Understanding the contribution of grass uplands to water quality

DEFRA project: WQ0121

DRAFT REPORT

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Contents

List of figures

List of tables

Executive summary

There are approximately 2 million hectares of upland grass in the UK. In Wales 73% of the land area is rough grazing. Upland grasslands are a vital source of high quality waters for public water supply and high status ecological habitats. Approximately 70% of the UK water resource comes from uplands and is generally considered to be of high quality with low resultant water treatment costs. Upland waters provide dilution for pollutant discharges downstream, are nursery areas for fish, especially salmon, and generate income for many rural businesses.

Although pollutant inputs from uplands are generally much lower than from lowlands, upland waters are more sensitive to pollutants since aquatic habitats are likely to be adapted to low contaminant thresholds. Consequently, small increases in some pollutants can have a disproportionally large impact on ecological quality. These effects are thought to be partially responsible for declining populations of some protected species, for example the vendace.

Soil type, location and climate are all important controllers of water quality in upland grass areas. However, due to a lack of studies we currently do not have a good understanding of how these factors control pollutant mobilisation, transport and flux. For example, we have data on the deposition of atmospheric nitrogen, but as yet are unable to determine when and where this nitrogen will be released to surface waters; nor do we understand the causes of increased DOC losses from the uplands, despite having information on the carbon content of upland soils. Conversely we know that phosphorus is a problem in some upland waters, but do not know what the contribution of grasslands is to phosphorus loads in receiving waters.

Whilst there are long-term water quality data sets for upland catchments, including those dominated by grasslands, the suite of measurements often reflects specific interests such as acidification so that other key parameters are absent. Furthermore, many of these data sets are not specific to upland grassland, but include catchments with significant areas of other land cover types which will contribute to the water quality signal measured at the catchment outlet. Those data sets which exist specifically for upland grasslands are mostly short-term records with limited information about the inter-annual variation in water quality parameters. This is a significant problem, as they are likely to be highly variable and as a consequence the impact of management activities or a changing climate may be difficult to detect. Failure to gain a better quantification of fluxes and pathways will impact on our ability to develop and monitor compliance with programmes of measures for the Water Framework Directive.

Few studies have attempted to quantify the potential impact of climate change on water quality, particularly in the uplands. This is a result of limited data and the highly heterogeneous nature of climate over uplands. However, anticipated changes include increased water temperature and reduced dissolved oxygen, decreased dilution capacity of receiving waters, increased erosion and diffuse pollution, photoactivation of toxicants, changing metabolic rates of organisms, increased eutrophication and greater prevalence of algal blooms – all of which could lead to exceedence of water quality standards. Lack of water at low flow periods could ultimately severely limit abstraction opportunities in the uplands and in downstream waters.

Some upland management activities are potential threats to water quality for example: overstocking, slurry spreading, stock access to streams, sheep dipping, static supplementary feeding, outdoor lambing, herbicide use and removal of riparian shading. However, there has been very little research into the impacts of these activities on water quality.

All the data sets identified in this review had limitations for classifying and mapping upland grassland at a national scale. No single water quality data set is available for creating such a classification although most have useful elements. Many of the schemes lack high-resolution sampling intensity, as well as some of the key parameters (acidity, nitrogen, DOC, P, suspended sediments, pathogens, pesticides and herbicides); none of the data sets have information on pathogens, herbicides and pesticides. New survey information is required in order to establish an upland grassland water quality classification.

The primary factors influencing the relationship between upland grassland and water quality are summarised in matrix form in the report, which links key pollutants to management activities. Gaps in the matrix indicate lack of knowledge and uncertainty of impacts; content reflects known pollution issues. Although there is little supporting research for any of the issues identified in the matrix, four key implications of gaps in knowledge for the WFD are identified:

1. Although the impact of drains on upland water quality is poorly understood, their extensive coverage and role in promoting hydrological connectivity mean that contributions from artificial pathways may be critically important in sediment and pollutant delivery and represent a very significant gap in knowledge.
2. We are uncertain of the actual water quality of headwater streams and have little data for verifying the risk-based characterisation of water body status.
3. Where water bodies fail to meet WFD objectives the cause is not always clear, partly as a result of poor data and monitoring in feeder streams. Monitoring activities may need to be extended to provide a background against which to monitor change in status.
4. Deriving programmes of measures for failing upland water bodies will require further research to determine the causes of failure.
5. Targeted field research could help develop the evidence base to support generic programmes of measures for upland grasslands known to improve water quality.

Changes in the Common Agricultural Policy may lead to reduced grazing pressures and improved upland water quality. However, threats include a reduction in the rural labour force leading to reduced maintenance of features that may help to protect water quality, for example stream bank fencing. Provision would need to be made to safeguard this in any future Entry Level Stewardship (ELS) scheme. There are other pressures outside of CAP that may counteract the benefits which may accrue from the reform of CAP. For example, the need for increased food production to meet a growing population may lead to farming intensification.

Suggestions for the revision of the Environmental Stewardship menu of options that would provide benefit to water quality in the uplands include: making **consistent advice** available to farmers within the ELS scheme; **targeting** measures to address particular catchment water quality objectives; **funding** for capital works, e.g. stream bank fencing; **research** to understand what the key issues are for a catchment; and determining **the effectiveness** of different mitigation measures. The report also identifies options that are not currently included in Environmental Stewardship, or are included in the higher level scheme but could be made more widely available.

1. Introduction

1.1. Definition of upland grasslands

There is no formal definition of upland grasslands for the UK; they have variable vegetation and soil types and are found in areas with different climatic conditions. Upland areas are generally defined as areas above 300 m altitude (Reynolds and Edwards, 1995), but using a broad representation of climate and habitat type they can be considered to range from areas of high ground in the south-west of England down to sea level in Scotland (Orr et al., in review). Figure 1 shows the area considered upland for the purposes of this study. Within these upland regions the grassland communities can vary considerably. The range of upland grassland types has been defined in the National Vegetation Classification (Rodwell, 1992 and Table 1). In general, upland areas tend to be cooler and have a higher rainfall than lowland areas; soils tend to be organo-mineral and although geology is very variable, many upland areas are found on acid bedrock.

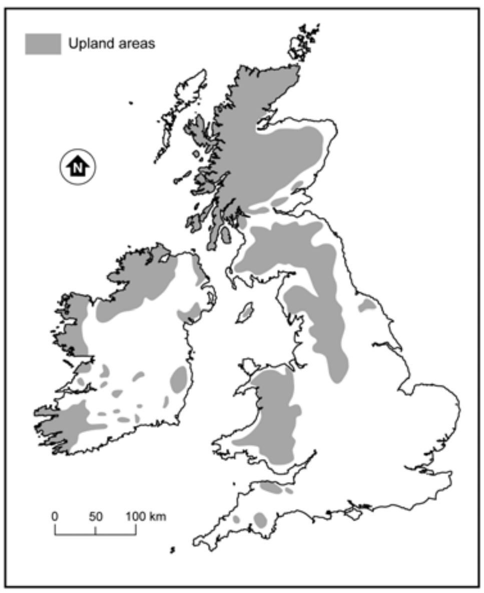

Figure 1 Upland areas of the UK (after Averis et al., 2004)

Upland grasslands are found mainly in northern and western parts of the UK; there are approximately 2 million hectares (ha) of upland grassland and a further 180,000 hectares dominated by bracken (RSPB, 2000). In Wales, 73% of the land area is accounted for by rough grazing (Figure 2). Although economic returns from hill farming are relatively small, they are important to local economies, and yet upland grasslands are an often overlooked component of agriculture in the UK.

Table 1 Grassland communities found in upland areas. Some grassland communities of very restricted distribution or low grass cover have been excluded

Community		**Description**
MG3	*Anthoxanthum odoratum-Geranium sylvaticum* grassland	Traditional upland hay meadow, restricted to northern England
MG7	*Lolium perenne* leys	Species-poor productive grassland
CG9	*Seslaria albicans-Galium sterneri* grassland	Limestone hill pasture restricted to northern England
CG10	*Festuca ovina-Agrostis capillaris-Thymus praecox* grassland	Herb-rich grassland found throughout upland Britain on calcareous bedrock
CG11	*Festuca ovina-Agrostis capillaris-Alchemilla alpina* grass-heath	Alpine pasture, largely restricted to Scotland
CG12	*Festuca ovina-Alchemilla alpine-Silene aculis* dwarf herb community	Alpine grassland confined to higher peaks of the Scottish Highlands
U2	*Deschampsia flexuosa* grassland	Grazed community of upland fringes
U3	*Agrostis curtisii* grassland	Upland fringe grassland restricted to south-west England
U4	*Festuca ovina-Agrostis capillaris-Galium saxatile* grassland	Common upland calcifuge grassland used for rough grazing
U5	*Nardus stricta-Galium saxatile* grassland	Hill grazing common throughout north and west Britain
U6	*Juncus squarrosus-Festuca ovina* grassland	Widespread upland grassland characteristic of moist soils with a high peat content
U7	*Nardus stricta-Carex bigelowii* grass-heath	Arctic grassland largely restricted to high mountains
U13	*Deschampsia cespitosa-Galium saxatile* grassland	Tussocky grassland characteristic of the coldest areas of Britain
U14	*Alchemilla alpine-Sibbaldia procumbens* dwarf-herb community	Low open turf found on ground irrigated by snow-melt
U16	*Luzula sylvatica-Vaccinium myrtillus* tall-herb community	Ungrazed community restricted to inaccessible slopes
U17	*Luzula sylvatica-Geum rivale* tall-herb community	Ungrazed community restricted to inaccessible slopes
U18	*Cryptogramma crispa-Athyrium distentifolium* snow-bed	Widespread throughout western Highlands
U20	*Pteridium aquilinum-Galium saxatile* community	Bracken dominated community found up to moderate altitudes

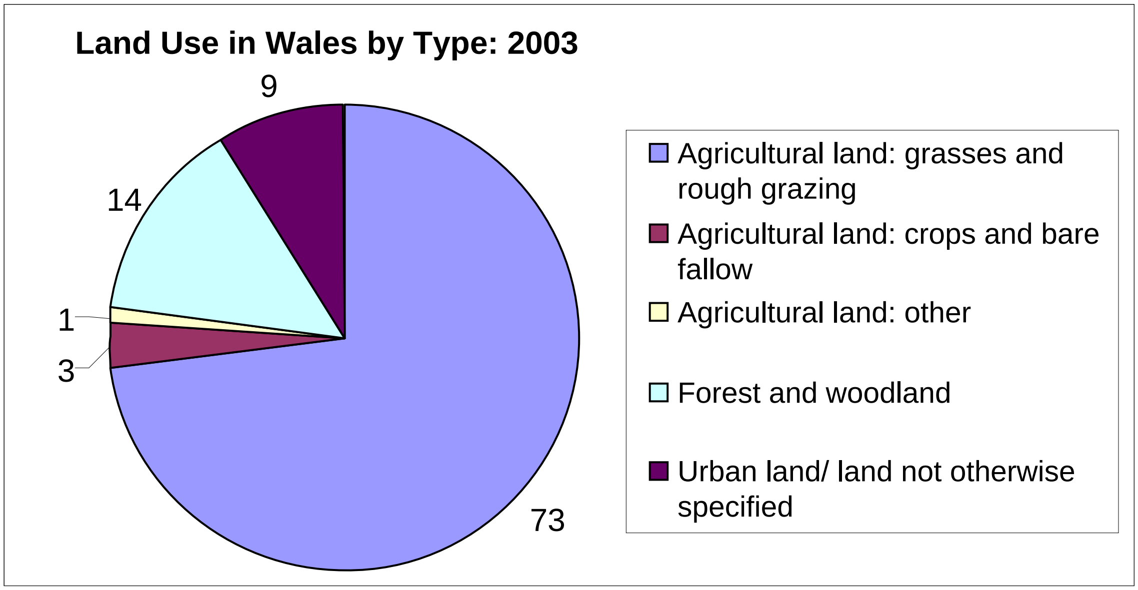

Figure 2 Land use in Wales in 2003 (National Assembly for Wales)

Agriculture in the uplands relies heavily upon semi-natural grassland and heathland habitat, which is interspersed with smaller areas of in-bye land (hay or silage) for cropping, sheep and cattle (mainly beef or sucklers). For the purposes of this study, upland grasslands are considered to be those characterised by rough grazing with little or no agrochemical inputs (open fells or out-bye) and limited land drainage, plus improved grassland, possibly with field drains (enclosed fields or in-bye). The dominant activity is sheep grazing.

1.2 The value of upland grasslands

Uplands, including upland grasslands, provide vital services many of which are essentially free, low cost or taken for granted. These include *provisioning services* such as food, water, hydropower; *regulating services* that affect climate, floods, disease, wastes and water quality; *cultural services* that provide recreational, aesthetic and spiritual benefits; and *supporting services* such as soil formation, photosynthesis and nutrient cycling (Millennium Ecosystem Assessment, 2005; Duigan, 2004). Society, while buffered against environmental changes by culture and technology, is fundamentally dependent on ecosystem services (Millennium Ecosystem Assessment, 2005).

Upland grasslands are especially important for water quality. For example, approximately 70% of the UK water resource comes from uplands; it is generally considered to be of high quality and as a result minimises water treatment costs. Upland waters are critical in providing a dilution effect for pollutant discharges downstream. Headwaters are also nursery areas for fish, especially salmon, generating income for many rural businesses. Upland grasslands and the waters they help to maintain also support unique and highly sensitive ecosystems. A number of threatened or endangered birds and plants depend on these ecosystems. The upland terrain means that they have been protected against development, allowing populations to survive.

Uplands have an important role as carbon sinks, with large quantities of carbon stored in upland soils, and historically have been important for providing us with natural resources, especially minerals.

Population growth, pollution and direct and indirect impacts from climate change threaten the continuing supply and low cost of the ecosystem services provided by the uplands.

1.3. Pollutants in the uplands

The uplands are not a large source of pollutants. Compared to lowland areas, pollutant inputs per unit area are likely to be considerably lower. This is because large areas of uplands have a low population density, land is managed at a lower intensity than in lowland areas and there is little industry. There is, however, a wide range of water pollutants that can potentially come from upland grasslands. Each of these pollutants will be described briefly below and discussed in relation to land management practices in Section 2.

Nitrogen – Nitrate (NO_3) and ammonia (NH_4) are the main forms of nitrogen pollution, but dissolved organic nitrogen (DON) is also a potential pollutant. Nitrogen is mainly transported in the dissolved fraction by leaching drainage and overland flow. Nitrogen contributes to eutrophication and acidification. In addition, NO_3 is implicated in methemoglobinemia (blue baby syndrome) and NH_4 is toxic to fish at concentrations dependant on the water pH. Levels of nitrogen input are likely to be much lower in upland areas than lowland areas.

Phosphorus – Phosphorus (P) also contributes to eutrophication of waters and soils. Phosphorus is frequently transported in a particulate form bound to sediment, providing a long-term store of phosphorus that becomes available to plants over time. As with particulate phosphorus, dissolved phosphorus contributes to eutrophication, although in this case it is rapidly available to algae. Upland grasslands are not likely to be a major source of phosphorus.

Acidity – Acidification is a reduction in the pH of surface waters. It is caused by land-use changes (e.g. afforestation), fertilisation and atmospheric deposition of nitrogen and sulphur. Acidity is a serious threat to water quality in the uplands, and many upland waters are anthropogenically acidified (Batterbee et al., 2004).

Suspended solids – Suspended solids (including eroded sediment) are predominantly delivered to receiving waters by overland flow. They increase the turbidity of receiving waters and are frequently associated with the transport of other pollutants. Fine sediment infiltration and smothering of gravel bed rivers can also damage salmon eggs and reduce oxygen supply to interstitial habitats important for invertebrates.

Metals – Heavy metals are potentially toxic to aquatic organisms and their presence increases water treatment costs. Metals of particular concern are copper, zinc, cadmium, arsenic, lead, and iron. Metals reach waters by direct deposition, or are transported there after mobilisation due to acidification and erosion of contaminated soils and sediments, which is a particular problem in the uplands (Rothwell et al., 2006). Drainage from abandoned metalliferous mine workings is also a potential source of heavy metals.

Dissolved organic carbon – Dissolved organic carbon (DOC) is associated with water colour and although not directly harmful, it can react with chlorine in transit to produce trihalomethanes. Trihalomethanes are potential carcinogens and their concentration is governed by law in the UK (Hsu et al., 2001). Furthermore, removal of DOC is costly.

Faecal indicator organisms – Faecal indicator organisms (FIO) include bacterial and protozoan pathogens such as *Cryptosporidium* and *Salmonellae*. They are frequently transported with suspended solids. FIO originate from animal excreta either directly or in manures and slurries applied as fertilisers. These may be deposited directly into surface waters or transported in overland flow. Pathogens pose health threats to wildlife, bathers and water supplies.

Biocides – Biocides include pesticides, herbicides and fungicides. They can be transported in overland and subsurface flow. Biocides are potentially harmful to a wide range of biota and can bioaccumulate higher up the food chain.

Sheep dip – A form of biocide, sheep dip reaches watercourses mainly by accidental spills and runoff from hardstandings. Guidelines exist for safe disposal on land away from watercourses. It is particularly toxic to aquatic invertebrates and is also responsible for fish kills.

Veterinary medicines – Veterinary medicines may enter waters by leaching, following the application of contaminated slurries and manure onto the land, or by direct deposition of faeces into the watercourse. Veterinary medicines and/or their metabolites are potentially toxic (Jones et al., 2004).

