Land management and achieving good water quality

Spray, Christopher

Published in:
Agriculture and the environment VII: Land management in a changing environment: proceedings of the SAC and SEPA Biennial Conference

Publication date:
2008

AGRICULTURE AND THE ENVIRONMENT VII

Land Management in a Changing Environment

Proceedings of the

SAC and SEPA Biennial Conference

Edinburgh

26-27 March 2008

Organised in association with

Scottish Crops Research Institute

The Macaulay Institute

Institute of Professional Soil Scientists

British Society of Soil Science

Edited by

Karen Crighton
SAC

Rebecca Audsley
SEPA

Success through **Knowledge**

Agriculture and the Environment VII

Land Management in a Changing Environment

Proceedings of the SAC and SEPA
Biennial Conference

in association with

Scottish Crops Research Institute
The Macaulay Institute
Institute of Professional Soil Scientists
British Society of Soil Science

held in

William Robertson Building
George Square
Edinburgh University
Edinburgh
26-27 March 2008

Edited by

Karen Crighton
SAC, Auchincruive, Ayr KA6 5HW, UK

Rebecca Audsley
SEPA, 5 Redwood Crescent, Peel Park, East Kilbride G74 5PP

2008 Conference Organising Committee:

Mark Aitken, SEPA Stirling
Rebecca Audsley, SEPA East Kilbride
Bruce Ball, SAC Edinburgh
Karen Crighton, SAC Ayr
Brian D'Arcy, SEPA Perth
Ian Dickson, SAC Edinburgh
Catherine Maguire, SEPA Stirling
Andrew Tibbs, SAC Edinburgh
Allan McBride, SAC Edinburgh
Stuart Somerville, SAC Lanark

Associated Organisations:

Dick Birnie, Macaulay Institute
Howard Davies, SCRI
Dick Thomson, IPSS
David Rimmer, BSSS

Acknowledgements:

The Organising Committee would like to thank the following for their assistance in the paper review process:

Helaina Black, Macaulay Institute
Jonathan Bowes, SEPA
Carole Christian, SAC
Bill Crooks, SAC
Karen Dobbie, SEPA
Claudia Erber, SEPA
Vincent Fitzsimmons, SEPA
Mark Hallard, SEPA
Janette Macdonald, SEPA
Scott Mathieson, SEPA
Eric McRory, SEPA
Bob Rees, SAC
Janet Shepherd, SEPA
Alex Sinclair, SAC
Pete Smith, University of Aberdeen
Christine Watson, SAC
Evan Williams, SEPA

Contents

Foreward

WAC McKelvey, SAC

Theme 1: Managing Land in a Changing Environment

Chaired by M Gill, Scottish Government

Theme 2: Soil Functions and Soil Protection

Chaired by A Branding, Scottish Government

Theme 3: Land Management and Achieving Good Water Quality

Chaired by A Kirmond, Environment Agency

Theme 4: Greenhouse Gas Emissions from Land and Mitigation

Chaired by C Spray, SEPA

Poster Presentations

FOREWORD

It is my pleasure as Chief Executive and Principal of SAC to welcome delegates to the 2007 SAC/SEPA Conference on behalf of both SAC and the Scottish Environment Protection Agency (SEPA). This is the seventh conference in a series that began in 1995 and it is encouraging that each successive conference has attracted a higher attendance, which clearly indicates the value placed on the meeting by international, UK and Scottish environmental and land management practitioners.

In particular I would like to extend a special welcome to land managers in the audience whether farmers or others - this conference is all about you and the decisions that you face today and tomorrow when evaluating alternatives in running your rural businesses, maintaining our landscapes, providing the high quality agricultural produce for which Scotland is justifiably renowned, contributing to healthy and vibrant rural communities while enhancing and protecting our very special environment.

The continuing success of this conference is partly due to the staff at SAC and SEPA who organise the event so professionally, but also to the input of the many speakers and poster presenters who have contributed over the years. We are also very pleased to have the involvement of the Scottish Crop Research Institute and the Macaulay Institute, as well as the British Society of Soil Science and the Institute of Professional Soil Scientists who have provided two keynote speakers, Dick Thompson and David Rimmer from whom we will be hearing in the first session.

It is clear that environmental issues are becoming daily news in mainstream life. The issues may be as far ranging as wind farm proposals, flooding, carbon-offsetting for air flights or the safety of our bathing waters. This conference will concentrate on Land Management issues and in drawing up the programme we have aimed for and, I believe, achieved, a balance between policy and science. We see this conference as forward-looking and targeted on supporting those whose responsibility it is to formulate the most appropriate strategies and to implement them correctly at a practical level.

This conference has four main cross-cutting themes which are: the changing environment, soil protection, water quality and atmospheric protection.

Any new directives, strategies or policies must be forward looking as our planet faces global warming in future decades. We must all become aware of the varied challenges our activities present to our global environment. It may be farmers who are at the focus of the emerging soil strategies, but farming is not the only land activity to have impacts on our environment. Recent flooding in England points to problems in housing, commercial and infrastructure developments. However, alleviating floods caused by high rainfall on farm land can be improved by better catchment planning and by flow attenuation, but the costs of implementing the required measures have to be considered. I am glad that the economics and the science will both be addressed at this conference, as drivers for changing policy and land use must have an economic dimension.

All our activities on land have an effect on surface water, ground water, and the atmosphere. The cumulative effect of these individual activities can become highly significant. We already have EU directives to control many of the possible impacts on the water and air environment, but one common issue is the interface of the activity with the soil surface. It should be no surprise that the EU, UK and Scottish Governments are now looking at soil directives and strategies. This conference will address the issue of the emerging soil strategies to protect soils from harm and the subsequent derived benefits.

The impact of agricultural practices on water quality has been examined by teams looking at both the livestock and the arable sectors. Projects have looked at focused areas and recommended Best Management Practices to mitigate the impacts. Cost-benefit analyses can help in the development of tools to enable policy makers to evaluate the relative merits of varying approaches. SAC's own Environmental Focus Farm project is aimed at testing out some of these strategies in priority catchments to identify effective, practical and affordable strategies for farmers.

Greenhouse gas emissions from land and from land management practices link back to the changing environment. Scottish research in this field will give insights into emissions rates and also, crucially, into developing strategies for mitigation. The techniques to be described and proposed have world-wide application which befits a topic with global impact.

Previous conferences have tended to focus on single issues such as the Water Framework Directive, Waste Directive, Nitrate Directive and Diffuse Pollution. This conference looks at our land use interactions in an holistic way. Our global environment is changing dramatically. We must now acknowledge the rate of the change and evaluate ways to mitigate the negative changes and to reinforce the positive ones.

While the conference is set in Scotland, change is being experienced world-wide in different ways. I hope that you will hear and see much that you will be able to take back to your own country that will help to reduce any negative impacts that particular land management practices may be having on our common global environment.

Professor WAC McKelvey
Chief Executive and Principal, SAC

SOIL SCIENCE AND POLICY: ISSUES BEHIND THE NEED FOR BETTER LAND USE PLANNING AND SOIL MANAGEMENT

TRE Thompson[1] (President of IPSS) and DL Rimmer[2] (President of BSSS)

[1]National Soil Resources Institute, Cranfield University, Cranfield, Bedfordshire, MK43 0AL, UK, E-mail: d.thompson@cranfield.ac.uk; [2]School of Civil Engineering and Geosciences, Newcastle University, Newcastle upon Tyne, NE1 7RU, UK

SUMMARY

Soil science is an applied science that is focussed on improving our understanding of soils and how they function, but it is dependent for much of its funding on answering policy-related questions. The history of the discipline is reviewed in order to establish which policy questions have influenced its development, and to provide the background for a forward look to the policy issues that will be important in the next half century. While initially the needs of agriculture and subsequently concerns for the environment have been the policy drivers, we predict that the challenges ahead will require policy-driven research that gives equal weight to both food production and environmental protection in a world where energy supplies and land resources will be in short supply.

INTRODUCTION

This paper is in two parts. In the first part we consider long-term trends in the development of soil science as a discipline and the links to land use and land management policies in the UK. We address the question of where we are currently at, in relation to soil science and policy issues. In the second part we look to the future, describing upcoming societal challenges for land use, their policy implications, and the role of soil science in providing inputs to assist policy development and implementation. We address the question of what the future holds in relation to soil science and policy issues.

POLICY-DRIVEN SOIL SCIENCE: THE PAST AND THE PRESENT

Any discipline develops through the research that is carried out; and this is certainly true of soil science. But how much soil science research has been policy-driven? Because the discipline is an applied one, the answer is that most of it probably was. Research carried out in government-funded organisations, such as ADAS and SAC, and projects funded by government departments were clearly policy-driven. Those research organisations funded through the Research Councils, such as the Macaulay and Rothamsted, and the projects that they funded might have been less overtly policy-driven, but a close examination will show that only a very small proportion were truly ‘blue-skies’. The Audit of UK Soil Research (Drew Associates, 2003) is a useful source of information for those wanting to check these assertions.

Soil Science and Agriculture

Through much of its history, soil science in the UK has been closely linked to agriculture. From its beginnings in the 19th century, when it was part of the discipline of agricultural chemistry, and pioneers such as Lawes and Gilbert were experimenting with artificial fertilisers at Rothamsted, through to the immediate post-World War II period, when considerable resources were devoted to increasing agricultural productivity, soil science has delivered research results that have rapidly led to improved practices in UK farming.

Significant dates in the history of the discipline include 1924, when the International Society of Soil Science was founded (one of its early conferences was held in Oxford in 1935), and 1947 when the British Society of Soil Science was founded. Through much of the 20th century, soil science researchers in the UK would have sought funds either from the then Ministry of Agriculture, Fisheries and Food (MAFF), the then Scottish Office Agriculture and Fisheries Department or from the Agricultural and Food Research Council, and these organisations would have determined the research agenda to a large extent.

This contribution of UK soil scientists to improving agricultural productivity was not limited to their own country. There was also a considerable overseas dimension with work going on in many Commonwealth countries, with the aid of funds from the Overseas Development Agency. In the 1950s and 1960s many early career soil scientists from the UK spent time overseas before returning to work in our universities and research institutes.

Soil Science and the Environment

In the 1970s, the environmental impact of increased agricultural productivity became increasingly apparent. For example, the realisation that the increase in nitrate concentrations of many rivers was paralleled by the increased use of nitrogen fertilisers on agricultural land, and that the pathway from one to the other was all too easy to understand (Addiscott *et al*., 1991). So the agricultural policy driver for soil science research was slowly but surely replaced by an environmental one.

The decline in importance of agriculture was not only due to its perceived impact on the environment, but also to the well-publicised over-production of food across Europe, leading to ‘food mountains’ and ‘wine lakes’. Ultimately this led to changes in the subsidies paid to farmers through the EU Common Agricultural Policy from support for food production, to support for ‘set-aside’, and most recently to support for environmentally-sensitive land management.

During this period of change, MAFF and the Agriculture and Food Research Council disappeared and were ultimately replaced by Defra (Department for Environment, Food and Rural Affairs) and BBSRC (Biotechnology and Biological Sciences Research Council). Changes also occurred in Scotland, ultimately leading to the formation of the Scottish Government. These funders, together with NERC (Natural Environment Research Council), are now important in determining the direction of research for soil science. Where once the primary goal was increased food production, it is now combating the threat of climate change or controlling pollution (see, for example, NERC, 2007).

The Need for Soil Protection

In 1970 a report commissioned by MAFF raised for the first time concerns about possible soil degradation in the UK. This was ‘Modern Farming and the Soil’ (Strutt, 1970), in which were highlighted problems of soil erosion and the damage to soil structure caused by intense agricultural production involving heavy machinery. Following its publication much research was undertaken to gather further data to understand which soils were susceptible to degradation, and under what conditions. It also initiated discussion in the UK about the need for a soil protection policy. Similar concerns were evident in other European countries, leading to the European Soil Charter (Council of Europe, 1972) and subsequent work on soil functions (e.g. van Lynden, 1994).

The next significant UK government-sponsored report on soil protection did not appear until 1996, when, in its 19th Report, the Royal Commission on Environmental Pollution (RCEP) addressed the ‘Sustainable Use of Soil’ (RCEP, 1996). This report recommended that the UK develop an explicit soil protection policy. This had a very long gestation; but finally appeared in the form of the First Soil Action Plan for England and Wales 2004-2006 (Defra, 2004). Parallel work was undertaken by the devolved government in Scotland, which is developing its own soil protection policy, and has also co-ordinated a well-funded soil research programme in its research institutes and universities.

Another consequence of the Royal Commission report was a research programme directed at the development of indicators of soil quality and their use for the long-term monitoring of the state of the UK’s soil resources (Loveland and Thompson, 2002). This has been a difficult task and the work is still on-going.

At the European level, there has been the development of a Soil Framework Directive (CEC, 2006), which has yet to be agreed by the member states at the time of preparing this paper. This Directive will complement the Water Framework Directive (WFD) which has been in force since 2000 and which has had a major impact on environmental protection, including many aspects of land management.

The ‘Multi-functional’ Approach to Soil Protection

The basis of environmental protection policies has developed over the years. Air pollution in the 1950s was an obvious health hazard and policies aimed at improving air quality were based on minimising risk to human health. A similar approach was initially adopted for water quality, because of the need to provide safe drinking and bathing water. But unlike air, water is also a habitat and the latter also needs to be protected. In the Water Framework Directive the principal indicator of water quality became the ecological status of the body of water, with secondary consideration given to safety of the water supply. This approach embodied in the WFD is a ‘multi-functional’ one. It attempts to reconcile the different functions that any body of water performs, i.e. it provides water for drinking, water for other domestic and industrial uses, and is a habitat for plants and animals.

The multi-functional model for water appears relatively simple, when compared to soil. The list of functions attributed to soil usually include its role: (i) in preserving cultural heritage, (ii) as a reserve of biodiversity, (iii) as a medium for the production of food and fibre, (iv) as a regulator of environmental flows, (v) as a source of raw materials and (vi) as a foundation for construction. Although soil is not part of our diet, it can represent a health hazard if severely contaminated and accidentally ingested. So there is a requirement for ‘clean’ soil, as much as there is for clean air and clean water, and the need to preserve the multi-functional nature of soil is currently the basis of most soil protection policies (Defra, 2004; CEC, 2006).

But can soil be multi-functional? Only certain of the six functions listed above are compatible. It is possible to imagine an area of grassland under which lies a buried archaeological site with the grass utilised by grazing animals, while the soil maintains its biodiversity and acts as a temporary reservoir for incoming rainwater. In this example the first four of the six functions are clearly compatible with each other; while it is difficult to imagine a situation in which the other two functions could be compatible with all or any of the first four. We would argue that it is impossible to reconcile the functions of source of raw material and/or foundation for construction with any form of soil protection.

