middle of the margin or at the burn edge of the margin. In support of this finding, paired t-tests indicated that abundance of Arionidae and Limacidae slugs, homopteran bugs, earthworms, leatherjackets, spiders and predatory beetle larvae did not differ significantly between pitfalls established in the middle of the buffer strip and those established at the burn edge.

Where margin attributes (i.e. grass height, soil penetrability, % plant trash, distance from field and the total margin width) were found to be related to invertebrate abundance (i.e. Limacidae slugs, homopteran and heteropteran bugs, spiders and predatory beetle larvae) stepwise multiple regressions were conducted to investigate the relationship further. Data were log transformed to normalise where required and only factors showing significant correlation ($P \leq 0.1$) were included in the models.

The number of spiders, homopteran bugs and Limacidae slugs were related solely to the widths of the margin which accounted for 41.8, 29.3 and 28.76% of the variation in the abundance of these species respectively (Table 18). Buffer width, in addition to grass height, was also strongly related to the number of heteropteran bugs with 65.3% of the variation in heteropteran abundance being accounted for by these 2 factors. Again more heteropteran bugs were found in the widest buffer strip areas and long grass was found to negatively influence the abundance of these species. The abundance of all four of these invertebrate groups was found to increase as the widths of the buffer strip increased. 77.1% of the variation in predatory beetle larvae could be accounted for by the

Table 18. Multiple regression indicating influence of environmental attributes on the activity abundance of spiders, heteropteran and homopteran bugs, predatory beetle larvae and Limacidae slugs. R^2 is the co-efficient of variation.

Environmental factor	Spider	**Hetero-ptera**	**Homo-ptera**	**Predatory Beetle larvae**	**Limacidae Slugs**
Margin width	T=2.98 $P < 0.05$ (+ve)	T = 3.73 $P < 0.01$ (+ve)	T=2.36 $P < 0.05$ (+ve)	-	T=2.33 $P < 0.05$ (+ve)
Distance from field	NS	NS	NS	-	NS
Grass Height	NS	T = 1.90 $P < 0.1$ (-ve)	NS	-	-
% plant trash	-	-	-	T=-6.16 $P < 0.001$ (-ve)	-
	R^2= 41.81%	R^2= 65.32%	R^2= 29.37%	R^2= 77.06%	R^2= 28.76%

percentage of plant trash, with the highest abundance of larvae occurring in ground with a high percentage of trash.

Conclusions and recommendations from invertebrate study

Each of the survey points along the buffer strip was similar in that they consisted of relatively tall and dense vegetation dominated by only one or two species of grasses. This lack of heterogeneity in both vegetation type and structure means that a relatively limited number of invertebrate species and groups currently occur within the buffer strip. In addition, the lack of any active management means that the ground beetle assemblages occurring within the strip are not as would be expected given the dominance of grasses within the buffer strip. Instead, the assemblages are generally more characteristic of arable situations and hence the grassland buffer strip are added little in the way of invertebrate species richness when compared to the surrounding arable land.

A greater amount of heterogeneity in terms of vegetation type and structure is required in the buffer strip in order to increase the diversity of invertebrates occurring there. To this end, active management is required to reduce the dominance of individual grass species, increase the diversity of both other grass and broad-leaved vegetation species and provide a more open sward structure. It will, however, be important to ensure that the type and timing of the management applied is such that the resulting vegetation structure does not detract from the buffer strips primary function of mitigating the movement of soil from the adjacent fields into the watercourse. It is our understanding that this aspect of the buffer strip is especially important during and immediately after cultivation of the arable fields (hence in spring and/or autumn depending on crop type). Bearing in mind the need to ensure sufficient growth of vegetation during those periods, it is recommended that:

- A cutting regime be introduced within the buffer strip to open up the vegetation, reduce the soil nutrient status and thereby encourage less of a dominance of individual grass species. Given the need for a tall sward in autumn, through the winter and at the start of spring, it is recommended that in the first instance the vegetation be cut and removed from the buffer strip once during mid to late July each year. This should allow for sufficient regrowth to occur prior to the peak of cultivation activity in the neighbouring fields in the autumn while also limiting the effects on any bird species which may be nesting within the sward. The cut vegetation should be retained in a swath on the buffer strip.

- Measures be introduced to open up the sward at the base and provide colonisation and establishment opportunities for other vegetation species. It is suggested that this could be achieved by chain-harrowing parts of the buffer strip immediately after the vegetation has been cut and removed. It may be that this need only be conducted in the year in which the cutting regime is first instigated, although it may also be that this harrowing needs to be repeated every second year. The resulting growth, vegetation composition and structure of the harrowed areas will determine whether additional passes are required following the first year of harrowing. It is recommended that harrowing is only applied to part rather than all of the buffer strip and that the areas adjacent to both the field and the watercourse are left unharrowed. The width of the harrow should be used to establish alternating harrowed and unharrowed areas along the buffer strip, though the width of the strip will dictate how many of these harrowed areas can be incorporated at each point of the buffer.

Adjacent field

Harrowed strip within buffer strip

Harrowed strip within buffer strip

Watercourse

It is recommended that the effectiveness of this management regime be reviewed in the autumn of the second year after the cutting regime is applied, i.e. after two bouts of cutting and removal of vegetation and one bout of chain harrowing. This review need not entail a reassessment of invertebrate communities but rather could be based on a visual assessment of the sward with the view that a greater diversity of both sward composition and structure is what is being aimed for.

Food resources for birds would be enhanced by planting winter seed bearing crops such as rape or kale. The increased cutting would also enhance accessibility, which is poor in some seasons at present in many parts of the buffer.

CESSNOCK WATER

Identification of suitable reaches for evaluating impact of fencing/buffer strips

The original objective of field work in the Cessnock catchment was to compare the water quality in contrasting sub-catchments in the Cessnock Water which had/had not been subject to improvements under the SEERAD Bathing Water works project. The proposed works for this project were divided into steading and field pollution prevention works and the decision was made that the participating farms in the Cessnock catchment would only receive the money for steading works, not catchment works, such as fencing. Therefore we modified the plans and looked for suitable reaches in the Cessnock catchment which were characterised by high and low amounts of stock access to water. Ideally we wanted reaches in the upper part of subcatchments (so that the input from upstream would be small) with little influence of farm steading and other sources of pollution. We wanted reaches that would represent different levels of stock access, identified by the presence or absence of fencing along stream margins. On 9th July 2003 AV walked over several potential farms and associated watercourses with the SAC area advisor, Andy Leggate.

Initial assessment of levels of livestock access to water were assessed using data collected for the whole Cessnock catchment in summer 2002 (Mouat, 2002). Following groundtruthing (see above) we identified one reach, above and below Barneighthill (Figures 20, 21a), where only a short section of watercourse was accessed by cattle, the rest being either fenced, or inaccessible. Figure 22 shows the results of previous point sampling data for 10 sampling points in the Cessnock catchment for faecal coliforms in summer 2002, which suggested that that below Barneighthill gives faecal coliforms counts lower than the average, at least at low discharges. However, some of the fields containing stock were adjacent to the watercourse and a drain carrying overflow from a sump collecting farm steading water was also identified and sampled. Moreover, there was a farm and roads above this reach, so incoming water would already be potentially polluted.

A second reach was identified, above and below Langside Farm (Figure 21b). Two streams draining this land join below the farm steading, one of which runs through mainly steep sided, wooded or well buffered riparian areas, except for one field, which is split by the watercourse. The riparian zone in this field is about 4-6m wide, contained rough vegetation (rushes, thistles etc) and several crossing points for livestock. The second stream rises adjacent to the farm steading and carries some steading runoff water, storm water from fields to the north and west of the farm and exhibits very severe stock access below the steading, up to its confluence with the first stream. We therefore identified 3 sampling points: above the farm, below the farm but above the confluence with the stream draining the farm steading, and below this confluence.

We identified a third reach, above and below Wynds, where little livestock influence was evident above the reach, but the reach consisted of only one field, adjacent to the steading, where overnighting of dairy cows, which have at least two drinking water access points to the stream, occurred. There was open access to the watercourse, with two major crossing points and two tracks draining into the stream along this reach.

Results for this reach are not reported, but to enable local information about discharges to be determined, a submersed and an atmospheric control pressure transducer (diver) were installed above Wynds reach. In this reach a river section with a good rectangular profile was identified, enabling flow estimation and scaling to give discharges at the other sites in the catchment. Flows over the period from July to September 2003 were mainly very low.

Results and Discussion

Sampling of storm events

Table 19 summarises storm events sampled in Cessnock reaches, during 2003/4

Table 19. Storm events sampled in Cessnock reaches.

	Sampling sites	**Flow measurement**	**Analysis SAC**	**Analysis SEPA**
21-22 Sept 2003	AL, FBL AW, BW AB,BB	Stage below Wynds		T,SS,DOC, NH_4, NO_3,OP,TP,pH,EC ,FE,FC,TC
29-30 Sept 2003	AL, FBL AW, BW AB,BB	Stage below Wynds	SS,DOC, NH_4, NO_3,pH,EC, FE,FC,TC	
1-2 Nov 2003	AL *, BL,FBL	Stage below Wynds	FE,FC,TC	T,SS,DOC, NH_4, NO_3,OP,TP,pH,EC
18-19 Nov 2003	AL,BL,FBL	Stage below Wynds	FE,FC,TC	T,SS,DOC, NH_4, NO_3,OP,TP,pH,EC
1-8 April 2004**		AV/FM at Langside	FE,FC,TC PH, SS, NH_4, EC	
8-14 April 2004**		AV/FM at Langside	FE,FC,TC	
14-20 April 2004**		AV/FM at Langside	FE,FC,TC	

AL= Above Langside, BL = Below Langside, FBL=further below Langside
AW= Above Wynds, BW=Below Wynds
AB= Above Barneighthill, BB=Below Barneighthill
FE = faecal enterococci, FC= faecal coliforms, TC= total coliforms
T= turbidity , SS=suspended solids, DOC=dissolved organic carbon, NH4 = ammonium N, NO3=nitrate N,OP = orthophosphate, TP = total phosphate, EC = electrical conductivity
* sampler failed ** events collected under SEERAD BMPs project

Event 1. September 21-22, 2003

This large event was hand sampled, below Langside, above and below Wynds and above and below Barneighthill. Due to hand sampling , and the difficulties of night work (failure to find the above Langside sampling site!), and travel time from Edinburgh, the sampling does not cover the whole event in all cases. Samples were analysed by SEPA

E.Kilbride for FIO's (Total Coliforms,Faecal Coliforms, Faecal Enterococci), pH, TOC, COD, NH4, NO3, NO2, TSS, Turbidity, Conductivity, SRP and TP.

Event 2. September 29-30, 2003
This was a much smaller event, sampled using 4 samplers from SAC Auchincruive, deployed above and below Langside and above and below Barneighthill. SEPA E.Kilbride labs were unable to take the samples, so they were analysed at SAC Auchincruive.

Events 3 and 4. November 1-2 and November 18-19, 2003
These events were only captured at Langside. 3 samplers were deployed (see map). On 1-2 November, the sampler above Langside failed, and the one below the input from the side stream draining the steading failed just as peak discharge approached. On 18-19 November, 3 samplers were present and the samples were analysed by SAC labs.

Figures 23-26 summarise water chemistry and microbiology during these events. The unfenced reach (Langside) is close to the top of the subcatchment, so pollutant concentrations upstream of the reach are low, so the impact of the reach on water quality is clear. In the fenced reach (Barneighthill), there is a larger input from upstream, including a large increase in concentration before the stage rises, probably related to animal movement during milking. This made unequivocal conclusions from the data difficult to reach. However it is clear that the fenced reach at Barneighthill is still generating an increase in pollution load, which is in fact larger than that of the unfenced reach at Langside. This may be due to overflow of a sump collecting steading drainage, which means there is a significant steading component to the increase. The increase in concentrations of ammonium and suspended solids in the unfenced reach are quite modest (250 μg/L NH_4-N is the class A maximum value for the Scottish Rivers Classification scheme). However it should be borne in mind that animals had recently been housed for the winter, so fresh faecal deposition was limited to young stock which were still outside. The only other storm event captured for the period from July to September 2003 (when livestock were still out), was on September 20-21). This had a > 1 year return period, and no clear increase in concentrations across the reaches were discernible, because discharges were so high.

The data from the Cessnock reaches has been summarised in Figure 27. The dominant effect for all chemical determinands presented is that of the size of the event, with flow weighted mean concentration increasing as scaled event discharge increases. There is some evidence of lower SS, DRP and TP above the farms studied, but these differences are much smaller than the effect of discharge. It is not possible to identify a difference in response of the unbuffered reach (Langside) compared with the fenced reach (Barneighthill). At high discharge one might expect source exhaustion to cause a dilution of the pollutant concentrations, but there is no evidence of this, although there is evidence that the pollutant concentration is asymptoting towards a maximum for some pollutants. For the faecal coliforms, there is evidence for each event of higher concentrations below the reach, but a much weaker relationship with discharge over all events, because fresh animal inputs decreased over the sampling period of September to November.

FIELD PLOTS, BUSH ESTATE

Buffer strip plots

Two 7m wide x 17m long plots were established on a sandy loam arable drained plots at Bush Estate, Midlothian. One had a 10m grass (>5 years old) buffer, with 7m bare soil above the buffer, one had 17m of bare soil without a buffer. Experiments were carried out in two MSc projects (Zhang, P (2004), Zhang, L. (2004)). Two days prior to each experiment, slurry was applied to a 2m x 2m area at the top end of each plot and two days later, artifical rainfall was applied (nominal volume of 2000L) at either very high intensity using a slurry tanker, or low intensity (see table 20). The high flow rate is comparable with the highest discharge considered in the sensitivity analysis on buffer strip efficiency (summarised in Table 6). There was significant kinetic energy in the application, leading to dispersal of surface soil and transport of high concentrations of sediment and faecal coliforms into the buffer strip, or control unbuffered area. Samples were taken at four cross sections down the buffer strip, with three replicate points across the cross section and up to five times at each sampling position. Samples were collected from raised boardwalks set up prior to the experiments (Figure 28).

Table 20. Experimental conditions during buffer strip plot work.

Experiment No.	1	2	3	4
Date	May-19-2004	May-26-2004	Jun-2-2004	Jun-8-2004
Field conditions	Buffer strip	Buffer strip	Non-buffer	Non-buffer
Flow rate	Low	High	Low	High
Duration of runoff	3h 24m	690seconds	2h 34m	300seconds

Figure 29 summarises the results. They confirm that the suspended solids concentration declines more strongly across a 10m buffer strip than the faecal coliforms. The suspended solids concentration entering the buffer strip at high flow (about 2L/m/sec) is higher than at low flow (0.05 L/m/sec) at the input side, but declines more rapidly, because the particle size is coarser.

Particle tracing experiments

A series of tracing experiments (James, 2003) was carried out using fluorescent sand grains of two particle sizes, coarse (>100μm) and fine (<100μm), to assess the effect of grassed buffer strips and plot treatments on soil transport from arable plots. The sand grains were placed on ploughed, rolled, buffered and non-buffered plots. A technique of photographing the fluorescent tracer under ultra violet (UV) light was developed to enable assessment of the particle distances travelled. Following rainfall events (natural and simulated), the new positions of the tracer particles were recorded using the photographic technique. In addition, surface runoff and sediment transport into the troughs at the base of each plot were observed. The tracer was found to move down the slope in response to increasing rainfall, with fine particles transported more and the furthest. It was concluded from the lack of surface runoff recorded, that tracer movement might have occurred by localised runoff or rainsplash erosion. The tracer material was transported more and further down rolled slopes than ploughed slopes, indicating that rolled plots are more susceptible to erosion than ploughed plots. The presence of the buffer caused a significant reduction in the movement and distance of particle transport, implying that buffer strips are effective in reducing sediment transport by rainsplash erosion.

DISCUSSION OF FIELD WORK RESULTS

The main observations from the work at Green's Burn were as follows:

- Subsoiling of headlands, an increased emphasis on spring cereals, and grubbing of the soil after potatoes are in-field BMPs which are already practiced in the Greens Burn catchment;
- Before buffer strip installation in the catchment the mean event total P concentration (TP) (for 3 significant events) was 1.1 ±0.6 mg TP/L and after installation (3 significant events) it was 0.5±0.16 mg TP/L. For SS the values were 233 mg/L ±115 and 94±62 respectively;
- most sediment trapping occurred before the buffer or in the first few metres of buffer;
- sediment retained in the buffer contained a lower P content and organic matter content than the soil from the field;
- Where flows were concentrated, sediment trapping was not very effective: efficiency of P removal from concentrated flows from West Hillfoot field was only about 35%;
- Drainage bypass routes across/under buffers, which may seriously undermine buffer strip effectiveness, were identified.
- The impact of such bypass routes is indicated by the difference between the TP load in the 29/11/03 event at sampler B (0.49 kg P) and sampler C (1.24 kg). The two samplers were separated only by two fields on one side, and one field on the other (Figure 7), one of which delivers water via a surface runoff drain from a heavily manured winter barley field.
- Biodiversity of the buffer strip was not greatly enhanced compared with the arable area.
- No significant trend of improved invertebrate diversity in Loch Leven was observed, in the region of the outlet of the Greens Burn.

The main observations from the work in the Cessnock catchment were as follows:

- Studies were inconclusive in evaluating the effect of fencing on faecal coliform pollution, because of the impact of other sources of pollution such as steading sump overflow;
- There was a clear impact of the dairy farming on pollutant concentrations in both the fenced and unfenced reaches studied.
- For all pollutants studied, the dominant factor controlling the pollutant concentration was the scaled discharge, and a close to linear relationship between chemical pollutant concentration and discharge occurred .

The main observations from the field plot work was as follows:

- Faecal pollutant capture is less efficient than inorganic sediment capture, both at high and low discharge.

There is evidence from the data that the buffer strip may delay transport of sediment to watercourses, possibly by providing a temporary reservoir, which may be overtopped at some point in the storm event. Hence for small events (return periods of several per year) buffer strips may be more effective than for large events. For example, the high, and delayed sediment loads at sampler A may be because of delayed release from a reservoir at the corner of field 27 (West Hillfoot) (see Figure 30 for concept). This would be in accord with the theory, described in Chapter 2, of buffer strip inundation during large events. The field concerned is known to erode sufficiently badly for water and sediment to eventually cut across the buffer strip untreated, in the right conditions.

These observations suggests that good field margin management (fencing, hedgerows, maintaining an uneven boundary between cultivation and margin) is as important and effective as maintaining buffer strips for sediment removal. The field margin is also an area where food resources for birds would be enhanced by planting winter seed bearing crops such as rape or kale.

Biodiversity of the Green's Burn buffer would be enhanced by more active management (eg grazing of the buffer, leaving bare soil areas, more frequent cutting). However this may impair effectiveness as a pollution control measure. The increased cutting would also enhance accessibility to the public, which is poor in some seasons at present, because of the height of the vegetation.

The poor efficiency of removal of finer colloidal material and of microbial pollutants when not sorbed onto larger sediment particles has been observed by others. Braskerud (2001) found that even on clay soils, much of the clay size fraction was collected by in-stream wetlands in aggregates of about 6μm diameter. This size class will not be effectively removed by field margins or buffer strips (see Table 6), and instream wetlands or temporary storm water impoundments at the field margin may be a more appropriate way of achieving removal of particles of this size, and of enhancing the temporary storage depicted in Figure 30. Such a measure has been designed and built at West Hillfoot field, Wester Gospetry, as part of a SEERAD funded BMP evaluation project. However it is clearly more desirable to prevent the mobilisation of colloidal material further up the treatment train, by continuing to promote spring cereal sowing/winter stubble, early establishment of winter cover and headland subsoiling.