Although pollutant inputs from upland areas are generally much lower than from lowland areas, upland waters are generally more sensitive to pollutants. Upland aquatic habitats are likely to be more adapted to low contaminant thresholds. Consequently only small increases in some parameters can have a disproportionate impact on ecological quality.

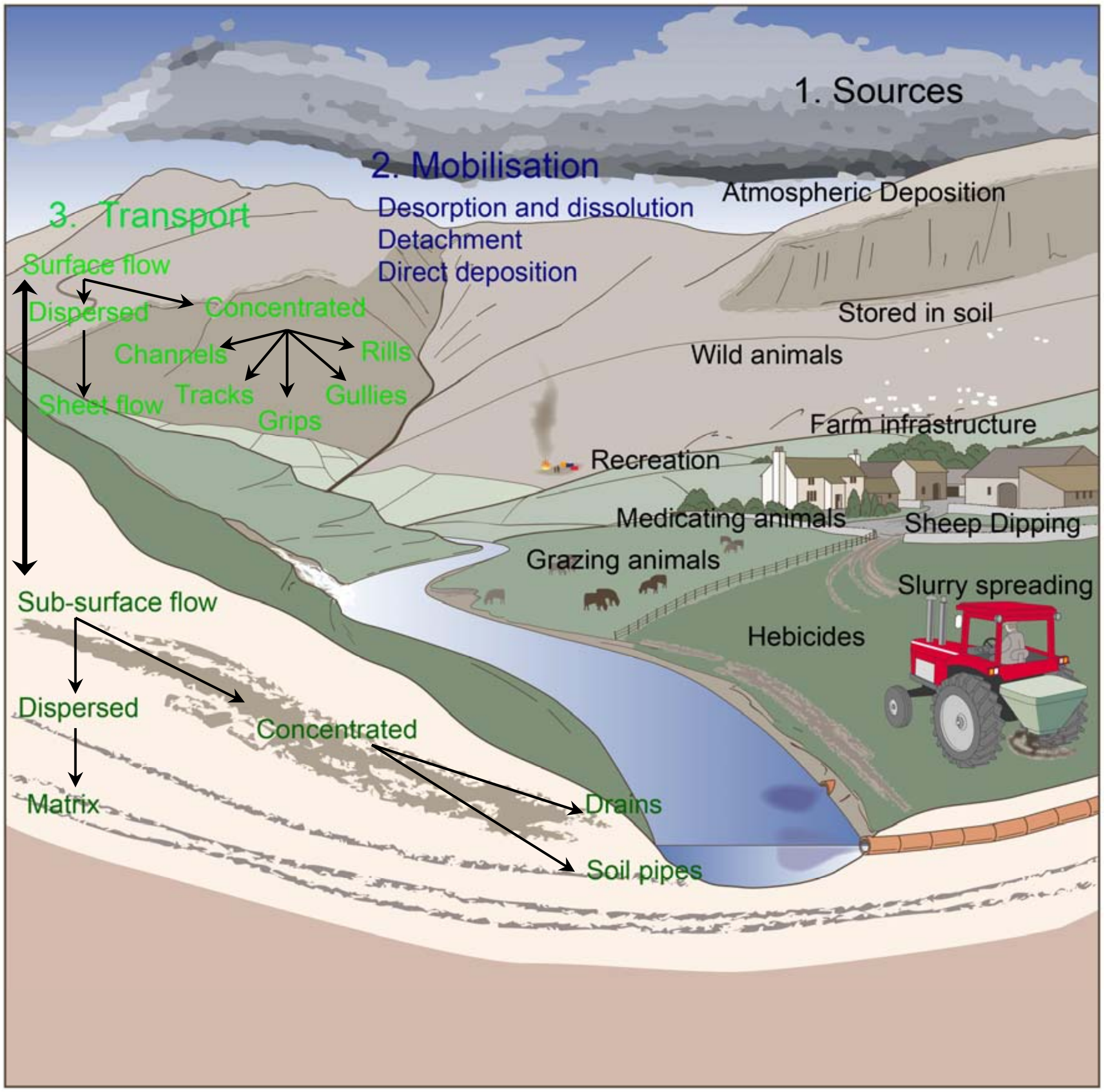

Figure 3 Conceptual diagram showing the sources, mobilisation and transport of pollutants in upland grasslands

Sources of pollutants associated with upland grasslands are identified in Figure 3, and include sources associated with farming, recreation and wild animals. These pollutants can be mobilised by desorption and dissolution (i.e. are in a liquid phase), detachment (physical erosion) and direct deposition.

Once mobilised through erosion or chemical detachment processes, pollutants may be transported in overland flow or leached through the soil, before being transported through drainage networks in association with sediments, as colloidal material or in solution. Evidence of erosion by surface processes such as overland flow is often clearly visible in upland grasslands, with gullies and erosion scars being prominent features in many upland catchments (e.g. Bassenthwaite). Erosion is poorly quantified at a national scale, but localised studies have indicated the extent of the problem (e.g. Evans and

Warburton, 2005), which may be exacerbated by poor stock management and compaction by farm vehicles.

Leaching – where substances move in solution through or out of the soil – is responsible for the transport of many dissolved pollutants both horizontally and laterally through the soil. Other pathways include naturally occurring subsurface soil pipes and artificial extensions of the drainage network, such as ditches, moorland grips and subsurface drains. The relative importance of different drainage pathway contributions to water quality is poorly quantified for the uplands; more information is available for lowland catchments (e.g. Heathwaite and Dils, 2000; Walling et al., 2002).

1.4 Water quality and upland farming

Waters in upland regions consist primarily of oligotrophic pools, lakes and headwater streams, with ecological communities that are highly sensitive to changes in nutrient inputs. Water quality is generally good, as a result of low-intensity agricultural practices and low population, although many upland waters in the UK are impacted by acidification (Batterbee et al., 2004). There is also widespread evidence of increasing nutrient enrichment of upland lakes in the English Lake District (e.g. Bennion et al., 2000; Barker et al., 2005) and Wales (e.g. Environment Agency Wales, 2007). These effects are thought to be partially responsible for declining populations of some protected species, for example the vendace (Winfield et al., 2003).

National trends in chemical and biological water quality in the UK are available and clearly show that areas dominated by uplands (and with lower populations), e.g. Wales, are generally of higher quality (Figure 5). In contrast to England, biological water quality in Wales is relatively lower than chemical water quality and has not been improving at the same rate over recent years and may in fact be in decline. It is unclear how much of this is due to historic effects and how much is due to current pressures on water resources. The relatively poorer biological quality of Welsh waters is a matter of concern because the Water Framework Directive (WFD) requires 'no deterioration' in ecological status, which is perhaps reflected more accurately by biological water quality, rather than chemical water quality.

Atmospheric deposition has been and continues to be a major pressure on upland water quality. However, concern has been expressed recently about the impacts of agriculturally derived diffuse pollution. For example, of 40 priority catchments selected in Defra's Catchment Sensitive Farming Initiative, eight are upland catchments with a variety of nutrient and sediment related issues, and several of which contain SSSIs.

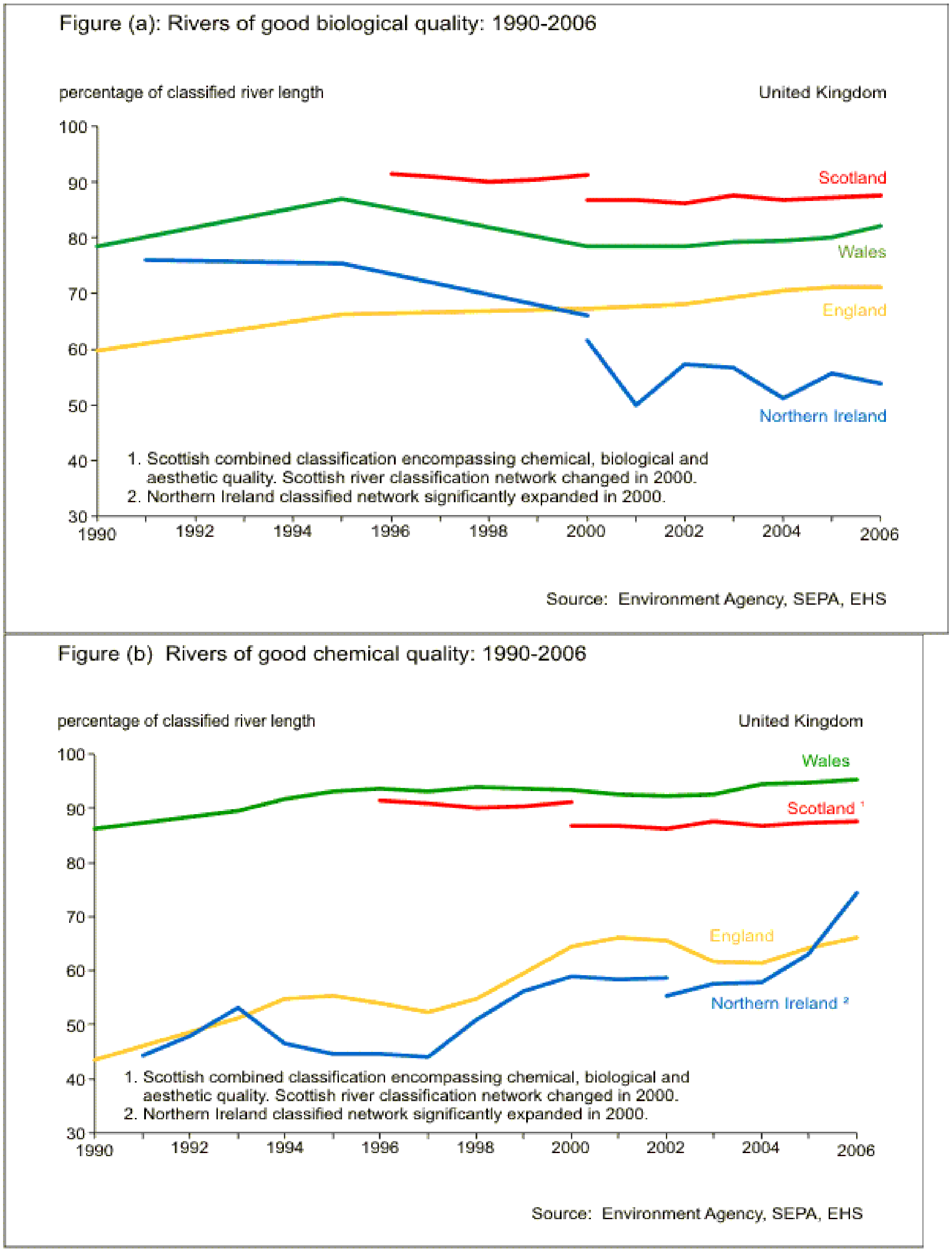

Figure 4 Rivers of good (a) biological and (b) chemical water quality between 1990 and 2006

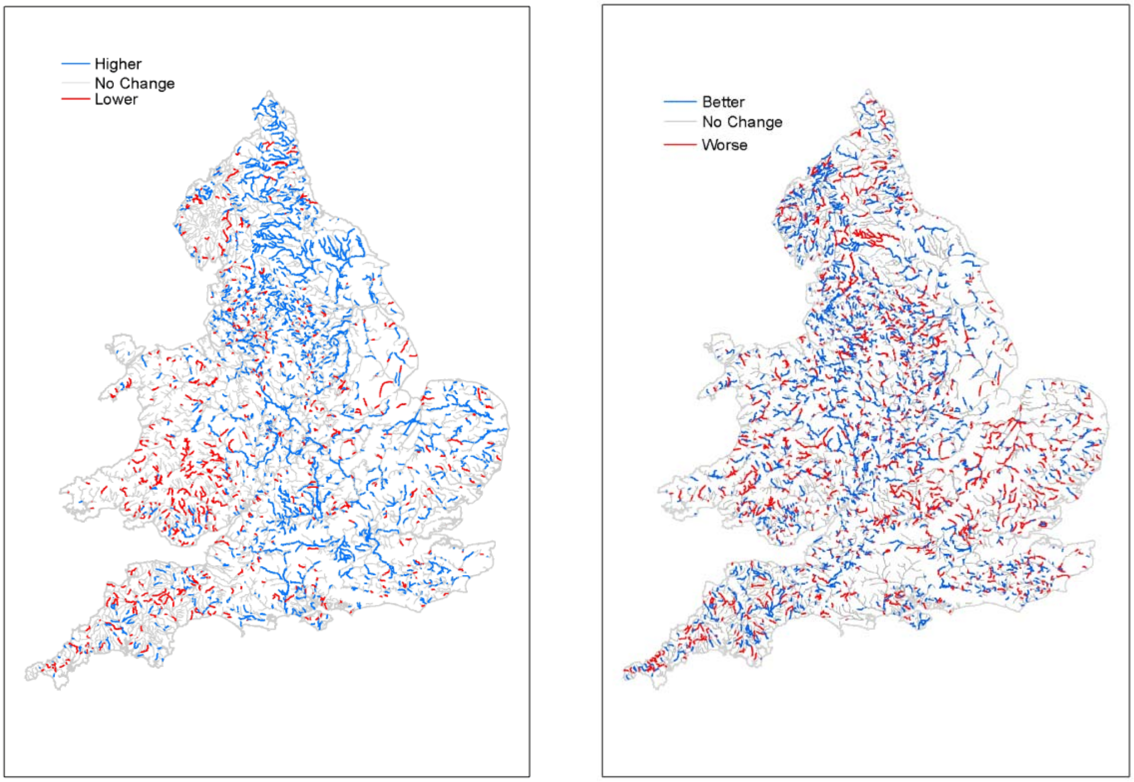

Figure 5 Changes in biological water quality (left) and phosphate concentrations (right) between 2000 and 2006

There is a limited literature dealing directly with the impacts of upland farming practice on water quality and limited knowledge of the inter-annual variation in water quality parameters in small upland catchments. They are likely to be very variable and as a consequence the impact of management activity may be obscured by inter-annual variation. The majority of studies reporting on the quality of water draining from upland areas have focused on other environmental issues, for example acidification (Reynolds et al., 1986), and have assumed that upland agriculture is a relatively benign activity compared with other forms of upland land use, such as plantation conifer forestry. As a result, most studies have relied on monitoring water quality at the outlet of upland catchments and providing a fairly generalised picture of the water quality effects of upland agriculture. For example, in the Dee catchment in Scotland it was apparent that upland sites dominated by moorland had lower concentrations of phosphorus and sediment than sites where agriculture was more intense (Stutter et al., 2007). More detailed investigations of the implications of agricultural practice for the ecology of the uplands have largely been dominated by terrestrial ecologists, who have been concerned with the impacts of grazing intensity on above- and below-ground biodiversity and nutrient cycling (see Bardgett et al., 2001; and Bardgett and Wardle, 2003). Some studies do exist which consider the impact of intensive grazing of lowland grassland areas of the UK and elsewhere in the world – for a review see Bilotta et al. (2007). These studies can indicate how upland grasslands are likely to behave. The lack of previous work in this area means that Section 2 of this review relies on observations made by the study team on farm visits and in discussion with upland farmers, regulators and water utility companies, as well as related literature.

The following activities are seen as potential threats to water quality from upland farming systems and are recognised by the farming community as having wider environmental impacts (IEEP and LUC, 2003):

- overstocking and poor shepherding
- slurry spreading
- poor nutrient management
- stock access to streams
- sheep dipping
- land improvement, including re-draining
- static supplementary feeding
- outdoor lambing
- herbicide use

Additional and largely unrecognised threats include water abstraction, increased water temperature and lack of riparian shading.

As the in-bye is more intensively managed than the out-bye, there is potential for higher threats from the in-bye areas than from the out-bye. However, even in in-bye land, management is generally of a lower intensity than lowland areas.

Section 2 will evaluate the impact of different farm and land management practices on water quality.

2. The impact of key land management practices on water quality

The impact of different land management practices on potential pollutants is outlined in Table 2, is briefly described below and in more detail in the following sections. Here, major direct and indirect (e.g. by increased erosion) impacts on water quality for important pollutants are identified for major upland grassland land uses. This table draws on evidence from published papers and reports together with expert opinion, but also highlights that there are some areas that have received little or no research attention. None of the land uses investigated have been thoroughly researched in the uplands of the UK, and although there is some evidence there is clearly a need for further research into the impacts of grassland management on water quality.

The table shows that high-intensity grazing generally has a negative impact on water quality, as does winter grazing (and year-round grazing). The impact of animal type is very difficult to quantify. However, there is evidence to suggest that sheep are more likely to increase surface compaction and increase overland flow (Carroll et al., 2004a). As sheep are currently the most extensively farmed animal in the uplands this is an important factor, although this may change in the future, as animal numbers are strongly driven by agricultural policy. The introduction of nutrients, whether through fertiliser, manure or supplementary feed, is also likely to have a negative impact on water quality. There are positive steps that can be taken to improve water quality, including preventing stock access to streams and reducing the use of static supplementary feeding.

As with grazing, grass cropping is unlikely to have negative impacts on water quality when it is not intensively managed, but the addition of fertilisers, manure and biocides are likely to have a negative impact on water quality.

Recreational use of the uplands is also unlikely to impact on water quality where it is not intensive; however, areas that experience high visitor numbers, such as the National Parks, are likely to experience soil erosion and associated problems and an increased incidence of fire.