When there are pressures on land use, how can the conflicting functions of soil be reconciled? If there is a pressing need for more houses, does this become more important than growing food or preserving biodiversity? If we assume that some areas of land on particular soil types are more suitable for house-building than others, does such information carry any weight in planning decisions? If particular soil types are more susceptible to damage when used for arable crop production, does this knowledge have any influence on the land management decisions of UK farmers?

Preserving the multi-functional nature of soil is good as an objective. In practice however it is difficult to apply at the individual soil level and is better employed as one input to the establishment of regional policies for more sustainable patterns of land use. Since its emergence as a discrete branch of science, the study of soil has therefore addressed a range of societal requirements and contributed to the well-being of society and of the natural environment. But what will its role be in the future?

FUTURE DIRECTIONS AND PRIORITIES

This second part is a forward look and seeks to establish the drivers of land use and management and the key environmental and societal priorities that will challenge future soil scientists. What will be the policy objectives of the future that will drive soil research and soil management practice?

The Global Context

Modern communications and logistics mean that the UK is subject to world trends and Table 1 identifies the key challenges that society will face over the coming decades.

Table 1: Future world challenges with implications for soil use and management

Challenge	Implications
World population growth	Greater pressure on/erosion of natural resources; greater competition for natural resources - energy, food, timber, minerals
Economic growth in the developing world	Greater competition for natural resources as above
A changing climate and therefore environment	Greater uncertainty; increased resource degradation in selected areas; pressure to reduce energy consumption

The world's population in 2050 is predicted to have increased by 50 per cent from its current 6.6 billion to approximately 9 billion. While Europe's population is predicted to drop, this is counter to the trend in all other continents with the steepest increase in developing world nations.

This will express itself as increased demand for food, timber, minerals and land during a period when the world stock of land will shrink due to degradation and marine inundation of the main river deltas and other low-lying coastal areas. These challenges are summarised in the Treasury's Public Policy Challenge 5 'Increasing pressures on our natural resources and global climate from rapid economic and population growth in the developing world and sustained demand for fossil fuels in [these and] advanced economies' (HM Treasury, 2006). The global context in which UK's future land policy needs to be assessed is therefore one of increasing demand, shrinking capacity, and much increased uncertainty at all spatial and temporal scales.

Challenges for the UK

Three issues that will impact significantly on how the UK land resource will be used and managed in future are discussed in more detail.

Climate change

It is widely accepted that the world's climate is changing due to man's impact on greenhouse gas concentrations in the atmosphere (IPCC, 2007). The latest IPCC reports conclude that increased atmospheric carbon dioxide concentrations and warmer temperatures will most likely increase crop yields in higher latitudes where moisture is not as limiting while reducing them at lower latitudes (Easterling *et al.*, 2007). However the majority of UK food production is rain-fed and the yields of a significant proportion of grain and seed crops and of grasslands are limited by soil moisture reserves. The productivity of these areas may be depressed by future climate change. In the driest parts of the UK, in addition, water resource constraints either due to drought or increased demands for drinking water may limit the availability of water for crop irrigation. With some of the largest areas of Grade 1 and 2 land at or below 5 m above sea level, a significant proportion of the country's most productive soils are at risk from coastal flooding exacerbated by the North Sea storm surge effect.

Rising temperatures, changes in the seasonality of precipitation and a potential increase in the frequency of extreme events also have implications for soil properties and process rates (Parry *et al*., 2007). The focus has been on changes in soil organic carbon and the influence of nitrogen and moisture availability on carbon sequestration particularly in organic soils under natural vegetation (Fischlin *et al*., 2007). However there is evidence of losses from all farmland soils other than those with the very least organic matter (Bellamy *et al*., 2004) which probably result from changes in land use and management as well as changes in climate.

Land use change

Agriculture dominates UK land use with over 70 percent of all land used for some form of production (Defra, 2006). Despite this, only one third of food consumed in the UK is home-produced (Carnus, 2004), although UK agricultural production is equivalent to 62 percent of UK food consumption. While urban land in the UK only accounts for 6 percent, it represents 10 percent of England and is strongly biased toward the south and east and the most productive soils because of the historic pattern of land settlement. Urban land has increased by a tenth since 1998 and a step change increase is scheduled for the south and east of England through to 2016. For the UK, increased demands for development land are running ahead of any population growth which has, until now, been static. Demographic trends, migration and changing expectations are translating into a continued pressure for more land to be developed. Household numbers in England are expected to grow from 20.9 m in 2003 to 24.8 by 2021 (DCLG, 2006). At the same time, policy mechanisms for protection of the ‘Best and Most Versatile’ land are ineffective and other factors are governing development decisions. While the focus on brownfield redevelopment has been effective for housing, it does not apply to industrial and retail developments.

Environmental regulation

Modern agriculture relies heavily on large inputs of fertilisers and the application of sophisticated crop protection compounds. As we have already seen, post-war agricultural research was immensely successful in enabling farmers to achieve yields that were previously unimaginable. However such yields are at a cost. As part of Europe, the UK is subject to European environmental policy which is putting increasing pressure on member states to regulate pollution particularly of water. The quality of natural waters is now to be achieved through implementation of a framework regulation for achieving good ecological status for water bodies - the Water Framework Directive. Infraction proceedings over nitrate levels in water have led to further proposed measures which are currently the subject of public consultation. Detection level-based pesticide limits in drinking water are equally challenging to the main arable cropping systems that rely on autumn establishment and therefore have crops in the ground over the winter when rainfall exceeds evapotranspiration. A Voluntary Initiative has been established to promote best practice in the hope that this will lead to a reduction in the frequency of water pollution events. Following its widespread presence in drinking water sources, the selective graminicide Isoproturon will be suspended in June 2008. Carbetamide, a control for black grass (Alopecurus myosuroides Huds) in oilseed crops for which no practical alternative exists, is also under scrutiny. Loss of these compounds and the ratcheting up of restrictions on nitrogen fertilisation threaten the sustainability of broad acre crop production. The timetable for compliance with the Water Framework Directive relates back to the ratification date of December 2000. By 2012, operational programmes of counter-measures must be in place for all river basins and the main environmental objectives of the directive must be met by 2015. A sister directive focused on soil is currently under discussion (CEC, 2006).

The Implications for Soil Research

Society faces a future of increased uncertainty. The challenge of achieving food security while maintaining the quality of the natural environment is significant. The uncertainty and lack of understanding of future climatic effects on the functionality of UK soils and land use systems adds to the challenges Government and land managers face.

A number of groups have reviewed future priorities for soil research (Drew Associates, 2003; the UK Soil Research Advisory Committee (SRAC, 2007) and NERC (2007) has just published its strategy for 2007-2013. The UK Audit of Soil Research (Drew Associates, 2003) saw opportunities for soil research to contribute to the big contemporary environmental issues of climate change, pollution and remediation and sustainable land management. They saw opportunities such as the application of in situ sensor techniques, but reported on the lack of investment in areas such as pedology, soil mineralogy and surface chemistry. They recommended an audit of the UK soil research skills base and reported on a widespread agreement that future soil research should be multidisciplinary.

The UK Soil Research Advisory Committee (SRAC, 2006) held a recent meeting to identify future horizons for soil research. The key question identified related to how soil systems function and interact with other biological and environmental systems on Earth. Soils were seen as the most complex system, because of their structural heterogeneity over many scales. The main challenge was seen as the understanding of soils as dynamic interactive systems building on existing infrastructure and contemporary programmes across the environmental and socio-economic spectrum. The concept of a multi-site Soil Observatory as a resource for understanding soil functioning at scales ranging from molecular to landscape was discussed.

Although the last ten years has seen an unprecedented number of soil policy initiatives both at European and national levels their effectiveness has been hampered by the lack of agreed indicators of quality or even simple information on the state of soils. No headline sustainable development indicators exist for soil quality and soil is therefore not on the sustainable development agenda. The new Defra action plan on ecosystem services (Defra, 2007) has identified the need for better information on the capacity and condition of natural resources such as soil.

A Future Agenda for Soil Science

We believe that the agenda within the context of this conference is very clear. The understanding of soil processes, of how these operate in different soils and of how they react to, and can be exploited by, management and their response to other influences such as a changing climate is central to our future survival. The priority within agriculture is to develop more efficient, low input:high output systems of production that have minimum impact on water, soil and atmospheric quality.

This is the case not just because of the need to comply with current environmental regulation but to reduce energy consumption in the supply of basic resources such as food and clean water and to secure our food supplies in an increasingly populated world. Just as food security was the driving force in the post-war period, so it will be again but in a world with strong incentives to minimise energy consumption. This will not be achieved without a return to production-oriented research, a topic that was abandoned in the 1980s with the perception of apparent food surpluses in Europe despite shortages elsewhere.

Soil management has a key and fundamental role to play in achieving this objective. Raising public and Governmental awareness of the importance and value of soil and of soil science is a critical first step. Within soil science, it is our duty to explain the central role that soil and its sustainable management have in the future provision of food, clean water and other ecosystem services in an energy-efficient way and in sufficient quantities to meet national and global needs.

REFERENCES

Addiscott TM, Whitmore AP and Powlson DS (1991). Farming, Fertilisers and the Nitrate Problem. CAB International, Wallingford, UK.

Carnus T (2004). International trends in primary production and sustainability. Three case studies: UK, Netherlands, Australia. Background report prepared for the Parliamentary Commissioner for the Environment. Wellington: PCE.

CEC (2006). Proposal for a Directive of the European Parliament and of the Council establishing a Framework for the Protection of Soil and amending Directive 2004/35/EC, 2006/0086 (COD). Commission of the European Communities, Brussels.

Council of Europe (1972). European Soil Charter, Council Resolution (72)19, 30 May.

DCLG (2006). Communities and Local Government Statistical Release 2006/0042.

Defra (2004). Soil Action Plan for England and Wales 2004-2006. Defra Publications, London.

Defra (2006) E-Digest Statistics: Land Use and Land Cover. http://www.defra.gov.uk/environment/statistics/land/ldurban.htm#ldtb4

Defra (2007) Securing a healthy natural environment: an action plan for embedding an ecosystems approach. Defra Publications, London.

Drew Associates (2003). Audit of UK Soil Research. http://www.defra.gov.uk/environment/land/soil/pdf/audit-report.pdf, last viewed 14 Nov 2007.

Easterling WE, Aggarwal PK, Batima P, Brander KM, Erda L, Howden SM, Kirilenko A, Morton J, Soussana J-F, Schmidhuber J and Tubiello FN (2007). Food, fibre and forest products. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. ML Parry, OF Canziani, JP Palutikof, PJ van der Linden and CE Hanson (Eds). Cambridge University Press, Cambridge, UK, 273-313.

Fischlin A, Midgley GF, Price JT, Leemans R, Gopal B, Turley C, Rounsevell MDA, Dube OP, Tarazona J and Velichko AA, (2007). Ecosystems, their properties, goods, and services. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. ML Parry, OF Canziani, JP Palutikof, PJ van der Linden and CE Hanson (Eds.). Cambridge University Press, Cambridge, UK, 211-272.

HM Treasury (2006). Long -term opportunities and challenges for the UK: analysis for the 2007 Comprehensive Spending Review. HMSO, Norwich.

IPCC (2007). Summary for Policymakers. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. S Solomon, D Qin, M Manning, Z Chen, M Marquis, KB Averyt, MTignor and HL Miller (Eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Loveland PJ and Thompson TRE (2002). Identification and Development of a National Set of Soil Quality Indicators, R&D Project Record P5-053/PR/02. Environment Agency, Bristol, UK.

NERC (2007). Next Generation Science for Planet Earth, NERC Strategy 2007-2012. Natural Environment Research Council, Swindon, UK.

Parry ML, Canziani OF, Palutikof JP and Co-authors (2007). Technical Summary. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. ML Parry, OF Canziani, JP Palutikof, PJ. van der Linden and CE Hanson (Eds.). Cambridge University Press, Cambridge, UK, 23-78.

RCEP (1996). The Sustainable Use of Soils, 19th Report of the Royal Commission on Environmental Pollution. HMSO, London.

SRAC (2006). Horizons in soil research: synthesis. Unpubl.

Strutt N (1970). Modern Farming and the Soil. Agricultural Advisory Council, HMSO, London.

Van Lynden GWJ (1994). The European Soil Resource: current status of soil degradation in Europe: Causes, impacts and need for action. ISRIC, Wageningen, Council of Europe, Strasbourg.

THE LEGAL FRAMEWORK FOR LAND (AND WATER) MANAGEMENT – PERCEPTIONS AND OPPORTUNITIES

P Howsam

Centre for Natural Resource Management, B.42a, Cranfield University, Bedfordshire, MK43 0AL, UK, E-mail: p.howsam@cranfield.ac.uk

SUMMARY

With a title such as Land Management in a Changing Environment there will naturally be much focus on the inter-relationship between the physical environment and anthropogenic activities. This paper takes a different perspective and will address the legal environment for land management. However because it is not sensible to talk about land management without regard to water management, both will be considered. It aims to review the legal framework in which land management has to operate and asks the questions - Is it fit for purpose? Will it's implementation be fit for purpose?

Key components of this legal environment are the Soil and Water Framework Directives. The paper will provide a legal interpretation of the requirements of the Directives and suggest the likely implications for different stakeholders in different circumstances, including differences for those in England, Wales, Scotland and Northern Ireland. This interpretation will rely to some extent on a review of past and current legislation dealing with land, soil and water, including rights and duties under private law. Also considered are other factors such as public perception, political priorities, human and financial resources and other components of a legal framework such as administration, regulation and the judicial system.

A STERN WARNING: DIFFICULT DECISIONS ON CLIMATE CHANGE

D Moran

SAC, West Mains Road, Edinburgh, EH9 3JG, UK,
E-mail: Dominic.Moran@sac.ac.uk

SUMMARY

Climate change is now a scientific fact, and its potential consequences present dilemmas for decision-making by governments trying to evaluate the risks of potentially catastrophic outcomes that may occur generations into the future. What to do about mitigating greenhouse gases is relatively clear, although cost considerations suggest a rational approach that compares mitigations options with the benefits they bring in terms of avoided damages. This paper outlines the evolution of thinking on the economics of greenhouse gas mitigation, which has been largely influenced by the contribution of the Stern Review (The Economics of Climate Change). While some of Stern's assumptions remain controversial, the report has provided considerable impetus for action by UK government, which is now committed to an ambitious programme of emissions reduction. Such obligations have implications for sectors responsible for emissions, and Scottish agriculture is implicated as a major contributor to the national greenhouse gas account.

INTRODUCTION

The world is getting warmer and so is the political, economic and ethical debate about how to address environmental impacts that could potentially undermine global growth and wellbeing for generations. It is impossible to divorce climate change policy relating to land use and agriculture from the wider scientific agenda that predicts warming scenarios, and that provide the basis for global, national and regional impact assessments. These assessments in turn give rise to the present value damage cost calculations that are the basis of the best estimate for the shadow cost of carbon emissions, the so-called price of carbon, which is set to figure prominently in policy decision-making (Defra, 2007). More immediately, emissions and damage scenarios provide the impetus for international agreement needed to address climate impacts specifically, Kyoto and successor agreements.