Although the field data from the Cessnock water are difficult to interpret, they do suggest that unless steading BMP's are implemented, it is unlikely that major impacts of field margin based BMPs such as fencing will be readily observed. Moreover the field plot data suggest that such BMPs are probably effective mainly by preventing stock access rather than by the buffer strip actions of filtration, infiltration and sedimentation described in Chapter 2. We have observed that where Rural Stewardship schemes have been used to implement riparian fencing in the Cessnock catchment, there are often gaps left to allow stock access to water. Unless these access points are well managed, or off stream drinking water supply is provided, this is unlikely to lead to an improvement in the pollution loads, although it may alter the biodiversity value of the riparian zone.

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4. APPLICATION OF BMPS TO SCOTTISH CONDITIONS

SPECIAL FEATURES OF SCOTLAND TO BE CONSIDERED

The special features which might make consideration of site suitability for BMP's different in Scotland, compared with England include:

- higher OM content of soils;
- cooler in spring (leading to more need for fertiliser in early spring);
- glaciated landscape leading to weakly structured soils, imperfect drainage and undulating landforms;
- high recharge (>300 mm) and lower summer soil moisture deficits(<100mm) even in arable areas;
- many sensitive or oligotrophic fresh water bodies;
- bathing water contamination more likely because of low summer soil moisture deficits;
- significant use of corrals for livestock overwintering;
- difficulties with slurry/dirty water management in livestock industry stemming from climatic and soil conditions;
- central role of livestock (dairy cattle, beef cattle, sheep) in Scottish Agriculture;
- central role of tourism in Scottish economy;
- presumption of open access to countryside.
- different system of agri-environmental subsidies currently and likely future different requirements under GAEC as a result of CAP reforms.

The four regions of Scotland and their predominant climate features and resulting agriculture are:

- North-West (NW); extremely wet but relatively mild temperatures; extensive grazing and crofting.
- North-East (NE); dry but also cold; mixed arable/grazing.
- South-West (SW); wet and warm; mainly intensive grazing.
- South-East (SE); dry and warm; predominantly arable cropping.

In Table 20 we have formed a judgement of the most appropriate BMPs that will find application in these four regions of Scotland, in accordance with the predominant agricultural systems prevalent in those regions. In Table 21 we identify specific issues that would need to be considered for the application of the various BMP measures in Scotland, along with potential approaches to assessing suitability of sites and soils on a national basis. Of these measures, we have identified two measures that are of wide interest and applicability for Scotland, and in the next section we explore the spatial distribution of the measures that would be appropriate. The two measures to be considered are (i) the use of buffer strips, and (ii) conversion from winter to spring cereals, in areas of high risk of erosion, with close proximity to water and high proportion of winter cereals.

Table 20. Applicability of possible control methods to reduce water pollution by diffuse sources from agriculture, for the four regions of Scotland shown. +++ = widely applicable; ++ = applicable in some areas; + = little applicability. Farming systems are arable (A) or livestock (L)(grazing) with either mixed farms or manure used on arable farms (L/A).As commented before split – planning, in-field, riparian, steading. Within each split groupings could be made by treatment train ie input reduction, run-off speed etc.

Control Method	Treatment Train position*	Farming System	Region			
			North West	**North East**	**South West**	**South East**
Planning tools:						
Crop/grass nutrient management planning	1	A/L	++	+++	+++	+++
Crop pesticide management planning	1	A & L	+	++	++	+++
Restricted spatial and temporal application of crop inputs	1	A & L	+	++	++	+++
In field tools:						
Over-winter ground cover	3	A	+	++	++	+++
Spring sowing and early seedbeds	2,3	A	+	++	+	++
Avoid the use of tramlines in winter	2,3,4	A	+	++	+	++
Strip cropping	2,3,4	A	+	++	+	++
Stocking rates	1,3	L	++	+	+++	+
Minimal cultivation	2,3,4	A	+	++	+	++
Contour management	2,3	A	+	+	+	++
Feed composition	2,3,4	L	+	++	+++	++
Manure application rate	1	L/A	+	++	+++	++
Manure treatment	1	L	+	+	++	+
Closed spreading period	1	L/A	+++	++	+++	++
Soil incorporation of manure	4	L/A	+	++	+++	+++
Cultivation for soil stability (inc. Sub-soiling)	4	A	+	+++	+	+++
Field margin						
Barrier ditches	5	A	+++	++	+++	++
Vegetative barrier strips and fences	5	A & L	+	++	++	+++
In stream:						
Reed-beds	6,7	A & L	+++	++	+++	++
Constructed wetlands	6,7	A & L	+++	++	+++	++
Riparian buffer zones	6	A & L	+++	++	+++	++
Restricted access of livestock to waterway	6	L	+++	++	+++	++
Reduced watercourse maintenance	6,7	A & L	++	++	++	++
Settling ponds	6,7	A & L	+++	++	+++	++

*Treatment train order: 1=reduce inputs,2=detain water,3=prevent soil dislodgement,4=prevent runoff,5=field margin,6=stream margin, 7=in stream

Table 21. "Scotification" of BMPs and possible approaches to assessing suitability criteria.

In field measures

Control Method	**"Scotification" issues**	**Approach to assessing suitability criteria;**
Over-winter ground cover (excluding stubble)	▪ Late establishment and cold soils leads to poor ground cover (esp in NE) and potential pollution swapping. ▪ Wet soils and poor structure lead to difficulties with cover crop incorporation in spring (esp. on glacial till soils with wetness limitation)	▪ LCA 3.2 or better ▪ Identify median spring cereal harvest date ▪ Identify Autumn/winter degree days minimum ▪ Adequate work days in Jan/Feb. ▪ Soil suitability for cover crop growth and incorporation
Strip cropping	▪ Unlikely to be widely applicable due to shape of contours and may end up with concentrated flows. ▪ Better to aim at grassing of zones of concentration within fields	▪ Slope <7° ▪ Develop criterion based on slope form
Stocking rates	▪ Runoff in summer months is common eg in Ayrshire (Vinten et al, 2004) ▪ Controlled management of track runoff desirable	▪ Avoidance of intensive stocking on steeply sloping fields (>15°) and convex slopes
Feed composition	▪ No Scottish element	▪
Manure application rate	▪ Sloping fields common	▪ Avoidance of slurry spreading on high risk slopes
Manure treatment	▪ Composting completely impractical in slurry based systems, without importing straw. Not unique to Scotland	▪ Highlight composting potential in parishes with more than a given specified level of cereals (straw available)

Control Method	**"Scotification" issues**	**Suitability criteria;**
Closed spreading period	▪ NVZ closed periods for Scotland differ from E and W. ▪ Need to achieve cross compliance with BW regulations	▪ Overlay NVZ and E.coli vulnerability maps (NIRAMS I and II) to identify areas where pollutant swapping an issue
Soil incorporation of manure	▪ Injection requires dry soils; drainage + backfill not desirable more than undesirable – not recommended if drains ▪ Therefore plough?	▪ Freely drained soils or imperfect drainage with no backfill on drains
Cultivation for soil stability	▪ Tied ridging not suitable where most of runoff is post harvest ▪ autumn cultivation after row crops is more difficult in Scotland. ▪ Many soils suited to subsoiling	▪ Establish whether potatoes/veg are followed by autumn cultivation ▪ Avoid soils with high Groundwater tables ▪ Map of soil suitability for subsoiling ▪ Develop classification of soil suitability for subsoiling
Minimal cultivation	• Scottish soils are suitable (Ball, pers. Comm)	▪ Map of soil suitability ▪ Cross compliance : reduced runoff might mean more bypass flow
Contour management	▪ Unlikely to be widely applicable due to shape of contours, may end up with concentrated flows. ▪ Better to aim at grassing of zones of concentration within fields (esp natural stream courses that have been removed)	▪ Identify uniform slopes, <= 7° ▪ Doesn't work on steep slopes
Crop nutrient management planning	▪ Higher OM content ▪	▪ Identify Parish N surpluses from NIRAMS ▪ Target these for 4 Point Plan /Waste Management
Crop pesticide management planning	▪ Longer period required for degradation	▪ All farms using pesticides
Restricted spatial and temporal application of crop inputs	▪ No Scottish element	▪
Leaving stubble overwinter + spring sowing and early seedbeds	▪ Temperature/moisture constraints – expand?	▪ Comment – widely applicable and highly recommended?
Avoid the use of tramlines in winter	▪ Winter spraying needs ▪ early fertiliser	▪

Field Margin methods

Control Method	"Scotification" issues	Suitability criteria;
Barrier ditches	Overloading more likely due to higher rainfall; Relatively flat field margin needed	LCA 2s or 1
Vegetative barrier strips and fences	The balance of hydrological pathways from land to water courses differs, affecting the potential for buffers strips to mitigate against pollution. Doesn't work on steep slopes Or where flow is concentrated	Uniform slopes, <=7° Assessment of extent of flow concentration from landform needed
Reed-beds Constructed wetlands	No Scottish element Fewer rural Scottish households on mains sewerage	
Riparian buffer zones	There may be more existing riparian vegetation, esp afforestation. This is a subject for further research.	Uniform slopes, with LCA 3.1g or better (7°) Doesn't work on steep slopes
Restricted access of livestock to waterway	Drinking water supplies needed first in livestock areas	Availability of suitable alternative drinking water (surface water+header tank, or groundwater source)

Watercourse methods

Control Method	Scotification	Suitability criteria
Reduced watercourse maintenance	Greater risk of flooding Lower pressure on floodplains etc for housing	Risk of damage during flooding?
Constructed wetlands		Risk of damage during flooding?
Settling ponds		Risk of damage during flooding?

MONITORING PROTOCOLS FOR EVALUATION OF BMP FUNCTION

The experimental and theoretical work in this report highlight the problems of evaluation of BMPs in the field because of the following features:

- Event based response
- Infrequent events ($\approx$ 1-2 per year) control performance
- Multiple pollutants
- Lack of replication
- Shortage of control or benchmark measures
- Uncontrolled/unknown management
- Seasonality and annual variation in response
- Variation in response with season, time since implementation, climatic and weather inputs

Having spent a year chasing storm events around Scotland, I think the following principles are important to consider when planning a monitoring strategy:

1. Flow proportional sampling rather than spot sampling is necessary to characterise BMP response.
2. Triggering of sampling should be automated, or routine.
3. Assessment is necessary over all seasons and several years.
4. Farms and landscapes must be thoroughly investigated to ensure all potential pollutant sources and BMP bypass routes (eg field underdrainage, field surface drainage and steading drainage) have been identified.
5. Imprecision of data is of secondary importance – over significant events parameter values often vary by a factor of 10 or more. A stronger emphasis should be put on capturing multiple events,.
6. Inaccuracy of data due to sample deterioration during storage should be dealt with by using robust determinands such as TP and SS, rather than labile determinands such as FIO's and soluble P, where possible.
7. Upstream and downstream (or inlet and outlet) water quality both need to be quantified.
8. Relative, not absolute measures of pollution abatement should be used, where possible (ie find a control, either in time or space, against which to compare the BMP).
9. We think there may be reason to reassess the use of SRP and TP as the main P categories. It is soluble plus relatively fine, colloidal and suspended P which

- will be readily transported to waterbodies
- will be less efficiently captures by buffer strips and other BMPs
- will contribute most to eutrophication.

We are discussing with SEPA a recommendation of using a <20 μm fraction for both SS and TP analysis, on top of the standard <1mm SS and TP done by SEPA, for event based samples.

We believe that a cost effective approach to sampling would be to use ISCO or equivalent samplers, equipped with AV/FM modules, linked to Flow Integration software, so that flow proportional sampling occurs continuously. Sampling should be into multiple bottles, averaging say 12-24 per week, which can be bulked if no events of interest occur. Sites should be serviced once per week. If flow data suggest an event analysis is warranted, then the individual samples can be analysed. If not, then the samples can be bulked to give data for analysis of baseline seasonal behaviour, or rejected. For assessment of sediment control, TPEC, DOC, SS are not subject to major sample deterioration and should be the determinands of choice. As a proxy for FIOs, DOC, and ammonium can be useful, where samples are stored more than two days, a protocol for correction of die-off of FIOs could be developed using first order inactivation kinetics, or events older than two days are discarded. Samples from events which occurred in the 2-3 days prior to a weekly visit would require only small corrections. Although this routine approach would mean more field visits through the year, there would be a considerable saving in out of hours work, in uncertainty as to laboratory capability and availability, and bulked samples taken regularly would allow seasonal loadings to be evaluated, which is not possible with short term campaigns.

One risk is that some farm installed BMPs are not being effectively utilized or maintained by some farmers and therefore the monitoring data could underestimate the effectiveness of properly utilized and maintained measures. Therefore detailed knowledge on the implementation of the BMP on the ground is required including any unique site factors impacting on the effect of the BMP and, just as important, the ability to assess whether the current management of the installed remediation measure by the relevant farmer is appropriate and as intended by the design. Farm audits will therefore need to be carried out. The part of the farm audited will depend on the BMP and may include any of the following:

- An assessment of whether the new measures are being used correctly and effectively by the farmers.
- New waste systems as a result of the BMP currently in place for slurry, FYM and dirty water, principally covering the effects and the farmers operation and maintenance of the new bmps.
- Farmyard drainage audit as a result of the new BMPs .
- Whether a RAMS, Pesticide, Erosion, Farm nutrient budget, FWMP or fertiliser plans are being used and adhered to.
- An assessment of pollution pathways from fields, tracks, drains, buffer strips, cow tracks, culverted crossings and installed water supply areas etc.

DEVELOPMENT AND APPLICATION OF A RULE-BASED GIS APPROACH TO DETERMINE THE LIKELY EXTENT AND LOCATION OF THE DEPLOYMENT OF BMPS IN SCOTLAND.

Introduction

As indicated in previous chapters, there are a range of BMPs which can be deployed to help control or minimise the effects of diffuse pollution from agriculture on water courses. Some of the diffuse pollutants are generic in nature and are common to a number of agricultural land uses, for example the leaching of nitrate to water, although the source of the nitrate may differ between cropping and livestock systems. Other pollutants are more specific to certain farming activities such as the transfer of soil sediment and attached particulate phosphorus being associated with arable cropping whereas the transfer of FIOs is explicitly linked with animal manures and excreta.

In Scotland, different farming systems have evolved largely as a response to the constraints that the biophysical environment imposes on plant growth and the timing of land management operations such as ploughing, harvesting and grazing. The spatial pattern of farming systems in Scotland is strongly regionalised, reflecting the spatial pattern of climate, soil, topography and interactions between them across the country. Technological advances, plant breeding and distortions caused by economic incentives have created some deviations in the 'natural' pattern of farming systems, but they are relatively minor.

The development of a spatial framework, based on land use, within which an appraisal of the relative importance of different BMPs might be judged, would be a valuable tool for two main reasons. Firstly, it could be used to identify the potential for the deployment of different BMPs, or a suite of BMPs, in different parts of the country. This would assist the development of a targeted approach to BMPs and to help identify priorities for their implementation. Secondly, it could provide an objective framework for a monitoring strategy to test the effectiveness of different BMPs.

Data

All Scottish farmers are obliged to fill in the Agricultural and Horticultural Census form in June every year. This is often referred to as the 'June Census'. A wide range of information is required including the total area of the holding, the areas of different crops (including different ages and uses of grass) and the areas of rough grazing and woodland. In a similar way, the numbers of different livestock types (dairy and beef cattle, sheep, pigs, poultry, goats and other livestock are also detailed. These data are then aggregated up to the parish level, of which there are 891 in Scotland. The cropping and animal data for each parish therefore provides a good indication of the predominant farming system within it. This allows each parish to be classified into a predominant farming type, based on the application of a series of decision rules.

Methodology

A series of decision rules were compiled and these act as a series of 'sieves' on the Farm Census Data (Table 22). If a parish fulfils the first condition, then it is excluded from the subsequent sieves and so on, until all the parishes have been classified into one, and only one, of the farming types. In essence, the rules have been devised to identify an increasing intensification of land use practice; the first condition identifies those parishes where extensive grazing predominates compared to the last one that identifies where both arable and spring sowing are the predominant activities.

Table 22. Decision rules to derive predominant farming system within Scottish parishes.

Condition			Description of farming type
Does parish have > 70% rough grazing? (**RG on map)**	——	YES	Predominantly sheep, some cattle enterprises.
NO ↓			
Does parish have between 30 and 70% rough grazing **(RG +PG)**	——	YES	Predominantly upland cattle farming, some sheep
NO ↓			
Does grass (> 5 years old) exceed grass < 5 years old + crops? **(G)**	——	YES	Predominantly cattle, dairying in some specific areas
NO ↓			
Does parish have grass (< 5 years old) > 40% of crops + grass (<5 years old)? **(YG)**	——	YES	Mixed farming, grassland based
NO ↓			
Does parish have spring crops > 75% of crops? **(SC)**	——	YES	Arable farming, predominantly spring sown
NO ↓			
Arable farming, > 25% winter sown. (**WC)**			

Results

The classification of the 891 parishes into the six broad farming types is shown in Table 2. There are no data for three of the urban parishes.

Table 23 Areas of different agricultural land use types across Scotland (after methodology in Table1)

Predominant agricultural cover types	Number of parishes	Area (square kilometres)
Rough grazing	191	31014
Rough grazing + permanent pasture	180	8624
Grassland – permanent pasture predominant	152	4450
Mixed – cropping, ley and permanent pasture	40	1028
Spring sowing predominant	105	2638
Spring and winter sowing predominant	220	4814
TOTAL	888	52568

Table 23 illustrates that much of the agriculture in Scotland is livestock based and of an extensive nature. Although difficult to compare precisely, these figures compare well with the published SEERAD data which shows that:

- Approximately two thirds of the agricultural land of Scotland is rough grazing
- Of the remaining third, half is permanent pasture
- Of the remaining area (approximately one sixth), there is a 2:1 crops:rotational grassland split.

These proportions are remarkably robust over a number of years, an indication that the farming pattern in Scotland is well established.

Table 23 indicates the predominant agricultural land uses within each class, but in reality they all contain smaller proportions of the other land uses. Table 24 indicates the 'average land use' across all the parishes within each land use category in Table 24 and Table 25 provides a more detailed picture of the proportions of different crops in the two most intensively managed categories.

The trends illustrated in Table 24 make, at a general level, intuitive sense. The rough grazing and cropping areas are almost mirror images of each other – as the proportion occupied by the one increases, the other decreases. Similarly, the proportions under improved pasture are highest in the 'middle ground' between the extensively managed rough grazing and intensively managed arable land.

The main differences between the two categories in Table 25 are not surprisingly, the balance between spring and autumn-sown crops. Table 24 also indicates that the proportion of land which is not cropped is lower in the parishes with > 25% winter sowing.

It must be recognised that these summaries hide some internal variation between the parishes in each land use category. This variation is highest in the 30-70% rough grazing category where in a small number of parishes for example, the areas under cropping and rough grazing are relatively large (both over 30% of the parish) and almost equal in size. A scoping study such as this cannot cater for anomalies like these (which are governed by the physical nature of these parishes) but they are worth bearing in mind.

The distribution of the different cover types in Table 23 is illustrated in Figure 31. By far the most extensive category, rough grazing predominates in the Highlands, Southern Uplands, the Western Isles and Shetland. The next category, rough grazing + permanent pasture, occupies much of the rest of what is commonly thought of as 'upland' Scotland and in most locations immediately abuts the area of rough grazing. Similarly the grassland – permanent pasture predominant category (Table 23) is often found adjacent to the previous category, suggesting a gradual intensification of farming activity 'down the hill' as management opportunities increase.