Table 2 also examines landscape features that are important for the transfer of pollutants. Tracks provide an important pathway for the rapid transport of sediment and pollutants to water bodies. Boundaries can also provide pathways for concentrated flow, especially where animals tend to congregate close to them; however, they also play an important role in preventing stock access to watercourses.

Little is known about upland semi-natural and improved grassland systems where sheep grazing dominates and pollutant concentrations and fluxes are much smaller than in lowland agricultural systems. Much can be learned by inference from lowland studies; however, better quantification of fluxes and pathways is required for the uplands in order to fully understand the effects of grassland management on the quality of naturally oligotrophic water bodies. Even for relatively well studied intensively managed grasslands, the relationships between on-site impacts of grazing animals on soils and vegetation and the downstream effects on water quality are poorly quantified, with little support from research (Bilotta et al., 2007).

A number of key practices for water quality are discussed in further detail below.

Table 2 The state of knowledge of land-use impacts on pollutant losses from grass uplands

	Grazing						
	Density	Timing	Animal type	Fertiliser	Manure management	Outdoor lambing	Supplementary Feed
Sediment	↑ with ↑ intensity Bilotta et al., 2007; Strebel et al., 1989	↑ with ↑ grazing period and winter grazing McDowell, 2006b	↑ surface runoff with sheep, less likely to poach Betteridge et al., 1999; Crofts and Jefferson, 1999	n/a	?	?	Potential ↑ with static feeding
Nitrogen	↑ with ↑ intensity Kurz et al., 2005	↑ with ↑ period Kurz et al., 2005	↑ with sheep McDowell, 2006a	↑ with fertiliser addition Preedy et al., 2001	↑ with manure addition Heathwaite et al., 1998	?	Potential ↑ with static feeding
Phosphorus	↑ with ↑ intensity Kurz et al., 2005	↑ with ↑ period Kurz et al., 2005	↑ with sheep (per animal unit) McDowell, 2006a	↑ with fertiliser addition Preedy et al., 2001	↑ with manure addition Heathwaite et al., 1998	?	Potential ↑ with static feeding Kirkham, 2006
Metals	↑ with ↑ erosion Rothwell et al., 2005	↑ with ↑ erosion Rothwell et al., 2005	↑ surface runoff with sheep (per animal unit) Betteridge et al., 1999	↑ with ↑ acidity Dise et al., 2001	↑ with ↑ acidity Dise et al., 2001	?	↑ with ↑ erosion Rothwell et al., 2005
Faecal Indicator Organisms	Possible ↑ with ↑ intensity Sturdee et al., 2007	Possible ↑ with ↑ period Sturdee et al., 2007	↑ with young animals Sturdee et al., 2007	n/a	↑ with manure spreading Vinten et al., 2004	↑ with outdoor lambing Sturdee et al., 2007	Potential ↑ with static feeding
Biocides	↑ with ↑ erosion	↑ with ↑ erosion	n/a	n/a	n/a	?	↑ with ↑ erosion
Sheep dip	↑ risk with ↑ intensity Virtue and Clayton, 1997	n/a	Sheep only	n/a	n/a	?	n/a
Veterinary medicines	↑ risk with ↑ intensity Jones et al., 2004	↑ with ↑ period Jones et al., 2004	?	n/a	↑ with manure spreading Burkhardt et al., 2005	↓ with outdoor lambing	Potential ↑ with static feeding
Acidity	?	?	n/a	↑ with N and S additions as with deposition	↑ with N and S additions as with deposition	n/a	n/a
Carbon and Colour	No change with intensity Worrall et al., 2007	?	?	↑ with fertilisation Weaver and Reed, 1998	↑ with manure Parks et al., 1997	?	Potential ↑ with static feeding

↑ indicates an increase, ↓ a decrease and ? insufficient or contradictory evidence. Direct effects are coloured yellow and indirect effects are coloured blue. Where there is some corroborating research, example references have been included.

Table 2. The state of knowledge of land-use impacts on pollutant losses from grass uplands (continued)

	Grass Cropping		Recreation	Landscape features	
	Silage	Hay	Recreation	Tracks	Boundaries
Sediment	Low losses	Low losses	↑ with increased visitor pressure McEvoy et al., 2006	↑ with tracks and paths	↑ surface runoff at boundaries McDowell et al., 2004 ↓ if prevents access to streams Scrimgeour and Kendall, 2002
Nitrogen	↑ with nutrient addition Cuttle et al., 1996	Low losses	?	↑ transport with tracks and paths Hooda et al., 2000	↓ if prevents access to streams Scrimgeour and Kendall, 2002
Phosphorus	↑ with nutrient addition Cuttle et al., 1996	Low losses	↑ with increased erosion Quinton et al., 2001	↑ with tracks and paths	↓ if prevents access to streams Scrimgeour and Kendall, 2002
Metals	Low losses	Low losses	↑ with ↑ erosion Rothwell et al., 2005	↑ with ↑ erosion Rothwell et al., 2005	? due to erosion
Faecal Indicator Organisms	↑ with manure application Vinten et al., 2004	Low losses	n/a	↑ transport with tracks and paths Hooda et al., 2000	↓ if prevents access to streams Scrimgeour and Kendall, 2002 ↑ with animal congregation at boundaries McDowell et al., 2004
Biocides	↑ with herbicide application	Low losses	n/a	↑ transport with tracks and paths Hooda et al., 2000	?
Sheep dip	n/a	n/a	n/a	↑ transport with tracks and paths Hooda et al., 2000	n/a
Veterinary medicines	↑ with manure application Jones et al., 2004	Low losses	n/a	↑ transport with tracks and paths Hooda et al., 2000	?
Acidity	↓ with liming	n/a	n/a	↑ transport with tracks and paths Hooda et al., 2000	n/a
Carbon and Colour	Low losses	Low losses	↑ with ↑ erosion	↑ with tracks and paths	?

↑ indicates an increase, ↓ a decrease and ? insufficient or contradictory evidence. Direct effects are coloured yellow and indirect effects are coloured blue. Where there is some corroborating research, example references have been included.

2.1 Grazing

2.1.1 Overgrazing and poor shepherding

Out-bye land is almost exclusively grazed by sheep, with some supplementary feeding practised. In many areas, this land is common land or owned by large landowners like the water utility companies or the National Trust. This often means that grazing is well controlled. However, grazing intensity and timing has changed considerably in upland Britain, with market forces and government and European Common Agricultural Policy (CAP) subsidies resulting in increased animal numbers and year-round grazing (Holden et al., 2007a; Emmett and Ferrier, 2004) (Figure 6). The numbers of grazing animals recorded in the parish agricultural census give some indication of the potential pressures on water quality. Data from the parishes of the Lune catchment show the huge increases in sheep numbers for this upland catchment from the mid-nineteenth century, with a more rapid rise in recent decades (Figure 7). Comparing the livestock numbers with the amount of land available indicates grazing intensity (Figure 8). For comparison, these figures far outstrip the Environmentally Sensitive Area scheme recommendation of stocking levels not exceeding 1.5 ewes per hectare (minimum of 0.3 ha per sheep).

In the near future, the loss of the hill farm payment from 2008 and the switch to Environmental Stewardship (ES) is likely to reduce sheep numbers.

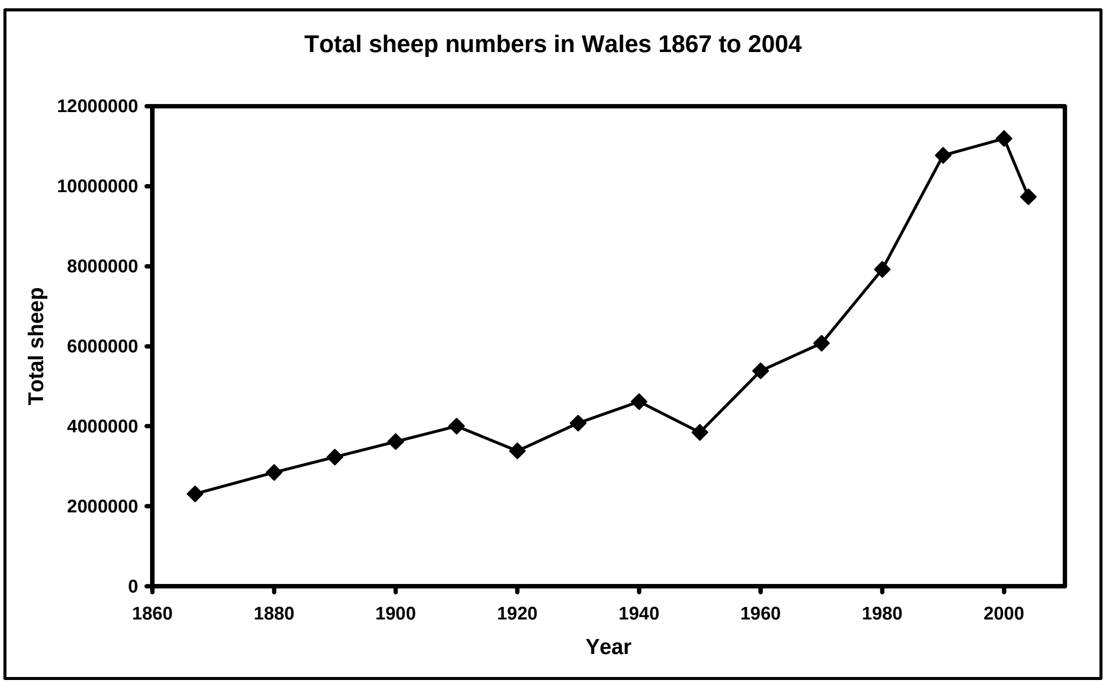

Figure 6 Total sheep numbers in Wales from 1867 to 2004 (Redrawn from data provided by Prof. Gareth Edwards-Jones, School of Environment and Natural Resources, Bangor University)

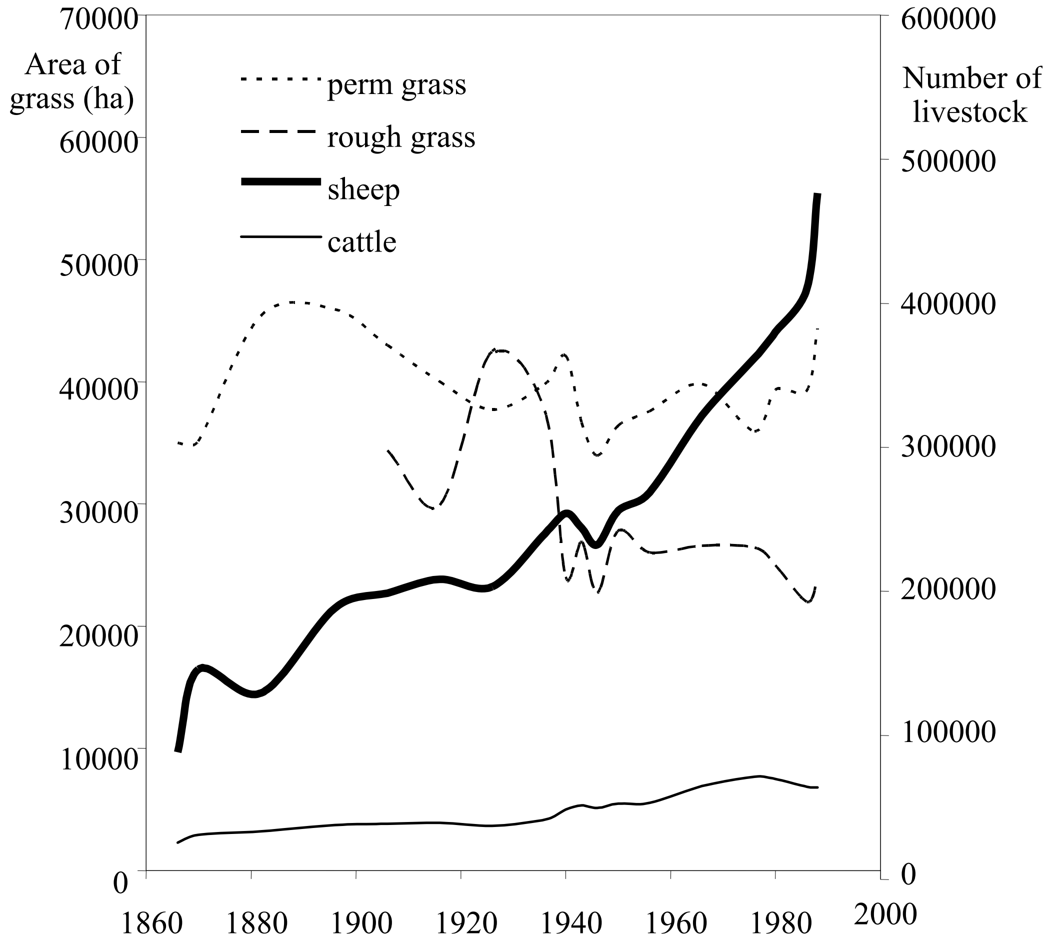

Figure 7 Cumulative agricultural statistics for the Lune catchment from 1860 to 1998 (from Orr and Carling , 2006)

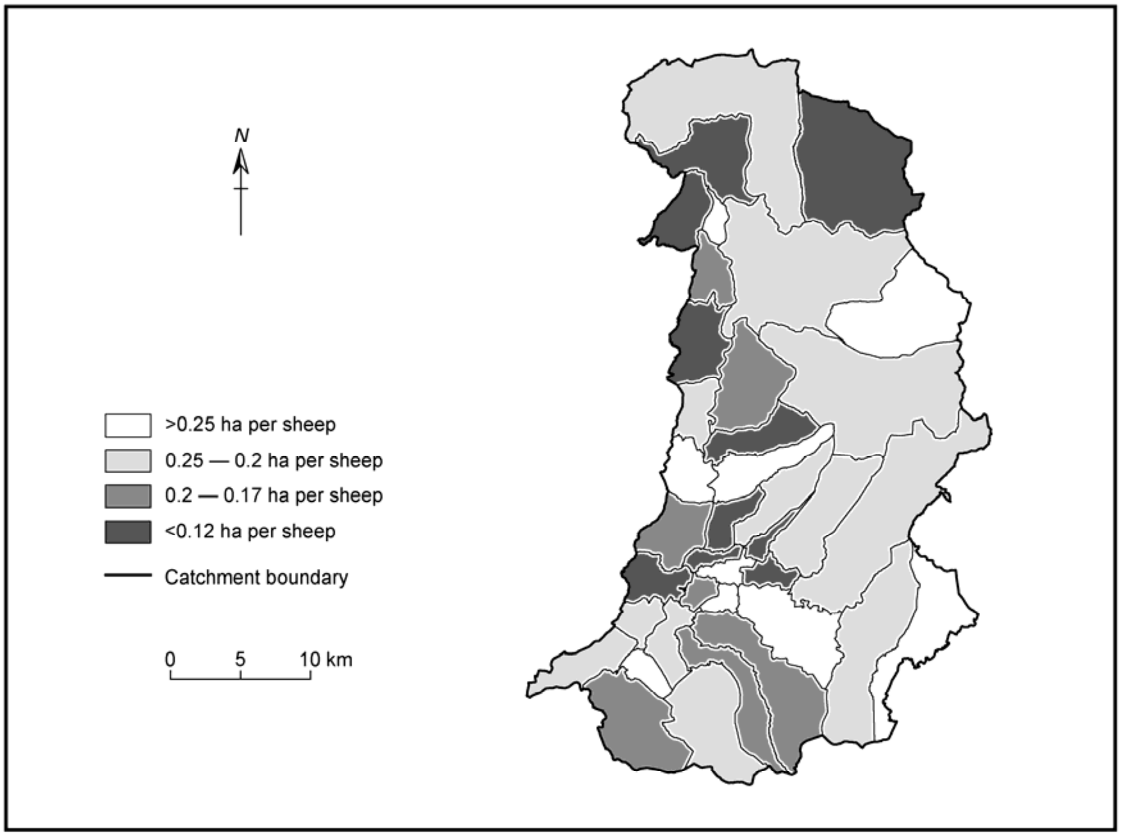

Figure 8 Maximum areas of grazing available for sheep in the Lune catchment, north-west England (from Orr and Carling, 2006)

Some out-byes now have stock on them throughout the year and although mean grazing densities are lower than in the in-bye, sheep are prone to concentrate where there is better quality grazing when not shepherded well (Evans, 1998). Where this occurs, vegetation cover may be removed and the soil compacted, resulting in greater surface runoff generation and soil erosion (Bilotta et al., 2007; Holden et al., 2007b) transporting nutrients, sediment, faecal contaminants, veterinary medicines, metals (Rothwell et al., 2005) and organic matter. Surface runoff will also transport nutrients associated with the soil and dung (nitrate, ammonium and phosphorus), faecal contaminants (Tyrrel and Quinton, 2003), veterinary medicines and particulate organic matter.