While it is possible for countries to free-ride on international climate initiatives, the fundamental public good nature of the problem provides the imperative for cooperation to affect the stabilisation of greenhouse gas concentrations. Within the confines of externally determined commitments, governments have flexibility as to how to regulate emissions and how to pro rate responsibility across emitting sectors. In most countries this process has targeted energy generators and heavy industry, which are more obvious targets for monitoring. Emissions reduction in other sectors is by implication voluntary, although in the case of agriculture this situation is likely to change.

In contrast to mitigation, there is considerable flexibility in how countries and sectors can adapt to climate change. Land managers can choose whether or not to adapt, depending on anticipated costs, or expected damages, which in turn emerge from the best assessment of downscaled impacts. The adaptation debate has highlighted the private nature of adaptation decision-making. In the case of agriculture it is worth noting that collective uncoordinated private decisions may yet have public good consequences for the environment. This distinction does not appear to feature largely in adaptation literatures.

This paper focuses on mitigation choices in the context of existing evidence on warming and the likely emergent policy agendas driving the need to reduce emissions. The discussion is guided by findings of the Stern Review, which was highly influential in setting the parameters of the economic case for action on climate change; specifically the observation that given the best damage estimates, and how we treat the future, it would be cost-effective not to postpone mitigation. The paper scopes the steps from this conclusion to action in the land based sector in Scotland.

EMMISSIONS AND WARMING SCENARIOS

The global debate about whether warming will happen has now become a question of how much warming and the accuracy of recent scenarios from Intergovernmental Panel on Climate Change (IPCC)[1]. IPCC Fourth Assessment Report, published in 2007 paints a vivid picture of the uncertainties. Figure 1 shows various estimates of the increase in global average temperature in 2100, compared with today. Temperature is on the vertical axis: on the horizontal axis are the IPCC's six scenarios for future greenhouse gas emissions, broadly speaking, emissions are increasing from left to right. The numerous dots, circles and triangles are individual estimates from computer models of the climate system, while the thin vertical lines are ranges of estimates from one model. The horizontal black lines are the average estimates and the ranges of uncertainty respectively from the full 'ensemble' of IPCC models. The best-case scenario is a small increase in temperature of about 1ºC. A small increase like this will not be without consequences (e.g. for indigenous Arctic communities, low-lying Pacific islands and coral reefs). Low temperature scenario like this would likely be more manageable in most parts of the globe.

But the worst-case scenario is a vertiginous 6.5ºC increase in temperature, well over the 2ºC "dangerous" warming threshold. This would be entirely unprecedented, comfortably more than the difference between temperatures today and temperatures during the middle of the last ice age, when, simply as a statement of fact, northern latitudes were under hundreds of meters of ice (remember this is a global average: on land we are looking at well over 10ºC warming). Not only will the consequences of 6.5ºC warming be severe on those dimensions we understand better (e.g. significant losses in global crop productivity, risks of coastal flooding), we run a high risk of triggering the various abrupt and global-scale system changes that we fear, including the well-advertised collapse of the Greenland and West Antarctic Ice Sheets, rapid slow-down in the North Atlantic circulation, but also less well-advertised shifts in the monsoon system and the El Niño Southern Oscillation, which could cause a sudden failure in rains in some places, and widespread flooding in others. It is not clear how we can adapt to these changes. Even if we could, it is likely to be costly, in no small part because at this rate of change, the climate system is warming faster than the turnover of capital assets like buildings.

[1] UK specific impacts resulting from emissions scenarios are downscaled by UKCIP, which has now released new scenarios under UKCIP08 http://www.ukcip.org.uk/climate%5Fimpacts/ukciP08_Trends.asp

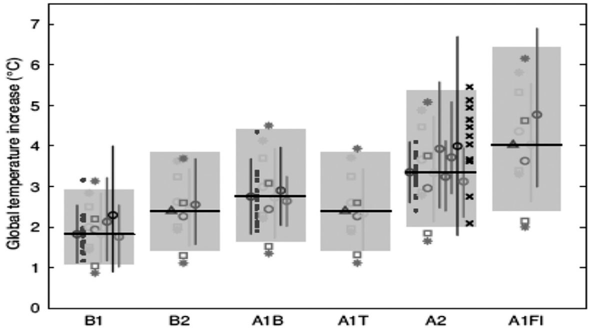


Figure 1: Temperature predictions across the range of IPCC emissions scenarios (source: IPCC 4th Assessment)

WHEN?

If warming is incontrovertible the question of which scenario remains highly uncertain. The alarming prospect of high end warming is as likely as low end scenarios, and climate scientists appear unable to be more specific. This uncertainty combined with specific time horizons for damages, partly explain why policy can be so apparently reticent in the face of scientists' calls for increased precautionary investment in research, mitigation and adaptation. This scientific uncertainty is also prone to be treated as a long term problem; or at least one that can be pushed over any normal political time horizon.

Added to this national inertia is a fundamental problem, the world is not a single country. Thus dealing with the emissions reductions required by the science[2] is highly complicated collective (in)action particularly failure of leadership by US perceived injustice and indifference.

In this difficult political context the UK could be interpreted as having embarked on an exercise of attempting to seize moral leadership in the area. By demonstration of its own efforts it (Tony Blair) is hoped to exert moral suasion on the other parties, so getting them to come into line. The Scottish Government has intimated that it would like to go further still. Part of the impetus for this was provided by the Stern Review.

The Stern Review of the Economics of Climate Change, advanced a compelling case for not treating greenhouse gas mitigation in the same way as other public investment decisions. In normal cases governments are justified in treating long term costs, which arguably climate change has become by scientific default, as less important by discounting. This is the extent to which we as a society choose to discount future costs and benefits simply because they are in the future and (if they are far enough in the future) happening to different people and not ourselves.

[2] A broad consensus is that avoiding dangerous climate change will require rich countries to cut emissions by at least 80 per cent, with cuts of 30 per cent by 2020. Emissions from developing countries would peak around 2080, with cuts of 20 percent by 20-50 (United Nations, 2007).

In the UK, the Treasury recommends a specific rate (3.5%) for general use. But given the global and temporal dimensions of the climate problem, the Stern Review deviates from this treatment of the future, and this is implicit in the crucial parameter assumption of the pure rate of social time preference. If like Stern, you choose a value near (but not quite) zero (just enough to account for the possibility that there will be no one around in the future, or at least no one in a position to care about our current choices on global warming), you reach the conclusion that immediate action to fix global warming is justified. If, like most of Stern's critics (e.g. Nordhaus, 2007), you choose a rate of pure time preference around 3 per cent, implying that the welfare of people 90 years (roughly three generations) in the future counts for about one-sixteenth as much as the welfare of people alive today, you conclude that we should leave the problem to future generations. So, responses to the Stern Review are a form of sensitivity analysis of his results. If you don't care (much) about future generations, you choose a higher discount rate and do not do anything (much) about global warming.

While there is still plenty of dispute about the economic costs of doing nothing, relative to stabilisation, the median estimate has been revised sharply upwards relative to the modest growth costs of doing something now. Put simply, the balance of evidence makes immediate action look like a good insurance bet. The clarity of this message has seen numerous governments changing their previous stances on mitigation.

POLICY AND THE LAND BASED SECTOR

The UK has an international target (under the Kyoto agreement) to reduce its greenhouse gas emissions by 12.5% by 2008-2012. It also has two more ambitious domestic goals - to reduce carbon dioxide emissions by 20% by 2010 and to reduce them by some 60% by around 2050. The latter confirmed in the Energy White Paper, published in May 2007. The Scottish share of these cuts is somewhat complicated by the administrative leeway afforded by reserved and devolved powers. The Scottish Government is currently consulting on this issue[3]. In the UK the consequences of these commitments are only binding for certain industries. Without specific regulation, non regulated industries can continue to free-ride.

However, what is clear is that an inventory of major polluters reveals that a significant share from agriculture. Estimation of Green House Gas (GHG) emissions associated with Scottish agriculture is hampered by methodological and data problems but current inventory figures used in formal reporting suggest that – taking a broad definition - agriculture is a significant source of emissions (Moxey, 2007), accounting in total for approximately 1/4 of total Scottish emissions, expressed as carbon dioxide equivalents (CO_2e). Within this, agriculture contributes less than 1/5 of total carbon dioxide (CO_2) but over 3/4 of all nitrous oxide (N_2O) and 1/2 of all methane (CH_4) emissions.

Agricultural emissions arise mostly as CO_2 from the conversion of land to cropping (nearly 1/2 of total agricultural emissions), N_2O from the application of fertiliser

[3] Climate Change: consultation on proposals for a Scottish climate change bill http://www.scotland.gov.uk/Publications/2008/01/28100005/0

and manure to soils (1/4), and CH_4 from enteric fermentation in ruminant digestion (1/5). Against this, agriculture also represents a GHG sink, with sufficient CO_2 being sequestered by crop and grassland to offset nearly 1/5 of agricultural emissions – bringing its net contribution to Scottish emissions closer to 1/5 than 1/4.

In the UK, agricultural emissions are currently outside any regulation, although as government signals an intention for all sectors to take responsibility, with a preference for introducing market based mechanisms for affecting cost-effective reductions.

In theory a market based approach could mean taxing emissions on polluting inputs and or a trading regime, which may include a link to the exiting EU Trading Scheme. Both mean that polluters can be faced with a price for their emissions. In the case of a tax, the rate would most likely be set to reflect the marginal damage cost of emissions - a price which is now notionally set by the social cost of carbon 25/ $tC0^2e$ by 2015. In a trading regime the price of permits is very much dependent on the demand and supply conditions imposed on the participants. The market based approach has properties which in theory lead to least cost mitigation. Both methods represent a notional transfer of the property right to pollute, and in both cases, the cost of maintaining that right can be palliated by more or less favourable allowances to lower marginal tax rate or initial pollution quotas.

In practice a new tax instrument is unlikely top be politically attractive. On the other hand, the application of a trading instruments is fraught with difficulties related to the significant monitoring, and transactions costs involved in introducing and running a regime. As shown in recent work for Defra (Nera, 2007), the structure of farming is not conducive to an efficient trading regime, with small farms emitting less than 600 t CO_2 per /year accounting for 90% of sector emissions. This contrasts to the profile of ETS participants where a much smaller number of large emitters helps minimise transactions and monitoring costs.

Command and control and voluntary measures offer alternatives to market based methods. These approached include the use of cross compliance emissions requirements within agri- environmental schemes. This approach combined with use of codes of practice and KT have furthered progress in the management of other diffuse pollutants.

In evaluating these alternatives government will still be seeking to affect mitigation efficiently. In other words, and as a rule of thumb if the social cost, or damage caused by a tonne CO_2 equivalent is approximately £25, society should not be spending more than this on abatement. More technically abatement strategies need to look across industries to apply the principle of equalising the marginal cost of abatement across sectors. So an important research agenda comes down to working out whether agricultural emissions are least cost.

LEAST COST MITIGATION IN AGRICULTURE

How much regulatory pressure should we expect agricultural emissions to come under? Aside the political aversion to introducing further regulation, the answer should depend partly on the cost to mitigate emissions from agriculture relative to any other industry. Industry wide estimates of the marginal cost of emission abatement are in fact difficult to derive, but recent evidence compiled by Hanley (2007) suggests a range that suggests agricultural abatement measures lying below the benchmark set by the social cost of carbon. Analysis by Nera (2007) suggests that these measures include the use of maize silage, reducing some livestock stocking rates, afforestation, improving milk yields and larger centralised anaerobic digestion. However, given the size of enterprises are relatively small, the rewards to participants as benchmarked by the social cost of carbon may mean that incentives to participate in any voluntary schemes are low.

Table 1: Marginal abatement costs for CO_2 e

Sector	Costs per tonne CO_2 eq.	Comments
Industry	£14	Current EU ETS price.
Housing	negative	Based on UK wide data
Transport	Not known	No Scottish research available
Renewables	£11 - £49	Depends on whether on- or off-shore wind and whether replaces coal or gas
Agriculture	£10	Can deliver up to 1 Mt/yr, but based on US/EU data
Biofuels	£75 >	No current estimate of Scottish land area likely to convert
Forestry	£4-£12	Assumes additionality
Carbon capture and storage	Not known	

CONCLUSION

There is dispute that a well-designed set of policies could greatly reduce global CO_2 emissions at very low cost. The global cost estimate in the Stern Review is marginally lower than average at 1 per cent of GDP, but it would be hard to find any serious analyst claiming costs much higher than 3 per cent. These are once off changes in levels corresponding to a once-off loss of between a few months and one year of improvements in material living standards. It is intuitively hard to see how risking the worst case outcomes of climate change to avoid such a small economic cost could possibly be justified. This line of thought is bolstered an ethical value judgement which has much do with how we decide to weight the potential welfare of future generations. With this high future damage weighting on the balance of costs and benefits it makes sense to act now. Emphasising the implicit carbon price provides a signal for how and when mitigation can be efficient and this has motivated action at national level and in the UK, this has furthered the use of market-based approaches to managing pollution.

Future regulation of agricultural emissions represents a progressive transfer of property tights (to pollute) from the sector. This is in keeping with the need to supply public goods (controlling a negative externality) from the sector. Farmers can be expected to be rewarded for compliance in this supply of greenhouse gas mitigation. Within the agricultural mitigation portfolio, some reductions can be delivered at lower cost than others, and more research needs to be conducted to determine the marginal cost of greenhouse gas abatement options.

ACKNOWLEDGEMENTS

This work was funded by the Scottish Government Rural and Environment Research and Analysis Directorate (RERAD).

REFERENCES

Defra (2007). The Social Cost Of Carbon And The Shadow Price Of Carbon: What They Are, And How To Use Them In Economic Appraisal In The UK. http://www.defra.gov.uk/environment/climatechange/research/carboncost/pdf/background.pdf

Financial Times (2007). Why the climate wolf is hard to kill by Martin Wolf. 4 December 2007. http://www.ft.com/cms/s/0/fc338d2a-a292-11dc-81c4-0000779fd2ac.html

Hanley N (2007). The Economics of climate change policy in Scotland, paper David Hume Institute, Edinburgh.

Moxey A (2007). Reviewing and developing agricultural responses to climate change, report to RERAD.

Nordhaus W (2007). Critical assumptions in the Stern Review on Climate Change, Science Vol. 317, 201-202.

Nera (2007). Market Mechanisms for Reducing GHG Emissions from Agriculture, Forestry and Land Management, report for Defra.