The mixed – cropping, ley and permanent pasture category is of relatively limited extent, being largely confined to NE Scotland where mixed farming is the traditional management system but which has reduced in size as farms became more specialist and cereal-orientated. Lastly, the two cropping categories are restricted entirely to the eastern side of the country with winter cereals, in general terms, increasing in importance towards the south.

This agricultural land use classification has been broken down into the four regions described above and adds some detail about the distribution of farming systems and relative importance of each within them (Table 26). There are some interesting contrasts between the regions, but it should be noted that all the BMPs or control methods in Table 20 are likely to have some role in every region. The main points to be drawn from Table 26 are:

- North-west. Predominantly extensive grazing, but with a small area of relatively intensive arable farming around the inner Moray Firth.
- North-east. A relatively even spread of farming types, with an increasing intensity of operation with decreasing altitude and increasing proximity to the sea. Much of the cultivated land, even at the higher altitudes is ploughed relatively regularly, reflecting the interaction between the relatively dry climate of the area and predominantly freely drained coarse-textured soils.

Table 24. Scottish parish classification; mean agricultural land use balance within different parish types

Criterion	Farm type	%Total Crop	%Rough Grazing	%Grass <5yrs	%Grass >5yrs
> 70% rough grazing	Sheep predominant	0.7	92.0	1.4	5.8
30-70% rough grazing	Sheep + cattle	10.1	50.9	9.2	29.8
Grass > 5 Years old < rotational grass +crop	cattle	10.5	15.0	15.6	58.9
Grass <5 years old > 40% of rotational grass + crop	mixed	30.0	13.4	27.2	29.4
Spring sown crops > 75% of crops	arable, >25% spring-sown	58.4	9.2	15.6	16.7
Winter sown crops > 25% of crops	arable, < 75% spring-sown	68.9	5.4	9.8	15.9

Table 25. Breakdown of cropping within the arable sector

	%Set Aside	%Winter Cereals	%Spring Cereal	%Potatoes	%Other crop	%Vegetables	%Soft Fruit
Spring sown crops > 75% of crops	11.5	16.3	60.3	4.9	4.2	1.8	0.5
Winter sown crops > 25% of crops	10.6	40.4	37.5	5.5	3.2	2.0	0.3

- South-west. Extensive grazing, particularly throughout Argyll, and improved grassland, particularly in Ayrshire, Lanarkshire and the SW predominate. Cropping is limited to the far north-east of the region. The lack of cropping land reflects the soil management difficulties caused by the interaction between the relatively wet climate and moisture retentive soils.
- South-East. Agricultural land use is either relatively extensive grazing (rough grazing + permanent pasture) or intensive arable. The area of improved grassland is relatively small.

Table 26. Areas (square kilometres) of agricultural land use types in the Scottish Regions

Predominant agricultural land use types	Region			
	North-west	North-east	South-west	South-east
Rough grazing	16926	1543	7887	4657
Rough grazing + permanent pasture	1663	864	3533	2555
Grassland – permanent pasture predominant	571	64	3318	495
Mixed – cropping, ley and permanent pasture	86	693	139	110
Spring sowing predominant	217	1788	3	630
Spring and winter sowing predominant	82	1224	94	3414
TOTAL	19555	6176	14881	11861

Best Management Practice Scoping – Substitution of autumn sown cereals with spring sown

One of the principal mechanisms that causes diffuse pollution is the direct transfer of soil to water by physical processes. The sediment itself is not a pollutant *per se*, but it can cause siltation, but it can also carry large amounts of particulate phosphorus which contributes to eutrophication of surface waters. Soil movement occurs primarily when the soil does not have full crop cover, so the land use types with large cropping percentages are most at risk. The categories ‘spring and winter sowing predominant’ (68.9%), ‘spring sowing predominant’ (58.4%) and to a lesser extent ‘mixed – cropping, ley and permanent pasture’ (30%) are those where significant proportions of the land are bare for some period of the year and at increased risk of erosion.

Spring sowing also coincides with the driest period of the year in Scotland (Meteorological Office 1981, Met Office web site), particularly in eastern Scotland. The period from February to April inclusive contributes approximately 20% of the area’s annual rainfall whereas in the autumn period (September to November) it is almost 30%. Ground preparation and sowing will be subject to fewer delaying rainfall events in spring and ground conditions will be progressively improving (the opposite is the case in autumn), reducing the risk of run-off. In addition, late sowing of autumn sown crops often leads to partial crop cover for a number of months over the winter period, whereas spring cereals achieve full ground cover much more rapidly.

Lilly et al (2002) have developed a classification that assesses the inherent geomorphological risk of soil erosion by overland flow. The factors used are slope, standard percentage runoff and soil surface texture. Different classifications have been developed for mineral soils and soils with organic surface horizons and each have been classified into three risk soil erosion risk classes; low, moderate and high. The classification does not take into account the frequency of heavy rainfall events nor management factors.

This dataset has been used to provide an assessment of the risk of soil erosion within the three agricultural land use categories where soil tillage is a common activity. The results are presented in Table 27. The main points are:

- As might be expected, most of the soils are mineral soils. The area of soils with organic surface horizons is likely to be the rough grazing component. It is interesting to note that the proportion of organic soils in each land use class display a similar trend to that of rough grazing in Table 24.
- In all three land use categories, the predominant risk class is moderate. Although clearly the risk is less than on the high risk land, there is a still an unquantified risk of soil movement, which largely depend on precise temporal factors such as the co-incidence of rainfall events with bare soil.
- The high risk area accounts for approximately 5% of the mineral soils. From the data, there is no way of knowing whether these areas are tilled every year, but these areas should not be ploughed frequently.

Table 27. Soil erosion classes in Agricultural Land Use Categories with tilled land

		Agricultural Land Use Category		
Soil Erosion Classification		'spring sowing predominant'	'spring and winter sowing predominant'	'mixed – cropping, ley and permanent pasture'
	Low	764	1553	232
Mineral soils	Moderate	2422	3928	1045
	High	138	266	88
	Low	31	50	11
Organic soils	Moderate	153	112	127
	High	64	66	70
Rock and scree		0	< 1	< 1
Unstable slopes		2	3	1
Built-up areas		196	275	29
TOTAL		**3770**	**6254**	**1603**

Erosion risk has been closely correlated with autumn sowing of cereals as the ground is left partially bare for a much longer period than with spring sown crops and over the wettest period of the year (Spiers and Frost 1985). In addition the soil has been prepared into a fine seed bed prior to sowing thereby increasing further the erosion risk. For this reason we have examined in more detail the relationship between winter sowing and erosion risk in each parish in these land use categories. The percentage of autumn sown cereals as a proportion of the total crops and grass area has been calculated and shown in Figure 32. There is a general increase in the area under autumn-sown cereals towards the south with the largest proportions in some of the parishes in East Lothian and Berwickshire. Most of the land however has less than 30% winter cereals and in NE Scotland large areas are below 10%.

The proportion of mineral soils assessed as having a moderate or high risk of soil erosion has been calculated in a similar way for each parish and presented in Figure 33. The predominance of the moderate and high risk classes is clear with only relatively small areas having more than 50% cover of soils with low erosion risk.

These two proportions have been multiplied together to produce a factor which essentially expresses the probability of winter cereals occurring on soils with a moderate or high risk of erosion (Figure 34); the higher the number, the greater the probability of coincidence of winter crops and moderate or high risk soils. Based on this analysis, the areas with the highest probability are in East Lothian, although they form a very small component of the whole area that has been analysed. Much of NE Scotland has a lower probability compared to the area from Strathmore southwards. This pattern would be expected given the cropping and soil erosion risk proportions displayed in Figures 32 and 33.

In those parishes where winter cereals are most commonly grown (Figure 32) and where the risk of soil erosion is relatively large (Figure 33), in most years, there will be ample opportunity to prepare the ground and sow spring barley without incurring a yield penalty through late sowing. In these areas, the field capacity period normally ends in early March (Meteorological Office 1981) and they also have the longest 'safe' accessibility period, without causing soil damage. This can be up to 240 days in the area to the east of Edinburgh. Indeed these are the most climatically favoured areas of Scotland from an agricultural production and flexibility perspective (Bibby et al 1991).

The conversion of land from winter cereals to spring cereals is estimated to reduce erosion risk by approximately 50%, but there is an associated economic cost to the farmer by so doing (Frost and Ramsay 1996). Despite higher costs, winter crops have higher yields and returns than spring sown cereals. Frost and Ramsey have indicated that a farm which was originally 50:50 spring:winter barley would stand to lose close to £100 per hectare/annum through conversion. These figures will need to be reviewed periodically given the uncertainty that will accrue from CAP reform and the relative profitability of different crops.

In 1999, there were 148,747 hectares of autumn sown crops sown in Scotland. If all of this was converted to spring cereals, the financial loss to Scottish agriculture would therefore be considerable. This however is very much a worse case scenario and considerable environmental benefits would still accrue if all land within say 50 metres of water courses were avoided for autumn sown crops. However, such a change would mean considerable disruption to farm management practices and in many circumstances might not be practically feasible.

Best Management Practice – efficiency of buffer strips

In this scoping study, it would be of value to know what the predominant pathways for pollutants in soils adjacent to water courses are within each of the different Agricultural Land Use Categories. By doing this, information on both the transport mechanism (based on an understanding of soil hydrological processes) and the potential pollutants associated with each farming activity (from the Land Use map) and which may be subject to transport can be linked.

To do this the Hydrology of Soil Types (HOST) classification (Boorman et al 1995) was used. Essentially this is a reclassification of original soil data using attributes relevant to water movement within and through the soil and based on 11 conceptual hydrological response models which describe the dominant pathways of water movement through the soil and substrate. The HOST classification has 29 classes although not all are present in Scotland. For this project each of these was aggregated into ‘wet’ and ‘dry’ categories and the area of each within 10 and 5 metre buffer strips along water courses was calculated.

Figure 35 illustrates the proportion of soils within the 10 metre buffers that have a drainage impedence (classified as wet) and therefore likely to have field drains that will reduce their effectiveness in preventing soluble or colloidal pollutants (e.g. nitrates and faecal coliforms) from reaching water courses. Areas with predominantly grassland farming have the highest proportion of ‘wet’ potential buffer strips and include much of Ayrshire and Central Scotland. Some areas with high proportions of winter cereals also have relatively high proportions, for example the Merse of Berwick, Lothians, Fife, parts of Strathmore and the Peterhead area. This could be indicative, at least in part, of the difficulties associated with spring sowing on soils with a drainage impedance where access can be difficult in spring. However, the increased profitability of at least some of the autumn-sown crops over spring-sown is also likely to be a factor promoting autumn sowing in these areas.

If all the water courses throughout the area displayed on Figure 35 had 5 or 10 metre buffer strips applied (a very unlikely scenario), then the ‘loss’ of land or ‘land take’ would be approximately 1.3 and 2.7 % respectively. However this does not equate directly with loss from agriculture as many of these areas will be in scrub or woodland already.

The maximum slope suggested for buffer strips to be effective is seven degrees (Table 21) and analysis of the slope range within the 10 metre buffers indicates that virtually all the land within this buffer width is less than this value (the rough grazing category has not been assessed). Indeed, over 99% of the slopes are three degrees or less. Although this result is intuitively correct – most streams and other watercourses within improved agricultural landscapes are on gentle slopes – it must be treated with a little caution. Due to the coarse resolution (a 50 metre grid) of the digital terrain model (DTM) from which the slope map was generated, many short, very steep slopes close to watercourses will not be identified and it is probable that the analyses presented here understates the area of land with slopes > 7 degrees within the 10 metre buffers.

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5. CONCLUSIONS AND RECOMMENDATIONS

1. Among the most effective field BMPs for Scottish conditions are nutrient budgeting, leaving winter stubble after cereal crops, cultivation after row crops, headland subsoiling, use of buffer strips, restricting water access by livestock, and constructed wetlands. Field margin and watercourse BMPs are not recommended as alternatives to in field BMPs, but may be important as additional options within the treatment train.

2. Site suitability for buffer strips should take into account of:
   - the slope (<7% is preferred);
   - buffer vegetation management (active management to promote a diversity of habitat is required to improve biodiversity);
   - likely flow concentration effects (where this is serious, impoundment measures or in-field measures should be undertaken in addition to, buffer strips).

3. Most sediment trapping occurs in the first 5-10m of buffer strips, so for inorganic sediment capture, a buffer width of > 10m is not warranted for sediment control, though may be of benefit for other reasons. In grassland areas, larger widths of livestock exclusion may be warranted for promotion of diverse vegetation and food resources for birds.

4. Biodiversity of the Green's Burn buffer would be enhanced by more active management (eg grazing of the buffer, leaving bare soil areas, more frequent cutting). However this may impair effectiveness as a pollution control measure. The increased cutting would also enhance accessibility to the public, which is poor in some seasons at present, because of the height of the vegetation.

5. Limited data suggest that the Greens Burn buffer strip may reduce total P and suspended solids loading by up to 50%, but the efficiency of removal for the colloidal fraction is lower.

6. In monitored reaches of the Cessnock catchment, Ayrshire, limited data suggest that without dealing with steading pollution issues, it may be difficult to observe the benefits of fencing and stock exclusion, as steading issues currently dominate faecal coliform loads.

7. Field plot data suggest that fencing and stock exclusion are probably effective mainly by preventing stock access rather than by the buffer strip actions of filtration, infiltration and sedimentation. We have observed that where Rural Stewardship schemes have been used to implement riparian fencing in the Cessnock catchment, there are often gaps left to allow stock access to water. Unless these access points are well managed, or off stream drinking water supply is provided, this is unlikely to lead to an improvement in the pollution loads, although it may alter the biodiversity value of the riparian zone.

8. Provision of drinking water supply off stream may be a more cost-effective pollution control measure than fencing and needs to be investigated separately. It is not known whether this will significantly reduce stock movement on the banks and into watercourses and further studies are recommended.

9. To ensure BMP effectiveness for pollution control, structural bypass routes (steading sump drains, surface runoff drains from fields) need to be taken into account and dealt with during audit.

10. Monitoring of BMPs needs to be event based, with an emphasis on robust, low degradability determinands such as suspended solids and total nutrient content. Continuous sampling stations, regularly serviced, and triggered on a flow proportional basis, are preferred. Inlet/upstream and outlet/downstream should be monitored.

11. The suitability of Scotland's farmed landscape for BMPs needs to take into account regional factors, especially:
    - the undulating landforms in many places, which promotes flow concentration,
    - the relatively wet climatic and soil conditions in many areas, which reduce the potential for the infiltration function of buffer strips, increase the frequency of sites with underdrainage bypass routes for leached pollutants and lead to greater risks with livestock and livestock waste management,
    - the occurrence of many sensitive or oligotrophic water bodies.

12. Although the full range of BMPs could be employed in the different regions of Scotland, there is a marked regional pattern to the distribution of different farming systems based largely on biophysical constraints. This pattern has been established over a large number of decades; technological advances, plant breeding and distortions caused by economic incentives have created some deviations in the 'natural' pattern of farming systems, but they are relatively minor. For this reason, different BMPs will predominate over others in different parts of the country.

13. The GIS scoping study proved of value:
    - to screen the country to identify areas where different BMPs might be best employed,
    - to identify areas where demonstration sites could be established,
    - to act as a framework to establish a monitoring scheme to test their effectiveness
    - to test scenarios, for example to indicate the potential scale and impact of a switch to spring sowing. Nearly 70% of land classified as 'spring and winter sowing dominant' occurs on land with a medium or high risk of soil erosion. The areas where this factor is highest are along the east side of Scotland, from the Laurencekirk area southwards with the highest probability occurring in a number of parishes in East Lothian.

14. The use of GIS could be usefully extended to:
    - identify the potential of a number of other BMPs identified on Table 21, specifically within a Scottish context.

- quantify the potential contribution that buffer strips and other potential BMPs might make to sediment load reduction in areas identified by the GIS as benefiting from intervention.
- to investigate the relationship between the range of pollutants addressed in the Screening Tool project (Provision of a Screening Tool to Identify and Characterise Diffuse Pollution Pressures: Phase II) within the different land use types identified here and the suite of BMPs that might be employed.

15. If all the watercourses in Scotland had 5 or 10 metre buffer strips applied (a very unlikely scenario), then the 'loss' of land from agricultural production would be approximately 1.3 and 2.7 %. Note that this does not equate with loss from agriculture as many of these areas will be in scrub or woodland already. The actual loss of land based on actual implementation is likely to be much less than this.

6. ACKNOWLEDGEMENTS

We thank SNIFFER for funding the work reported here. We acknowledge the contribution of SEPA's analytical laboratories (especially Malcolm Smith and Bill Craigs from SEPA Edinburgh, and David Milne from SEPA East Kilbride) and SEPA's field staff (especially Paul Dawson and Bryce Syme of SEPA East Kilbride).

We thank the farmers that co-operated in providing time for discussion, access and helpful information (Angus Bayne from Wester Gospetry, David and Cecil Foster at Langside).

We thank Jeanette MacDonald and Bryan D'Arcy of SEPA's Diffuse Pollution Initiative for their enthusiastic interest and constructive contribution to the direction and reporting of this project, and others of the Steering group (Alan Frost, Craig Campbell) for comments on a draft of this report.

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Figure 24. Water chemistry and microbiology, above and below Langside and Barneighthill, Cessnock catchment, during storm event 29-30 September 2003.

Figure 1. Example of sediment transport across Green's Burn buffer strip.

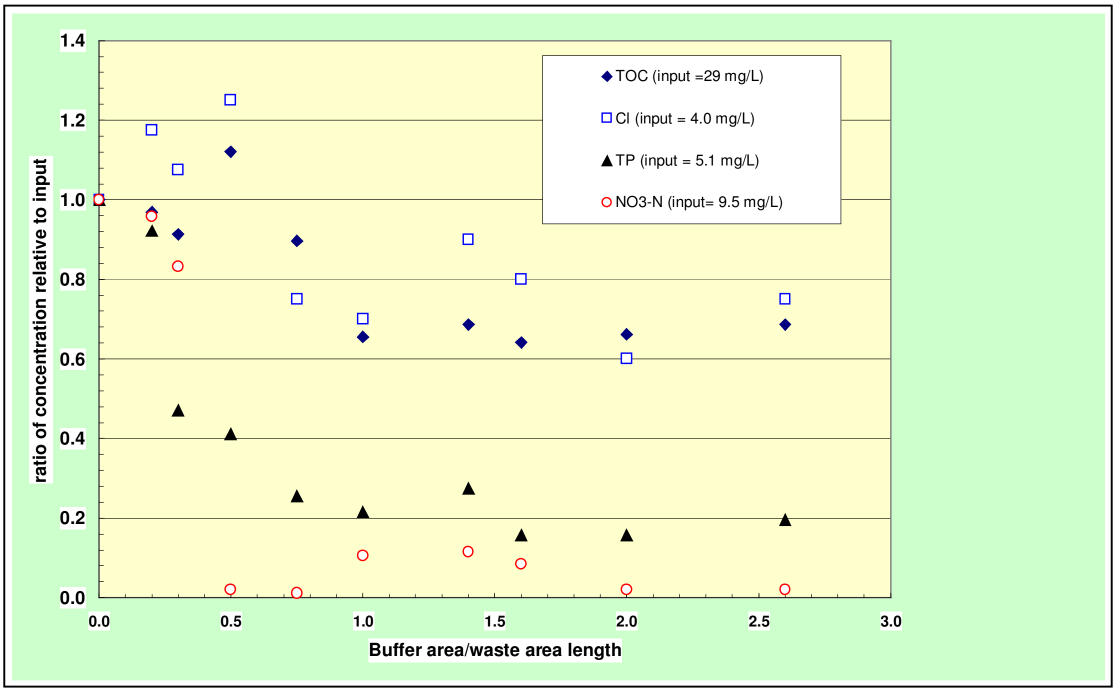


Figure 2. Removal of pollutants from runoff from poultry manure treated plots during transfer across a buffer strip.