Grasslands have been implicated in the failure of some beaches to meet bathing water standards (Vinten et al., 2004) and are thought to be partially responsible for delivering fine sediment and phosphorus to upland lakes, e.g. Bassenthwaite Lake (Orr et al., 2004). However, there are few studies in the literature which have experimentally determined the impact of grazing pressure (number of animals and duration of grazing) on the loss of contaminants. Surveys in the uplands found that grazing animals were the principle cause of soil erosion (McHugh et al., 2002a) and work in the Bassenthwaite catchment (Orr et al., 2004) showed that the area of bare ground had increased by 4% from 1970 to 2000, although it cannot be determined whether sheep grazing was responsible. Studies at Bleham Tarn in the Lake District support the link between grazing and soil erosion (van der Post et al., 1997). There is direct evidence that overgrazing is affecting soil structure and function in upland soils (Emmett and Ferrier, 2004). The impacts of sheep grazing on soil physical properties are reviewed by Carroll et al. (2004a), who point out that the results of studies in the literature are not consistent. Carroll et al. (2004a; 2004b) demonstrated that infiltration rates were significantly lower and soil compaction was significantly higher with high stocking densities at Pwllperian, Wales (Carroll, 2004a). Similar results were found in Moor House National Nature Reserve (Worrall et al., 2007). Other sites investigated at Snowdonia and Pontbren in Wales gave a less clear picture, suggesting soil properties have an important role to play in determining the impact of stocking density (Carroll, 2004a). This decrease in infiltration combined with a loss of vegetation cover at high grazing intensity can lead to increased overland flow (Burt and Gardiner, 1984a and 1984b) and nutrient transport (Kurz et al., 2005).

Different animal species affect the physical properties of the soil surface (and hence infiltration) differently. Although all grazing animals have the potential to compact soil, cow hooves tend to cause a large amount of surface disturbance with upward and downward movement of the soil. Sheep cause a greater degree of surface compaction (Betteridge et al., 1999). In addition, the grazing of different species has different impacts on vegetation and can considerably change the sward composition and structure (Holden et al., 2007a). However, the consequences of this for water quality have not been investigated.

Studies at catchment scale have not provided conclusive evidence of a link between grazing pressure and faecal indicator organisms. For example, work by Hunter et al. (1999) in an upland catchment in Derbyshire with both improved, semi-improved and rough grazing could not find a relationship between the intensity of grazing and the concentration of faecal coliforms in a stream. Sturdee et al. (2007), working in the English Lake District, also found it difficult to associate the contamination of waters by *Cryptosporidium* oocysts with grazing intensity. In their work they considered four micro-catchments dominated by improved land (in-bye); steep slope grazing; grazed wet moorland; and enclosed woodland, fenced for the previous 20 years to prevent livestock and deer access. All micro-catchments had low electrolyte concentrations, close to neutral pH and low nutrient status. However, *Cryptosporidium* oocysts were found at all sampling sites and especially at the in-bye site where animals collected for lambing, dipping and other livestock operations. The study found that most of the oocysts were shed from small wild mammals (28%) rather than livestock, although calves (15.7%) and lambs (8.1%) were also significant sources. Adult livestock (1.8%) and large wild mammals (4.8%) were less important. Large numbers of oocysts were also collected from overland flow and from runoff along farm tracks, demonstrating the importance of overland flow as a potential pathway for oocyst transport to the stream.

Grazing animals excrete urine in patches, and as the number of animals increases there is a greater chance of patches overlapping. This increases the possibility of nitrate leaching. By increasing the grazing intensity on a given area of land there is an increased potential for nitrate leaching (Pleasants et al., 2007; Strebel et al., 1989). The length of the period of grazing and the grazed species (and their feed) will also affect the amount of leaching. This is partly because the nutrient content of the faeces varies between species (Williams and Haynes, 1995). McDowell (2006a) investigated contaminants in overland flow from dung and found losses of total phosphorus, nitrogen and sediment were greatest from cattle, but that dissolved reactive phosphorus and *E. coli* was greatest from sheep.

There has been little investigation of the impact of the timing of grazing on water quality in the UK. In New Zealand, winter grazing has been shown to increase sediment losses by up to 75%, although this had little impact on phosphorus losses (McDowell, 2006b). This research was not conducted in an upland system, so we do not know how applicable it would be in UK upland grasslands. It is likely, however, that through increased erosion risk many of the problems associated with grazing animals will be magnified in the winter, and without supplementary feeding (see Section 2.1.4) animal health is put at risk (Croft and Jefferson, 1999). One of the few studies to take place in upland areas of the UK was carried out at the Bronyydd Mawr Research Centre near Brecon, Wales, at an altitude of 335 m. Leaching losses, estimated from suction samplers, of N ranged from 0.1 to 226 kg N ha^{-1}. This was attributed to stock not being evenly spread over the site. Where the sheep congregated, leaching losses were highest (13–24 kg N ha^{-1}) (Cuttle et al., 1998). Nitrate losses were not correlated with stocking rate; this is in contrast to work on a lowland (35 m altitude) site near Aberystwyth (Cuttle et al., 1998), which received 200 kg N ha^{-1} of fertiliser, and a ryegrass/white clover (*Trifolium repens* L.) pasture, which received no fertiliser at all. Cuttle et al. found that nitrate losses were similar at 6–34 kg N ha^{-1} y^{-1} and 2.5 kg N ha^{-1} y^{-1} from the fertilised and non-fertilised pastures respectively, and were positively correlated with the number of lamb grazing days between late June and the end of the grazing period. Cuttle et al. also found that there was no direct influence of using rotational rather than continuous grazing, although, as he points out, this is in contrast to work in New Zealand (Brock et al., 1990).

It is clear that overstocking will reduce vegetation cover and damage soil structure, and that this is likely to lead to increased runoff and contaminant transport. However, there has been insufficient study of the impact of stocking density on soil physical conditions and the link to runoff generation and contaminant transport in the uplands. The impact of the timing of grazing has received even less attention. There is an urgent need for such studies to be carried out across a range of soil and vegetation types. This should be integrated with the need to develop climate change adaptation measures that can reduce flood flows and augment drought flows.

2.1.2 Access to watercourses, tracks and boundaries

Allowing stock direct access to streams is a contributing factor in the transfer of pollutants, allowing nutrients, organic matter and bacteria to enter streams directly. Sturdee et al. (2007), Kay et al. (2007) and Oliver et al. (2007) all highlight the presence of stock in streams as a mechanism for polluting upland streams and rivers with faecal contaminants. Allowing direct access to streams also reduces riparian vegetation and increases bank erosion. This is potentially the most serious impact and studies across the USA have shown a reduction of up to 77% in stream bank erosion with the removal of livestock (Scrimgeour and Kendall, 2002). Fencing riparian areas to prevent stock access could potentially reduce stock impact, although in out-bye areas this is not always a practicable solution. It should be noted that excluding stock from riparian areas can deliver a wide range of other benefits, including increases in biodiversity.

There is potential for boundaries to create their own problems. Some animals will spend more time at boundaries, which can result in increased compaction and erosion (McDowell et al., 2004). However, Carroll et al. (2004b) did not observe any difference in compaction and infiltration when they investigated this in Montgomeryshire, Wales. They also found that the use of tree shelter belts had a very positive influence on infiltration rates in grazed pastures, with infiltration rates increasing within two to six years of establishment.

If they are connected, tracks may allow a rapid transfer route to watercourses. This applies not only to man-made tracks but also to sheep tracks on hillslopes (Hooda et al., 2000). If tracks and gateways are poorly sited they can have a negative impact on water quality.

2.1.3 Sheep dipping

Sheep dipping has undeniable benefits for animal health, but also presents a risk to water quality, especially if the dipping is not conducted with due care or if sheep dip is inappropriately disposed of. Organo-phosphate dips represent the greatest threat to water quality and these have already been banned, but some organisations, such as the Salmon and Trout Association, would like a complete ban on all sheep dips.

Sheep dips have been identified in waters in several upland areas, for example River Tweed catchments (Virtue and Clayton, 1997), the Grampian region of Scotland (Littlejohn and Melvin, 1991) and parts of Wales where the Environment Agency found that poor operational practices were widespread (Environment Agency, 2007). Dips can be responsible for killing fish and reducing stream macroinvertebrates (Virtue and Clayton, 1997). In order to minimise losses of sheep dip chemicals to the environment, it is essential to follow codes of good agricultural practice; this includes not using soakaways to dispose of dip (Hooda et al., 2000) or disposing directly into a waterway, avoiding poor siting of dipping facilities near to waterways (Virtue and Clayton, 1997) and allowing time for the sheep to stop dripping before they are moved to holding areas (Sinclair et al., 2007). Although there is clear guidance on sheep dipping available to farmers, this is not always adhered to and it may not be sufficient to prevent dip entering water bodies. Problems have recently been reported in Wales (Environment Agency, 2007), but the continuing extent and severity of the impact of sheep dipping on water quality are unclear. Prevention measures are known, but investment in infrastructure and short-term holding areas will help improve practice and reduce the chance of accidental spills.

2.1.4 Supplementary feeding

Supplementary feeds are used where grazing is not sufficient to meet the dietary requirements of the animals. It is generally used in winter (December to March) and can be used to supply energy, proteins and minerals.

Supplementary feeds can take different forms and the impact of them on water quality varies greatly. In upland valley meadows, traditional practices involve supplemental winter feeding with hay. This replaces nutrients removed by a summer cut and is an important part of the grassland management system (Jefferson, 2005). More commonly, supplementary feeding involves the supply of hay, commercial feeds (such as

Rumevite), cereal- or protein-based concentrates and mineral licks. These all introduce minerals and nutrients to the grassland via increased N and P content of animal excreta.

Supplementary feeding can take the form of static feeding or it can be moved around to many or few locations. Static feeding may take place on hardstandings. Kirkham (2006) reports areas of 5–20 m^2 of bare ground during winter months around supplementary feeding areas, with poaching covering a similar area. Kirkham also presents evidence of increased overgrazing in the vicinity of supplementary feeds and an increased concentration of dung. The impacts of animals concentrating in one area are discussed above (Section 2.1.1) and include potential for increased leaching and overland flow. These effects are likely to be most apparent where fixed feeding stations are used.

Kirkham (2006) makes the following recommendations for minimising the environmental impact of supplementary feeding: choose a supplement that is suitable for stock requirements; choose mineral licks with low phosphorus content and only use them when needed; and consider whether the stock type and timing of grazing is appropriate in sensitive habitats such as SSSIs.

If supplementary feeding areas are connected hydrologically with waterbodies, supplementary feeding clearly has a potentially negative impact on water quality: a poorly located supplementary feed area has the potential to act as a direct source of nutrients, FIO, sediment, veterinary medicines and DOC, as well as other pollutants mobilised with eroded sediment. Despite this, this area has received little research attention. Without further research, the degree to which supplementary feeding is a source of pollutants is unknown, as is the fate of these pollutants. Advice is available to farmers, but the evidence base for this advice is not strong.

2.1.5. Herbicide use

Herbicides are used in grasslands for both grazing and grass cropping systems; this tends to be for targeted weed control for species such as ragwort (*Senecio jabobea*), thistles (*Cirsium* spp.), dock (*Rumex* spp.) and bracken (*Pteridium aquilinum*). Application may take the form of spot treatment or large-scale spraying. Guidelines are available to farmers on the use of herbicides and alternative methods of weed control (e.g. DEFRA, 2007; Croft and Jefferson, 1999) and herbicide use by farmers is not extensive in upland areas.

There has been no research relating herbicide use in upland grasslands to water quality. Research in row and combinable crop systems has highlighted the main pathways for losses as being spray drift, overland flow and leaching of biocides and their metabolites. The degree of spray drift will depend to a large degree on the application method. Herbicide application on a large scale is unlikely to be common in the uplands, except for bracken control, where herbicides are commonly spread over large areas using aerial application techniques and spray drift could be an issue .

In lowlands, Vincent et al. (2007) compared the transport of herbicides (those targeting arable weeds) in different land uses. They found that there was similar mobility within the soil between grassland and combinable crops. This means that there is potential for herbicides to reach watercourses through leaching, although upland soils may behave differently.

Overland flow does occur in grasslands and is a potential problem where poor vegetation cover and tracks provide suitable conditions. Spraying for bracken can leave areas of bare litter, especially at low grazing intensity (Pakeman et al., 1997). All bare areas present a considerable erosion risk.

2.1.6 Poor nutrient management

The careful management of nutrients is an important part of farm practice and essential for both in-bye grass production (for grazing and silage) and protecting water quality. Nitrogen, as nitrate, is vulnerable to leaching and transport to surface and groundwaters by subsurface flow, including drains. Phosphorus binds more strongly to soil particles and is therefore more vulnerable to transport by overland flow, although recent evidence from arable situations suggests that substantial amounts of P are also lost with colloids via drainage systems (Deasy, 2007) and this may also be the case in drained upland grassland (CEH unpublished data; 2007).

Upland soils are generally low in P and upland aquatic systems are P limited. However, where farming exists, P inputs are likely, particularly in the in-bye. A modelled budget for a hill sheep farm supporting 694 Blackface ewes indicated that approximately 0.28 kg P ha^{-1} yr^{-1} were being retained within the farm (Haygarth et al., 1998). The largest component of the budget was represented by P recycled through the plant-soil and plant-animal-soil systems. The largest input flux to the farm was in the form of P fertiliser to 47 ha of improved grassland, amounting to 8.57 kg P ha^{-1} yr^{-1}, whilst outputs to water from the same area amounted to 1.5 kg P ha^{-1} yr^{-1}. In contrast, losses to water from 238 ha of rough grazed *Molinia* grassland exported 0.54 kg P ha^{-1} yr^{-1}.

Upland farmers manage the inputs of nutrients from fertilisers, slurries and manures. These are normally applied only to in-bye areas, although there are some instances of the fertilisation of out-bye. The British Survey of Fertiliser Practice does not provide statistics for upland farms, but does give the average application rates of inorganic fertilisers to grass in sheep and cattle farms (which are likely to be similar to in-bye applications) as 44 kg N ha^{-1} and 13 kg P ha^{-1} (Goodlass & Welch, 2007). According to Goodlass and Welch (2007), applications of N typically take place during the spring and early summer when the grass is actively growing, with applications of P taking place in the early autumn.

Where nutrients are applied in the form of slurries and manures, practical reasons (such as proximity to the farmyard or being better draining) may lead to some fields receiving more manure or slurries than others. Farmers may also not always consider the nutrient content of slurries in their nutrient planning. This can lead to some fields becoming enriched in N and P. However, no studies could be found of the nutrient content of in-bye areas.

Incidental losses of fertilisers and slurries are also likely to occur on upland farms. These losses occur when an overland flow event follows the application of fertiliser, slurry or manure to the soil surface. However, any evidence for this is anecdotal rather than based on quantified work. However, we can hypothesise that due to the higher frequency and intensity of rainfall in upland regions (e.g. Malby et al., 2006), these areas are more vulnerable than lowlands to this type of nutrient transfer.

2.1.7 Outdoor lambing

There have been no studies of the impact of indoor versus outdoor lambing on water quality in the uplands. However, there are some potential benefits of indoor lambing, including a reduction of poaching due to lower stock numbers outside during the lambing period. Lambs are also a source of *Cryptosporidium* oocysts and keeping them indoors may reduce the risk of *Cryptosporidium* and other FIOs reaching surface waters. The Scamp project (United Utilities, 2007) has given farmers grants to enable them to construct indoor lambing sheds. Indoor lambing carries a higher risk of disease so may not be popular with farmers. If there is a hydrological connection, there may also be an increased risk of veterinary medicines entering watercourses.

2.2. Grass cropping

There is very little information published on the impact of upland grasslands managed for hay and silage on water quality, but where meadows are not intensively managed their impact is likely to be relatively low. Indeed, hay harvest is promoted as a land use that can be used in place of grazing to reduce soil losses (Haan et al., 2006). The high percentage of vegetation cover and undisturbed soil structure mean that sediment losses (and associated pollutants) are likely to be low. Management practices that maintain high levels of soil organic matter, litter and plant cover improve infiltration (Neath et al., 1990). Nutrient (nitrogen and phosphorus) losses are likely to be very low where systems are not managed intensively.

Aftermath grazing (following hay or silage removal) tends to be at a low intensity and although there is an increased pollution risk associated with grazing animals, as long as they do not have direct access to a watercourse the low-intensity nature of this grazing means that the impact is likely to be minimal.

Management for silage tends to be more intensive than traditional hay management, involving the use of manure spreading, artificial fertilisers and herbicides. The impact of nutrient amendments on grasslands has been discussed above in Section 2.1.6. and herbicide use in Section 2.1.5. These impacts would be seen in grasslands managed for grass crops as well as for grazing.

2.3 Slurry spreading

The production of slurry is largely associated with dairy and beef cattle. Slurry is stored and spread onto in-bye fields during the winter months and after they have been cut for silage or hay in the spring and summer. There are two major risks to water quality associated with slurry spreading: that a surface runoff event occurs soon after slurry has been spread; and that soils accumulate nutrients because the nutrient content of slurries is not always taken into account by farmers. The latter point is dealt with under nutrient management (Section 2.1.6).

Losses of slurry in surface runoff and the potential for faecal contamination of surface waters has been described by a number of authors (Heinonen-Tanski and Uusi-Kämppä, 2001; Vinten et al., 2004). Losses of faecal organisms will be lower the longer the period between application and runoff event. Studies in lowland Scotland (Vinten et al., 2004) found the *E. coli* concentrations (14 c.f.u. ml^{-1} or 0.03% of estimated faecal input) in drains where slurries had been applied (36 m^3 ha^{-1}) to plot experiments were lower than the 14 c.f.u. ml^{-1} or 0.4% of estimated total *E. coli* inputs over the grazing from sheep grazed (4 ha^{-1}) on similar plots. This is attributed to greater die off in the slurry during storage. However, during a surface runoff event that occurred shortly after the spreading of the slurry the situation reversed, with 48 c.f.u. ml^{-1} found in the small amount of surface runoff from slurry plots, compared with 6 c.f.u. ml^{-1} from grazed plots. There is also likely to be an increased incidence of veterinary medicines reaching the watercourse. Veterinary medicines persist in the soil and can be lost by both overland flow and leaching, but as they have received very little research attention the potential for pollution is not fully understood (Jones et al., 2004).