The Stern Review of the Economics of Climate Change. http://www.hm-treasury.gov.uk/independent_reviews/stern_review_economics_climate_ change/sternreview_index.cfm

United Nations (2007). Human Development Report Fighting Climate Change: Human Solidarity in a Divided World. http://hdr.undp.org/en/media/hdr_20072008_en_complete.pdf

REDUCING *ESCHERICHIA COLI* O157 RISK IN RURAL COMMUNITIES

H Gordon, G Edwards-Jones, J Roberts, A Wossink, C Jones, C Hunter, D Chadwick, I Ogden, D Jones, O Rotariu, N Strachan, S Erdem, D Rigby, J Farrington, P Teunis, R Kosmider, P Williams, P Cross and K Killham

Universities of Aberdeen, Bangor, Manchester and London, E-mail: k.killham@abdn.ac.uk; IGER North Wyke; Veterinary Laboratories Agency; RIVM

SUMMARY

This paper reports on the distribution of *E. coli* O157 across a sheep stock transfer farm in Aberdeenshire in order to identify the different hydrological pathways by which the pathogen is transported. This was carried out as the first stage of a RELU project to identify possible mitigation strategies to reduce risks of O157 infection in rural areas.

INTRODUCTION

Infectious intestinal disease causes substantial morbidity being responsible for over 300 deaths and 35,000 hospital admissions annually (Lewison, 1997). The Food Standards Agency estimate that 4.5 million people in the UK suffer annually from food poisoning. Our previous work has estimated that the cost to the UK economy is £350 million (FSA, 2000). Recent, high profile outbreaks of human diseases caused by food borne pathogens such as *E. coli* O157 have highlighted a serious lack of knowledge and understanding about the factors which determine both the population dynamics and dispersal of these pathogens in rural and agricultural environments (Defra, 2003). These pathogens primarily enter the food chain from contaminated meat products, however, there are an increasing number of environmental outbreaks associated with consumption of contaminated water and contact with soil and livestock. This paper focuses on the pathogen *E. coli* O157 which has been targeted by the FSA as posing a major risk to human health.

It has been estimated that between 10-40% of UK cattle herds are infected with O157 and that the pathogen has spread to a range of wild and domesticated animals (e.g. birds, rodents, deer) (Jones, 1999). *E. coli* O157 is introduced into the environment from farm ruminants primarily from direct defecation but also from agricultural wastes spread to land (Oliver *et al.*, 2005). These can directly contaminate the food chain during primary (slaughtering) or secondary food processing. Runoff, leaching and waste spreading within agricultural environments can contaminate water supplies, pastures, vegetable crops, stiles etc which can subsequently lead to animal re-infection and transmission to humans (Williams *et al*., 2005). The discovery of the capacity of human pathogens to persist in the soil for months (Ogden *et al.*, 2002), and even years, was a key scientific finding and a frightening revelation for society (Jones, 1999).

This paper aimed to determine the distribution of O157 strains across a sheep stock transfer farm in order to identify hydrological pathways by which the pathogen is transported. This was carried out as a first step towards identifying possible mitigation strategies to reduce risks associated with the pathogen in rural areas.

MATERIALS AND METHODS

Field Site - Field sampling was carried out on a farm in Aberdeenshire on two separate occasions for a period of twelve weeks, each between April and July in 2005 and 2006. During the first sampling session, faeces, soil and water samples were tested for O157. The second twelve week period followed the same sampling strategy as the first, but with the inclusion of drain water samples and the exclusion of soil samples.

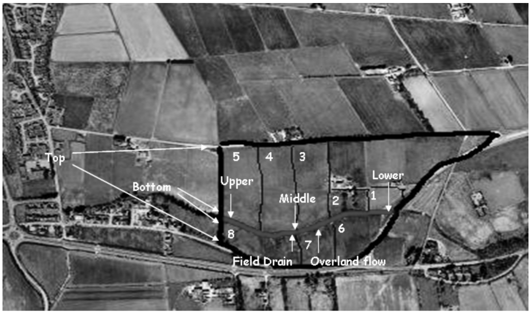


Figure 1: An aerial view of the farm detailing the stream sampling points and field numbering system. The bottom of the field is always the side of the field nearest the stream. Four overland flow traps were installed adjacent to the boundary between fields 6 and 7

The site was chosen for a number of reasons. Firstly, its close proximity to the laboratory meant that samples could be collected and processed rapidly. Secondly, the farm formed a mini catchment area with steep slopes leading to a small permanent stream. Thirdly, the land was leased to a sheep dealer which meant a constant change in animal stocks increasing the likelihood of O157 being detected. Finally, the farm had been studied in the year prior to this study and had tested positive for O157 on several occasions (Urdahl *et al.,* 2005).

The fields were labelled 1-8, with fields 1-5 on the south side of the stream and the remaining fields on the north side (Figure 1). In fields seven and eight, there was a 2500 m^2 area of marshland from which a shallow drainage ditch had been dug. The drainage ditch ran from the marsh directly into the stream and offered an opportunity to collect saturated overland flow continually throughout the study.

Sampling Strategy - Faeces were collected from each field in sterile, re-sealable, plastic bags. Each faecal sample consisted of at least 100 g of material collected from one pat. Only faeces that were still wet were collected. Once collected, the samples were transported directly to the laboratory where they were stored at 4 ± 1°C for a maximum time of 24 hours until processed for bacterial isolation.

Soil samples were taken and analysed for water content and bacterial load during the first session of fieldwork. The soil samples were collected during each visit from two fields chosen at random. The field was divided up using a grid system with cores being taken from the top, middle and bottom of the field, with the bottom being the side of the field nearest the stream. Soil samples were taken using plastic cores of 4 cm diameter and 15 cm length. The five individual cores were pooled to make one large sample for each section of the field. The soil samples were stored in sterile, re-sealable, plastic bags at 4 ± 1°C for a maximum time of 24 hours until processed for bacterial isolation.

During the first fieldwork session, water samples were collected from the upper (where the stream entered the farm), middle and lower (the point the stream left the farm) end of the stream (see Figure 1). A sample was also taken from the overland flow resulting from the marshland. The same locations were also sampled during the second session. In addition, subterranean drains that emptied into the stream were sampled at the point of exit. Between one and three litres of water was collected in sterile plastic containers at each visit. Also, during the second session, overland flow was tested for the presence of O157. Traps were installed down one side of field 6 and were checked for collected water at every visit. The traps were constructed from plastic dustpans with a hole drilled in the back to allow the collection of water. The hole was plugged with a tapered tube connector and was connected to a plastic container via a silicone tube. The dustpans were covered in electrical tape to ensure that any water collected was from the generation of overland flow and not rainwater. The water was stored at 4 ± 1°C for a maximum time of 24 hours until processed for bacterial isolation.

Culturing Techniques - During the first fieldwork session, the isolation of O157 from faeces and soil involved the pooling of samples. Five faecal samples were collected from one field so 5 g aliquots of each sample were combined to make a total of 25 g. Enrichment medium (buffered peptone water (BPW) supplemented with 8 mg l^{-1} vancomycin), was added to the solid sample in a stomacher bag at a ratio of 1:9 sample to BPW. The bags were then homogenised for thirty seconds each. As the soil had been pooled, 25 g were measured directly from the resulting samples. As many of the soil samples contained stones, homogenisation was by hand for a total of one minute as opposed to stomaching.

Water samples were initially prepared by centrifuging 450 ml for 15 minutes at 7,000 g at 4°C. The supernatant was removed and the resulting pellet re-suspended in another 450 ml of sample, and centrifuged again. The supernatant was again removed, leaving 5 ml to prevent target loss. The pellet was then fully re-suspended in BPW at a ratio of 1:9. During the second sampling session, the technique had to be modified for larger 1 l volumes which were spun for 30 minutes at 2,560 g and 4°C. As with the previous method, 5 ml was left to ensure any pellet was not lost. The pellet was re-suspended in BPW at a ratio of 1:9.

Water samples comprising small volumes, such as those recovered from the overland flow traps, required a slight variation on the above method. Samples were made up to 50 ml with sterile, distilled water and transferred into a 50 ml sterile centrifuge tube. This was then centrifuged for 20 minutes at 3,790 g and 4°C. The supernatant (5 ml) was collected, and BPW added at a ratio of 1:9. All samples were then incubated for 6 hours at 42°C to maximise target recovery.

Immunomagnetic Separation (IMS) - Two methods of IMS were used to isolate O157 (Ogden *et al*., 2001). For the faecal and water samples, automated IMS was carried out using the BeadRetriever™ instrument. For soil samples, the IMS procedure was carried out manually due to sediment settling over the magnetix beads which obstructed the automated process. Colonies were confirmed by latex agglutination.

Multiple-Locus Variable-Number Tandem-Repeats Analysis (MLVA) - MLVA is a technique developed to determine the Variable Numbers of Tandem Repeats (VNTRs). These repeats are a source of genetic polymorphism that can be used as a typing assay (Lindstedt *et al*., 2004). The VNTRs selected were situated on five genes and two intergenic regions and had repeated motifs ranging from 6 to 30 bp (Lindstedt *et al*., 2003). The method required an initial PCR step using two primer sets. The primers were named Vhec1-7 with the forward primers 5' labelled with fluorescent dyes (Lindstedt *et al*., 2004).

DNA was isolated by picking up to three identical colonies of the O157 (to ensure adequate amounts of DNA were extracted) and transferring to an Eppendorf tube containing 50 µL Sigma water. The tube was placed in boiling water for 5 minutes and mixed thoroughly before centrifugation at 15,700 g for 10 minutes. The supernatant was removed and transferred to a clean Eppendorf where it was stored at -20 ± 1°C.

The required PCR reactions were made using the Qiagen PCR multiplex kit. For each reaction, 45 µL of master mix was added to 5 µL of DNA. The reactions were performed for the following temperature profile: 95°C denaturation for 15 minutes followed by 30 cycles of 94°C for 30 seconds, 63°C annealing for 90 seconds and 72°C extension for 90 seconds followed finally for an extra 10 minute extension step at 72°C. Products were visualised on 3% TAE gel before being dried and sent to the Division of Infectious Diseases Control, Norwegian Institute of Public Health, Oslo, Norway, for capillary electrophoresis and subsequent analysis. This involved determining the allele number at each locus from chromatograms generated by the capillary electrophoresis (Lindstedt *et al*., 2004). A dendrogram was drawn by entering allele numbers into BioNumerics version 3.5 database as character values, and constructing the dendrogram using Categorical coefficients and the Ward algorithm (Lindstedt *et al*., 2004).

RESULTS

The results showed that there were nine different MLVA profiles forming three distinct clusters. The most common MLVA profile was 15-1-1-11-4-4-3. The profiles are generated by listing the number of repeats at each locus, allowing the strains to be classified as a series of numbers.

Table 1: MLVA results from forty-one isolates. L1-L7 are the loci detailed in Lindstedt *et al*. (2003). The numbers listed under each locus are the repeat numbers

Cluster	No. of different profiles	No. of strains	L1	L2	L3	L4	L5	L6	L7
I	5	22	13 14 15 16	2	1	11 12	4	4	6
II	2	16	14 15	1	1	11	4	4	3
III	2	3	0 9	1 2	2 9	6	0 6	4	6

A dendrogram was drawn and clusters were designated at 20% homology.

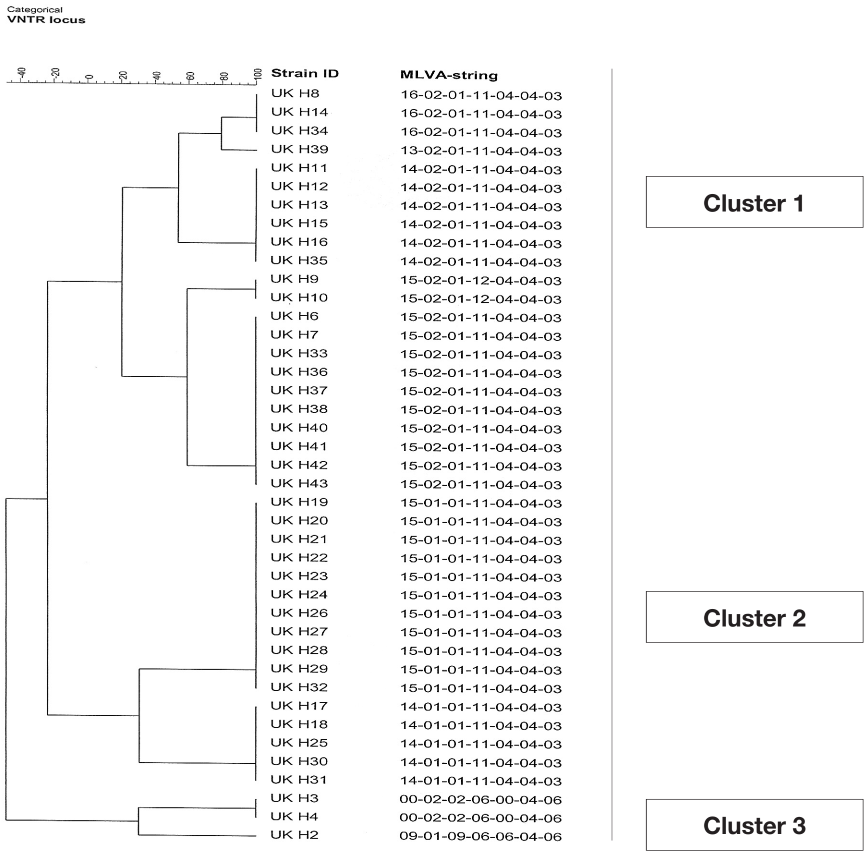


Figure 2: Dendrogram showing the relationship between strains of O157. All strains were isolated from faecal material with the exception of H2 from soil, H3 and H4 from the top of the stream, H17 and H18 from overland flow, and H27 and H28 from the lower end of the stream

Cluster I contains most isolates (22/41), all of which were from sheep faeces. Studies of a neighbouring farm upstream from the field site also found the same MLVA profile as here (14-2-1-11-4-4-3) in cattle faeces (Solecki, 2007) suggesting that the same strain of O157 can colonise both cattle and sheep.

Cluster II contains the isolates found in the overland flow (H17 and H18) and at the lower end of the stream as well as faecal isolates, indicating that one strain had exhibited direct transfer between these three matrices. It should be noted however, that the strain found in the overland flow was not the same as the strain found in the faeces the following week.

Cluster III consists of the isolates located in the stream (H3 and H4) on the 1 May 2006, twenty-one days before any other O157 isolates were found and the isolate found in the soil (H2) in 2005. The strain from the stream may have come from another farm upstream, but as the MLVA profile had not been seen previously, we cannot determine the source. The isolate from the soil (H2) is similar to an isolate found in sheep faeces (MLVA profile: 0-2-1-7-4-4-5) found on another farm in the local area (Solecki, 2007). The similar isolate was found in 2006 and may have been transported to the neighbouring farm from an infected animal.

When each sample was collected, the location was noted. By comparing this information with the dendrogram, it was possible to identify the location of each particular strain on the farm.