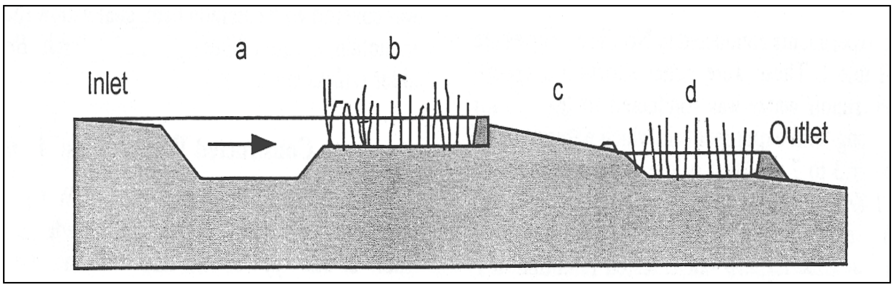


Figure 3. Section through a constructed wetland, comprising; sedimentation pond areas (a), vegetated filter strips (b) and overflow zone (c) and secondary collecting basin (d)(from Uusi-Kämppä *et al*., 2000).

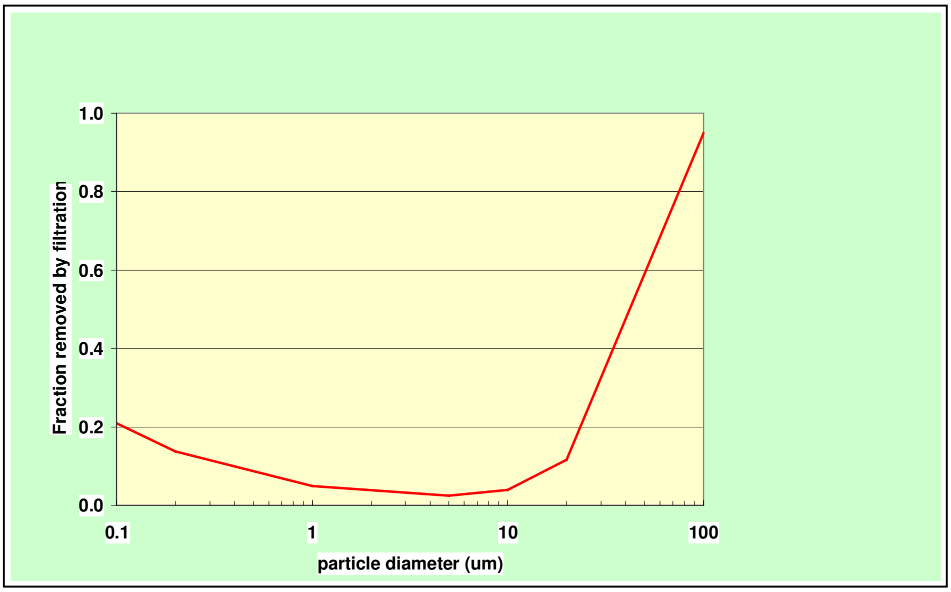


Figure 4. Effect of particle size on filter efficiency by interception and diffusion during flow through buffer (see Table 7 for conditions).

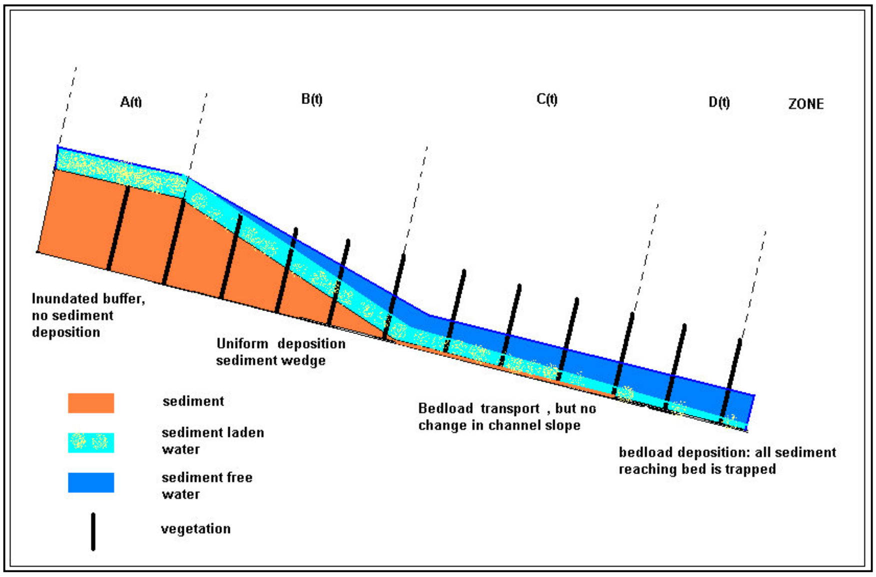
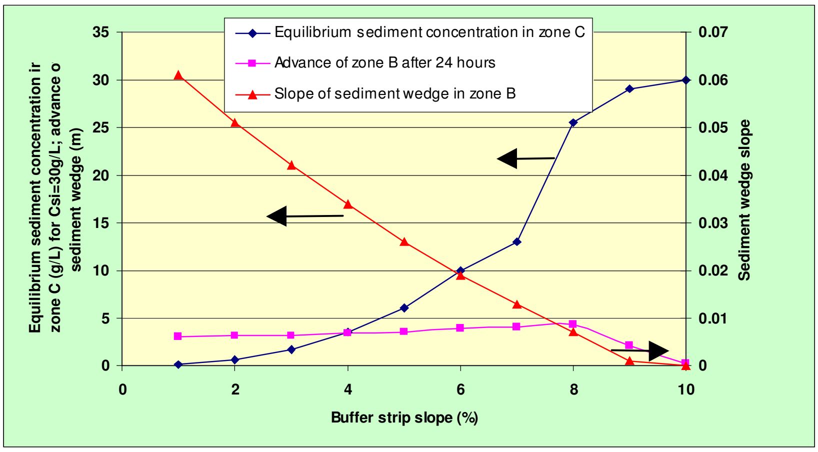


Figure 5. Functional zones within a buffer strip during sedimentation.

a)



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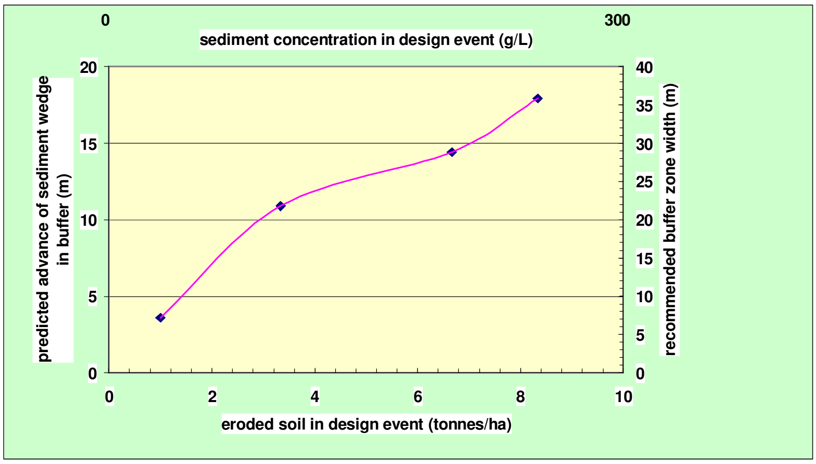

Figure 6. Predictions of sediment removal by buffer strip, according to model of Barfield et al. (1979) linked to flood frequency analysis for Scottish conditions.(a). Effect of buffer slope on sediment wedge slope, advance and equilibrium sediment concentration for an inlet concentration of 30 g/L and discharge of 1.25 x 10^{-4} m^3/m buffer/second. (b). Effect of sediment concentration on advance of sediment wedge in buffer. Same conditions as (a), slope = 5%.

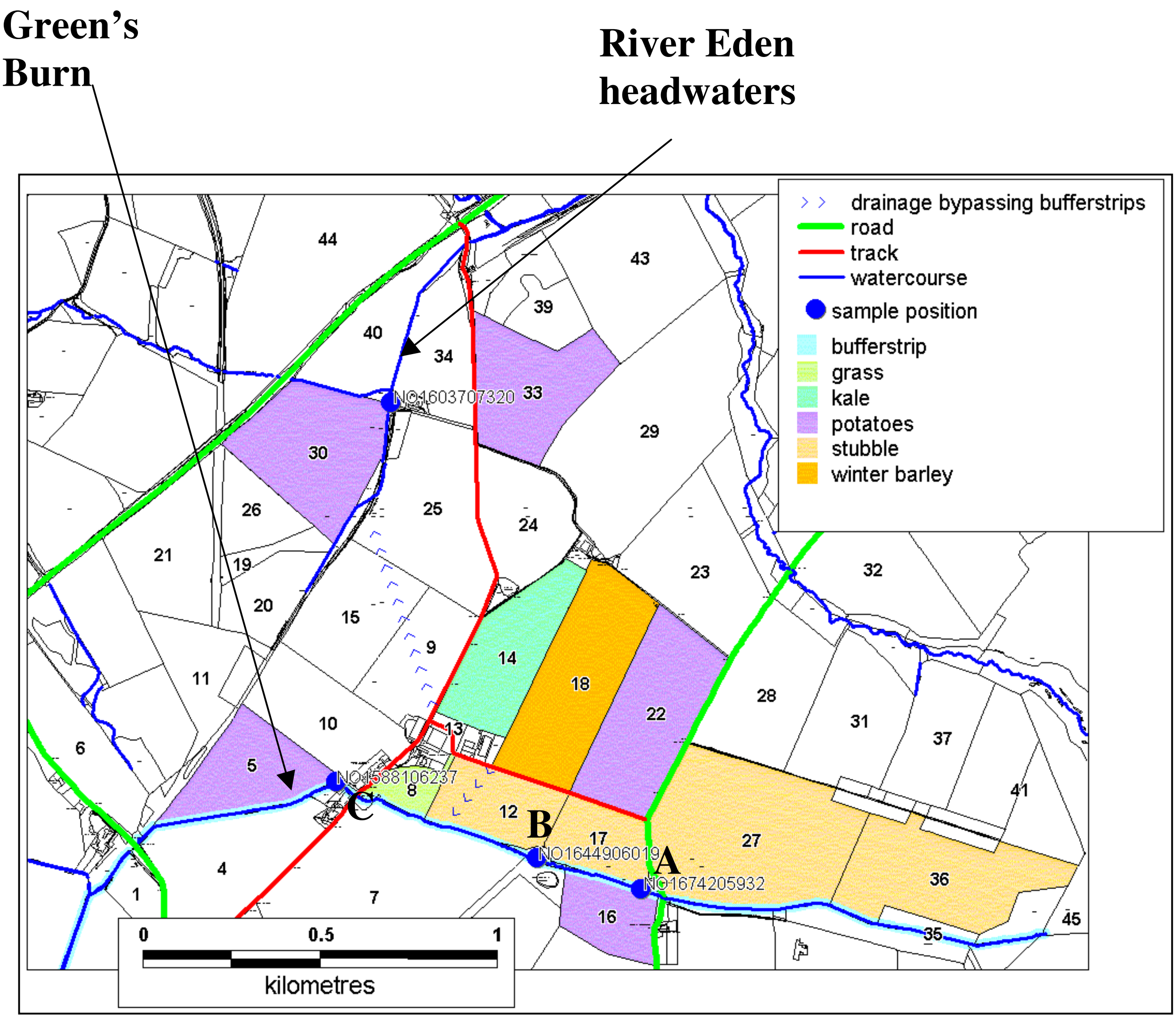

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Figure 8. Sedimentation at field margin, October 2002 and in Green's Burn buffer, February 2003.

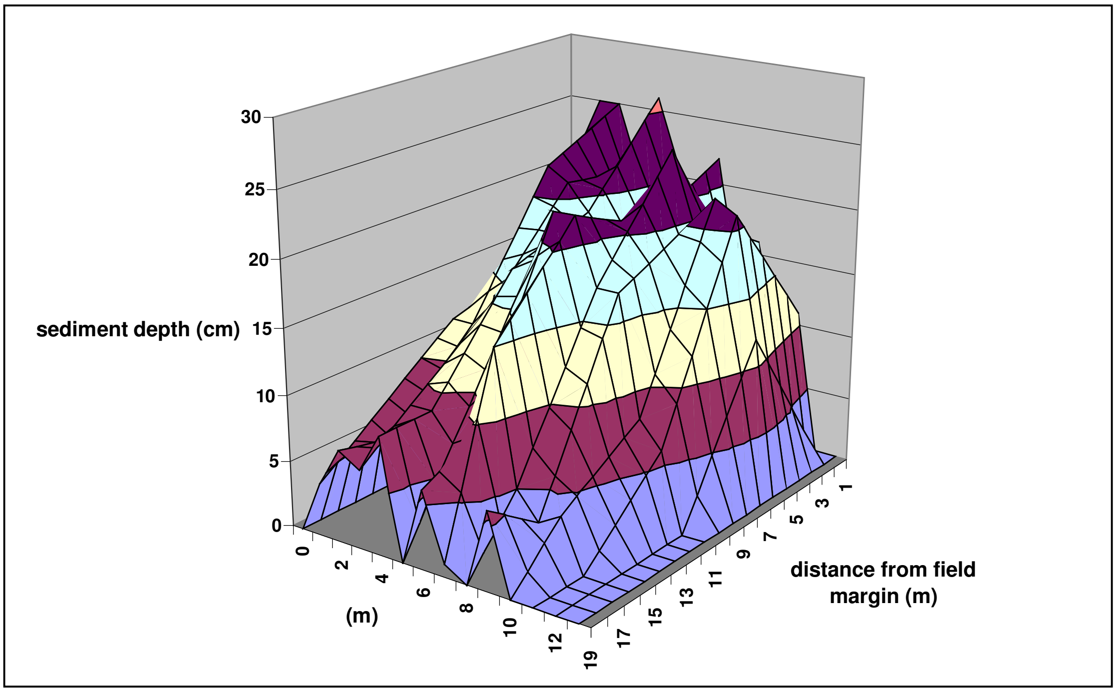
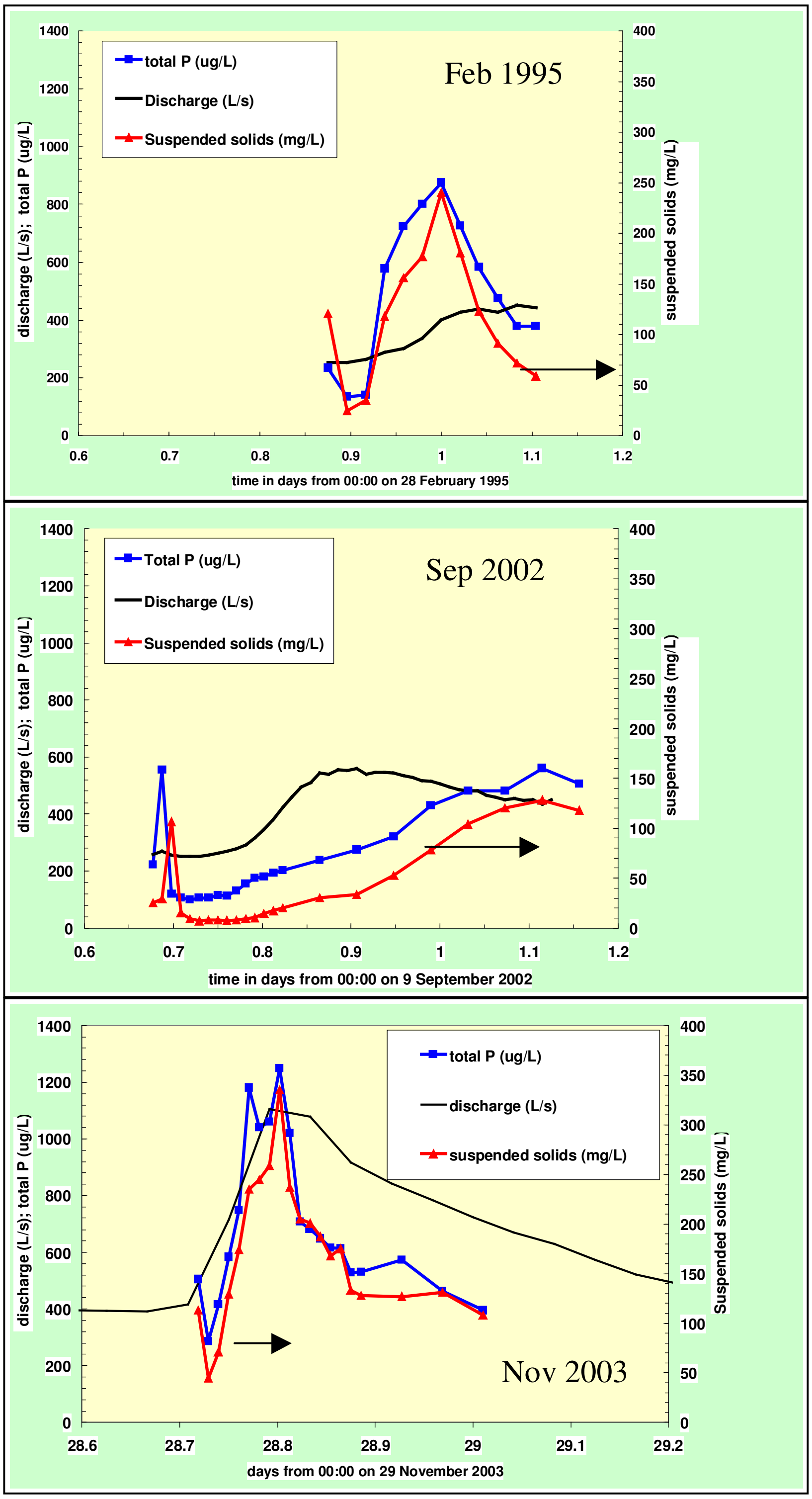


Figure 9. Profile of sediment wedge across Green's Burn buffer strip, February 2003.