No studies could be found that quantified the losses of N and P from upland grasslands with slurry applied to them. However, evidence does exist from other parts of Europe and from lowland grasslands in the UK. Work in Finland (Turtola and Kemppainen, 1998) where cow slurry was applied in autumn, winter and spring showed that, in the four years of grass ley, losses from slurries were highest in the autumn and spring, with 11% and 13% of the applied N and 17% and 59% of the applied P lost after autumn and winter applications of slurry. When overland flow occurs, N and P losses tend to be high from grasslands that have had slurries applied to them. Applications of slurry (50 m^3 ha^{-1}) to a grassland soil in Devon during May followed by four simulated rainfall events of 12.8 mm in 35 minutes once a day on days 3 to 6 after the slurry applications resulted in much lower losses of TN (3.3%) and TP (0.3%). However, losses from slurry plots were higher than those from farmyard manure plots. The presence of 10 m buffer strips beneath the plots reduced losses of N by 75% and P by 10%, with most P remaining in the particulate or dissolved organic fraction (Heathwaite et al., 1998).

Slurries are also a source of readily oxidised organic matter (measured as BOD), ammonium, phosphorus and nitrate. Nevertheless, working on a spring barley crop, Parkes et al. (1997) found that when slurries infiltrate through the soil the BOD concentration can fall to 1% of the applied concentration, 20 times lower than the concentrations in surface runoff quoted by Parkes et al. for a study conducted by Sherwood and Fanning (1981). Peak concentrations of nitrate and ammonia were also high, particularly from plots receiving the greatest volumes of slurry (35 m^3 ha^{-1}). Drainage water nitrate-N concentrations in Parkes et al.'s (1997) study peaked at 28 mg l^{-1}.

The application of slurries to soils in saturated or frozen conditions is likely to increase the risk of surface water pollution. The code of good agricultural practice for water (Ministry of Agriculture, 1998) distinguishes between frozen hard soils and soils which have been frozen for less than 12 hours and are likely to thaw during the day. Slurry spreading is permitted on the latter but not the former. Many upland farmers cannot avoid spreading when conditions are unsuitable, as they have insufficient storage. In many parts of the uplands, it is common for farmers to spread on frozen soils: besides preventing damage to the structure of the soil, the idea is that the slurry will infiltrate as the soil thaws during the afternoon. There has been limited research into spreading slurries on frozen ground, although work in arable systems suggests that spreading when there is a light frost to 0.1 m can be beneficial; for late winter spreading deeper frost penetration can increase the ammonia contamination of drain flow (Parkes et al., 1997).

2.4 Land drainage

The extent of artificial drainage is poorly quantified in the UK, but it is believed to be extensive in the uplands, both in terms of surface grips and subsurface mole and tile drains (Holden et al., 2004*)*. The number of land drainage schemes in England and Wales increased considerably in the 1960s and 1970s, with the rate of artificial drainage exceeding 100,000 ha yr^{-1} at its peak between 1972 and the mid-1980s (Robinson & Armstrong, 1988). Drainage was implemented to increase productivity in uplands by improving the land for sheep grazing and grouse. Upland land drainage increased dramatically after the Second World War as a result of government grants. Other factors which contributed to the increase include the introduction of plastic pipes and mechanised installation techniques, the outcome of MAFF research demonstrating the benefits of drainage, political and economic pressures, and the Strutt report which identified that drainage was essential for many soils to achieve their agricultural potential (O'Connell et al., 2004). Due to concerns about the impact of upland drainage on water colour and on flooding, a proportion of upland surface drains have now been blocked; new drains are not usually created and re-drainage (reinstating naturally or artificially blocked drains) seems to be quite rare. The first sites were blocked 18 years ago but there has been a dramatic increase in blocking in the last five years. The rationale for blocking is to put upland sites into favourable condition, and failing that favourable management. Furthermore, there is no evidence that surface drains fulfilled the aims of their installation (Stewart & Lance, 1983).

At the national scale, artificial drainage pathways are poorly delineated, and outlets, if known, are rarely monitored. However, locally, individual farmers often retain their drainage maps so that it is possible to reconstruct the history and distribution of land drainage within individual holdings (Hadzilacos, 2004). These records rarely exist for surface drains, possibly because of their visibility from the land surface and the generally featureless landscape. Aerial photography can be used to identify upland surface drains, although the success of this is dependent on vegetation cover: drains can be obscured by mature heather. Furthermore, aerial photography, particularly during extended periods of dry weather, will often reveal the outline of subsurface drainage networks, the latter appearing as darker green linear features on colour photographs. One aerial photo study of the large AONB in the North Pennines found grip lengths totalling 8500 km in an area of 2000 km^2 (www.northpennines.org.uk). Most blocked surface drains are mapped. However, the maps are kept by various bodies (Natural England, RSPB, Peatscapes) in various forms (from paper maps to GIS layers) and there is no central archive (even within the same organisation).

Artificial drainage extends the natural drainage network, and may increase the mobilisation of sediment and pollutants and therefore have long-lasting hydrological effects, with implications for water quality (Holden et al., 2006). However, upland drainage channels also interrupt surface flow pathways, thus reducing runoff distances and potentially reducing erosion and transfer of sediment and pollutants. These changes in flow pathways have different effects in different catchments depending on the catchment characteristics, i.e. slope, ditch design and position in the channel network, vegetation and soil type (Holden et al., 2006).

Limited water quality data are available from drained upland catchments, and the studies that have been undertaken do not always classify the catchment vegetation type. Furthermore, upland catchments are often mosaics of different vegetation types and therefore cannot be isolated as 'grassland' catchments.

In addition to the impacts of drainage related to changes in hydrological connectivity, the effects of the associated lower water table impacts upon water quality. Surface drains can be significant contributors of sediment: a drained sub-catchment which constituted 7.3% of the total catchment area contributed 18.3% of the total catchment sediment yield (Holden et al., 2007a). In addition, several studies from peatland areas indicate that surface drainage increases DOC concentration (Mitchell & McDonald, 1992; Clausen, 1980). Increased leaching of ammonium, nitrate, calcium, magnesium and potassium has also been noted following surface drainage (Miller et al., 1996; Sallantaus, 1995; Lundin, 1991; Burt et al., 1990; Freeman et al., 1993).

In the light of the recent trend of drain-blocking, several studies have examined the impact of blocking on water quality, primarily DOC. These indicate that blocking reduces DOC flux by reducing flow, and the capacity for DOC production by decreasing the aerobic zone (Gibson, 2006; Armstrong et al., 2007). Blocking may also change water transport pathways: flow may originate from deeper peat in unblocked grips and therefore exhibit different water quality. Wallage et al. (2006) examined the DOC and colour of soil water at different depths from an intact slope, drained slope and drain-blocked slope and found significant differences in the colour to DOC ratio, thus indicating different water sources. Research by Baker et al. (in press) also found that DOC composition changed after ditch clearance confirming that blocking, either natural or deliberate, affects DOC. While data exist for the impact of blocking on soil water and drain water, there is limited understanding of the impact at the catchment scale. Given the dearth of information on in-stream processing of DOC, no robust inferences can be made regarding the impact of blocking at the river catchment scale. How far downstream these changes in runoff are propagated is a significant research gap identified by a recent study on land use impacts on flooding (O'Connell et al., 2005).

Although there is relatively little information available from upland catchments, drains in lowland catchments are known to be important contributors of sediment and pollutants (including P and pesticides) to the stream network, particularly in soils where macropores interact with the drainage system to produce an efficient transport route from hillslopes to stream (e.g. Dils and Heathwaite, 1999; Chapman et al., 2001). Limited data from upland improved grassland systems indicate that subsurface drains can act as significant conduits for nutrients and suspended sediments if there is a nutrient source (CEH unpublished data, 2007), and there is evidence which indicates that drains in upland soils may promote the release of N to runoff (Roberts et al., 1984). Although their impact on upland water quality is poorly understood, the extensive coverage and role of drains in promoting hydrological connectivity mean that contributions from artificial pathways may be critically important in sediment and pollutant delivery and represent a very significant gap in knowledge.

2.5 Recreation

Recreation is a very important economic activity in the uplands. Of the 13 national parks in England and Wales, 11 contain upland habitats; the Lake District National Park receives approximately 12 million visitors every year, while the Peak District National Park receives 30 million (McEvoy et al., 2006). Grasslands form an important component of this, and are mainly utilised for walking and shooting. Recreation can place a considerable strain on resources, with potential negative impacts on water quality. High visitor numbers place an increased strain on infrastructure and increase the risk of fire and erosion.

Footpath erosion is a problem that has long been recognised in popular upland areas, and a considerable amount of resources are invested in maintaining footpaths. There are few studies monitoring footpath erosion, but evidence from Scotland indicates that it is increasing, due to increased visitor pressure (Davidson and Grieve, 2004).

As footpaths are compacted and vegetation cover is reduced by walkers, soil is exposed to erosion and the paths become conduits for concentrated water flow. When this occurs there is further potential for erosion. Surveys of the Bassenthwaite catchment revealed high erosion rates on many paths, with almost 50% of paths on slopes being vulnerable to active erosion; mountain biking and fell walking in winter were particularly damaging (McEvoy et al., 2006).

Erosion of footpaths has the same impact on water quality as trampling by sheep. The eroded sediment enters waterways and carries with it nutrients (nitrogen and phosphorus), metals and carbon. The footpaths themselves also provide concentrated flow networks, allowing pollutants to be transported rapidly and at a greater capacity.

3. What are the relative contributions of topography, soil type and location to water quality in upland grasslands?

3.1 Topography

Topography can be considered an important control on water quality through its influence on soil type, erosion and hydrology. Steep slopes have thin soils with poor vegetation and are prone to erosion, for example through gullying (e.g. Chiverrell et al., 2007); but they are also areas where surface runoff is generated as water flows over, rather than infiltrates, the soil. Steep slopes may therefore be important source areas for sediment and associated pollutant transfer. Shallow slopes are often poorly drained, and may provide ideal conditions for the development of deep peats, which are particularly vulnerable to erosion (see Section 3.2).

The extent to which hillslopes act as source areas for sediment and pollutants depends on the degree of coupling between hillslopes and channels (e.g. Harvey, 2002). Where slope channel coupling is high (headwaters), or channel margins (including drains) are actively eroding, the potential for connectivity to sediment-associated pollutant source areas is high. However, both source areas and runoff pathways are variable (Evans and Warburton, 2005), and connectivity varies with runoff characteristics, slope and topography (e.g. Burt and Butcher, 1985). Upland streams and lakes usually have low nutrient status and may be highly sensitive to changes in stream chemistry. In addition, the amount of time taken for rainfall to travel through a catchment controls the retention of soluble contaminants. Catchments with short flushing times will deliver brief, intense contaminant pulses to downstream waters, whereas catchments with longer flushing times will deliver less intense but more sustained contaminant fluxes (Kirchner et al., 2000).

3.2 Soil type

Upland soils in the UK are organic or mineral, and include peats, peaty gleys, acid brown earths and related soil types formed over glacial till. Upland soils are predominantly acidic and of low nutrient and base status. Organic soils (peats) are particularly prone to erosion by wind and water, because of their light texture (e.g. Holden et al., 2007b). When waterlogged, soils are vegetated by wetland plants, which protect the soil surface. However, when soils dry out, crusting occurs and vegetation dies, leading to bare areas of soil that are vulnerable to erosion by both wind and water. Organic soils are important sources and sinks of carbon. The transfer of DOC, trihalomethanes and water colour are particular issues of concern for water quality in the UK (Worrall and Burt, 2005). Peatlands, which have been sinks for heavy metals in the past, have also been found to be sources of contaminants to aquatic systems (Rothwell et al., 2007). As a result, there have been a number of studies on erosion and runoff water quality from peatlands, particularly in the South Pennines and northern England (e.g. Worrall and Burt, 2004). However, the causes and mechanisms of soil carbon loss reported recently (e.g. Bellamy et al., 2005) are not yet clear for upland soils (Orr et al., in review). Mineral soils are not traditionally thought to be vulnerable to erosion, principally because they are well vegetated, therefore little water quality research has been carried out on these soil types. However, this assumption is now being challenged, as it is recognised that erosion does not take place only through surface pathways and that water quality is influenced in a number of ways. Work by Orr et al. (2004) suggests that 8% of a large area of the northern Lake District is experiencing erosion, where more than 50% of the eroded soils are mineral. McHugh et al. (2002) also identified problems with erosion in the uplands, related particularly to mineral soils.

The extent of erosion in both organic and mineral soils, as indicated by areas of bare ground, gullies and erosion scars, appears to be increasing from 4% by area in the 1970s to 8% by area in 2000 (e.g. Orr et al., 2004). However, rates of erosion over bare ground vary considerably, and these can also be areas of deposition (Whitehouse, 1978). Measurements of bare ground are not able to indicate the rate of erosion or depth of soil loss, or take account of erosion occurring over vegetated ground (McHugh, 2000; McHugh et al., 2002). Although McHugh's approach can present a broad spatial picture of erosion, it only identifies source areas and does not consider the connectivity of hillslope source areas to drainage networks and receiving surface waters, or link erosion loss to timescale (Warburton et al., 2003) – all critical parameters for understanding potential water quality impacts.

Soil type strongly influences drainage pathways. Many upland catchments with poorly drained soils are dominated by flashy storm responses, which indicate the importance of rapid transfer of runoff to the stream. The principal runoff pathway is usually thought to be surface runoff (e.g. Burt and Gardiner, 1984a and 1984b), which can contain high concentrations of sediment and pollutants. However, in peatland catchments, natural subsurface pathways in the form of soil pipes can act as preferential pathways for transfer of runoff to the stream (e.g. Jones, 2004). Soil pipes are thought to contribute between 10 and 50% of streamflow (Holden and Burt, 2002; Jones and Crane, 1984), but their importance in sediment and pollutant transfer is not yet known. Research from lowland catchments has shown that considerable amounts of runoff, sediment and pollutants can be transferred to the stream via preferential flow pathways, particularly in soils where macropores interact with the artificial drainage system to produce an efficient transport route from hillslopes to stream (e.g. Chapman et al., 2001). It is likely that natural soil pipes fulfil a similar role to artificial drains, and in some catchments, pipes may deliver more sediment and pollutants to surface waters than surface pathways. There is little information available on the extent of natural pipes in mineral soils, and although soil piping is believed to be most significant in highly organic peat soils, estimates suggest that 30% of the UK may be covered by soils susceptible to piping (Jones, 2004).

In non-drained mineral grassland catchments, the dominant runoff pathways are likely to be either surface or subsurface runoff from variable source areas. Throughflow pathways, although less important in sediment and P transfer than surface runoff due to sorption of pollutants and trapping of sediment by the soil matrix (Dils and Heathwaite, 1996), are important pathways for leaching of substances, particularly N, through the soil. Recent work in an upland catchment containing grassland and dwarf shrub communities has shown that dissolved organic nitrogen is the dominant form of nitrogen in soil and shallow groundwaters (Lapworth et al., 2008). The study attributed much of the spatial variation seen in N speciation in soil and groundwaters to microbial processes, whilst redox controls were important in saturated flushes in the lower hillslopes and valley bottom areas.

Leaching is partly determined by soil texture and organic matter content, and partly by the pollutant leaching potential. In general, unmodified organic soils are less prone to leaching, but catchments with mineral soils are at greater risk (Helliwell et al., 2007; Evans et al., 2006). Nitrogen cycling within soils and vegetation is complex and incompletely understood, but surface water DOC, as a proxy for the catchment carbon pool, can indicate the C : N ratio and hence soil sensitivity to N leaching (Evans et al., 2006). There is further evidence to suggest that vegetation type can affect the relationship between soil C : N ratio and nitrogen leaching (Rowe et al., 2006). This work showed that acid grassland and deciduous woodland began leaching nitrogen at lower C : N ratios compared to heathland and coniferous forest. It was suggested that this might be related to the reactivity of the soil carbon pool, such that soils with a large proportion of recalcitrant carbon begin to leach nitrogen at a higher C : N ratio than those containing more labile carbon. Thus adaptive strategies might include ensuring the soil carbon pool is maximised, whilst acknowledging the role of vegetation type. It is known that nitrate concentrations in upland waters display strong seasonal and inter-annual patterns of variation which can be related to climatic controls driven by global scale processes such as the North Atlantic Oscillation (Monteith et al., 2000). However, a key uncertainty remains to be addressed in quantifying the relative roles of physical (hydrological) versus biological processes in controlling the short- and longer-term release of inorganic N from terrestrial systems.

In addition to its influence on erosion and drainage pathways, soil type has a strong influence on land use, and where soils are subjected to land management impacts, including artificial drainage, overgrazing, liming and fertiliser applications, downstream water quality may be affected (see Section 2).

3.3 Location

Locational factors, for example geographical area of the UK, altitude, aspect and proximity to heavily populated areas, may be important in water quality by determining the geology, soil type and hydrology, type of vegetation and rate of human-induced erosion. High-altitude areas have higher rainfall and more potential for erosion, as well as a greater proportion of organic soils. Aspect influences temperature differences and hence the extent and location of frost action, which is an important driver of upland erosion (Evans, 1990) and may be important in biophysical processes. Certain areas of the UK have vulnerable soil types; for example the North Pennines are covered in blanket peat, and the South Pennines in degraded blanket peat. The South Pennines are accessible from many of the UK's major cities and hence heavily affected by human activity. This includes accelerated erosion of accessible areas by walkers and other recreational users, and atmospheric deposition of pollutants from industry, both of which lead to upland areas being sources for water quality pollutants. Arguably the North Pennines were equally degraded but have recovered and re-vegetated more effectively; it is unclear whether climate or other factors may have a role in this.