A

B

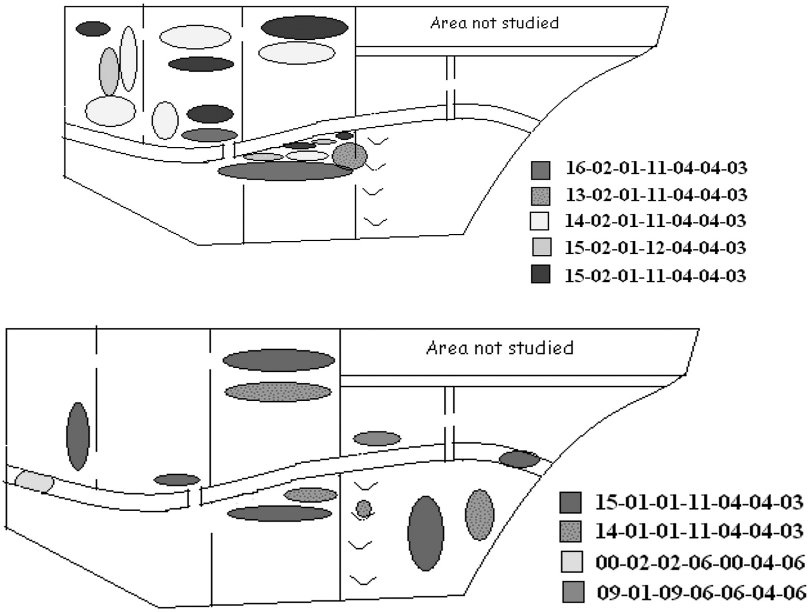


Figure 3: Diagrams showing the farm plan and locations that isolates were collected from. Diagram A contains the Cluster I isolates. Diagram B contains both the Cluster II and Cluster III isolates

DISCUSSION

Using MLVA, it was possible to determine which strains were common to the farm and which were isolated incidences. One strain from cluster I was also found in cattle faeces on a neighbouring farm (Solecki, 2007) illustrating that O157 can colonise both cattle and sheep. It is therefore possible that infected animals may have been transported to the second farm, subsequently spreading the strain. It may also be possible that the strain originated from the upstream farm, but this cannot be proven as the farm was not tested prior to this date.

A previous study at this field site observed that O157 was found in sheep faeces and in the stream water (Urdahl *et al*., 2005) which was able to confirm, using MLVA, that the same strain was washed from the fields into the stream. These strains (all from cluster II) were spread throughout the farm and were found in a variety of matrices including sheep faeces, overland flow and stream water. The same strain isolated from overland flow, proved for the first time that this hydrological pathway (in addition to matrix flow) was responsible for environmental transfer of O157.

In conclusion, a strain of O157 from sheep faecal material was found to be transported by matrix and overland flow to soil, streams and rivers. These hydrological pathways, particularly overland flow, need to be considered in future mitigation strategies designed to reduce the risk caused by O157 in rural areas.

ACKNOWLEDGEMENTS

The investigation carried out was funded by the BBSRC, and is now being extended through the RELU programme of the Economic and Social Research Council.

REFERENCES

Defra (2003). Economic Evaluation of Defra Policy on Food-Borne Pathogens in Live Animals.

FSA (2000) A report of the study of infectious intestinal disease in England.

Jones DL (1999). Potential health risks associated with the persistence of *Escherichia coli* O157 in agricultural environments. Soil Use and Management 15, 76-83.

Lewison G (1997). Gastroenterology in the UK: The Burden of Disease. Second Report of the Prism Project 9608, The Welcome Trust, London.

Lindstedt B-A, Heir E, Gjernes E, Vardund T and Kapperud G (2003). DNA fingerprinting of Shiga-toxin producing *Escherichia coli* O157 based on Multiple-Locus Variable-Number Tandem-Repeats Analysis (MLVA). Annals of Clinical Microbiology and Antimicrobials 2, 12.

Lindstedt B-A, Vardund T and Kapperud G (2004). Multiple-Locus Variable-Number Tandem-Repeats Analysis of *Escherichia coli* O157 using PCR multiplexing and multi-colored capillary electrophoresis. Journal of Microbiological Methods 58, 213-222.

Ogden ID, Hepburn NF and MacRae M (2001). The optimisation of isolation media used in immunomagnetic separation methods for the detection of *Escherichia coli* O157 in foods. Journal of Applied Microbiology 91, 373-379.

Ogden ID, Hepburn NF, MacRae M, Strachan NJC, Fenlon DR, Rusbridge SM and Pennington TH (2002). Long-term survival of *Escherichia coli* O157 on pasture following an outbreak associated with sheep at a scout camp. Letters in Applied Microbiology 34, 100-104.

Oliver JD (2005). The viable but nonculturable state in bacteria. Journal of Microbiology 43, 93-100.

Solecki O (2007). Personal Communication, University of Aberdeen.

Urdahl AM, Strachan NJC, Wasteson Y, MacRae M and Ogden ID (2005). A longitudinal study of *Escherichia coli* O157 contamination of a watercourse by farm ruminants at pasture. In Proceedings of Society for Veterinary Epidemiology and Veterinary Medecine, Nairn, Scotland, United Kingdom, 30 March-1 April. http://www.svepm.org.uk/posters/index.pfp?path_.%2F2005%2F.

Williams AP, Avery LM, Killham K and Jones DL (2005). Persistence of *Escherichia coli* O157 on farm surfaces under different environmental conditions. Journal of Applied Microbiology 98, 1075-1083.

LOCAL AND ADAPTIVE MANAGEMENT OF CATCHMENTS FOR THE PROTECTION OF WATER QUALITY: DRAWING ON INTERNATIONAL EXPERIENCE

LED Smith

Centre for Development, Environment and Policy, University of London (SOAS), Wye College, Ashford, Kent, TN25 5AH, UK, E-mail: L.Smith@soas.ac.uk

SUMMARY

Diffuse water pollution originating from agriculture and other rural activities poses challenges for public policy and requires innovative management approaches. Solutions ultimately require behavioural change on the part of land users and must be flexible and adaptive to stochastic catchment conditions and to long-term trends. Internationally new models of governance for land and water resource management are developing. This paper reports on a comparative analysis of international catchment management programmes and provides a synthesis of key programme elements and lessons of success. It is concluded that land management and diffuse sources of pollution have a local basis and protection of water at source necessitates the fostering of local instruments and participation of stakeholders in an adaptive process, supported by an enabling policy and regulatory environment. There is a need for local and wider legitimacy, and assimilation of responsibilities in partnerships that offer a unified and integrated catchment programme.

INTRODUCTION

Catchments (watersheds or river basins) are natural units for managing the volume of water abstracted and for protection of water quality because they comprise a convenient naturally defined topographic area from which water drains to an identifiable point. They are therefore an obvious base for integrated analysis and management of the quantity and quality of water, its economic use, and ecosystem functions. However, this principle should not be applied without appreciation of both the typically poor match between physically defined catchment boundaries and existing political and administrative boundaries, and the value of analyses of some processes and activities at a smaller sub-catchment scale. That use and protection of water resources must assume a catchment framework is increasingly recognised and, for example, the European Union Water Framework Directive (WFD) requires water resources to be managed by river basin rather than existing political administrative boundaries.

A parallel and similarly international development is the emergence of new models of governance in water management. Subject to increasing scarcity of the resource, water governance can involve contradictory, complex and interrelated problems that are characterised by high levels of uncertainty and a diversity of competing values and claims. These problems can defy definitive formulation of the problematic itself, or of rules for determination of a final optimal solution. They require a ‘process’ or adaptive management approach that seeks iterative improvements over time rather than a final resolution. The problems are typically intractable for a single agency working alone, but the establishment of effective partnerships and collective action must overcome the tendency of stakeholders to pursue narrow self-interests (whether land users or ‘single issue’ interest groups). The role for central government is to set the essential legislative and regulatory framework and to foster more democratic, participatory and adaptive approaches to problem solving and implementation; achieving this through local instruments and participation of stakeholders supported by sound scientific understanding and an enabling policy environment

The extent to which such developments are being codified in legislation is illustrated by the statutory incorporation of public participation into the production of river basin management plans by the WFD. Internationally ‘stakeholder participation’ has become central to debates about resource use conflicts, policy choice and implementation, the accountability of public and private agencies, and the internalization of external social and environmental costs by all actors. However, relatively little is known about how best to initiate and sustain such processes for catchment management for the control of diffuse rural pollution.

METHODS

To address these issues a scoping study was carried out which made an initial assessment of catchment management for water quality improvement. Specific research methods included literature review, individual and group interviews during study tours and site visits and comparison of catchment characteristics and management programmes. Regular meetings of the research team and two workshops that facilitated engagement with a wider network of stakeholders were held. The work focused on programmes implemented through change in land and water management practices on farms, and on the participatory and other governance arrangements necessary to implement these successfully.

RESULTS: BRIEF PROFILES OF CATCHMENTS STUDIED

New York City’s Watershed Protection Plan and the Cannonsville Catchment, USA

This programme protects the water supply for New York City (NYC) that mainly originates from three catchments in upstate NY: the Catskill, Delaware and Croton systems (approximately 5,100 km^2). The surface waters from the Catskill and Delaware systems in particular carry little sediment and are not filtered before delivery to about nine million consumers. Recognising the growing pollution pressures arising in the catchments, and that it owned, managed and could control only a small proportion (<10%) of the land supplying its reservoirs, NYC through its Department of Environmental Protection (DEP) entered into a Watershed Agreement with Federal, State and local stakeholders. Its purpose is to maintain ‘filtration avoidance’ and to resolve the conflict between economic growth in the watersheds and deterioration in water quality. Changes required for water quality protection are partly voluntary, partly regulatory and fully funded by NYC. They include land acquisition (willing-seller only) or purchase of easements on land use, regulations addressing waste-water treatment, septic systems and stormwater pollution, agricultural programmes, and partnerships to improve infrastructure, management systems and fund environmentally sustainable economic development.

The Cannonsville reservoir in Delaware County, NYS, suffers from excessive phosphorus concentrations and the need to reduce phosphorus loads in the streams that feed it imposes particular constraints on farming and other activities. This and other improvements in water quality are coordinated by the Delaware County Action Plan (DCAP). The premise that the local capacities of Delaware County are the best means to meet water quality goals reflects the autonomy granted to local government under the Home Rule provisions of the Constitution of New York State, and a historically established ethos of vigorous local democracy and community involvement. DCAP operates under the authority of elected local government and engages with all agencies involved in the watershed protection programme. Its operation encompasses: planning and economic development; community services (local water supply and septic systems); stormwater and highway runoff management; agriculture; forestry; stream corridor management; and monitoring, modelling and research.

Improvement in on-farm practices and water quality protection measures is achieved through DCAP and the Watershed Agricultural Council (WAC), a committee composed of farmers, agribusiness and NYC DEP representatives, which engages local research and extension services to work with farmers. The resulting Watershed Agricultural Programme operates through implementation of whole-farm plans prepared by planning teams with individual farmers, with the twin objectives of improving water quality while maintaining farm economic viability. NY State Water Resources Institute at Cornell University, the NYS Department of Environmental Conservation (DEC), USDA-NRCS, and NYC DEP provide research and monitoring activities. Effective coordination of the efforts of all these agencies has been a feature of the programme.

The Upper Susquehanna Coalition and Catatonk Creek Watershed, USA

The Upper Susquehanna Coalition (USC) is a network of county level natural resource professionals employed by local government, which formed to develop strategies, partnerships, programmes and projects to protect the headwaters of the Susquehanna River and Chesapeak Bay Watersheds. The USC is comprised of representatives from 15 counties in New York (NY) and 3 in Pennsylvania (PA). The USC members are Soil and Water Conservation Districts in NY and Conservation Districts in PA. All members have signed a Memorandum of Understanding for development of diffuse pollution control projects on a catchment basis. Funding for the USC operations and projects is obtained from federal, state and local sources. Most funds for planning and implementation are obtained competitively; either on behalf of the entire USC or frequently for one or more of its county members for sub-catchment projects.

The USC provides liaison support between state, regional and federal agencies and local planners and implementers, and has partnered with local, regional, state, federal, academic and non-governmental organizations to conduct projects at varying scales. Through its county members the USC networks with local catchment organisations, town and county public works and planning officials, farmers and other catchment stakeholders. A key aim is not to duplicate or compete with ongoing efforts at any level, but to integrate other programmes into an overall basin strategy.

The USC uses a Multiple Barrier Approach (MBA) for planning and implementation. This addresses issues (such as flooding, stream bank erosion or degraded fish habitat) at the source (e.g. headwaters), across the landscape, and in the stream corridor, as well as programmatically (e.g. regulations, training). Development of multiple projects at a range of scales to target the critical control point of a problem helps achieve tangible results over a wide range of funding levels.

The USC initiatives are significant for their potential contribution to the Chesapeake Bay Program (CBP), a multi-state/federal partnership that has been working toward restoring the Chesapeake Bay since 1983. The Susquehanna River contributes 50% of the fresh water to the Bay. Despite great efforts continued water quality impairments within the Bay led the federal Environmental Protection Agency (EPA) and the states to list over 90 percent of the Bay tidal waters as “impaired” due to low dissolved oxygen levels and nutrient pollution. Primary nutrient sources are sewage, cattle manure, inorganic fertilizer and atmospheric nitrogen deposition. Primary sediment sources are agriculture, stream bank erosion and construction.

The EPA will require a Total Maximum Daily Load (TMDL) for the Chesapeake Bay in 2011. To avoid this regulatory TMDL, the CBP has committed to correct all nutrient and sediment problems in the Chesapeake Bay and its tidal tributaries sufficiently to remove it from the list of impaired water bodies under the Clean Water Act by 2010 (requiring a reduction of more than 40 percent in nutrient and sediment loading). This provides a unique opportunity to test a voluntary approach to a regulatory necessity. The CBP defined the water quality conditions necessary to protect aquatic living resources (criteria for dissolved oxygen, chlorophyll a, and water clarity). It then assigned load reductions for nitrogen, phosphorus, and sediment needed from each tributary basin to achieve the necessary water quality. Each state is committed to developing and implementing tributary strategies to achieve these load reductions as “living documents” that will evolve as data resources improve. To develop its tributary strategies the New York State Department of Environmental Conservation (DEC) has partnered with the USC for local input and technical support, recognising that the USC is well suited to develop, implement and track many of the diffuse pollution aspects of the strategies

The Catatonk Creek watershed (390 km^2) provides one example of local action supported by the USC. ‘Citizens for a Controlled Creek’ is an initiative to promote sound river restoration. Flood attenuation and control of stream bank erosion were the initial concerns, but recognising that an integrated approach to catchment management could generate benefits beyond flood protection, with technical support from the USC the group now addresses wetland and stream bank restoration, natural stream design and improved management of diffuse pollution.

The success of such examples depends in large part on raising local awareness of issues through education and promoting local ownership of the problems. A major inroad into communities is via school children and, for example, ‘Project WET’ (Water Education for Teachers) is a programme and resource pack aimed at schools and educators across the US.