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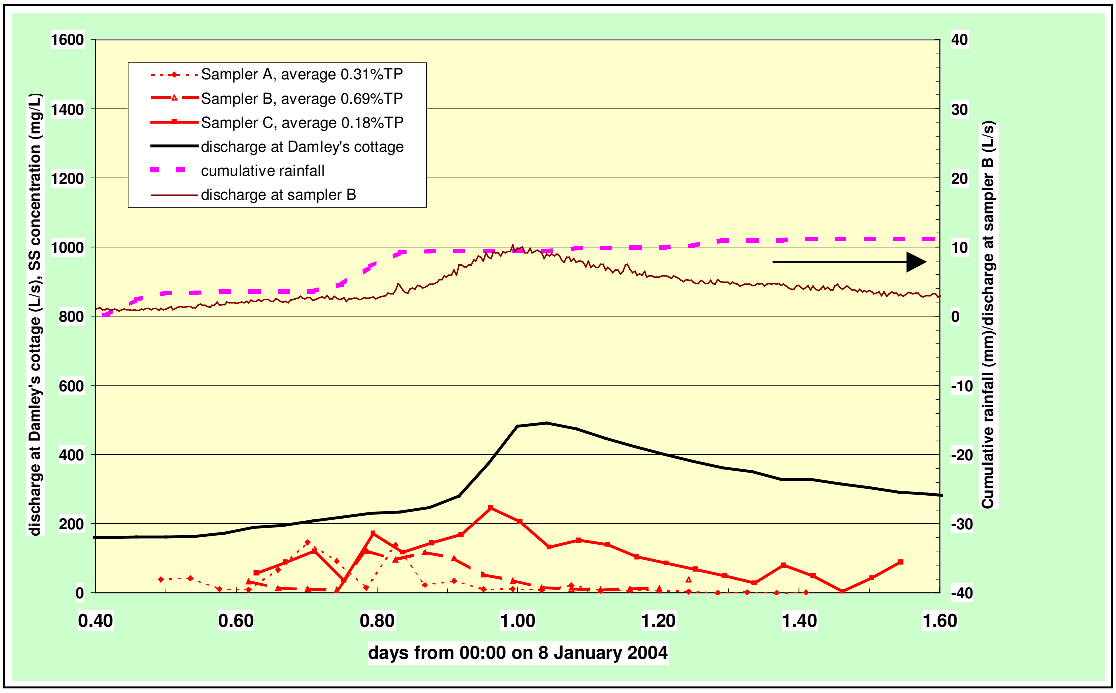
a)



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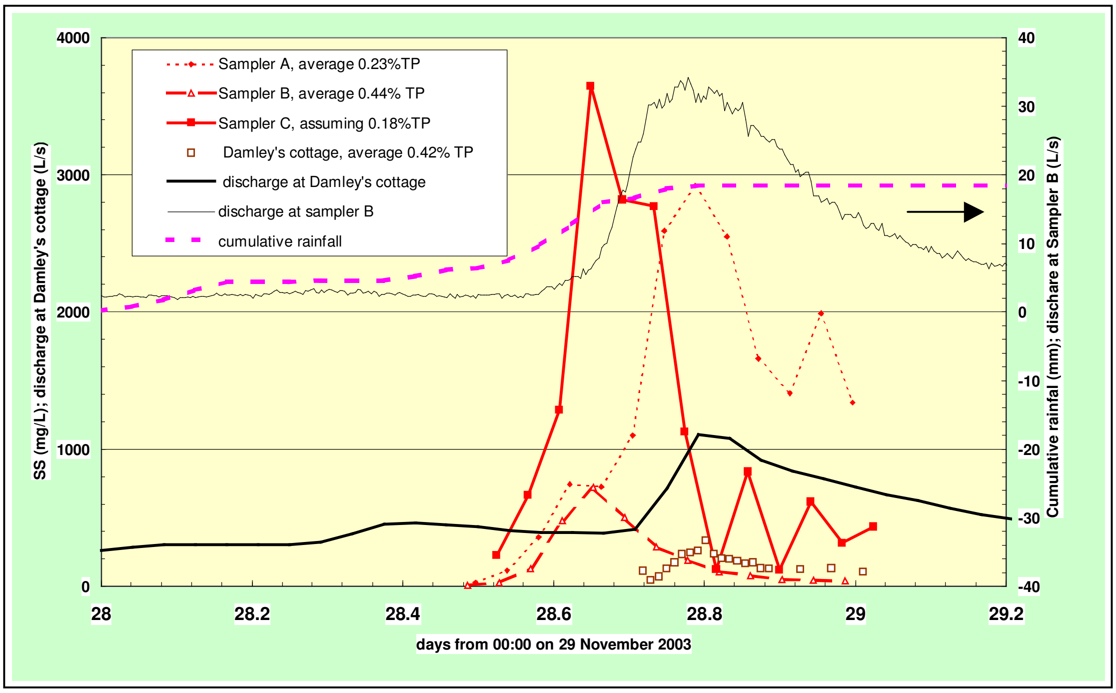


Figure 11. Comparison of storm events sampled at SAC samplers A, B and C. (a) 29 November 2003. (b) 8 January 2004. Suspended solids at three samplers and discharge at weir installed at sampler B are shown. Average Total P concentrations are also shown.

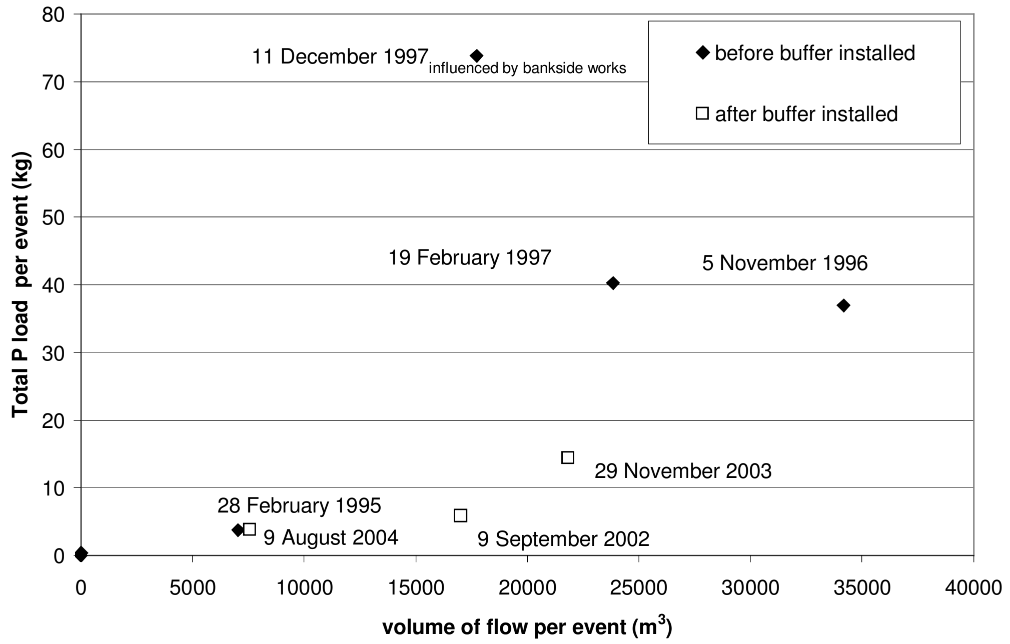
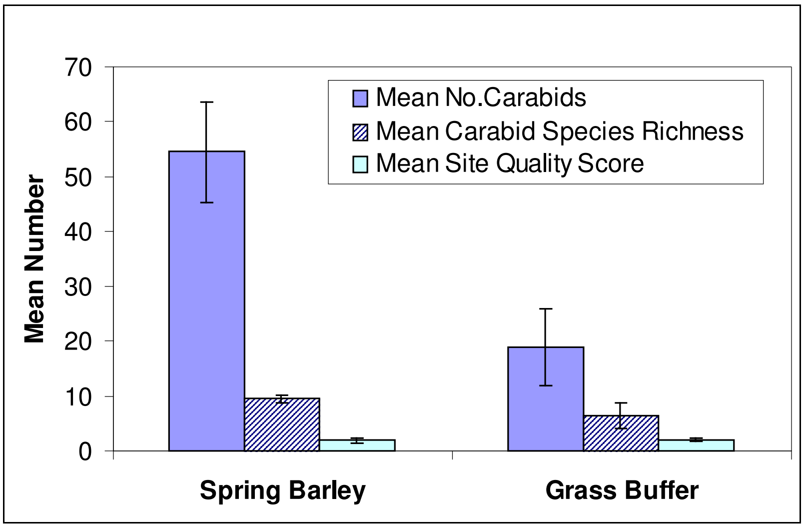


Figure 12. Comparison of estimated Total P losses from storm events measured at Damley's cottage, Green's Burn before and after buffer strip installation.

Figure 13. Bund with filter+low flow drain, plus high flow drain just below the top of the bund, design to slow down water discharge into buffer strip at 'hot spot' shown in Figure 8. Wester Gospetry. Both high and low flow drains discharge into a 20m wide trench, which discharges into the buffer strip.



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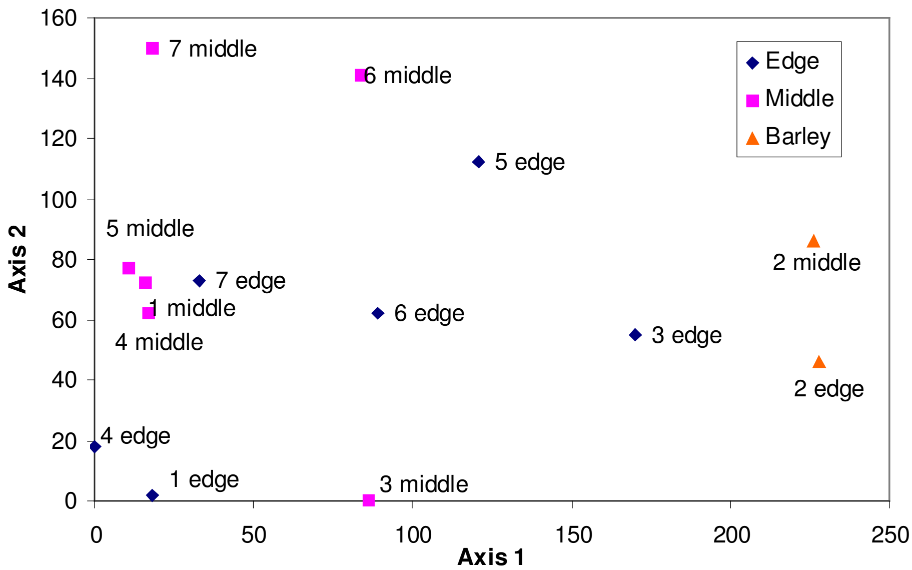

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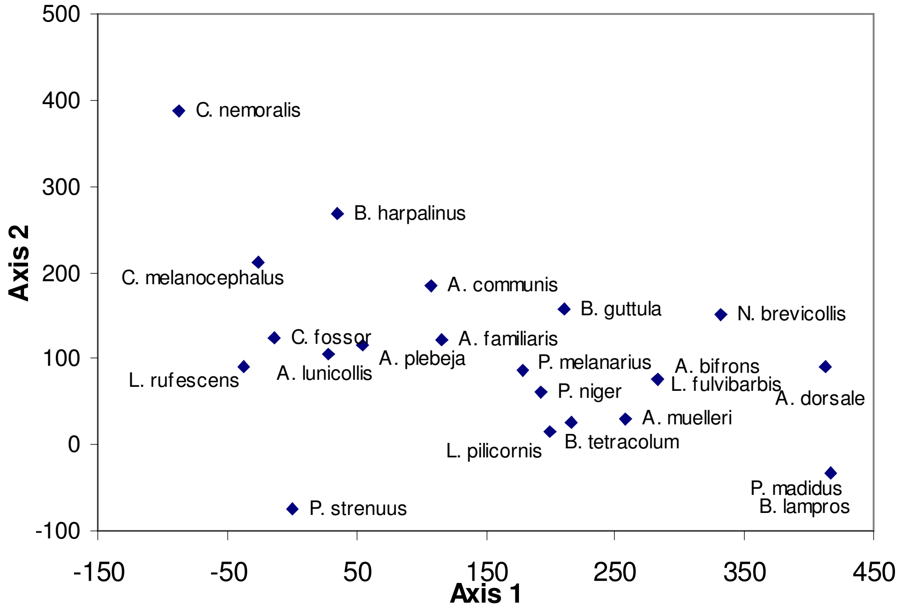

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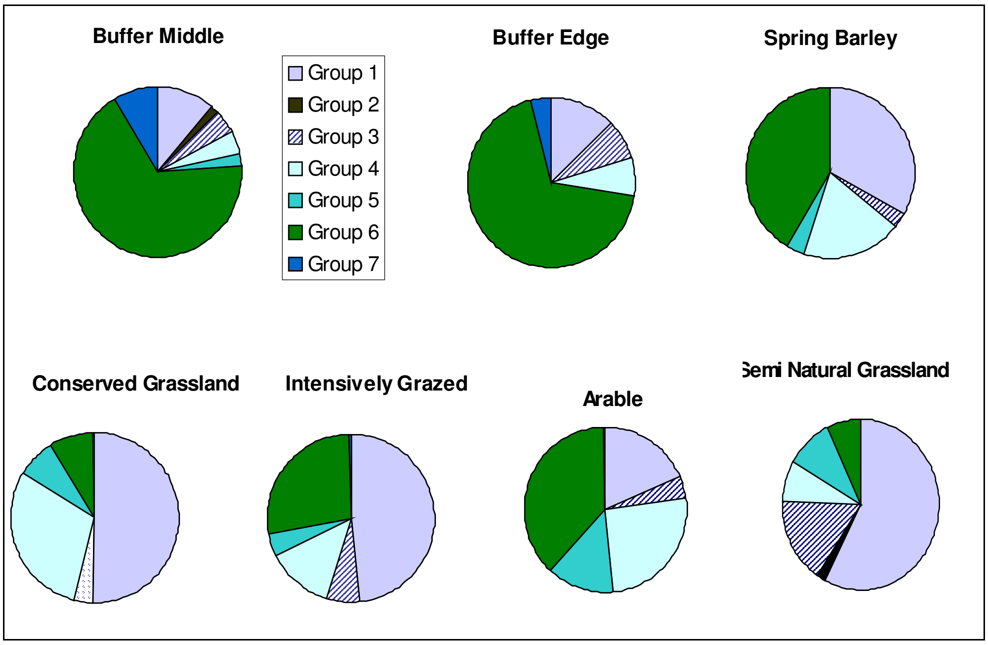

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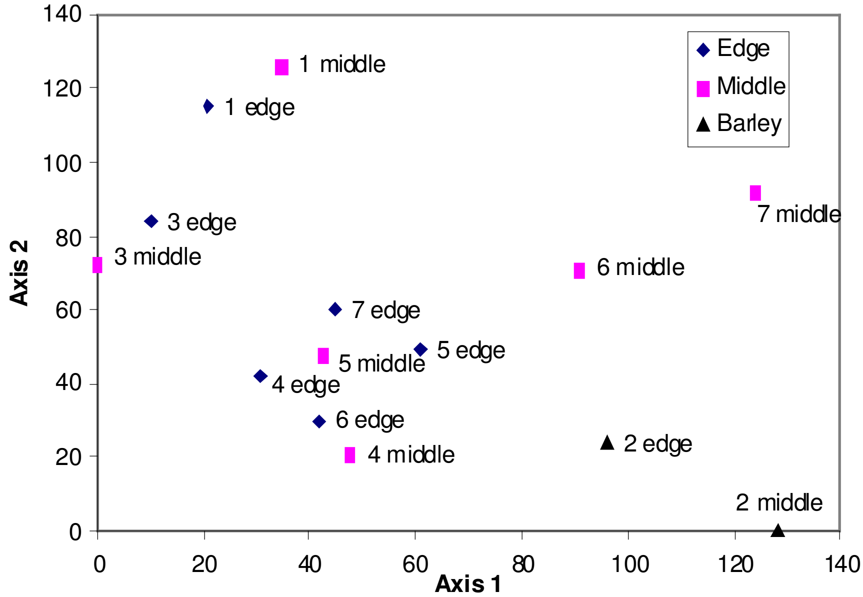

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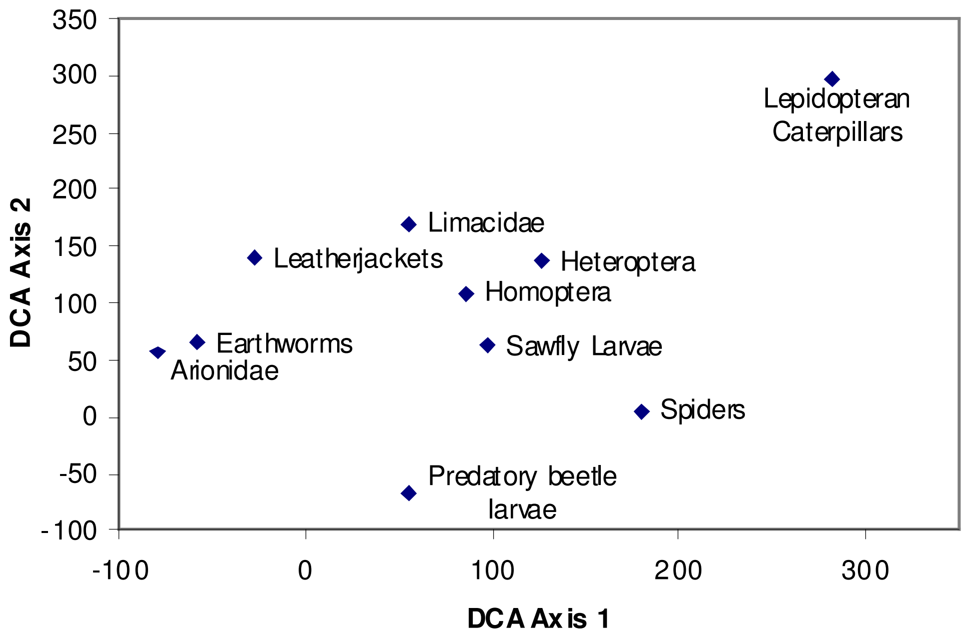

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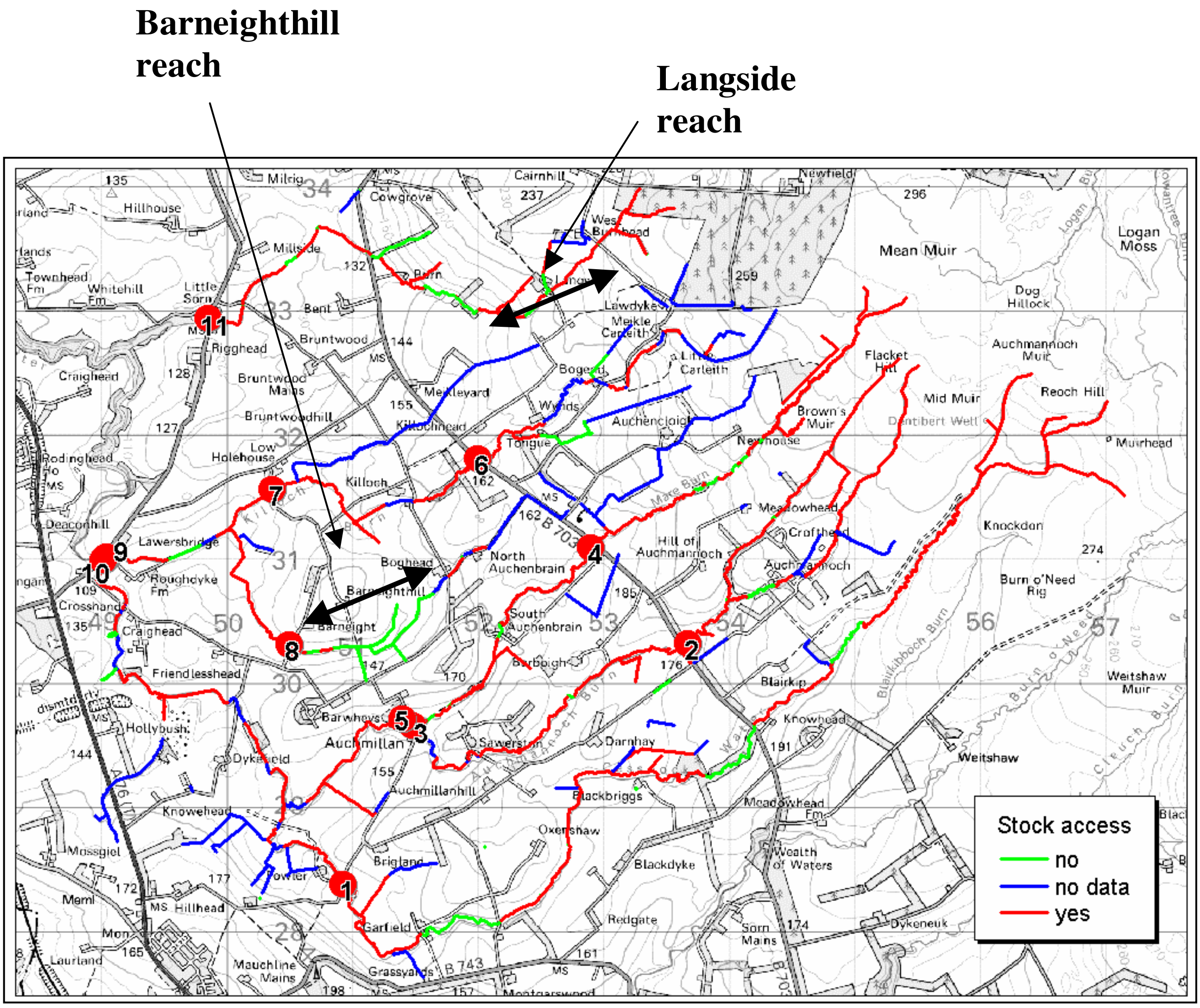