Upland erosion is widely distributed across the UK. A survey of 400 sites across England and Wales indicated that the worst-affected areas are the Pennines and mid and south Wales, while the Lake District (see Orr et al., 2004), the North York Moors, the Cheviot Hills and Bodmin Moor are the least affected (McHugh, 2003). McHugh et al. (2002) estimated that around 2.5% of the upland area surveyed was eroding, although rates are locally variable depending on topography and soil type.

4. Antropogenic perturbations

4.1 Climate change

Upland climates are highly heterogeneous and exhibit a wide range of natural variability. Nevertheless, evidence of recent climate change comes from observations at high-altitude sites across the globe, with increased winter rainfall and rainfall intensity (e.g. Barry, 2003; Beniston, 2003; Groisman et al., 2005; Pepin and Losleben, 2002), and temperatures increasing more rapidly than at lowland sites, particularly through increases in minimum (nocturnal) temperatures (Bradley et al., 2006). In the UK uplands, winter precipitation has shown large changes – in parts of western Scotland totals have increased by 60–100% since 1960 (Barnett et al., 2006). There is evidence of more rapid warming (Holden and Adamson, 2002) and more marked precipitation changes in uplands (Malby et al., 2006). Winter rainfall intensity has increased over high ground (Fowler and Kilsby, 2007) and rain shadows may have weakened (Malby et al., 2006). Although the record is too short for trend analysis, overall mean winter rainfall and river flows have increased in western parts of upland Britain since the 1960s (Dixon et al., 2006; Wade et al., 2005; Wilby, 2006). There is also tentative evidence of long-term changes in snow cover and persistence in UK uplands (e.g. Harrison et al., 2001; Johnson, 2005; Watson et al., 2004).

There have been few studies that quantify the potential impact of climate change on water quality, particularly in the uplands. This is a result of limited data and the highly heterogeneous nature of climate over uplands. However, anticipated changes include increased water temperature and reduced dissolved oxygen, decreased dilution capacity of receiving waters, increased erosion and diffuse pollution, photoactivation of toxicants, changing metabolic rates of organisms, increased eutrophication and greater prevalence of algal blooms; all of these could lead to exceedence of water quality standards (Wilby, 2004; Wilby et al., 2006). Lack of water at low flow periods could ultimately severely limit abstraction opportunities in the uplands.

Climate impacts upon upland water quality directly through rainfall, the intensity of surface and subsurface flows and changes in temperature. Upland water bodies are highly responsive to elevated air temperatures (e.g. Durance and Ormerod, 2007), which often combine with low flows, affecting dilution and dissolved oxygen concentration. Climate drives the flux of pollutants (see Section 4.2 on atmospheric deposition); for example, the amount and intensity of rainfall can have an impact on leaching rates of pollutants in soils (Ness et al., 2004), transfer of pollutants, dilution effects in receiving waters and rates of sediment delivery (e.g.Wilby et al., 1997). Drought periods can lead to soil drying, with direct and indirect consequences for water quality. In peaty catchments, enhanced water colour levels have been observed in post-drought periods (Naden and McDonald, 1992). Indeed, immediately following severe drought, colour levels may decrease and then increase during the recovery period (Watts et al., 2001). Long-term effects of drought may also be seen due to physical disruption of the peat pore structure following collapse of macropores (Gilman and Newson, 1980) and hydrophobicity (McDonald et al., 1991). In areas of ombrotrophic peats, such as the Pennines and Scotland, which have historically received high atmospheric loadings of heavy metals and sulphur, pulses of metals may be released to the water course due to drought-induced acidification (Tipping et al., 2003). Model predictions indicate that the size of the metal pulses will reduce rapidly over successive droughts as the supply of labile metals is exhausted. Atmospheric inputs of metals and sulphur have declined substantially in recent decades, and are unlikely to compensate for the drought-induced leaching losses (Tipping et al., 2003). Most of these observations apply to peat-dominated catchments. However, upland grassland may be developed on organo-mineral soils, such as podzols and gleys, with a substantial surface organic horizon. Thus similar, though less intense, effects may be expected following drought periods.

Water quality may also be affected indirectly by drought, due to changes in hydrological pathways induced by soil cracking, especially in areas of clay-rich soils. In a heavily grazed, improved upland grassland catchment in mid-Wales, the proportion of autumn and winter runoff generated as overland flow was substantially reduced following the dry summer of 2006 (Marshall et al., in review). During the summer, large cracks formed in the clay-rich soil, which allowed rapid transfer of surface water to drains and shallow groundwater. It was several months (spring 2007) before the hydrology of the site returned to pre-drought conditions. Although water quality was not monitored over the pre- and post-drought periods, unpublished data from the site have shown that overland flow is enriched in nitrogen and phosphorus which can be rapidly transferred to the surface drainage network. Infiltration in the drainage network may ameliorate water quality if there is capacity for the biomass and mineral sub-soil to remove nutrients.

Stream water chemistry is affected by the relative contributions from soils and groundwater, which depend to a large extent on catchment hydrology and climate (e.g. Neal et al., 1990). Indirect effects include temperature impacts on the rate of biophysical processes, influencing production and release of nutrients (e.g. Bardgett et al., 2005). High temperatures and reduced flows in summer increase the degree of eutrophication in lakes and rivers and exacerbate the effects of acid pollution (e.g. Schindler, 2001). Local factors, such as riparian shading, may significantly reduce water temperature locally, particularly in upland catchments where groundwater temperature has a smaller impact than in larger lowland catchments, thus buffering the impacts of high temperature on water quality (e.g. Malcolm et al., 2004).

Changes in amount and lie times of snow fall in UK uplands may have an impact on water quality. In England and Wales, this will only impact on the highest ground, but other sites may be affected by changes in freeze–thaw cycles and frost frequency (e.g. Soulsby et al., 2002a). Wind speed and storminess are important drivers of mixing in lakes where stratification leads to changes in water quality (e.g. George et al., 2004).

Climate also drives vegetation growth rates and changes in the composition of the vegetation community; this will have consequences for land managers and it is not yet clear what direct and indirect effects it will have on water quality. It is likely that the grazing period will increase, meaning animals are left on the out-bye land for more of the year. This would have a negative impact on upland water quality. Climate change will also have an effect on the relationship between livestock and diseases/parasite infestation, leading to the need for more biocide use; and if winters become milder and wetter, diseases and parasites may persist for longer.

4.2 Nitrogen deposition

The earth's atmosphere is approximately 80% nitrogen. In its gaseous form it is inert and unavailable for use by most organisms, but in its reduced and oxidised forms it is of great importance to plants. However, in these forms it is also a pollutant. The main nitrogenous air pollutants include nitric oxide (NO), nitrogen dioxide (NO_2) and ammonia (NH_3), which are dry deposited; and nitrate (NO_3^-) and ammonium (NH_4^+), which are deposited as wet deposition. Wet deposition occurs when soluble nitrogen compounds are dissolved in rain and cloud drops. It can also take the form of cloud droplet (occult) deposition and snowfall. Dry deposition consists of gasses and particles that are deposited directly to vegetation surfaces or the ground.

Nitrogen can have a number of potential impacts on soils and on plant communities, including acidification, an increase in soil nutrient status, an increase in nitrogen leaching, changes in the species composition and a reduction in the species richness (Stevens et al., 2004).

A number of studies have demonstrated that the fertilisation of grassland leads to an increased loss of nitrogen via leaching (e.g. Decau et al., 2004); however, the majority of these studies use very high nitrogen application rates that are not comparable to nitrogen deposition. In the UK, nitrogen deposition ranges from 5 to 35 kg N ha^{-1} yr^{-1}, but many experiments make N additions in excess of this. Phoenix et al. (2003) conducted a study using cores taken from semi-natural upland acidic and calcareous grasslands in the Peak District. They found that the addition of 35 kg N ha^{-1} yr^{-1} (almost doubling the ambient nitrogen deposition, but equivalent to an in-bye annual fertiliser application) resulted in very small (statistically non-significant) increases in nitrogen leaching. The majority of the additional nitrate supplied was immobilised by soil microbial processes. Even at higher nitrogen application rates, a large proportion of the applied nitrogen was immobilised in the soil. This means it is likely that upland grasslands are absorbing large amounts of pollutant nitrogen and reducing the amount that enters water courses. Reynolds et al. (1997) examined three grassland upland catchments in Wales and found all of them to be N sinks. However, Curtis et al (2005) have demonstrated that plant species change resulting from increased nitrogen deposition may also alter the N retention capacity of the system. Using a ^{15}N tracer to quantify N retention, Curtis et al. demonstrated that bryophytes and lichens were a major sink for deposited N in grassland and ericoid shrub-dominated communities across a nitrogen deposition gradient. The non-recovered fraction of the added ^{15}N (assumed to have been leached from the system) was closely correlated with the reduction in the ^{15}N recovered from the bryophyte and lichen pools. These organisms had the highest N retention efficiency and were important components of the total above-ground plant biomass.

Most studies concerned with investigating the factors determining N retention within ecosystems point to the soil N store as the dominant sink for N deposited from the atmosphere (Emmett, 2007). Thus parameters linked to the retention capacity of the soil are identified as the main controls on the response of nitrate leaching to changing N deposition. Of the many, often confounding, factors involved, soil type (especially carbon content) is likely to play a major role in the ability of a grassland to immobilise nitrogen (Rowe et al, 2006) (see Section 3.2).

Nitrogen and sulphur deposition both play an important role in acidification. This affects water quality both directly through acidification of surface waters, and indirectly as soils are acidified. Acid deposition has received considerable research attention (e.g. the UK Acid Waters Monitoring Network) and still remains one of the greatest threats to upland water quality (Monteith, 2004). Acidification can also result in increased mobilisation of metals (Dise et al., 2001), particularly aluminium which is harmful to aquatic organisms.

Topography and climate have an important role to play in nitrogen deposition in upland areas. The enhanced precipitation and cloud droplet deposition mean that upland areas of Britain receive some of the highest levels of N deposition in the country (Figure 9); of particular note are the upland grasslands of the Peak District, where nitrogen deposition reaches 35 kg N ha^{-1} yr^{-1}. The high levels of N deposition are a result of enhanced rainfall and the 'seeder-feeder' effect which enhances pollutant deposition in upland areas, especially those which experience significant periods of orographic cloud cover (Taylor et al., 1999). Vegetation also captures wind-driven cloud droplets very effectively, and as cloud droplets can contain much higher concentrations of ions than rainwater, this is a significant source of pollutant deposition to high ground (Reynolds et al., 1997).

Location is also an important factor: close to point sources (such as farms), deposition levels are elevated and can result in high levels of ammonia downwind of the source; diffuse sources (such as large cities and the road network) also play an important part in determining the level of N deposited (Negtap, 2001).

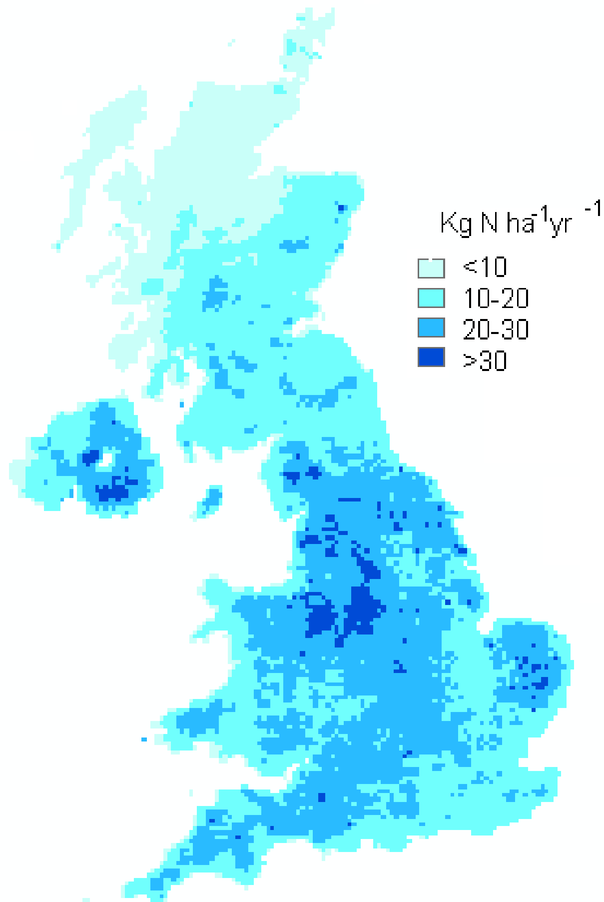

Figure 9 Atmospheric nitrogen deposition to heathlands and rough grazing (Data supplied by the Centre for Ecology and Hydrology)

5. Can we classify areas of upland grassland in relation to water quality?

<u>5.1 Mapping the type and distribution of upland grassland in England and Wales</u>

In order to classify upland grassland in relation to water quality, spatially explicit information is required as to the distribution, type and management of grassland across England and Wales. The mapping should relate to identifiable units of the landscape or parcels of land; simple distribution maps based on occurrence or dominance in a grid square are much less useful. It is also preferable to use a grassland classification that has broader application and relevance beyond water quality objectives. In this way, information derived for water quality purposes can be integrated with other data sets, for example soil monitoring, to meet conservation and land management objectives across a number of policy areas.

There are four land cover data sets in Great Britain which meet these criteria. These are: the ITE/CEH Land Cover Maps (LCMGB and LCM2000); the CORINE Land Cover Map (CLC2000); and the Phase 1 Survey designed by JNCC (JNCC 1990).

5.1.1 CEH Land Cover Maps

The Land Cover Map of Great Britain (LCMGB) was pioneered in 1990–92 to provide a land cover census of Great Britain compiled from satellite-derived remote sensing data. LCMGB recorded 25 land cover types in 25 m grid cells to give complete, though generalised, national coverage of the whole of Britain. LCMGB data were compared and contrasted, both qualitatively and quantitatively, with ground survey data from Countryside Survey 1990 in order to assess correspondence and accuracy, which ranged between 50 and 90%.

Land Cover Map 2000 refined the approach taken in 1990. Most importantly this involved developing new methodologies to provide a classification based on land parcels with vector boundaries rather than image pixels. Further refinements were added, including the use of ground reference 'training data' and contextual information such as altitude or soil characteristics, to improve the classification. LCM2000 has also been calibrated and validated against the field data acquired from Countryside Survey 2000 (CS2000). The classification used in LCM2000 provides 24 target land cover types, with the aim of achieving >90% allocation of parcels to the correct cover type. The target land cover types can be cross-referenced to the widespread broad habitat types used for the monitoring and maintenance of biodiversity under the UK Biodiversity Action Plan. The 24 target classes (Level 1) are subdivided into 27 subclasses (Level 2), within which further land cover variants are also recognised (Level 3).

Eight grassland cover types are distinguished at Level 3 of LCM2000 and their relationship to broad habitat type is shown in Table 3.

Table 3 Classification of grassland types within LCM2000 and their relationship to widespread broad habitats

<table>
<tr><th>Widespread broad habitat</th><th>LCM target class
Level 1</th><th>Code</th><th>LCM subclass
Level 2</th><th>LCM subclass
Level 3</th></tr>
<tr><td rowspan="2">5. Improved grassland</td><td>Improved grassland</td><td>5.1</td><td>Improved grassland</td><td>Intensive grazing, hay/silage cut, grazing marsh</td></tr>
<tr><td rowspan="2">Abandoned and derelict grasslands</td><td>5.2</td><td>Set-aside grass</td><td>Grass set-aside</td></tr>
<tr><td rowspan="2">6. Neutral grassland</td><td>6.1</td><td>Rough grass</td><td>Rough grass (unmanaged)</td></tr>
<tr><td rowspan="5">Semi-natural & natural grasslands & bracken</td><td>6.2</td><td>Managed neutral grass</td><td>Grass (neutral/unimproved)</td></tr>
<tr><td>7. Calcareous grassland</td><td>7.1</td><td>Calcareous grass</td><td>Calcareous (managed, rough)</td></tr>
<tr><td rowspan="3">8. Acid grassland</td><td rowspan="3">8.1</td><td rowspan="3">Acid grass</td><td>Acid</td></tr>
<tr><td>Acid with Juncus</td></tr>
<tr><td>Acid Nardus/Festuca/Molinia</td></tr>
</table>

LCM2000 therefore has the potential to identify and map eight grassland types across Britain as a basis for defining a relationship between upland grassland and water quality. Unfortunately, there are issues with the classification of grassland types within LCM2000 which have to be recognised. Improved grassland is the most extensive single-cover class, accounting for more than 25% of the UK land cover according to LCM2000 and

CS2000 (Fuller et al., 2002). Improved grassland was generally well classified by LCM2000, but Fuller et al. noted that there was some difficulty and controversy in defining 'improved' as distinct from 'semi-natural' grassland types. Semi-natural grasslands and bracken habitats also proved difficult to classify. Neutral, calcareous and acid grassland broad habitats have no consistent spectral characteristic by which to determine the acidity of the underlying soil. The ancillary data used in the classification (acid sensitivity map developed by Hornung et al., 1995) proved to be of limited value. Furthermore, much of the satellite imagery used in LCM2000 was recorded in May when emergent bracken cover was at a minimum. Thus many areas of bracken were recorded as the underlying acid grassland (Fuller et al., 2002).