The Groundwater Protection Programme of the Water Board of Oldenburg and East Friesland (OOWV), North West Germany

From groundwater the OOWV produces the domestic water supply for one million people living in a rural area of 7800 km^2. Its southern water catchment areas are located in a region of intensive agriculture where farmers compensate for low fertility sandy soils with high animal density and intensive use of organic and in-organic manures. A 'water-surplus' of 300 mm per year also results in high leaching rates. This combination of factors led to high nitrate concentrations in the production wells in the 1980s. In response the OOWV, together with local government (the District Administration of Weser-Ems), developed a comprehensive programme of groundwater protection based on cooperation between local partners.

The main elements of this programme are regulation, cooperation with farmers, promotion of organic farming and purchase of land and its afforestation, supported by scientific research, and public outreach activities. As part of the necessary regulatory framework the district administration designated the catchments as water protection areas within which legal restrictions could be placed on land use. Farmers receive compensation payments where these restrictions impose constraints on farm productivity and incomes. OOWV works with the local Chamber of Agriculture to provide extension advice to farmers and to encourage them to enter into voluntary agreements under which they receive payments for further specific on-farm measures such as reduced row spacing for maize and using less fertiliser. Particular efforts are made to promote adoption of organic farming with financial support for conversion and technical support for both production and marketing. "Groundwater protection forests" are an established practice in Germany and OOWV has purchased land in the water protection areas and handed it over to the State Forest of Lower Saxony for afforestation, mainly with deciduous trees. However, the scale of this is limited by adverse local reaction to large scale land use change.

Groundwater protection is financed by higher water rates paid by the water consumer. Currently the private water user pays 5 cents per cubic meter (industrial and agricultural use is charged at a lower rate), and 40 percent of this amount is used for groundwater protection and sustainable land use. Financial contributions to the programme are also gained from other sources such as environmental foundations.

The Drastrup Project, Aalborg, Denmark

Sixty percent of the residents of this city in northern Denmark are supplied with water by the Aalborg Municipality, through a public utility company run on a commercial basis in competition with private water suppliers. The water source is groundwater, abstracted from three areas to the south of the city and supplied untreated. In the 1980s it was identified that the quality of this resource was threatened by nitrate pollution (plus pesticides and other chemicals) and the Drastrup groundwater protection project was launched in 1992.

Land use change and the removal of polluting activities from designated groundwater protection areas are achieved through a unified approach to spatial planning. Land is converted from intensive agriculture to extensive farming and recreational areas through voluntary and compensated agreements for change in farming practices, voluntary land sales, or ‘land swaps’ for areas outside the protected zones. Gravel pits and waste disposal sites are also closed and restored through voluntary purchase agreements. Legislation allows for compulsory land purchase but this is regarded as a last resort. Public information campaigns encourage residents within the protection areas to make their homes and gardens pesticide and chemical free. To date at least 210 hectares have been converted to permanent grass or forest, and monitoring of wells has shown improvements in nitrate and pesticide concentrations. Finance for the project is provided by the Aalborg city council.

The Tamar Catchment in S.W. England and the Westcountry Rivers Trust

The Tamar catchment has an area of 930 km^2 and provides the water supply for the city of Plymouth. In the South West of England, as in other areas of the UK, diffuse pollution of water from agricultural sources has become a major concern. The Tamar has suffered from eutrophication and toxic algal blooms in dry years, while soil erosion, sedimentation and localised flooding are also significant.

A programme to manage diffuse pollution from agriculture through voluntary action by farmers is being implemented by the Westcountry Rivers Trust (WRT), an independent charitable trust. The aims are to protect fisheries, the riverine environment and public health, whilst sustaining or improving profitability for farmers and the economic viability of rural communities. WRT’s methodology involves systematic and integrated identification of environmental impacts at a catchment scale through the ‘ecosystem approach’, followed by practical and iterative engagement with local stakeholders to address these impacts at source. Once priority sub-catchments or target areas are identified, remediation involves awareness raising for river managers and farmers, and development of practical solutions at an individual site or farm scale. Advice to farmers is based on whole farm planning and is delivered pro-actively. Site specific management plans are developed, integrating advice on best farm management practices with an appraisal of options to improve land use, reduce costs, improve returns and meet conservation priorities. Wider education campaigns are also undertaken to raise public awareness. Funding is generated from charitable donations and from success in winning grant finance from the UK government, EU and foundations.

A cornerstone of WRT’s philosophy is working in partnership with individuals and organisations. Collaboration takes place with stakeholders ranging from individual businesses through to academic institutions, non-governmental organisations and government departments. In doing so WRT aims to circumvent sectoral interests and develop joint solutions to complex environmental problems. From its formation key partners have been the Environment Agency (providing influence and technical support), the Wetlands Ecosystem Research Group based at the Institute of Grassland and Environmental Research, North Wyke, Devon (joint development of WRT’s modus operandi and applied ecosystem approach) and agricultural consultants (agricultural expertise and joint development of win-win solutions offering environmental gain and economic benefits for farmers).

In existence for a decade WRT now delivers around £1 million worth of work every year. Achievements include: over 1800 farmers and landowners in the South West given advice and 1400 integrated land and river management plans developed;

over 200 km of vulnerable riverbank fenced; 16 wetlands restored/improved; over 74 km of ditches prioritised for re-vegetation; over 400 sites of accelerated erosion controlled; 450 demonstration sites developed and operational; over 180 sites of habitat improvement; over 50 buffer zones created; and 100 guidance sheets on best management practices have been developed and widely distributed.

The Broads in East Anglia and the Upper Thurne Working Group

In the drier east of England and subject to continual development pressures from farming and recreation, the Broads and its rivers face complex threats to water quality and the aquatic environment. As a result the area provides a model of multi-agency involvement and multiple stakeholder interests and competing priorities. The Norfolk and Suffolk Broads Act of 1988 established the Broads Authority and requires it to manage the Broads (an area of shallow lakes and waterways originating from medieval peat diggings). Guiding principles are: sustainability (social progress that recognises the needs of everyone); effective protection of the environment; wise use of natural and cultural resources; maintenance of economically and socially thriving communities; and working in partnership. The aim is for ‘living landscapes’: a long-term vision for the Broads as a more naturally functioning and biologically diverse wetland in harmony with its economic and social uses.

The Upper Thurne Working Group is a forum of stakeholders presently addressing the issues arising from tourism, recreation, urban discharge and agriculture for its catchment. It is multi-agency, incorporates public, private and voluntary sector stakeholder interests, and is tasked with producing a water management plan for the Upper Thurne. The group is independently chaired and includes English Nature, Environment Agency, Defra, Parish Councils, Hickling Broad Sailing Club, Drainage Broads, National Trust, Norfolk Wildlife Trust, and RSPB.

DISCUSSION: A SYNTHESIS OF ISSUES AND LESSONS

This study has observed catchment protection programmes that address diffuse pollution at source, and has drawn on the knowledge and experience of a diverse group of stakeholders. From this the following lessons appear to have wide relevance and applicability.

1. *Each catchment and all sources of pollution must be analysed in an integrated and holistic way, and environmental criteria must be integrated with the economic and social goals of those affected by change*. Given the interconnections between land, and surface and groundwater, to protect or enhance one area of a catchment whilst ignoring adjacent and particularly upstream areas is not a viable solution. Similarly a diversity of interests and priorities among all stakeholders must be acknowledged. Targeting only farming whilst ignoring other interests may limit credibility and success, and shared goals and cross-sectoral approaches are essential. This lesson is demonstrated in each of the cases considered above.

2. *A range of technologies and management options to reduce diffuse water pollution exist*. Despite some gaps knowledge is sufficient to support the design and implementation of diffuse pollution mitigation measures under a wide range of conditions. Two core principles have wide applicability: i) selection of measures requires site specific ‘whole-farm’ plans developed by locally trusted farm extension

agents who provide technical assistance; and ii) adoption of a multiple barrier approach to minimise use of pollutants, modify farm infrastructure and the 'landscape' between the source and stream to block pollution pathways (or minimise risk), and use of the stream margin as a final barrier. This lesson is particularly demonstrated by the programmes in the USA, in Germany and the work of the WRT.

3. *A target based approach and adequate research base are required*. For example, the approach to pollution control in the USA is comprehensive and target driven. Watersheds must meet pollutant loading targets from point and diffuse sources that do not exceed specified threshold levels set as Total Maximum Daily Loads. The thresholds are determined, with public participation, to ensure that specified uses of the water bodies are sustained. To meet them it is necessary to estimate the loadings of pollutants from all sources and quantify management options, whilst taking account of their economic, social and environmental consequences. This requires a knowledge base and monitoring system able to support ongoing decision making. This lesson is particularly demonstrated by the cases from the USA. In the EU the question of how the goal of 'good ecological status' under the Water Framework Directive will be translated into practical programmes of measures remains to be resolved.

4. *Land management and diffuse sources of pollution have a local basis and protection of water at source necessitates the application of sound scientific understanding plus the fostering of local instruments and participation of stakeholders, supported by an enabling policy and regulatory environment*. Catchment-scale projects must involve multiple agencies from different levels of government and from civil society, and partnerships and structures that result in effective cooperation, coordination and leadership at local level are essential. Participation works when it builds shared knowledge and the capacity for trust and collective action, but must be supported by a sound scientific base, an enabling regulatory and policy environment, adequate financing, and the necessary degrees of autonomy, accountability and legitimacy. This lesson is demonstrated to varying degrees by each of the cases above. More research is needed to determine the optimal governance structures and arrangements for country and location specific conditions and needs.

ACKNOWLEDGEMENTS

This study was financed by the Rural Economy and Land Use Programme, a collaboration between the Economic and Social Research Council, the Biotechnology and Biological Sciences Research Council and the Natural Environment Research Council, with additional funding from the Scottish Government and the Department for Environment, Food and Rural Affairs. The author acknowledges the contributions made to the study by Kevin Hiscock, Keith Porter, Hadrian Cook, Alex Inman, Jon Hillman, Patricia Bishop and Mary Jane Porter, but bears sole responsibility for any errors or omissions.

THE ROLE OF LAND MANAGERS IN NATURAL FLOOD MANAGEMENT

M Donaghy[1] and A Johnstonova[2]

[1]WWF Scotland, Little Dunkeld, Dunkeld, Perthshire, PH8 0AD, UK, E-mail: mdonaghy@wwfscotland.org.uk; [2]RSPB Scotland, Dunedin House, 25 Ravelston Terrace, Edinburgh, EH4 3TP, UK, E-mail: Andrea.Johnstonova@rspb.org.uk

SUMMARY

In Scotland, many people, and at least 100,000 properties, are at risk from flooding. Flooding is also predicted to increase in response to the effects of climate change. Scottish Ministers have introduced recently a flooding Bill into Parliament. This new legislation is intended to transpose the EC Floods Directive and deliver sustainable flood management (SFM). A major component of SFM places more emphasis on the roles of natural processes and land use practices. Flood plans describing flood processes will be produced for all areas in Scotland. A wide range of stakeholders will be involved in developing and implementing the plan. The plan will identify where, when and how within a catchment action could be taken to deal with the causes of flooding. Farmers, foresters and land managers, in conjunction with the other flood managers, will have a key role in producing the plans and delivering the natural flood management component of SFM measures on the ground. It is to be expected that national schemes will be developed by the Scottish Government to ensure that land-users will be compensated and rewarded at least for any negative effect that involvement has on the farm, estate or forest business.

INTRODUCTION

Scotland is renowned for, and in many ways defined by, its water. Its geographic location ensures a plentiful supply that shapes its characteristic landscapes and provides a superb resource, vital to the Scottish people, environment and economy. With increasing volumes of water predicted and arriving in more violent storms, we will all have to prepare to live with floods, rely less on outmoded techniques and do everything to work with nature to protect people and property. This will require a change in the way that we presently deal with flood management. It will require a suite of measures that reflects the complexity of catchment flood management. In addition, it will require a widening of the stakeholders involved in flood management to include those who live and work in the areas where floods arise.

EFFECTS OF CLIMATE CHANGE ON WEATHER

A recent report published by SNIFFER reviews climate trends in Scotland from 1961 to 2004 (SNIFFER, 2006). It indicates that we are already experiencing changes in our climate. For example, there has been a significant increase in average winter precipitation, with North Scotland experiencing an increase in winter rain of almost 70%, East Scotland 37%, and West Scotland 61%. Increasing trends were also noted in heavy rainfall, particularly in North and West Scotland, and an increase in average rainfall intensity in both East and West of Scotland. The UK Climate Impacts Programme (CIP) briefing (Tindall Centre for Climate Change Research,

2002) concludes that winters will become wetter, with increases in rainfall intensity and frequency, while summers may become drier. Therefore floods, which are currently considered 'extreme', will become more common in future. The report states that by 2080, winter precipitation in the west of Scotland could increase by 20%, and in parts of the east of Scotland the increase could be as much as 30%. Simultaneously, summer precipitation is projected to reduce by 30% - increasing the risk of flash flooding as water runs off dry ground more quickly. A medium-emission climate change scenario predicts that a 1 in 100 chance flood in any year is expected to become a 1 in 70 chance flood in any year by the 2020s, and to a 1 in 40-60 chance flood in any year by the 2080s (WWF, 2002). Rising sea levels are also one of the impacts of climate change and will lead to increased breaches of sea defences, loss of important estuarine and coastal habitats and damage to property. The UK CIP predictions for sea level rise suggest 0-10 cm for low emission scenarios, and 50-70 cm for high emission scenarios by 2080 (UKCIP, 2006).

TRADITIONAL APPROACHES TO FLOODING

Traditionally, the approach to flooding has been very reactive. Flood management tended to result in local authorities and government funding the construction of floodwalls or embankments to move water downstream to prevent inundation. Land-users and engineers have always tried to get rid of water fast, draining it off the land and into the sea in re-engineered rivers. However, no matter how much deeper and straighter rivers get, the threat of floods just keep getting worse. Hard engineering solutions are perceived as tried and tested, easier and quicker to construct than undertaking large scale catchment management. However, building ever higher walls and embankments is known to exacerbate the problem – by moving large quantities of water downstream and causing flooding elsewhere. It is all about dealing with the effects of flooding and not the causes. The current approach to flooding does not serve the public well. It is extremely expensive, isolates communities from their watercourses and become increasingly unreliable in the face of climate changes effects. Changing from a narrow, urban-only, concrete approach to a broader, inclusive, catchment-wide approach reflecting the nature and variety of the flooding processes puts us all in a better position for living with floods in the future.

WHAT IS SUSTAINABLE FLOOD MANAGEMENT?

Sustainable flood management is a process. It describes flood risk through a 'whole river' or catchment approach. It involves a wide range of stakeholders and defines their roles in flood management. Importantly, it provides many additional benefits beyond flood management. In the context of climate change, it offers huge advantages over the traditional methods of flood management.