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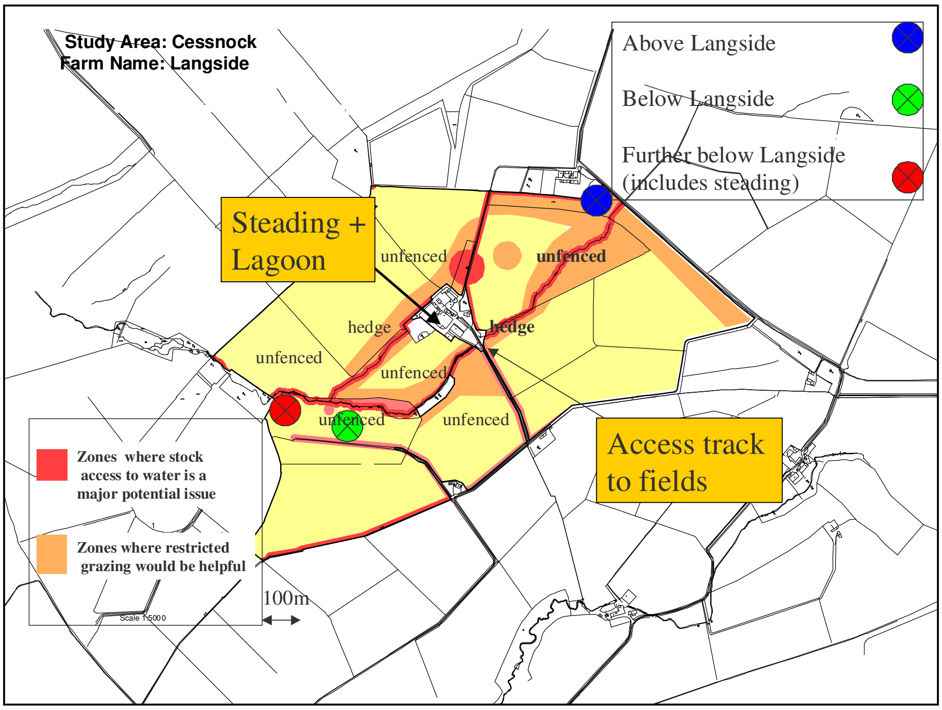


b)

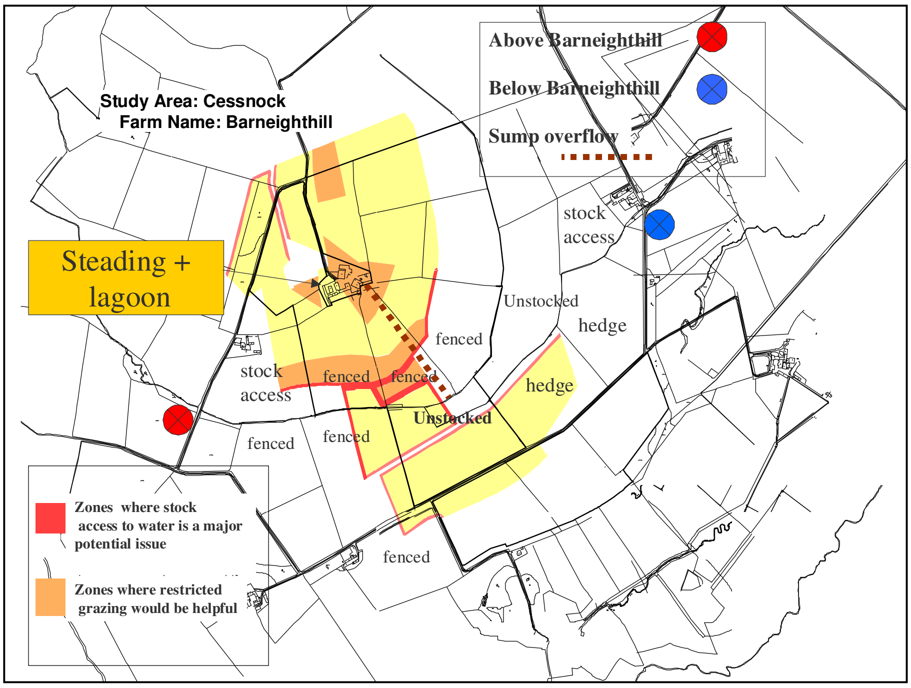


Figure 21. Details of reaches used for assessing the impact of field margin management. (a) Langside reach (high livestok access -unfenced). (b) Barneighthill reach (low livestock access – mainly fenced), showing sampling positions.

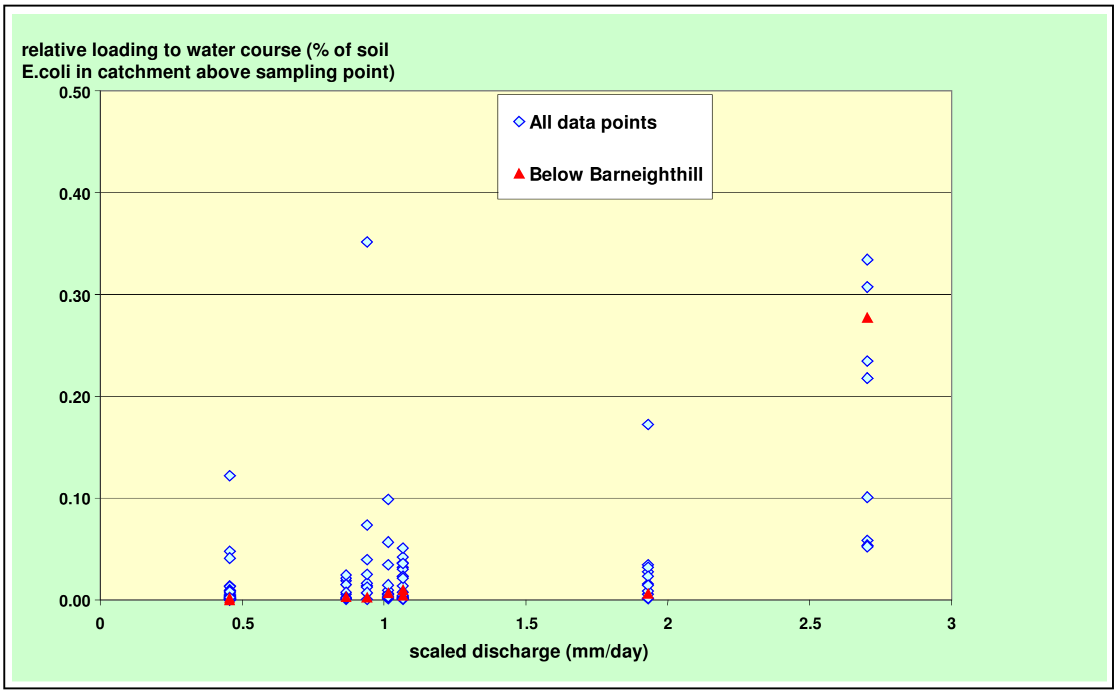


Figure 22. Point sampling data for summer 2002 in Cessnock catchment. Estimated daily E.coli load in stream discharge, relative to soil content in catchment above discharge point. Based on weekly spot samples and daily discharges.

Figure 23. Water chemistry and microbiology, above and below Langside and Barneighthill, Cessnock catchment, during storm event 21-22 Sept 2003.

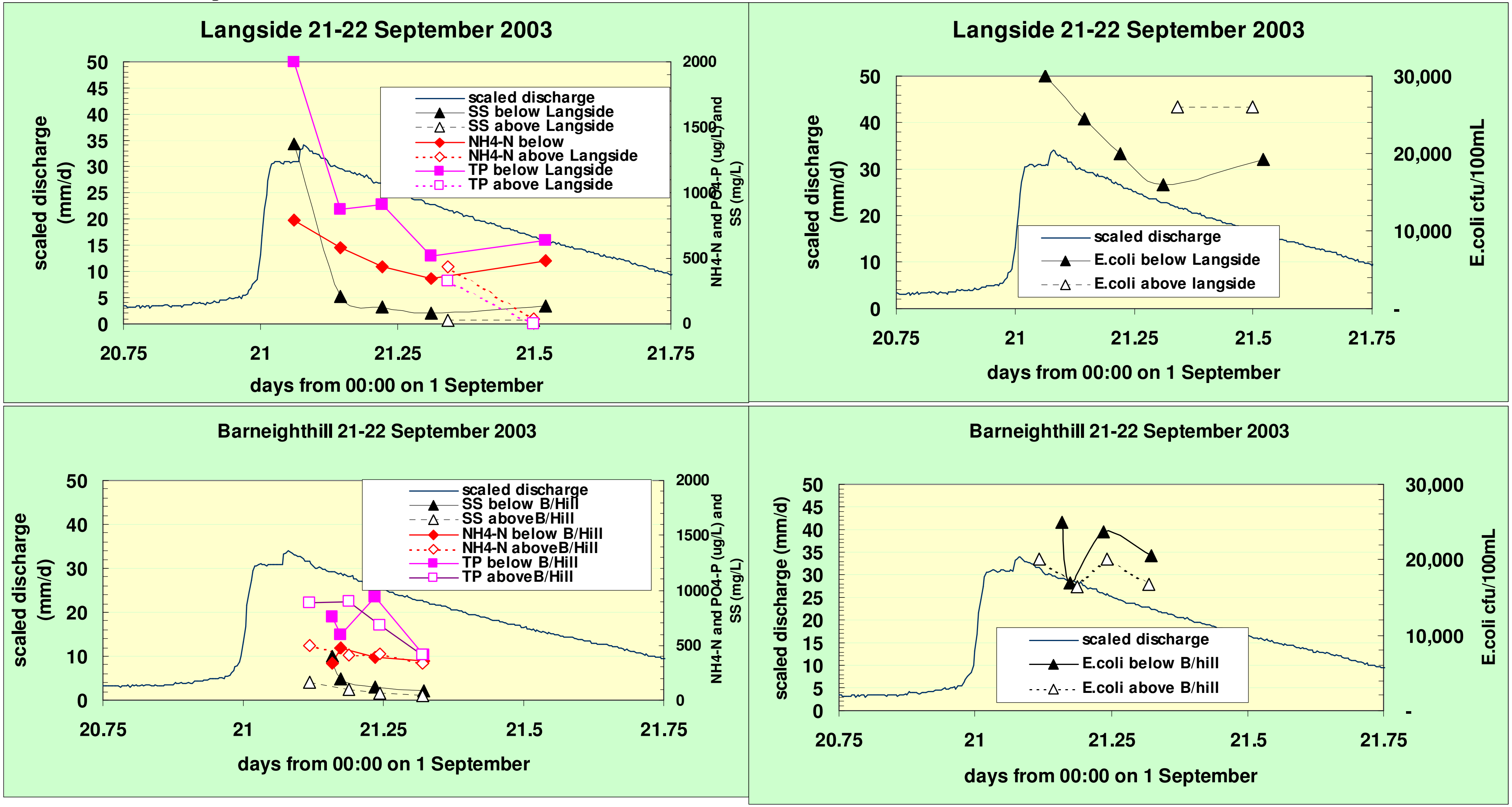


Figure 24. Water chemistry and microbiology, above and below Langside and Barneighthill, Cessnock catchment, during storm event 29-30 September 2003.

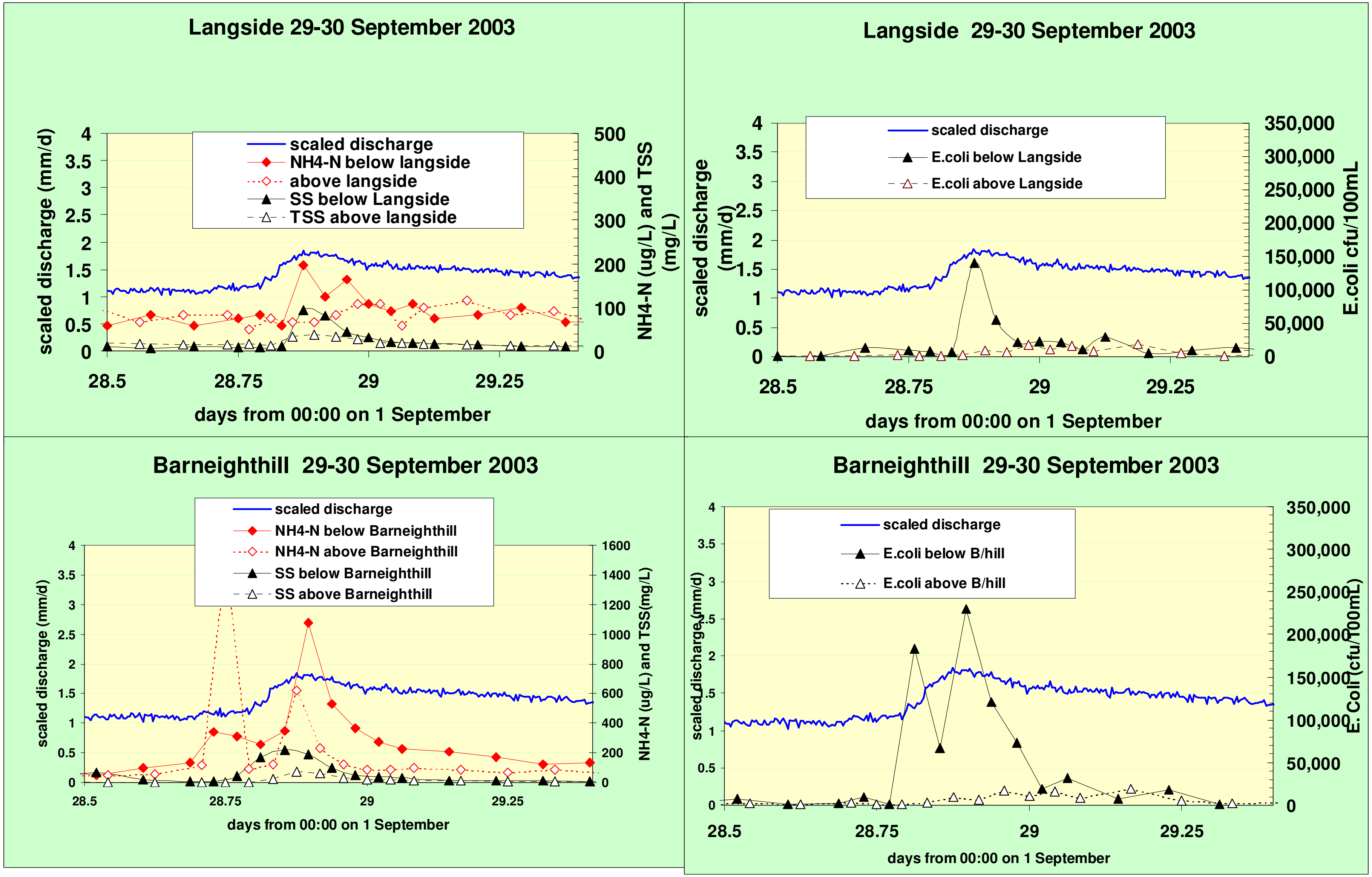
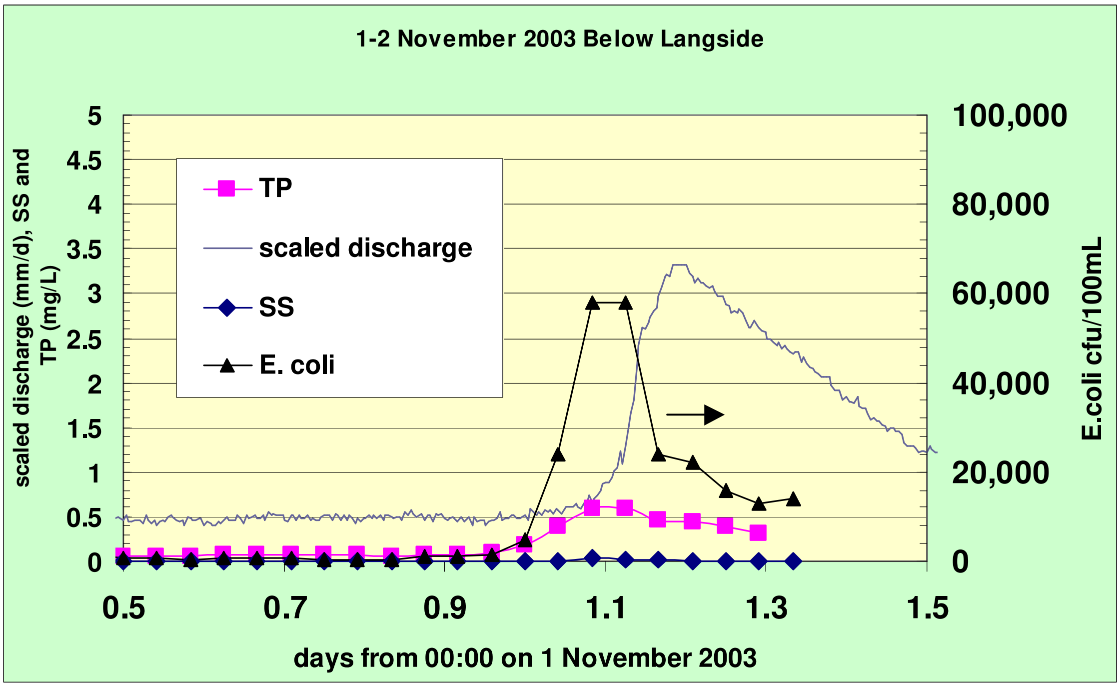


Figure 25. Water chemistry and microbiology, below and further below Langside, Cessnock catchment, during storm event 1-2 November 2003