Despite the limitations, LCM2000 has been used as the basis for mapping acid grassland broad habitat across Britain at 1 km square resolution in connection with critical loads for acidity and nitrogen (UKNFC 2003). To do this, the LCM2000 data have been refined using maps of species distribution at 10 km square resolution produced by Preston et al. (2002). These maps show the percentage of species in each 10 km square, making adjustment for the latitudinal gradient in species diversity in the UK (UKNFC 2003). A cut-off value for the percentage of species that best represent the key areas for the habitats was then applied following advice from habitat experts. For calcareous grassland, a cut-off value of 50% was selected; a cut-off of 40% was applied to all other habitats, which means that 10 km squares were selected where more than 40% of the species pool for the particular habitat was present. The 10 km squares selected using the species data were overlaid on the corresponding 1 km LCM2000 map. The 1 km LCM2000 squares falling within the 10 km squares were mapped to represent the habitat. HOST data at 1 km resolution were used to distinguish between wet and dry areas of acid grassland and dwarf shrub heath.

5.1.2 CORINE Land Cover Map of the United Kingdom (CLC2000)

CLC2000 is a pan-European land cover database providing an inventory of biophysical land cover using 44 land cover and land use classes representing the major surface types across Europe. It is produced jointly by the European Commission and the Member States and is an update of a similar map produced in 1990. The map is designed to be used at a scale of 1:100,000 and has a minimum mappable unit of 25 ha. The UK contribution to CLC2000 was produced by CEH by generalising the more detailed LCM2000. The generalisation process used a cross-tabulation of the thematic classes, a semi-automated spatial generalisation procedure and visual satellite image comparison using computer-assisted image interpretation tools. Although the map is more generalised than LCM2000, it has the advantage of covering the whole of the UK and is compatible with the maps of the EU Member States.

5.1.3 Phase 1 Mapping

The Phase 1 survey was developed to provide a rapid and standardised method for classifying and mapping wildlife habitats across large areas of countryside. Developed in the 1970s, the system was modified in the early 1980s for mapping habitats in SSSIs. The SSSI mapping system was considered to be too detailed for more widespread application across large land areas, and a simplified but compatible version was subsequently produced. A number of revisions were made to the original method in order to remove ambiguities and improve classifications. However, since 1982, all versions of the NCC Phase 1 habitat mapping system have used the same basic hierarchical classification system and are thus compatible with one another in most respects (JNCC, 1990). Phase 1 maps are available for the whole of Wales. In England, Phase 1 surveys have been completed in Cumbria and parts of West Yorkshire (Bonner, 1979). The more comprehensive SSSI mapping system was also used to map Somerset and Dorset (Swash et al., 1983; Howlett et al., 1983).

Phase 1 methodology is applicable to mapping specific habitat types, such as grasslands or woodlands, as well as larger areas of the countryside where the aim is to classify and map every land parcel. The classification is based principally on vegetation, supplemented by topographic and substrate data where vegetation is not the dominant component of the habitat (JNCC, 1990). The map data are supplemented by 'target notes' which record particular areas and features of interest. These can be used as a basis for selecting sites for more detailed follow-up Phase 2 survey.

Table 4 shows the grassland habitats identified by Phase 1.

Table 4 Phase 1 grassland habitat categories and National Vegetation Communities

Phase 1 habitat category	**subclass**	**National Vegetation Community***
Acid grassland	unimproved	U1-6
	semi-improved	U1-6
Neutral grassland	unimproved	MG1-5, 8-10
	semi-improved	MG1, 3-6, 11
Calcareous grassland	unimproved	CG1-14
	semi-improved	CG1-13
Improved grassland		MG6 & 7
Marsh/marshy grassland		MG8 & 10, M22-28
Poor semi-improved grassland		MG6

*These National Vegetation Communities are associated with, but are not necessarily confined to, these habitat categories

A number of caveats apply to the use and interpretation of Phase 1 maps. These include:

- limitations to accuracy, although error estimates are included in the protocol
- the minimum area for survey at 1:10,000 is 0.1 ha
- sites are only visited once so that some communities may be missed due to seasonal effects
- most importantly, significant habitat changes may have occurred since the mapping was completed

It is, however, a potentially useful tool for mapping upland grassland in areas where the survey has been completed and the maps have been digitised.

5.1.4 Choice of land cover data set – the way forward?

All four data sets have drawbacks for mapping the distribution of upland grassland in relation to water quality. The CORINE data are probably the least useful for an England and Wales assessment, as they are simply a more generalised version of LCM2000 and therefore offer no advantage. CORINE would, however, be useful for any extension of the assessment into Europe.

LCM2000 has some fairly major constraints in terms of distinguishing between different semi-natural grassland habitats, although it is claimed that at the target class level (Level 1) LCM2000 is likely, in general, to be 85% correct in its mapping (Fuller et al., 2002). There may, however, be significant local errors.

It is not known how extensive Phase 1 cover is for England, although data are available for the whole of Wales. One important issue is that some of the Phase 1 data are now more than 20 years old. This may not be a major problem for upland grassland areas where land use has probably remained broadly unchanged over this period. In marginal upland areas, however, it is likely that there have been subtle grassland management changes which might affect water quality. For example, it is probable that improved swards in many marginal areas are no longer maintained to the same level of productivity as formerly, reflecting changes in economic conditions and revisions to policy, such as removal of the lime subsidy and CAP reform. In many parts of Wales, for example, there has been a recent shift away from cattle grazing on improved upland swards and some ploughing of improved permanent grassland for fodder crops in response to CAP reform and environmental regulation.

One way forward might be to use LCM2000 with an altitude filter to define ‘upland’ as a baseline data set to identify areas and types of upland grassland. Subsequently, these areas can be validated to broad habitat level using Phase 1 data. Given the availability of Wales-wide digital coverage for Phase 1 and LCM2000, Wales might be a candidate area for testing this approach.

5.2 Factors influencing the relationship between upland grassland and water quality

At its most basic level, a classification of upland grassland in relation to water quality could be developed by selecting a suitable grassland cover data set, e.g. LCM2000, and assigning a water quality signature to each grassland category on the basis of ‘expert knowledge’, review of existing data sets, e.g. EA water quality monitoring, some form of water quality survey or a combination of these approaches. The previous sections of this review have, however, identified many complex and interacting factors that determine the relationship between upland grassland and water quality. Table 5 attempts to summarise some of the primary factors that influence this relationship.

Table 5 Summary of primary factors influencing the relationship between upland grassland water and water quality

	N	TDP	TP	Acidity	DOC/colour	Pathogens	Suspended sediments
Season	X				X		
Flow		X	X	X	X	X	X
Stocking density		X	X			X	X
Stock type	X	X	X			X	X
Management history/maintenance	X			X			
Farming calendar	X	X	X			X	X
Soil hydrology		X	X				
Artificial drainage	X	X	X		X		X
Atmospheric inputs	X			X	X		
Substrate geochemistry				X			
SOM content	X				X		

Notes:

1. Many of these factors interact; for example, the effects of flow on acidity are most clearly seen in winter when amounts of runoff are high, so there is a seasonal component, but this is not the primary factor.

2. Management history/maintenance – this seeks to capture land use history; for example, at Plynlimon some upland grassland areas received very heavy doses of lime in the 1930s. Although the treatments were not maintained, their imprint on water quality could be seen over 40 years later (Hornung et al., 1985).

3. Soil hydrology – this can be considered in terms of a simple division into freely drained upland soils (e.g. brown earths) and soils with impeded drainage (e.g. stagnohumic gleys)

4. Artificial drainage includes both moorland grips and under drainage

5. Substrate geochemistry – represents the geochemistry of either the bedrock or drift deposits as they affect base flow hydrochemistry and hence buffering capacity, for example (Reynolds et al., 1986).

6. SOM content – included to capture the influence of surface peat/organic horizon development

The factors listed in Table 5 can be broadly addressed in two ways: firstly by refining the grassland cover data using other geospatial information; and secondly by ensuring that the water quality 'calibration' data encompass time-varying influences such as flow, seasonality and farming calendar. Thus a minimum requirement for the water quality data would be sampling at high and low flow, preferably within each season.

Additional geospatial data sets can be employed to address many factors included in Table 5 (although issues of spatial resolution must be taken into account), and include:

- stocking density/stock type – agricultural census data
- soil hydrology – HOST classification

- atmospheric deposition – nationally mapped data derived from the UK Acid Deposition Monitoring Network and held by the National Focal Centre for Critical Loads
- substrate geochemistry – 'raw' geological data are of limited value as the rock types tend to be mapped by lithostratigraphic units rather than geochemical properties. Derived data such as the acid waters sensitivity map (Hornung et al., 1995) are likely to be more useful
- SOM content – NATMAP soils data

Factors such as management history and soil drainage are more difficult to quantify. Land management history is very difficult to assess remotely, requiring explicit local knowledge, although even this may fail where land has changed ownership. The location of surface soil drainage can be mapped from aerial photography, but this does not provide information as to the condition of the drains and whether they still contribute to the drainage network. Subsurface land drainage presents more even more of a problem. As noted earlier, individual farmers often have knowledge of the drainage network within their own holding, and aerial photography (particularly during extended periods of dry summer weather) will often reveal the outline of surface and subsurface drainage networks. However, it is not possible to remotely assess subsurface drain condition and functionality.

5.3 A review of some candidate data sets for investigating relationships between upland grassland and water quality

This review does not claim to be exhaustive, but illustrates the types of data set currently available for an analysis of the relationships between upland grassland type and water quality.

5.3.1 The PEARLS model

The PEARLS model developed by CEH (Cooper et al., 2000) is a spatially explicit modelling tool for predicting water quality at the catchment, regional or national scale. The approach involves dividing the target catchment into landscape classes which have combinations of features that influence the chemical parameter of interest. For example, for predicting acid neutralising capacity (ANC) the landscape features might include geology, soil type and land cover (forest, moorland, grassland, heath, improved grassland). It is assumed that there is a distinct statistical distribution of the chemical parameter for each landscape class by location. The statistical distribution is estimated using survey data from headwater catchments draining a single landscape class. These statistical distributions are used together with GIS landscape coverages to give statistical distributions of the chemical parameter throughout the catchment. Initially developed for acidification modelling in the Twyi catchment (Cooper et al., 2000), the approach has been applied to much of Wales, parts of Scotland and Dartmoor (Evans et al., 2001). The approach has primarily been applied to the prediction of acidification trends in upland catchments and therefore the survey data sets for defining landscape chemistry do not include phosphorus or suspended sediments. In practice, it is likely that the landscapes in PEARLS have not been defined in sufficient detail below broad categories such as conifer forest, grassland etc. However, it may be possible to re-examine the survey data sets in order to extract those sites representative of upland grassland in the broadest sense, and to redefine the land use component more explicitly in terms of grassland habitat type or management.

5.3.2 CS2000 and CS2007

The Countryside Survey is an approximately decadal audit of the countryside, which provides detailed information on habitats, landscape features, land use, soils and freshwaters using field survey data. There have been four surveys since the first in 1978 (1984, 1990, 1998, 2007), and from these it is possible to examine both the ‘stock’ of natural resources as well as change between surveys. The basis of the survey is the subdivision of the UK into 32 land classes derived according to the major environmental gradients encountered in the UK. In 1978, each land class was sampled using eight randomly selected 1 km squares in which all the features of interest were mapped. Subsequently, the number of squares has been increased and the land classes reclassified to 40 to allow for country level reporting. Within each 1 km square, a soil sample is also collected for chemical analysis from each of the five randomly distributed quadrats used for detailed vegetation mapping, and one watercourse is assessed for habitat features, habitat quality and aquatic macroinvertebrate fauna. A single water sample is collected from the watercourse for chemical analysis. In 1998, a total of 569 squares were visited (366 in England and Wales and 203 in Scotland). Within these squares, 425 watercourses were surveyed, of which 404 were flowing at the time of the visit and therefore sampled chemically and biologically. The survey for 2007 is currently ongoing, with a target of 629 1 km squares to be surveyed, of which 425 will be targeted for a survey of indicative stream water chemistry, where samples will be collected and analysed for pH, conductivity, alkalinity, soluble reactive phosphorus and total oxidised nitrogen.

CS2000 and CS2007 data can be reported in several ways including by broad habitat type, ITE land class, country (England + Wales combined and Scotland in CS2000 and individually in CS2007) and environmental zone. Soils can also be reported by soil type. Environmental zones (EZs) are an aggregation of the 40 land classes into a less complex grouping of six zones (Table 6). For the freshwater component of CS2000, the mean percentage cover of broad habitat type in a 20 m riparian strip for each stream subject to River Habitat Survey has been estimated, to provide information about the characteristics of each EZ.

The unique feature of CS is the synchronous collection of terrestrial habitat, soil and freshwater data from each 1 km square, which allows the various components of the survey to be related to one another. Thus, for example, soil chemical data (pH, loss on ignition, available-P, carbon and nitrogen) can be analysed to identify the characteristic soil chemistry associated with each broad habitat type.

Table 6 Countryside Survey environmental zones

EZ	Description
Environmental zone 1	Easterly lowlands of England and Wales
Environmental zone 2	Westerly lowlands of England and Wales
Environmental zone 3	Uplands of England and Wales
Environmental zone 4	Lowlands of Scotland
Environmental zone 5	Intermediate uplands and islands of Scotland
Environmental zone 6	True uplands of Scotland
Environmental zone 7	Northern Ireland

So far, water chemistry data for CS2000 have been reported by environmental zone (Furse et al., 2002) and the most relevant data for this project are contained in EZs 3, 5 and 6 (Table 7). The data show, not unexpectedly, that the sites in the upland EZs have the lowest nutrient status and are the most acid. In principle, it may be possible to take this analysis further and to examine the water chemistry of those sites in EZs 3, 5 and 6 with riparian zones dominated by grassland broad habitats. At this level of disaggregation, sample numbers may become too small for valid statistical analysis and Furse et al. (2002) stress that the chemistry data should only be regarded as indicative of the chemical characteristics of each site. Thus there is a danger of over-interpretation of limited data.

Table 7 Mean water chemistry data from CS2000 summarised by environmental zone (from Furse et al., 2002). Figure in brackets is the number of sample sites in each EZ

EZ	pH	Alkalinity	Conductivity	SRP
1 (68)	7.92	4.87	1087	510.0
2 (98)	7.72	2.56	609	139.8
3 (55)	6.87	0.63	146	14.7
4 (58)	7.62	1.73	326	51.9
5 (64)	7.05	0.70	197	14.2
6 (62)	6.86	0.35	67	7.1

5.3.3 G-BASE

G-BASE is the regional geochemical survey programme of the British Geological Survey (BGS). The sampling programme began in 1968 in the northern Highlands of Scotland, with the aim of producing maps to show the regional distribution of elements in stream sediments (BGS, 2006a). The programme is still ongoing in the more lowland parts of the UK, though the majority of the uplands have now been sampled (BGS, 2006b). The earlier surveys principally comprised stream sediment samples collected at a density of approximately 1 sample per 2 km^2. These were analysed for a wide range of major, minor and trace constituents, but unfortunately only pH, conductivity, uranium and fluoride were determined on water samples, reflecting the analytical technology available at the time. The exception to this is the survey of Wales and the West Midlands in which 13,444 water samples were collected from first- and second-order streams at a sampling density of one sample per 1.5 km^2. The samples were analysed for over 30 analytes, including pH, alkalinity, total dissolved P (by ICP-AES) and nitrate. The information is presented in map form in a geochemical atlas (BGS, 1999). More recently, soil samples have been added to the survey, particularly as it moves into more urbanised parts of Britain and into areas with limited surface drainage, such as the Chalk.

The G-BASE data set for Wales may provide a potentially useful tool for investigating the relationship between upland grassland and water quality. The sampling density is high and focuses on first- and second-order streams, so the potential for finding sites draining single grassland types is high. Unfortunately, sampling was focused on summer campaigns, which will have been dominated by low flow conditions. Only one sample was taken at each site and, furthermore, concentrations of nitrate are usually at their lowest during the summer months. These factors will impose limitations on the usefulness of the data set. Alongside the maps, the geochemical atlas provides a more detailed thermodynamic analysis of the data together with an assessment of factors affecting water chemistry (BGS, 1999). However, there has been little published analysis of relationships between land cover type and water quality derived using G-BASE data.

5.3.4 Welsh Acid Waters Survey data set

In 1995, over 100 headwater streams located in the acid-sensitive areas of Wales were sampled at monthly intervals and analysed for a broad range of chemical and biological determinands (Stevens et al., 1997). Land cover, soils and geology data were compiled for each catchment and used to analyse the factors determining the chemistry and freshwater ecology of acid waters in Wales. The catchments included a range of landscape types, including conifer forest, semi-natural grassland, improved grassland and moorland. Analysis of a subset of data lying within a single geological unit (Lower Silurian Llandovery) identified a very strong signal associated with the proportion of improved upland grassland in the sampled catchments (Hutchins et al., 1999). In this case, 70% of the variance in water hardness at high flows could be explained by percentage of improved grassland within the catchment. This provides an illustration of the potential use of upland grassland management (e.g. liming) to ameliorate acid water conditions generated by other land uses within the catchment, such as conifer forestry.