Sustainable flood management embodies a shift from our predominantly piecemeal and reactive approach to flood management towards a catchment-based approach that takes account of long-term social and economic factors and, together with a wide suite of measures, restores natural processes and natural systems to slow down and store water run-off. A typical sustainable flood management approach would include some or all of the following measures to lower flood risk in a catchment.

- **Planning** - avoiding development in flood prone areas
- **Flood Mapping** - identifying areas at risk and areas that are safe
- **Flood Resilience** - building or modifying properties to recover quickly from flood events
- **Education, advice and awareness raising** - raising the awareness, and improving the understanding of flooding issues in communities and advising on measures that can be taken to prevent or limit damage
- **Reservoir Management** - linking high quality weather information with reservoir storage
- **Building Removal** - removing properties which, for economic or practical reasons, cannot be protected
- **Flood Warning Schemes** - allowing quicker and better preparedness for flood events
- **Insurance Effects** - designating areas with lower or higher insurance premiums based on risk
- **Engineering** - Hard: constructing walls, embankments and gates; Soft: Sustainable Urban Drainage Schemes (SUDS)
- **Natural Flood Management** - Involving land-use practices and restoring natural processes

WHAT IS NATURAL FLOOD MANAGEMENT?

Natural flood management is an integral part of sustainable flood management. It is largely achieved by slowing the flow of water to rivers using natural water and land processes to lower flood risk to people and property further downstream. Within the sustainable flood management approach, it defines the role that farmers, foresters and estate owners have in flood management, within their catchments.

Much of it is achieved through land management. Techniques include restoring upland wetlands and reforesting gullies; replanting native riparian woodland, restoring lowland wetlands and bogs, and re-connecting rivers with floodplains and meanders.

It is a cost-effective means of achieving many objectives, including our biodiversity targets and obligations, the aims and objectives of the Water Framework Directive, improving recreational and well-being opportunities, buffering the effects of climate change, recharging groundwater systems and improving water quality. Such approaches have been shown to deliver social, economic and environmental benefits. Significantly, it offers a rare opportunity for urban communities to appreciate the effects of the role and function of land-use in rural areas upstream.

The effectiveness of these natural techniques has been extensively tested in a WWF Scotland demonstration project on the River Devon in Clackmannanshire (WWF, 2007a) and elsewhere in the UK and Europe. The River Devon project demonstrates that although the effects of river flooding are felt downstream, the causes of flooding actually begin upstream among fields, forests and gullies. Findings of the demonstration project and work done by RSPB in Insh marshes and elsewhere (RSPB, 2007) indicate that by restoring the functionality of rivers and uplands, it is possible to reduce the risk of flooding downstream in the long-term for a fraction of the costs of expensive, short-lived, hard-engineering (WWF, 2007b).

THE ROLE OF LAND MANAGERS IN MANAGING FLOOD RISK

The way land is managed can have significant effects on the run-off and storage capacity within a catchment. Integration of flood management into land use management and agricultural policies is essential for achieving the objectives of SFM and river basin management. Farmers have a key role to play in implementing natural solutions to flooding. However, this will require recognition of the value of natural flood management, and a full integration with the land-use management framework. This includes offering well funded land management schemes, usefully linked to the Restoration and Remediation process of the Water Framework Directive, redirecting support payments towards alternative from solely hard-engineering, and promotion of natural flood management through existing programmes and initiatives, such as the Scottish Rural Development Programme (SRDP). Tying support payments to innovative land management practices, such as the natural solutions to flooding would ensure wider public and societal benefits.

Flood defence and the drainage of farmlands have been actively encouraged by the EU Common Agricultural Policy (CAP) since the late 1940s, with the aim of increasing and securing food production. Applications for major drainage schemes can still be made today under the Land Drainage Act (Scotland) 1958 and 1930.

However, agricultural policy is changing and the emphasis is increasingly on diversification, the delivery of public benefits and environmental enhancement. Where flood banks are protecting marginally viable or even higher quality land, decisions need to be made on whether current farming practices are genuinely providing the widest benefits from that land or whether the public interest would be better served by a change in land management. The CAP has the potential to benefit sustainable flood management through support of natural flood management techniques; but measures may be limited by the funding and prioritisation process. Furthermore, Pillar 2 funding only offers short-term management agreement, which is not the most appropriate funding mechanism for long-term management of land for flooding.

The views of farmers and other land managers are obviously critical to implementing sustainable flood management. To encourage a positive approach, there is a pressing need for appropriate and targeted incentives to encourage restoration to more sympathetic, less intensive, agricultural management on land which can be used to lower flood risk to communities. There is an urgent requirement for an appropriate funding mechanism, combining compensation and reward. Redirecting flood scheme budgets from a wholly engineered approach to supporting the sustainable flood management approach is a major part of the solution.

ENSURING AN INTEGRATED APPROACH

The duty to adopt an “integrated approach” across land use policy is a critical opportunity. It is reflected in the aspirations of numerous recent government documents and initiatives, but key changes to policy and funding are required if sustainable flood management is to become a reality. The structure offered by the Scottish Government for River Basin Management under the WFD offers a ready made and appropriate framework for achieving those aims. A parallel or linked structure, based on the eight sub-basins plus two, would allow for participation, action at a catchment scale, collaborative applications, local disbursement of funds and an integrated approach.

NEW LEGISLATION – NEW OPPORTUNITY

At a recent Scottish Government Flood Summit in Perth, the Cabinet Secretary for the Environment and Rural Affairs, Richard Lochhead announced the Government's intentions to introduce in the Scottish Parliament a Bill reforming the current approach to flooding. This is a major opportunity to introduce a more sustainable approach to the management of flood risk, which works with, rather than against, the natural environment. The new Flooding Bill should aim to promote a concept of catchment management integrated with sustainable flood management, and require a long-term planning approach to reducing flood risks. Clear links need to be made with river basin management planning, and more emphasis needs to be placed on non-structural measures such as using natural floodplains to store water during floods. Farmers and land managers will have a more pro-active role to play in managing land for flooding, bringing important benefits to the communities and the Scottish economy.

CONCLUSION

Traditional methods of flood management are ineffective in the present and future climates. We need to develop and integrate our approach to river basin, flood and land management. Sustainable flood management offers an efficient way of dealing with floods and managing land in a way that benefits people and sustains their economies and their environment.

REFERENCES

RSPB (2007). Time for a Change.
http://www.rspb.org.uk/Images/sfmreport_tcm9-158603.pdf

SNIFFER (2006). A handbook of climate trends across Scotland.

Tyndall Centre for Climate Change Research (2002). Climate Change Scenarios for the UK.

UK Climate Impacts Programme (2006). Updates to regional net sea level change estimates for Great Britain, August 2006.
http://www.ukcip.org.uk

WWF (2007a). Slowing the Flow.
http://www.wwf.org.uk/filelibrary/pdf/slowingflow_web.pdf

WWF (2007b). Flood Planner.
http://www.wwf.org.uk/filelibrary/pdf/floodplanner_web.pdf

WWF (2002). Turning the tide on flooding.
http://www.wwf.org.uk/filelibrary/pdf/turningtide.pdf

BENEFITS AND 'POLLUTION SWAPPING': CROSS-CUTTING ISSUES FOR DIFFUSE POLLUTION MITIGATION

DR Chadwick[1], BJ Chambers[2], R Crabtree[3], S Anthony[4] and D Harris[4]

[1]IGER North Wyke, Okehampton, Devon, EX20 2SB, UK, E-mail: david.chadwick@bbsrc.ac.uk; [2]ADAS Gleadthorpe, Meden Vale, Mansfield, Notts, NG20 9P, UK; [3]CJC Consulting, 34 Southmoor Road, Oxford, OX2 6RF, UK; [4]ADAS Wolverhampton, Wergs Road, Wolverhampton, WV6 8TQ, UK

SUMMARY

A 'model-farm' approach was taken to estimate the impact of potential combinations of Diffuse Water Pollution from Agriculture (DWPA) mitigation methods on emissions of ammonia, nitrous oxide (direct and indirect), methane, energy-derived carbon dioxide (CO_2) and total CO_2 equivalents. The model farms were; arable, arable+manure, poultry, indoor pigs, dairy and beef enterprises comprising typical numbers of stock, land and inorganic fertiliser inputs. Gaseous emissions were expressed on a per hectare basis to enable simple scaling to the national (England) level. In scaling, we took account of trends in land use and livestock numbers, and potential combinations of DWPA mitigation methods that could be implemented simultaneously as a result of introducing additional advice, schemes, grants or a Water Protection Zone policy. Changes in N fertiliser use and livestock numbers, under a Business as Usual scenario prior to the implementation of supportive approaches or policy instruments in 2015 were predicted to result in significant reductions in ammonia and greenhouse gas emissions (in the range of 14-24%) compared with the year 2000 baseline. Additional supportive approaches or policy instruments implemented after 2015 were predicted to have little impact on national ammonia and greenhouse gas emissions.

INTRODUCTION

Any proposed mitigation method to reduce Diffuse Water Pollution from Agriculture (DWPA), i.e. nitrogen, phosphorus and faecal indicator organism (FIO) transfers to watercourses, has the potential to affect ammonia (NH_3), nitrous oxide (N_2O) and methane (CH_4) emissions. In order to make a more complete assessment of the impact of combinations of potential DWPA mitigation methods on the environment as a whole (i.e. to investigate secondary benefits/disbenefits), it is necessary to understand the impact they might have on gaseous pollutants. This study was designed to provide such a more holistic view.

METHODS

In order to quantify the effects of single and combined DWPA mitigation methods on gaseous emissions, it was necessary to: a) identify the direction of any effect (i.e. positive, negative or no-effect) and b) estimate the magnitude of the effect (i.e. percentage change from baseline). The initial unit of accounting was based on a 'model farm' approach, and it was therefore essential to determine the baseline emissions of the three gases from the model farms. The gaseous emissions data were expressed on a per hectare basis to enable simple scaling to the national (England) level (including rainfall and soil type influences) through multiplying by the land area attributable to each model farm type. In scaling the gaseous emissions, we took account of trends in land use and livestock numbers, and potential combinations of DWPA mitigation methods that could be implemented simultaneously as a result of introducing additional advice, schemes, grants or a Water Protection Zone (WPZ) policy.

Model Farm Descriptions

Descriptions of the typical model farms (arable, arable+manure, poultry, indoor pigs, dairy and beef enterprises) were required with enough information to provide sufficient detail to allow the estimation of gaseous emissions from the various sources: animal buildings, collecting yards, manure stores, inorganic fertiliser and organic manure applications, and grazing livestock (i.e. excretal returns and ruminant sources of methane).

Sources of Ammonia, Nitrous Oxide, Methane and Energy-CO_2 Emissions Data

The UK Ammonia Emissions Inventory (Misselbrook *et al*., 2006) was used as the source of emission factors (EFs) for NH_3, as the inventory is regularly updated with the latest published research and farm activity data and EFs for N_2O and CH_4 were taken from a range of sources, including IPCC default values and published data (e.g. Yamulki *et al*., 1998; Chadwick *et al*., 2000; Chadwick, 2005; Thorman *et al*., 2006). Each model farm type was also assessed in terms of its baseline energy use (Metcalfe and Cormack, 2000).

A spreadsheet was constructed for each of the model farms which quantitatively described the gaseous EFs from all sources. EFs were then multiplied by other appropriate factors such as, numbers of animals, time spent in housing versus time spent grazing, quantity of manure stored, inorganic fertiliser application rates etc. In this way, the total emission of each gas was calculated for each model farm and expressed on a per hectare basis to enable 'simple' scaling up to a national level.

As part of the scaling exercise, it was necessary to take into account the potential impact of different soil types and climatic conditions on emissions of NH_3, N_2O and CH_4. For simplicity, we assumed that soil and climatic conditions had negligible effects on CH_4 and NH_3 emissions, although we did include a higher EF for NH_3 emissions following slurry application in summer. However, as N_2O emissions following inorganic fertiliser N and manure spreading are driven by microbial processes directly controlled by soil moisture and temperature, we ran our model farms through the mechanistic UK-DNDC model (Brown *et al*., 2002) to produce soil and climatic scaling effects.

Estimation of Total CO_2 Equivalent Production

The CO_2 arising from energy use was added to the CO_2 equivalents arising from CH_4 emissions (21 times that of CO_2) and direct and indirect (associated with N emission/deposition and nitrate leaching) N_2O emissions (310 times that of CO_2).

Effect of WFD Mitigation Methods on Gaseous Emissions at the Farm Scale

The 44 on-farm mitigation methods compiled in the DWPA 'User Manual' (Cuttle *et al*., 2006) were collated into model farm datasheets and the secondary impacts of the 44 methods on NH_3, N_2O and CH_4 emissions, energy-CO_2 use (including diesel and fertiliser energy – but does not include electricity in buildings) and total CO_2 equivalents. The spreadsheets used to scale emissions to the farm scale were able to reflect the effects of individual mitigation methods by adjusting, for example, the number of stock, inorganic fertiliser N application rates, quantity of manure stored, the grazing/housing period etc. This exercise was repeated for all the model farms on two soil textures (clay loam and sandy loam) and three rainfall zones (high, medium and low).

Effects of Combinations of DWPA Mitigation Methods at the National Scale

The model farms were constructed to allow specific combinations of mitigation methods to be implemented for selected farm typologies. In order to scale from the model farms to the national scale, we calculated the percentage difference between the effects of the mitigation methods and the baselines for each farm sector. This percentage was then used to scale the sectoral baseline totals calculated by the UKAEI (NH_3) and IPCC (N_2O and CH_4) methods. These 'adjusted' sectoral values were then summed to generate the England totals.

We then tested a combination of mitigation methods as a result of introducing ***advice***, e.g. implementing the England Catchment Sensitive Farming Delivery Initiative (ECSFDI) across the whole of England; ***schemes***, e.g. extended Entry Level Stewardship; ***grants***, e.g. contributing to the implementation of high cost mitigation methods, such as increased slurry storage; or a Water Protection Zone (WPZ - similar to that defined in article 93 of The Water Resources Act, 1991) policy instrument. By multiplying the additive effects of the combinations of mitigation methods at the model farm scale by the corresponding land areas, the impact of the mitigation measure packages was scaled to the national level.

The methods that were invoked by the WPZ policy instrument were selected to be sufficient to achieve a *ca*. 50% reduction in P losses (although mitigation methods would also impact on other water pollutants) from each of the agricultural sectors and included other measures e.g. cultivation of compacted tillage soil, cultivating and drilling across slopes, leaving autumn seedbeds rough, establishing in-field grass buffer strips, reducing stocking rates, reducing the length of the grazing season, not applying fertiliser and manures to high risk areas (e.g. steep slopes, within 10 m of a watercourse) and at high risk times, re-siting gateways and solid manure heaps, establishing riparian buffer strips and constructed wetlands. Not only did we need to take account of the efficacy for each individual method for all the soil/climate combinations, but we also needed to assume a percentage of implementation by farmers according to soil/climate conditions.