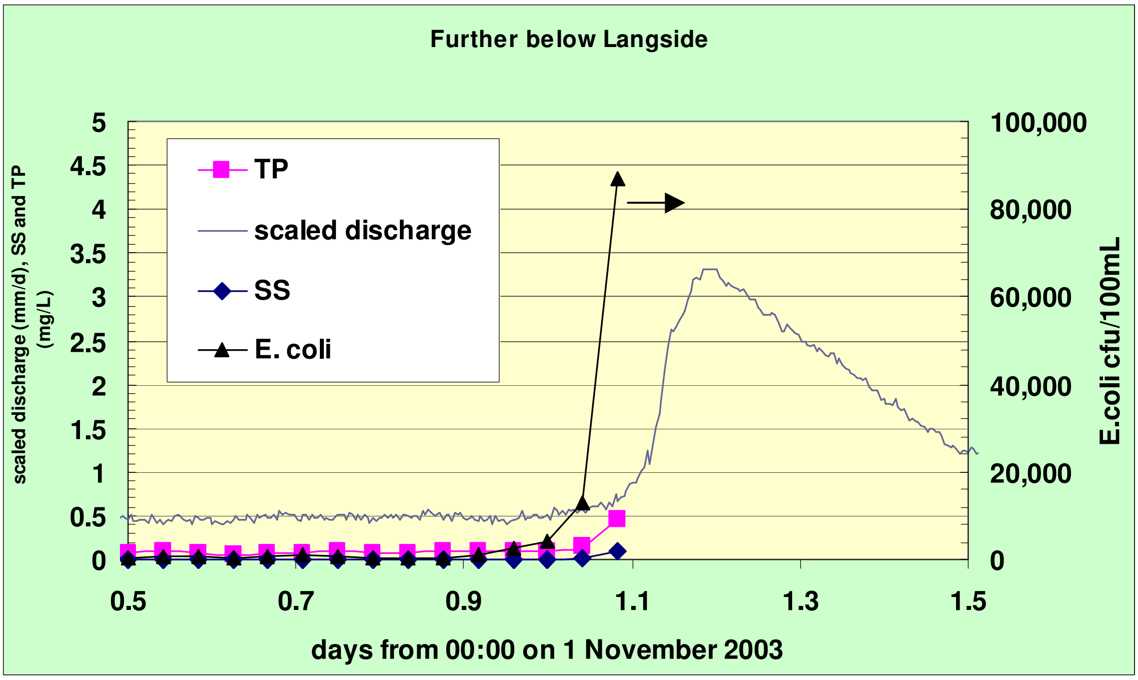
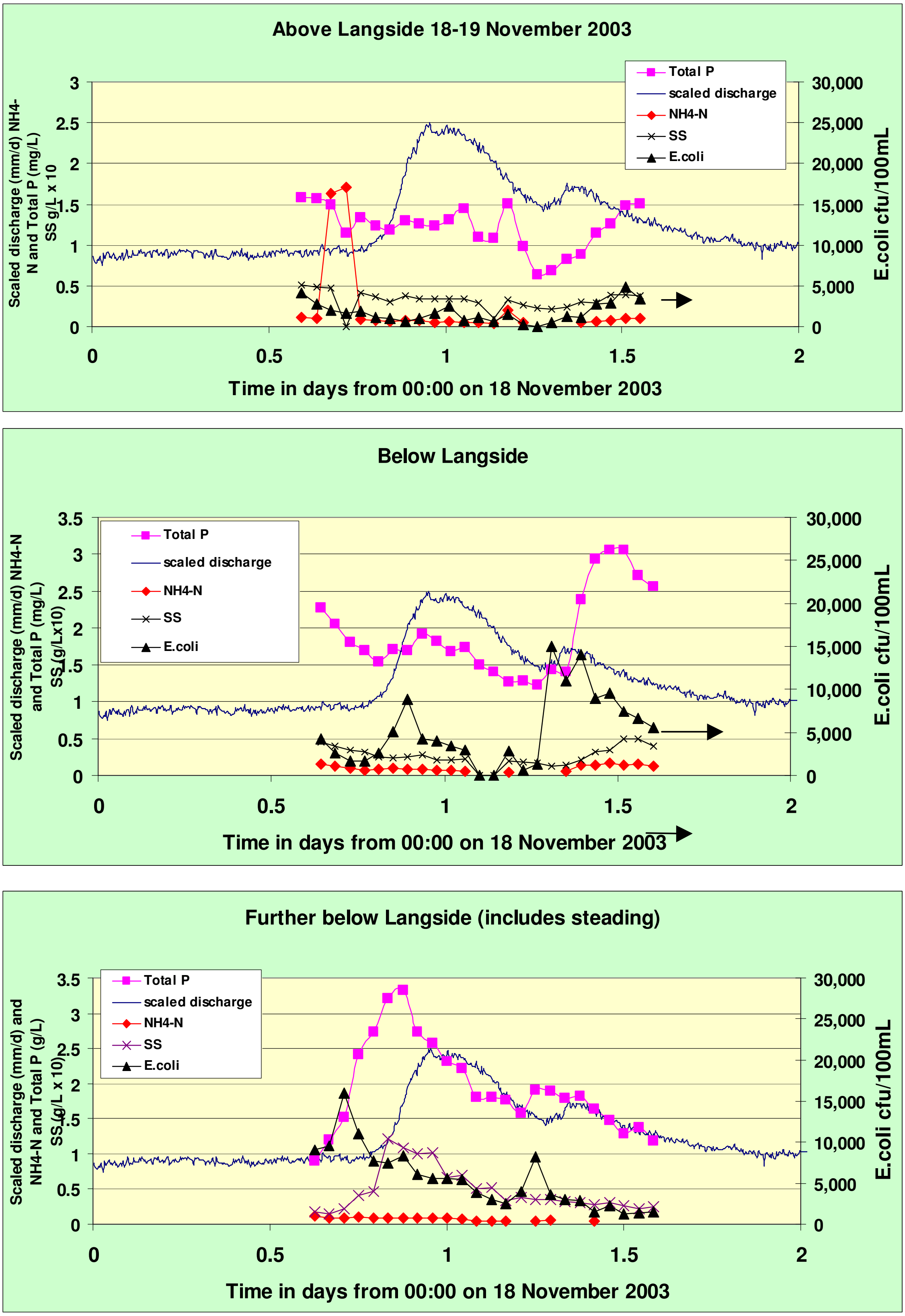


Figure 26. Water chemistry and microbiology, above, below and further below Langside, Cessnock catchment, during storm event 18-19 November 2003.



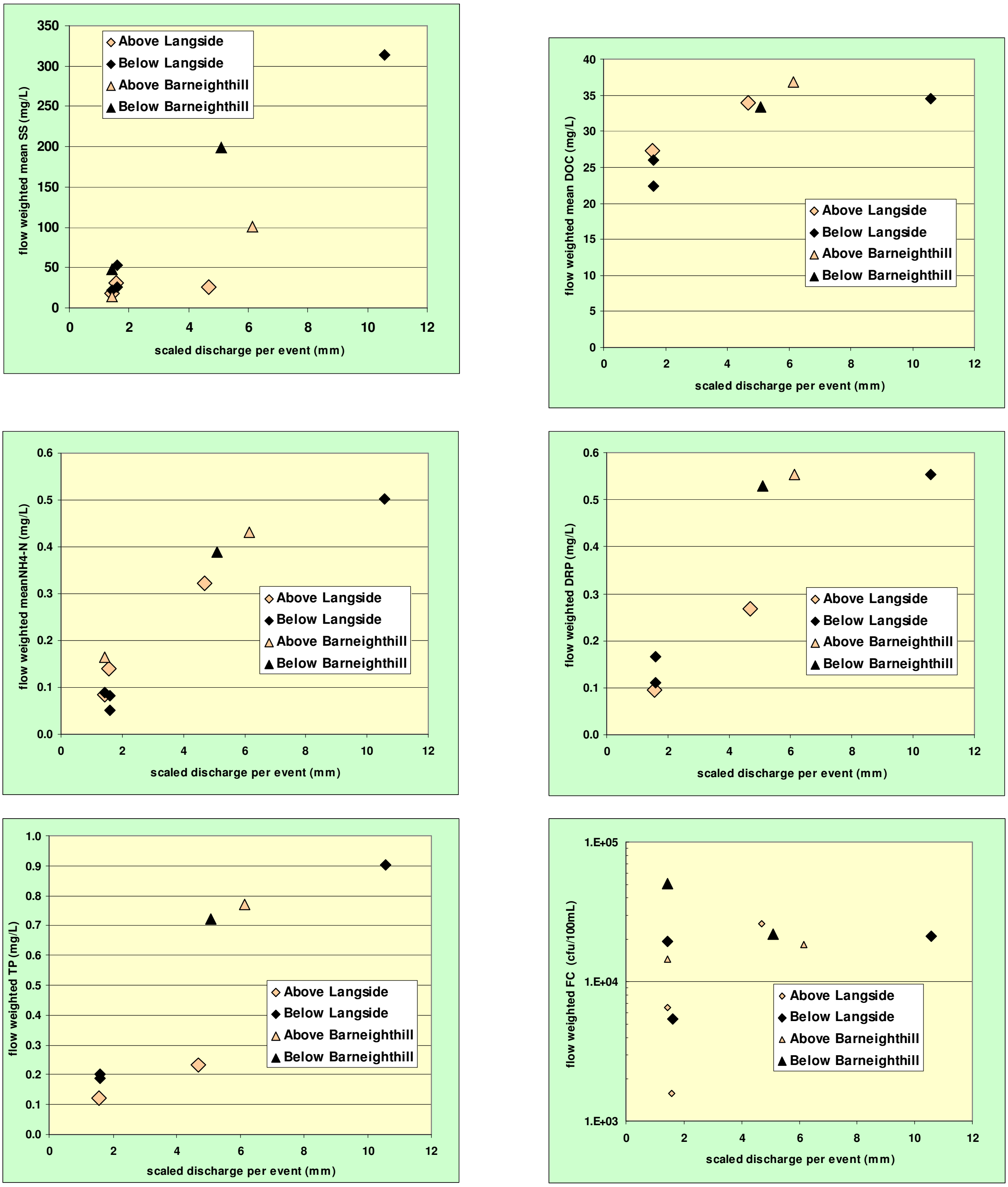


Figure 27. Summary data on storm events from Cessnock catchment showing the relationship between scaled event size (in mm) and mean event pollutant concentration. For SS, DOC, NH4-N, DRP, TP and faecal coliforms. Open symbols show values above the farmed reaches studied, closed symbols show values below the farmed reaches.

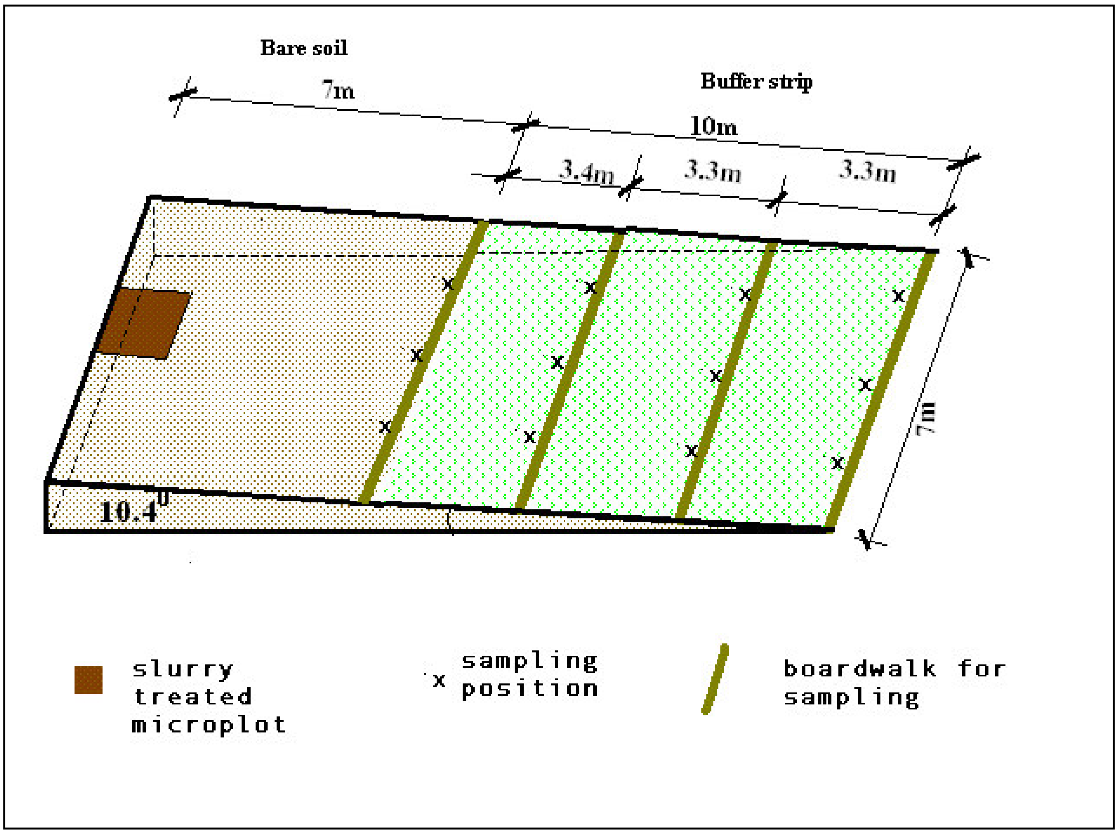


Figure 28. Layout of slurry treated microplot and buffer strip plot and sampling positions. Sandy loam soil, Bush Estate.

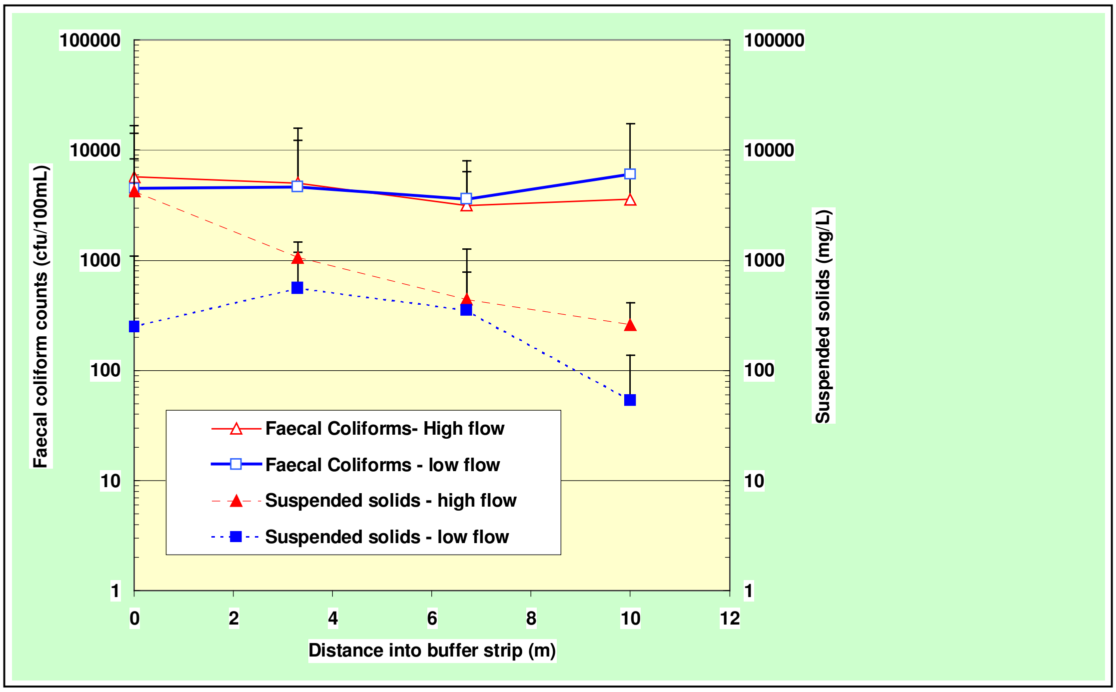
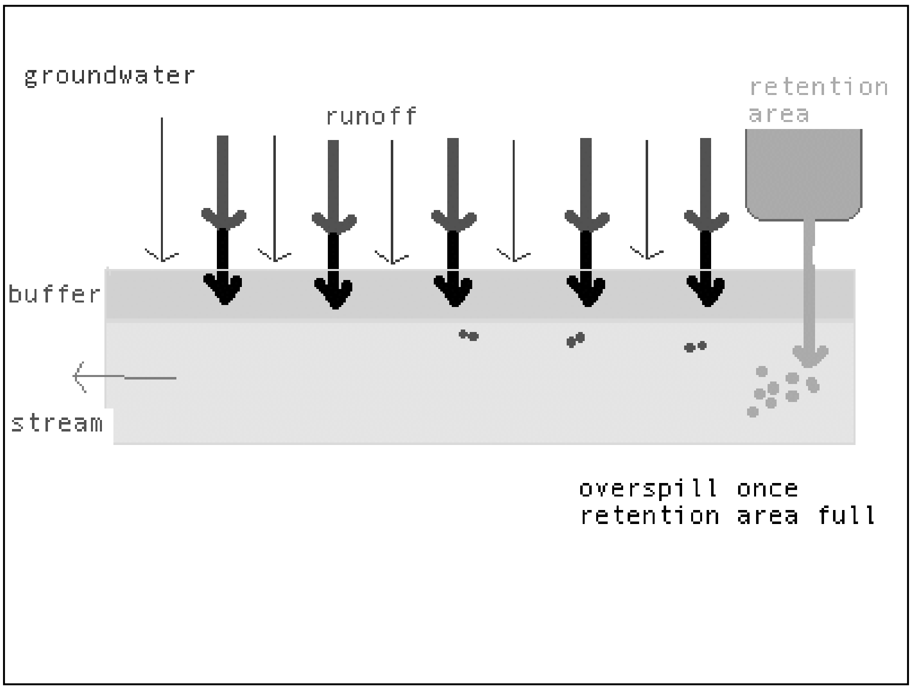


Figure 29. Geometric mean suspended solids and faecal coliform concentrations in water samples within a buffer strip collecting runoff from slurry contaminated bare soil plot to which either high intensity or low intensity irrigation has been applied. Error bars are +sd of geometric mean.

Figure 30. Conceptual diagram showing how the sediment peak in storm event chemistry could be delayed until after discharge peak, in the presence of temporary retention capacity at field margin (eg buffer strip).