5.3.5 Lake District data set

This data set comprises analyses of water samples collected during six synoptic surveys of 55 sites in headwater catchments draining the central English Lake District, along with associated catchment data (Thornton and Dise, 1998). The range of analytes included major cations, anions, silicon, alkalinity and pH. It does not appear that suspended sediments or phosphorus were determined. A significant finding from the work was the influence of agricultural improvement on water quality, especially in relation to acid buffering and nitrate concentrations. In their paper, Thornton and Dise (1998) simplify the land use categories from those in the LCMGB data set down to forest, upland vegetation and agriculture.

5.3.6 GANE lakes data set

The GANE lakes data set comprises hydrochemical (seasonal samples in one year) and catchment data for 80 lake sites in four regions of the UK with contrasting climate, soils, geology and geomorphology (Helliwell et al., 2007). The objective of the survey was to assess the key catchment attributes, including atmospheric nitrogen deposition, influencing the leaching of inorganic nitrogen in headwater catchments. Samples were collected at the lake outflow and analysed for a wide range of determinands, but did not include phosphorus or suspended sediments. Two main land cover types were identified in each lake catchment, namely moorland and semi-natural grassland. Further, more detailed examination of these data may allow individual grassland types to be identified.

6. What are the key pollution issues and how do they affect our obligations in relation to WFD?

The WFD, as a new piece of legislation for European waters, introduces regulation of water bodies previously largely unprotected by law. As a result, many of the smaller water bodies have not been monitored and water quality (ecological status) assessments are based on a risk assessment. In addition, there will be no baseline survey data against which to assess change.

Environment Agency water quality assessments to date have been on ‘main rivers’, a formal designation whereby the Environment Agency has greater powers and responsibility to maintain them. In general, the ‘main river’ network does not include the smaller headwater streams characteristic of much of the uplands. All WFD requirements apply to headwaters and tributaries in catchments greater than 10 km^2 and to lakes that are greater than 0.5 km^2 and those of any size that are of ‘conservation significance’ e.g. SAC, SSSI and other designated sites. For main rivers, Environmental Quality Standards (EQS) were established for lengths of river in all catchments which were reported as part of the General Quality Assessment (GQA). Monitoring water quality of lowland rivers, however, does not indicate headwater quality; for example, acidification effects were not observed in downstream reaches. Some of the upland lakes in north-west England have already been identified as ‘at risk’ or ‘probably at risk’ from diffuse pollution (Figure 10).

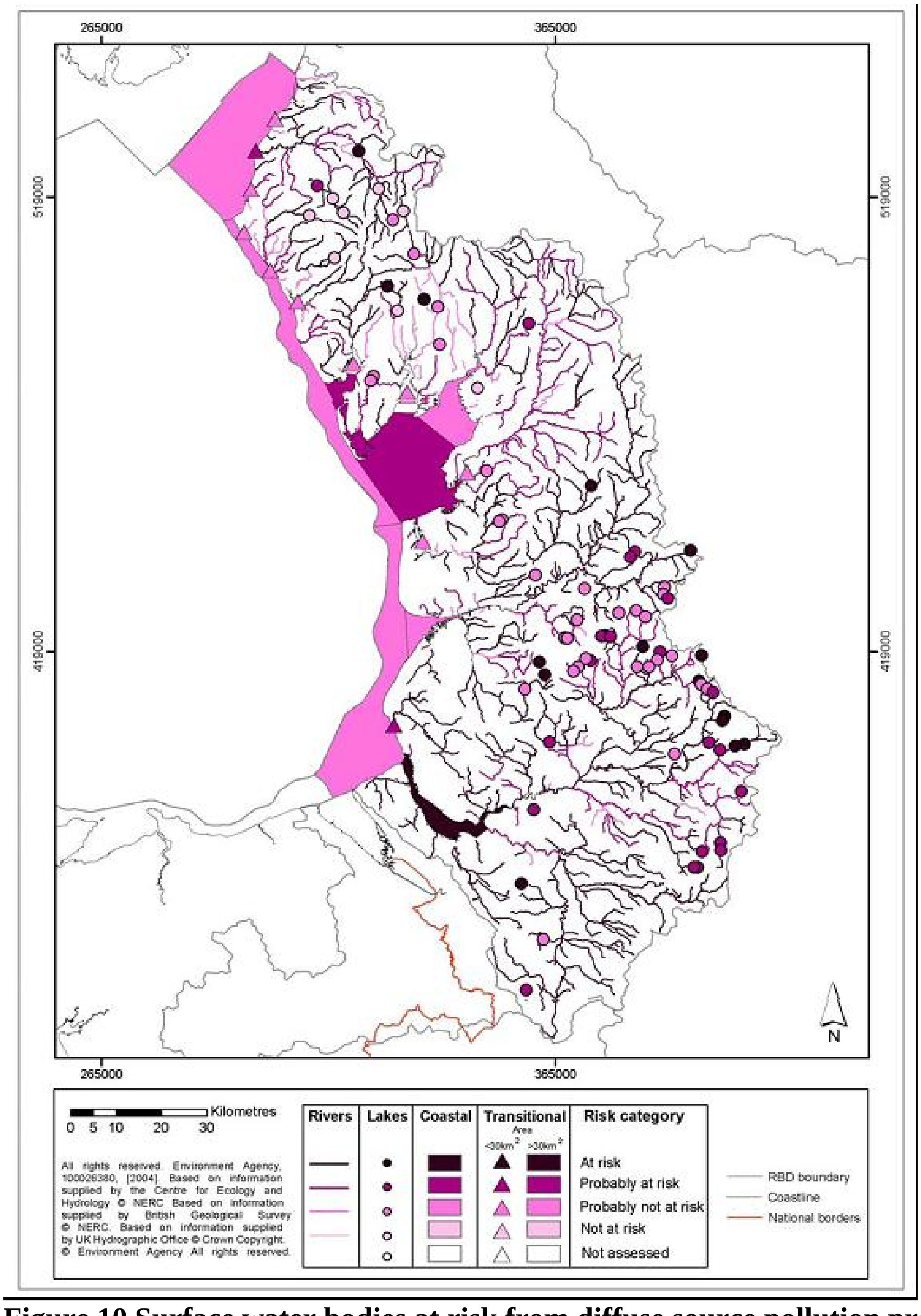

Figure 10 Surface water bodies at risk from diffuse source pollution pressures (North-West river basin district)

It should be noted that the existing water quality monitoring network is based on the presumption that water quality is only adversely impacted by human activity, specifically abstractions and effluent discharges. In general, water quality and the presence or absence of people are well correlated. However, atmospheric N deposition, increasing agricultural intensity and high stocking levels in upland grasslands may be contributing to eutrophication effects in sensitive headwaters. A greater understanding of nutrient budgets from upland grasslands could help with identifying, and more importantly deriving, suitable programmes of measures to combat eutrophication in upland waters.

Where upland water quality problems have been recognised, and in all cases where this has involved designated sites, local investigative monitoring may be undertaken. Natural England and Countryside Council for Wales report on the conservation status of designated sites, and where necessary local agreements for monitoring may be established with the Environment Agency. However, in general such investigations have not considered the nutrient contribution from upland grasslands.

There are four main implications of gaps in knowledge for the WFD:

(1) We are uncertain of the actual water quality of headwater streams and have little data for verifying the risk-based characterisation of water body status.

(2) Where water bodies fail to meet WFD objectives the cause is not always clear, partly as a result of poor data and monitoring in feeder streams. Monitoring activities may need to be extended to provide a background against which to monitor change in status.

(3) Existing water quality monitoring was largely designed to assess compliance with point sources of pollution and may need to be changed (for example to include continuous sediment sampling) to assess diffuse pollution water quality effects.

(4) Deriving programmes of measures for failing upland water bodies will need further research to determine the causes of failure. Water quality is generally affected by diffuse sources of pollution and so needs to take full account of hydrological fluxes of pollutants by integrating field monitoring with modelling studies (e.g. Soulsby et al., 2002b). Input parameters from upland grasslands are not currently available.

Some improvements to upland water quality may be experienced as a result of changes in the Common Agricultural Policy. Predicted changes in upland agriculture as a result of CAP reform have been identified by the Central Science Laboratory. Specifically they predict:

- reduction in suckler cows and extensification of beef
- reduction in and extensification of breeding sheep
- rising input costs and lower profitability
- abandonment of least-productive land
- loss of labour and skills – economic impacts
- greater financial dependence on environmental stewardship
- off-wintering of sheep

The move to more extensive systems is likely to reduce grazing pressures and therefore should improve water quality. There are, however, some threats to water quality, as extensification and the reduction in the rural labour force may lead to reduced maintenance of features that may help to protect water quality, for example stream bank fencing. Provision would need to be made to safeguard this in any future Entry Level Stewardship (ELS) scheme.

There are other pressures outside of CAP which may produce intensification rather than extensification in the uplands, for example the need for increased food production to meet a growing population. This may counteract the benefits which may accrue from the reform of CAP.

The reform of ELS will therefore need to take account of the unpredictability of upland futures and will need to be more flexible than the present schemes. Suggestions from the stakeholder workshop on ‘Understanding Water Quality in the Uplands’ held at Lancaster University in November 2007 included:

- **Advice** needs to be available to farmers within the ELS scheme, e.g. from catchment sensitive farming (CSF) officers or FWAG advisers. Advice is not currently part of the ELS. The advice also needs to be consistent, both over time and in different areas.
- **Targeting** of measures is needed to address particular catchment water quality objectives. Targeting advice on particular options has been shown to be very successful in influencing the uptake of measures in particular areas.
- **Funding** needs to include provision for capital works, e.g. stream bank fencing.
- **Evidence** is needed for:
  - Understanding what the issues are, for example is sheep dip a problem? Note that the perception of water quality issues is different between interest groups – fisheries may say sheep dip is the biggest problem, water companies may argue that it is colour and pesticides, others may argue for eutrophication.
  - Understanding the effectiveness of different measures, for example the effectiveness of grip blocking on reducing water colour, the effectiveness of mobile sheep shelters (Yorkshire Water has paid for these for some farms).

7. What land management options would provide net environmental benefits, but are not available on the Environmental Stewardship menu of options?

During the stakeholder workshop held at Lancaster University a number of land management options were identified that are not currently included in the Environmental Stewardship menu of options that would provide benefit to water quality in the uplands, or are included in the higher level scheme but could be made more widely available.

Fencing of small drainage ditches: small drainage ditches are potentially a very important conduit for pollutants. Fencing them to allow vegetative barriers to develop and prevent stock access would be beneficial to water quality. This would be most suitable as an upland ELS option. May need a capital works element.

Buffer zones: buffer zones for upland grasslands could be introduced. As a difference to current options to be suitable for improving water quality these would need to exclude animals – essentially riparian fencing option.

Maintenance of riparian fencing: this option would reduce stream bank erosion and prevent direct stock access to streams. It could be made available under ELS.

Gully blocking: this option would reduce erosion in gullies where it is actively occurring, particularly in peatland systems.

Gill woodland: native woodlands could significantly reduce sediment delivery and associated pollutants, as well as reducing surface water temperatures and creating habitat.

Removal or blocking of underdrainage: drainage provides an important pathway for the transport of some pollutants, especially phosphorus. Blocking or even removing underdrainage in upland areas would benefit water quality, reduce stream inputs at high flow and benefit biodiversity with a greater heterogeneity and area of seasonal standing water. The level of guidance involved and high capital input would mean this needed to be a Higher Level Stewardship (HLS) option.

Runoff management plans: runoff or infiltration management plans could be incorporated into the ELS scheme. This could include plans to disperse concentrated flows. Farmers may need specialist guidance, therefore the points should reflect the likely expenditure. Although guidance on development of runoff plans could come from some of the CSF projects, e.g. Basssenthwaite.

Resource management plans: similar to those in Tir Cynnal ELS in Wales, that identify assets on the farm that should be protected.

Wetland creation: in particular to help deliver runoff management and reduce pollutant connectivity to watercourses.

Scrub and woodland development in degraded habitats: some upland habitats have degraded to the extent that it would not be possible for them to be restored. Scrub and woodland could be developed (either by natural regeneration or planting) in these areas, benefiting biodiversity and potentially benefiting water quality. It is proposed that this would be an upland HLS option due to the level of guidance required and capital input.

Planting of clough woodlands: this is very likely to provide water quality and biodiversity benefits and could be provided as an upland HLS option.

Restoration of river channels and floodplain grazing: this would improve water quality by reducing erosion and depositing sediment and associated pollutants especially phosphorus, on the land. It would increase the frequency of floods on the land but reduce the severity. This option would only be suitable as an HLS option due to the guidance required. The payment level should reflect compensation for farmers for loss of productive land to flooding.

Peat stabilisation: peat erosion is a considerable problem in the uplands and an option for revegetation of bare peat would be beneficial to water quality, carbon storage and biodiversity. This would fit easily into HLS but would attain most benefit from incorporation into ELS.

Removal or narrowing of tracks: tracks can provide an important conduit for the rapid transport of pollutants. Removing them or narrowing them could improve water quality. This would provide most benefit as an ELS option.

Relocation of tracks: where tracks are located to run down slopes they transport water and pollutants very rapidly. Relocating tracks to run across slopes would reduce this. This would need to be done with some degree of guidance, as only tracks that currently pose an erosion problem should be relocated – therefore it should be a HLS option.

Footpath maintenance: maintenance of cross-drains on tracks is provided as an HLS option, but footpaths can also act as conduits for water flow and be a source of sediment themselves. General maintenance of footpaths and maintenance/installation of cross-drains could be provided to farmers and would provide both water quality and social benefits.

Zero fertiliser input: a no-input option to stop slurry spreading close to rivers would benefit water quality by reducing nutrient and FIO input. This would redistribute slurry spreading to areas where it presented less of a threat to water quality. This could be an ELS option, giving additional points to the current low-input option.

Designation and management of sheep post-dipping drainage areas

An option of no biocide (including sheep dip) application: this would improve water quality and could be made available under ELS. It would be especially beneficial in fish nursery areas (where even low dose pollution can lead to mortality in the marine phase of sea trout).

Reduced stocking density: overgrazing contributes to reduced water quality and biodiversity. An option for reducing stocking density in upland areas would have many benefits. This could be an option under ELS.

Balancing nutrient applications with atmospheric inputs: in areas of high nutrient deposition the low-input option of fertiliser input means the 50 kg N ha^{-1} yr^{-1} that farmers can add is combined with up to 35 kg N ha^{-1} yr^{-1} of atmospheric input. Off-setting against estimated atmospheric inputs would reduce total input and benefit water quality.

Rebalancing of ELS and HLS points: points could be rebalanced to target measures and encourage uptake, and measures used in combination to further improve water quality could be given bonus points.

Targeting of schemes: targeting of schemes and holistic catchment management would allow measures to be placed more effectively. This might be achieved in parallel with River Basin Management Plans required for the WFD.

Control option selection: to ensure some water quality options are included, for example using a menu of options where one option must be selected from each category.

8. Conclusions

Ensuring that the uplands are a source of good quality water is essential for the supply of water for drinking and industrial use in both the uplands and lowlands. Many upland water bodies are oligotrophic and consequently, more sensitive to changes in water quality than those in the lowlands.

Grass-uplands cover a large land area and undoubtedly have an important role to play in maintaining good water quality but linking management practices to water quality is currently difficult due to a paucity of studies investigating the sources, mobilisation and transfer of pollutants from this land use and the impacts of different management activities on this. This is a significant problem which will severely affect our ability to protect upland water quality, develop and monitor compliance with programmes of measures for the Water Framework Directive and adapt to a changing climate.

The lack of an evidence base for mitigation measures in the uplands suggests an urgent need for research into the impacts of management practices on water quality in the uplands. Although data exists for the lowlands there is very little similar data for upland management practices. As the uplands differ from lowlands in their management, vegetation, soils, climate, geology and hydrology extrapolating from lowland data can only ever give an indication of likely impacts. Throughout this report we have identified many knowledge gaps that need to be filled in order to assess the impacts of upland grasslands on water quality.

Particular management activities which impact on water quality and which require quantification include:

- grazing (when it results in overgrazing) including supplementary feeding
- drainage
- incidental losses of fertilisers and slurry
- the role of recreation

There are a number of measures which are appropriate for inclusion in the ES and HLS schemes (Section 7). However, there is a need to provide evidence regarding the impact of new and existing mitigation measures on water quality in the uplands.

All of the data sets identified had limitations for classifying and mapping upland grassland at a national scale. No water quality data set is available for creating such a classification. Ideally such a database would include all chemical parameters of interest (acidity, nitrogen, DOC, P, suspended sediments, pathogens, pesticides and herbicides) and have the appropriate sampling intensity in time and space. Thus for example, G-BASE in Wales has unparalleled sample point density, but is restricted to one summer sample per site. Conversely, the PEARLS survey is specifically designed to explore relationships between water quality and landscape types, accounting for hydrological and other time varying influences, but omits measurement of, for example, suspended sediments, phosphorus and pathogens. The CS2000 freshwater data set has too few analytes and limited scope for detailed interpretation in the context of the requirements of this project. None of the data sets have information on pathogens, herbicides and pesticides.

The next steps for developing a classification of upland grassland in relation to water quality will require:

- defining the coverage of upland grass specific to upland grassland types;
- undertaking more detailed and targeted analysis of the chemical data sets;
- plan and execute a water quality survey for upland grasslands to provide data directly for classifying upland grasslands impacts on water quality.

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