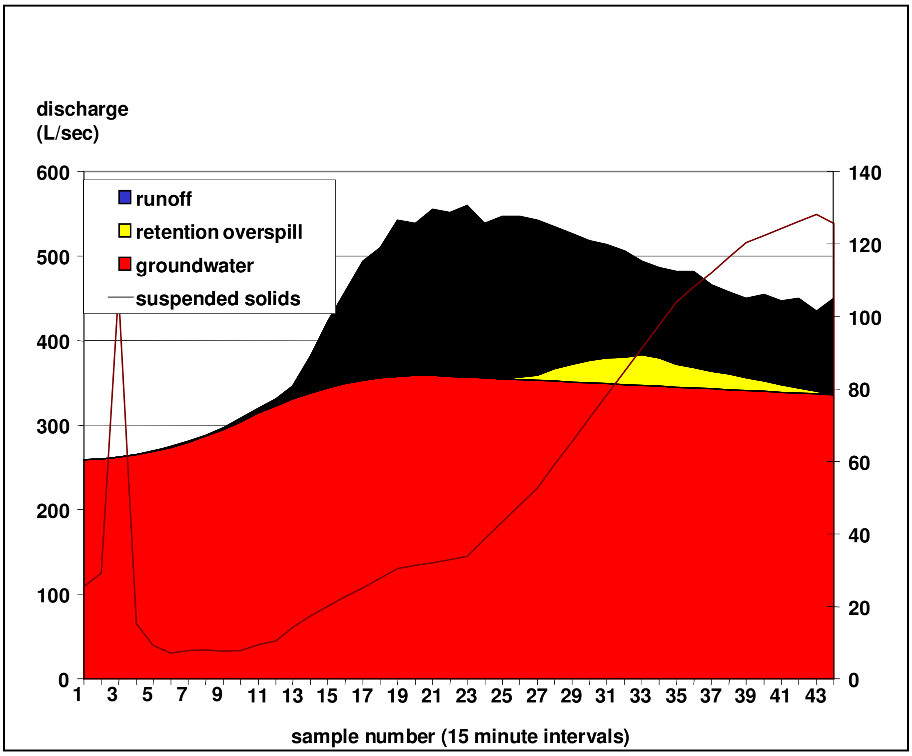


FIGURE 31

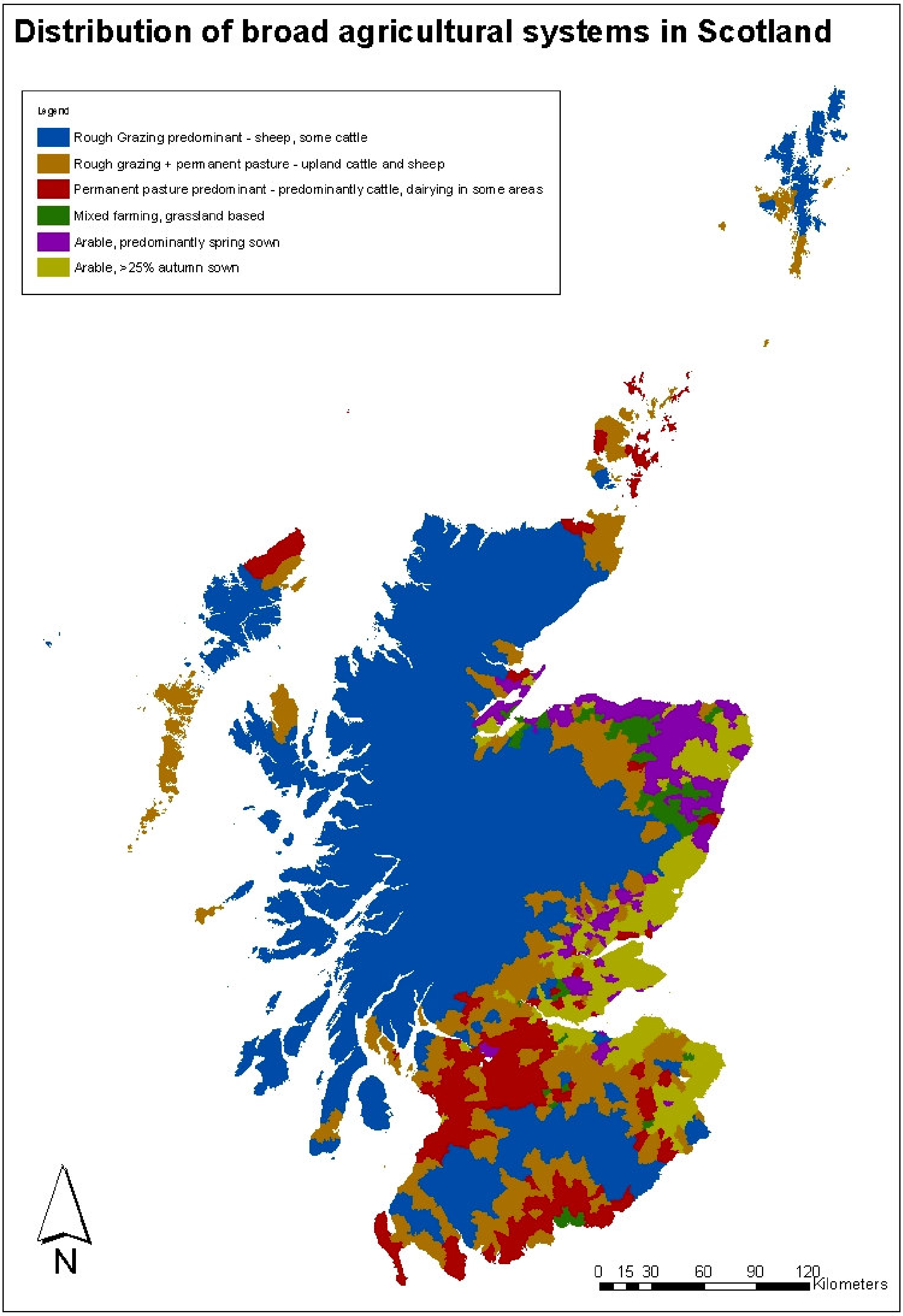

Distribution of broad agricultural systems in Scotland
Legend
Rough Grazing predominant - sheep, some cattle
Rough grazing + permanent pasture - upland cattle and sheep
Permanent pasture predominant - predominantly cattle, dairying in some areas
Mixed farming, grassland based
Arable, predominantly spring sown
Arable, >25% autumn sown
N
0 15 30 60 90 120
Kilometers

FIGURE 32

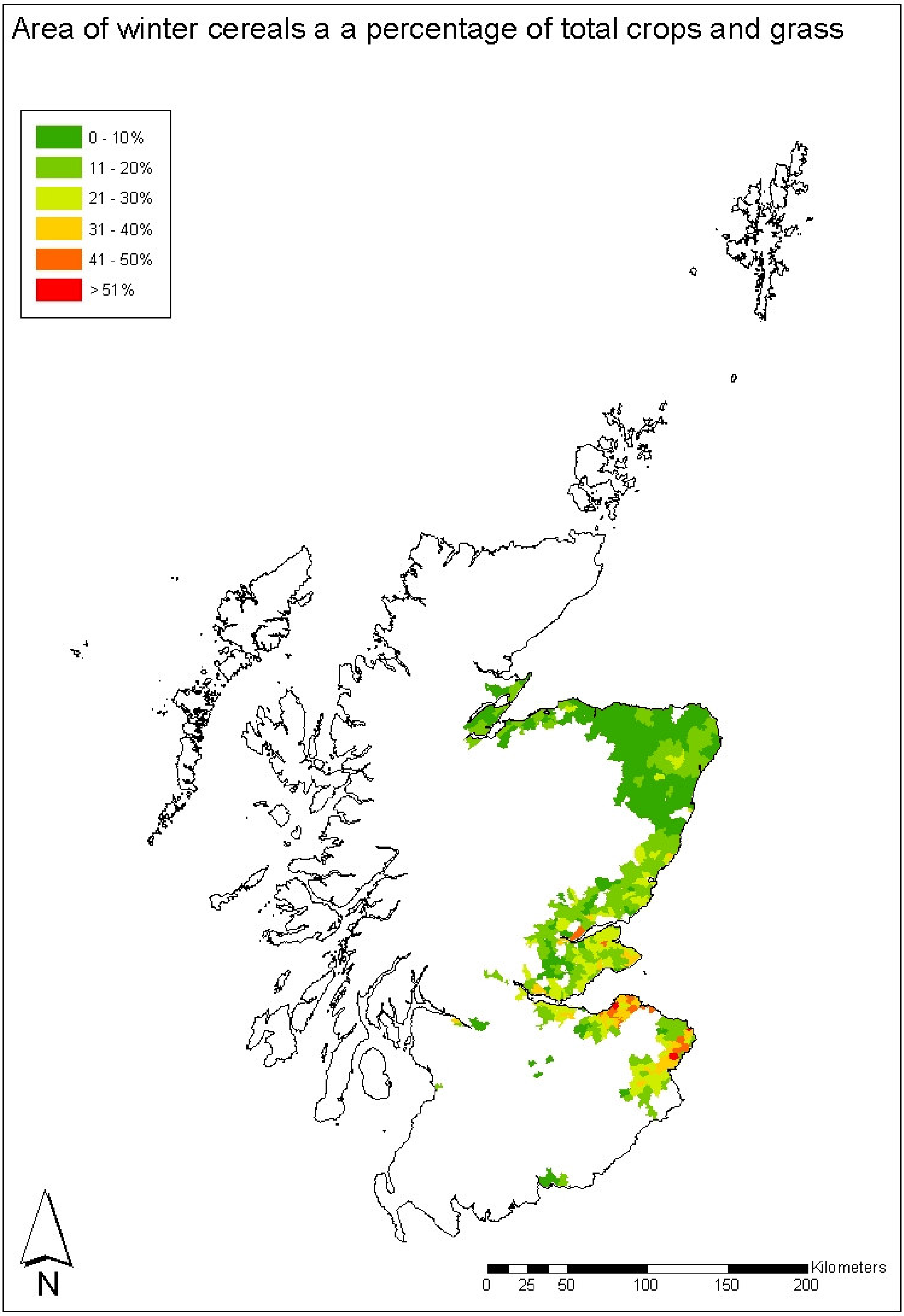

Area of winter cereals a a percentage of total crops and grass
0 - 10%
11 - 20%
21 - 30%
31 - 40%
41 - 50%
> 51%
N
Kilometers
0
25
50
100
150
200

FIGURE 33

Percentage of mineral soils assessed as having high or moderate risk of erosion

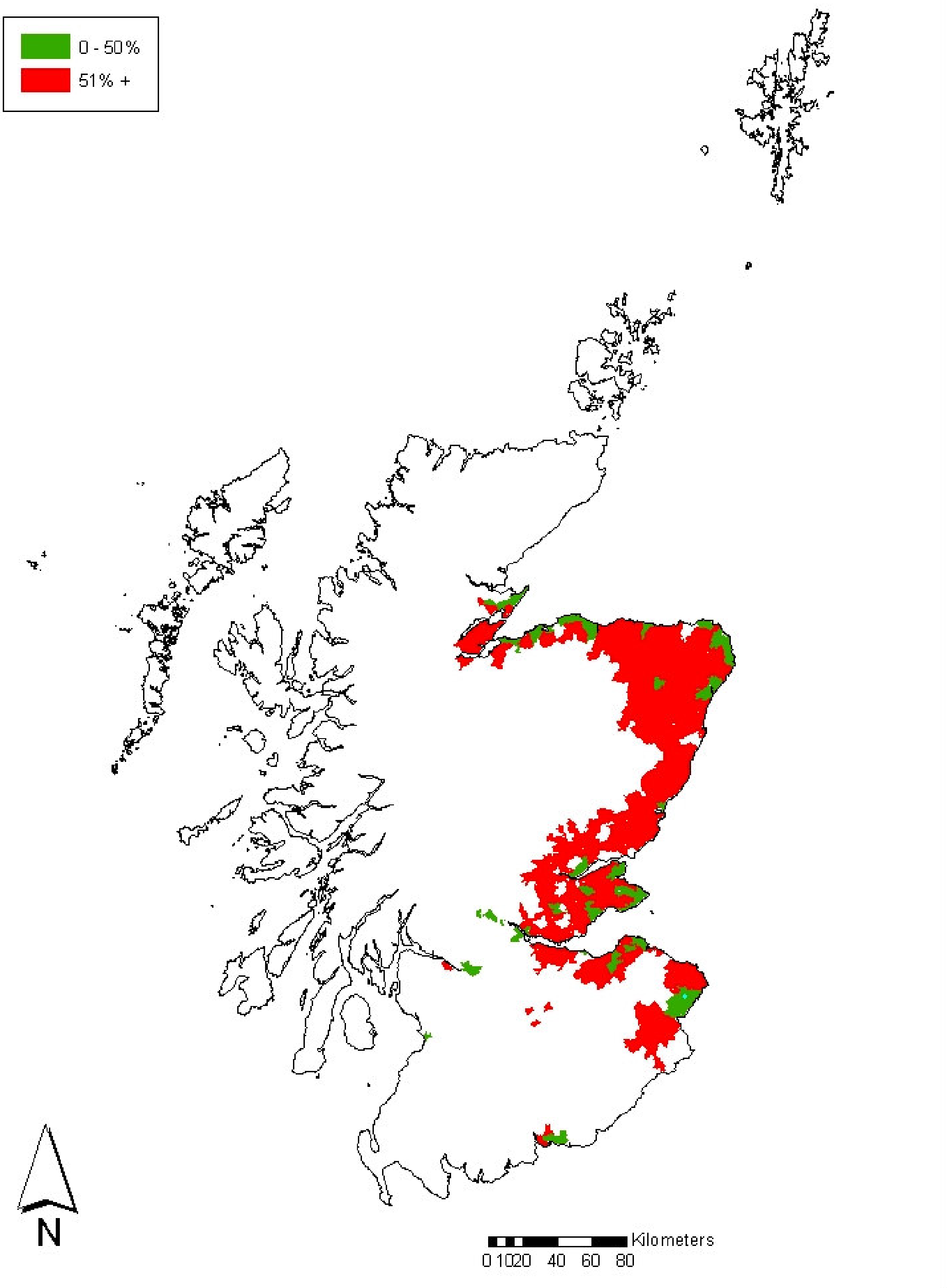


FIGURE 34

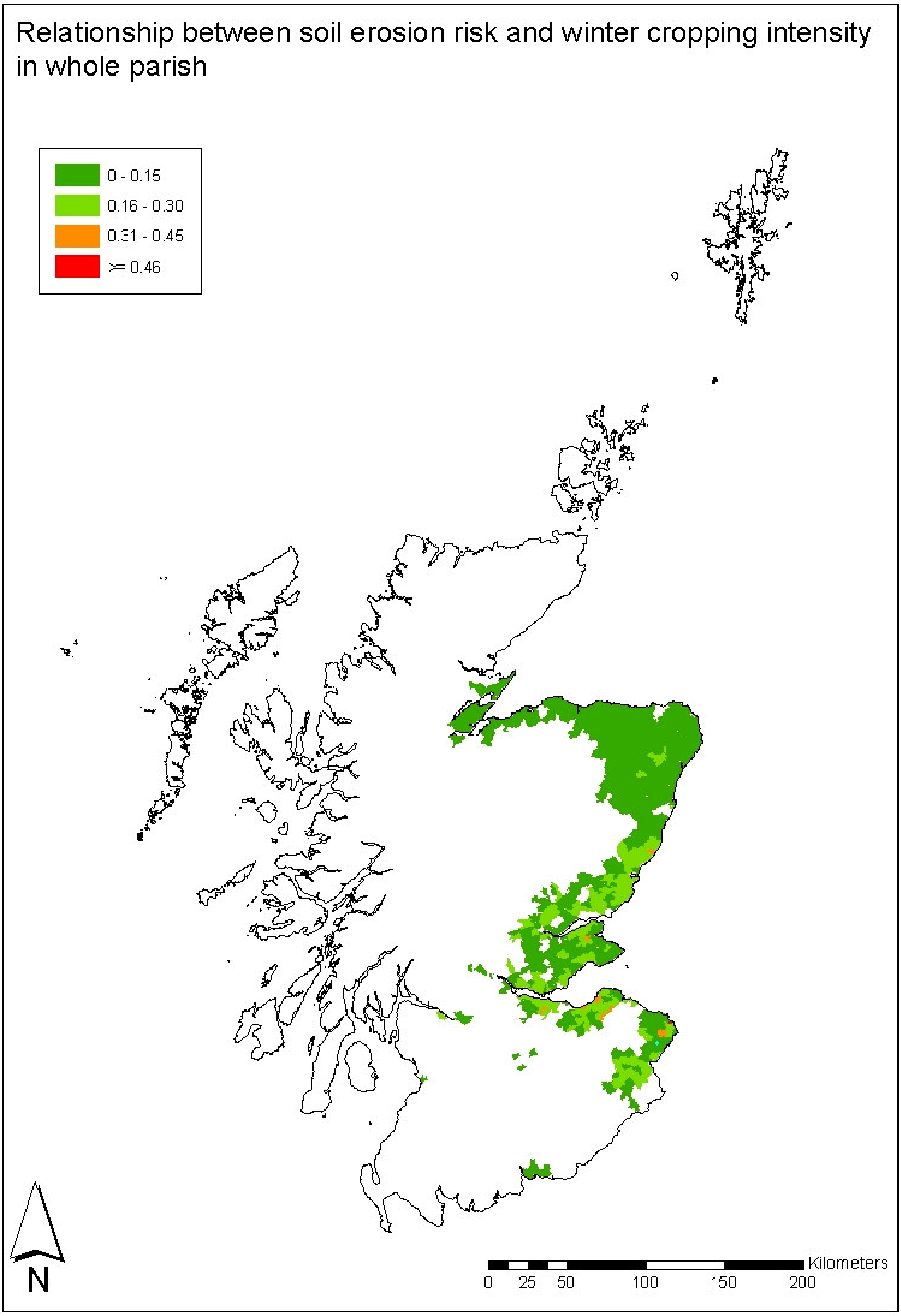
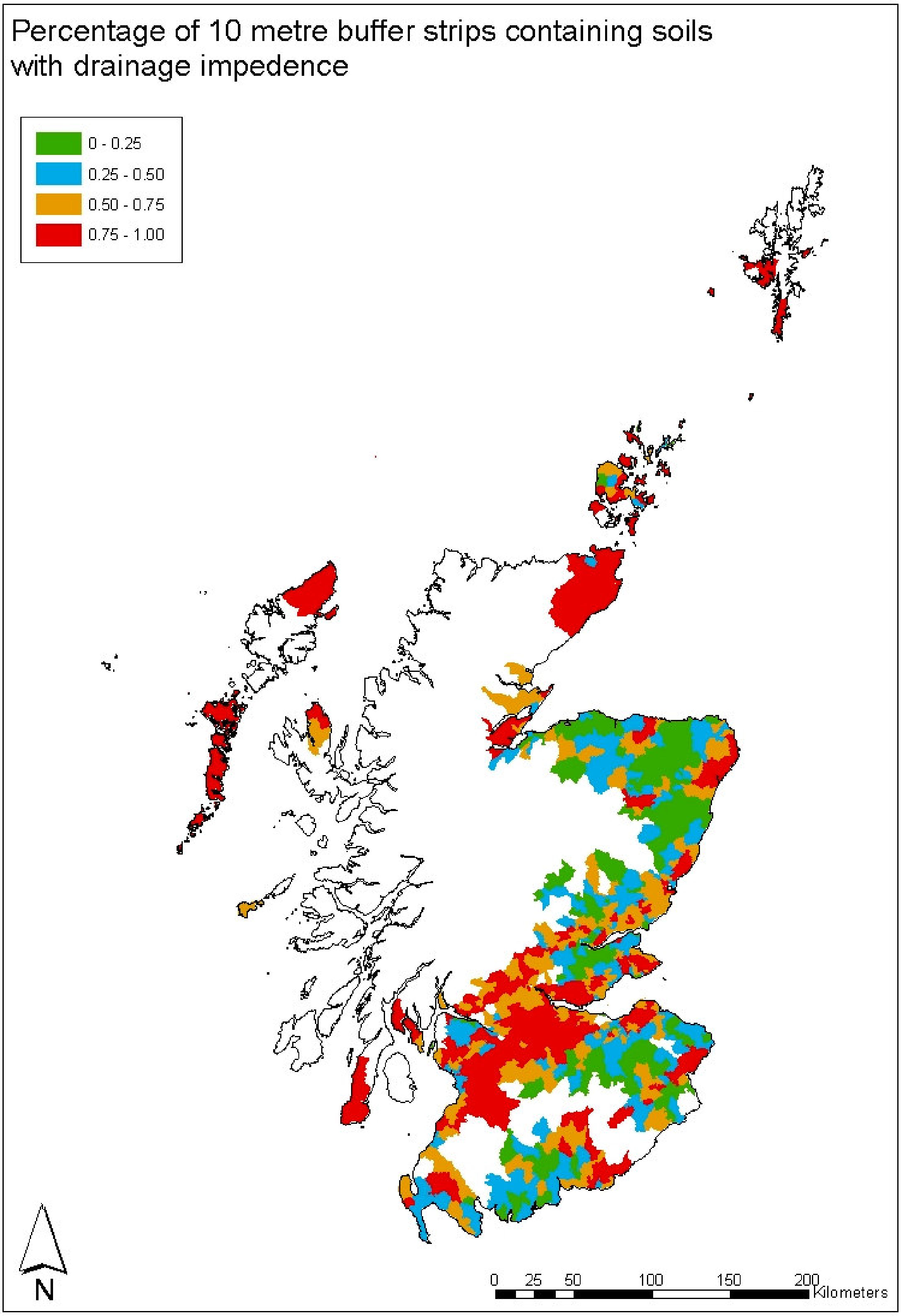

Relationship between soil erosion risk and winter cropping intensity in whole parish
0 - 0.15
0.16 - 0.30
0.31 - 0.45
>= 0.46
N
Kilometers
0
25
50
100
150
200

FIGURE 35



Percentage of 10 metre buffer strips containing soils with drainage impedence
0 - 0.25
0.25 - 0.50
0.50 - 0.75
0.75 - 1.00
N
0
25
50
100
150
200
Kilometers