In order not to over-estimate the impacts of additional advice, schemes, grants or policy instruments that would be implemented after 2015, it was necessary to determine what degree of reduction in diffuse water pollution would be likely as a result of existing/emerging policies (e.g. Cross-Compliance and the Nitrate Vulnerable Zone Action Programme 2002) and advice (e.g. via ECSFDI), between the years 2000 and 2015. Therefore, the 'modelling tool' was run for the chosen 'BASE' year (2015), using estimates of potential uptake and efficacy of a specific combination of methods and using information on potential changes in land use and animal numbers using the projections in the Business As Usual Phase 2 report (BAUII, 2006). In the BAUII report, the important changes to animal numbers by the year 2015 were predicted to be: beef cattle (-15 to -20%), dairy cattle (-25 to -30%), pigs (-10%) and poultry (+8.5%).

RESULTS

Model Farm Descriptions

Total N loading for each of the model farm types are summarised in Table 1.

Table 1: Total N loading to land on each of the model farms (includes fertiliser N, manure N and excretal returns)

Farm type	Arable	Arable+ manure	Broiler	Indoor pigs	Beef	Dairy
Total N (kg/ha)	165	204	316	315	173*	345**

*33 kg/ha from handled manure and 60 kg/ha from excreta deposited directly in the field
**90 kg/ha from handled manure and 65 kg/ha from excreta deposited directly in the field

Baseline Gaseous Emissions at the Farm Scale

Estimated emissions of NH_3, N_2O (direct), CH_4 and energy use (CO_2) from each of the model farm types are summarised in Table 2 (in the case of N_2O, soil and climatic scaling have not been included in this table). The model farms with livestock had the highest NH_3 emissions. Similarly, CH_4 emissions were highest from the dairy and beef farms due to rumen sources, with elevated emissions from the pig farms due to the pig slurry source and relatively small land area. Nitrous oxide emissions were lowest from the beef, arable and arable+manure farms, reflecting total N loadings rates compared with the pig, broiler and dairy enterprises (Table 1).

Table 2: Baseline NH_3-N, N_2O-N (direct) and CH_4 emissions and energy use for model farm types on a clay loam in a medium rainfall environment

Farm type		NH_3-N kg/ha	N_2O-N kg/ha	CH_4 kg/ha	Energy use t CO_2/ha
1	Arable	1.3	2.1	<1	1.3
2	Arable+manure	9.0	2.5	9	-
3	Poultry	72	4.0	3	3.4
4	Indoor pigs	55	2.8	64	3.4
6	Dairy	42	3.3	198	3.6
7	Beef	16	2.2	105	0.3

The model beef farm was estimated to use the lowest amount of energy and the model arable farm (largely due to N fertiliser manufacture and use) intermediate amounts of energy. The pig, broiler and dairy farms used the largest amounts of energy due to inorganic fertiliser N use and manure management activities.

Effect of DWPA Mitigation Methods on Gaseous Emissions from the Model Dairy Farm (Clay Loam Soil in a Medium Rainfall Climate)

Example 1. DWPA Method No. 13. Reduce the stocking rate on the farm by 50%. This method resulted in 50% less fertiliser N use and 50% less N excreted by the livestock. Hence, NH_3-N and direct N_2O-N emissions were reduced by 50% from 44 kg/ha and 3.35 kg/ha (clay loam/medium rainfall climate) to 22 kg/ha and 1.68 kg/ha, respectively. Methane emissions were also reduced by 50% from 198 kg/ha to 99 kg/ha.

Example 2. DWPA Method No. 14. Reduce the length of the grazing season by 5 weeks. This resulted in a greater period of time (ca. 20%) that the dairy cows were housed. Emissions of NH_3 are less from cattle grazing in the field than from housed animals due to more rapid infiltration of urine into the soil, where the ammonium-N becomes bound to soil, compared with during housing, when the urine remains on the surface of the concrete until it is physically removed by scraping or hosing down. Thus, NH_3-N emissions were increased by 8.3 kg/ha (across the whole farm area). No further NH_3 emissions were estimated from the greater quantity of slurry generated and stored, as it was assumed that the additional slurry was stored in the same storage tank and the surface area was therefore the same (NH_3 emissions from slurry storage are related to the exposed surface area and not the mass of slurry). Direct N_2O emissions were estimated to be reduced by 0.2 kg/ha (across the whole farm area) as a consequence of fewer urine patches being generated in the field (which are significant sources of N_2O), with the extra slurry collected during the additional five weeks housed period estimated to result in lower N_2O emissions than the urine patches. For CH_4, there was a small increase in emissions as a result of the additional slurry storage, which generated a further ca. 10 kg CH_4/ha/yr (across the farm).

Example 3. DWPA Method No. 25. Increase slurry storage capacity. This increased NH3-N emissions by 4.3 kg/ha as a result of an increase in slurry spreading in warm dry soil conditions in summer when slurry infiltration rates can be slow as a result of hydrophobic soil conditions. There was assumed to be no impact of slurry application timing on N_2O or CH_4 emissions.

Example 4. DWPA Method No. 30. Change from a slurry to a solid manure based system. This was estimated to result in a significant reduction in NH_3 emissions of 17 kg/ha N across the whole of the dairy farm area, due to substantially lower NH_3 emission during the housed period (when NH_3 emissions are known to be around one-third of those where cattle are housed on a slurry system) and associated lower NH3 emissions from spread FYM compared with slurry. Direct N_2O emissions were estimated to increase due to additional N_2O generated within the animal house and particularly during solid manure storage (by 1.15 kg/ha across the farm).

The model farm approach enables the effects of introducing an individual mitigation method to reduce the transfers of N, P and FIOs on emissions of environmentally damaging gases to be assessed and quantified, i.e. to identify ‘win-win’ and ‘pollution swapping’ situations.

Effect of DWPA Mitigation Methods on Gaseous Emissions at the National Scale

The impacts of Business as Usual predictions and advice, schemes, grants and the WPZ policy instrument to reduce P loss on NH_3, N_2O (direct and indirect), CH_4, energy (CO_2) and total CO_2 equivalents were investigated.

Business As Usual predictions for England, which took into account effects of existing/emerging policies (e.g. NVZs 2002 and Cross Compliance) and predicted changes to livestock numbers and land use, were estimated to result in a 14% reduction in ammonia emissions (from 145 kt NH_3-N in 2000 to 124 kt NH_3-N in 2015), a 23% reduction in methane emissions (533 kt to 411 kt), a 16% reduction in total nitrous oxide emissions (including direct and indirect emissions) (39.6 kt N_2O-N to 33.4 kt N_2O-N), a 23% reduction in energy-CO_2 (12,400 kt down to 9,600 kt CO_2) and a resultant 24% reduction in total CO_2 equivalents (from 35,100 kt to 26,800 kt CO_2e). The key pollutant targeted by the different advice, schemes, grants and WPZ policy instrument was P, and utilising our modelling framework we were able to demonstrate that the combinations of methods selected had little effect on gaseous N losses as ammonia and nitrous oxide, or on methane emissions beyond the 2015 Business As Usual scenario.

ACKNOWELDGEMENTS

We acknowledge the support of Defra in funding this work.

REFERENCES

BAUII (2006). Business as Usual Projections of Agricultural Activities for the Water Framework Directive: Phase 2. Final Report to Defra, pp.129.

British Survey of Fertiliser Practice (2003). Fertiliser Use on Farm Crops for Crop Year 2003. Defra/SEERAD. ISBN 1 86190 128.

Brown L, Syed B, Jarvis SC, Sneath R W, Phillips VR, Goulding KWT and Li C (2002). Development and application of a mechanistic model to estimate emission of nitrous oxide from UK agriculture. Atmospheric Environment 36, 917 928.

Chadwick DR, Pain BF and Brookman SKE (2000). Nitrous oxide and methane emissions following application of animal manures to grassland. Journal of Environmental Quality 29, 277-287.

Chadwick D (2005). Emissions of ammonia nitrous oxide and methane from cattle manure heaps: effect of compaction and covering. Atmospheric Environment 39, 787-799.

Cuttle SP, Haygarth PM, Chadwick DR, Newell-Price P, Harris D, Shepherd MA, Chambers BJ and Humphrey R (2006). An Inventory of Methods to Control Diffuse Water Pollution from Agriculture (DWPA) - User Manual.

Metcalfe JP and Cormack W (2000). The Efficiency Test (A model of energy inputs into organic and conventional farming systems). Organic Farming. Issue 67, Autumn 2000.

Misselbrook T, Chadwick DR, Chambers BJ, Smith KA, Williams J, Demmers T and Sneath R W (2006). Inventory of Ammonia Emissions from UK Agriculture 2005. Inventory Submission Report. Defra Report (AC0102), pp. 32.

Thorman R, Chadwick D, Nicholson F and Chambers BJ (2006). Nitrous oxide and di-nitrogen losses following applications of livestock manures to agricultural land. In: 12th Ramiran International Conference – Technology for Recycling of Manure and Organic Residues in a Whole-Farm Perspective (Ed. Petersen SO), Volume I, pp. 179-181, DIAS Report Plant Production No.122, August 2006.

The Water Resources Act (1991).
http://www.opsi.gov.uk/acts/acts1991/Ukpga_19910057_en_1.htm

Yamulki S, Jarvis SC and Owen P (1998). Nitrous oxide emissions from excreta applied in a simulated grazing pattern. Soil Biology and Biochemistry 332, 491-500.

PROTECTING THE NATION'S SOIL – PAST, PRESENT AND FUTURE

CD Campbell, W Towers, R Hough and HIJ Black

Soils Group, Macaulay Institute, Craigiebuckler, Aberdeen, AB15 8QH, UK, E-mail: c.campbell@macaulay.ac.uk

SUMMARY

Scotland's soil is one of its most important natural assets not only underpinning our agriculture and forest industries but also increasingly recognised for a wider range of environmental, economic and societal benefits. Our diverse geology and climate has given rise to a variety of soil types each with a different balance of functions and responses. It is often assumed soils change slowly over time being resistant to environmental change but we also now know that changes in vegetation through management or succession can change soils dramatically in just a few decades. The government is developing an evidence-based soil policy and research has been commissioned to look for evidence of change and examine how we might monitor soils effectively for the future. We are now re-sampling the National Soils Inventory of Scotland and by research at the laboratory, field and landscape scale investigating new ways of measuring, monitoring and understanding soil.

INTRODUCTION

Soils are essentially a non-renewable resource. They are one of Scotland's most important natural assets, most widely recognised for their role in supporting plant growth. However, soils also provide a vital link between the atmosphere, biosphere and underlying rocks and are increasingly recognised for a range of environmental, economic and societal benefits. Soils provide the basis of the agricultural and forestry industries, producing economic outputs through production of crops, livestock and timber. Agriculture alone sustains ~67000 jobs in rural Scotland and has a gross output of ~£2000 million (http://www.scotland.gov.uk/Publications/2007/05/15131914/4). Scottish agriculture is rightly proud of its quality brand and healthy soils are vital to protect the food chain from contamination as well as maintain nutritional quality. Soils also underpin our nationally and internationally valued and rare habitats. These include blanket peatlands, montane habitats, native pine woodlands and machair grasslands. Rural areas attracted 30-40% of overseas visitors in 2006. Soils protect air and water from the impacts of a range of pollutants. Most of the water that we use has flowed over or through soil. The acidity of rainfall can be neutralised and contaminants such as trace metals removed by adsorption to soil solids. Soils store significant amounts of carbon and Scotland's soils account for some 70% of terrestrial carbon in Great Britain (Milne and Brown, 1997). Consequently we need to ensure our soils do not become net emitters of greenhouse gases and further accelerate climate change. Warmer climates and/or more intensive land use could increase loss of carbon to the atmosphere. Soils also contain and sustain a vast wealth of biodiversity that underpins its many functions and many of these organisms may have biotechnological and pharmaceutical potential. Lastly, soils provide the foundations for buildings and roads the effects of which are largely irreversible. An area the size of Dunfermline (1,200 hectares) is contained within the built environment each year (Towers *et al*., 2006).

It is logical therefore that informed management and protection of soils will contribute to developing Scotland into a wealthier, healthier and greener nation. A wider and shared appreciation of their role and a greater understanding of how soils respond can also contribute to a smarter Scotland that is more responsive and more likely to change behaviors for a more sustainable future.

Scotland's Soils

Soil is often described as the most complex of our environmental media. It is formed from minerals derived from the basic geology modified by the actions of soil biodiversity, plant inputs, weather and climate, time and man. Compared to air and water it is an unmixed media that exhibits high spatial variability due to the fundamental nature of its formative processes. This variability sets challenges in managing, protecting and monitoring our soil resources.

The diverse geology and climate of Scotland has given rise to a wide variety of soil types (Soil Survey of Scotland, 1984). Variations in topography have caused further local-scale variation in soil types associated with slope and landform. The strongly maritime climate with cool temperatures, and rocks that are generally resistant to weathering, has resulted in soils that are more organic, more leached and acidic and wetter than those of most other European countries. Scotland contains greater proportions of podzols (23.7% of the land area), peat soils (histosols, 22.5%) and gleys (20.6%) than Europe as a whole. There is a contrast between soil types in the Midland Valley (primarily mineral soils) and those in the Highlands and Southern Uplands (primarily peaty soils).

This pedological diversity determines the balance of soil-related functions in our landscape. Although almost all soils produce above-ground biomass, the land cover map of Scotland shows that only ~25% of the area of Scotland is used for arable crops and improved grassland, with a further 17% under woodland. Arable crops are primarily grown in the eastern half of the country, with improved grassland in the south west. The remainder of the country supports semi-natural vegetation such as heather moorland, blanket bog and montane habitats many of which are of high conservation value. Consequently protecting such a soil resource and the balance of varied functions creates particular challenges.

PAST

We have been working the soils of Scotland for over 5000 years and this long-term relationship has contributed to the diversity of soils and the uniqueness of our landscape. For example, machair soils, with their unique chemical and biological characteristics, contribute to Scotland's historical island crofting culture. Heather moorlands, managed for grouse, have in part determined the nature of our peaty podzolic soils with historical signs of muir burning. More intensive agricultural use of soils is reflected in soils limed to an optimal pH of around 6.2 and relatively low, but fairly stable, organic matter levels in soils.

There has been a historical tendency to assume that soils change slowly over time, being resistant and relatively insensitive to environmental change. In an evolutionary sense this is partly true as soil formation and development can take many 1000s of years. In contrast we now know that changes in vegetation cover through management or natural succession can change soils much more quickly. For example, the growth of birch on heather moor has been shown to change the soil from a carbon (C) sequestering peaty podzol to a brown earth in less than 30 years (Mitchell *et al*., 2007). External pressures can also alter soils. Over the last century, we know that acid rain has lowered the pH of many of our soils as a result of sulphur and to a lesser extent nitrogen deposition, from industrial, often transboundary, sources while more localised contamination often reflects past industrial activities.

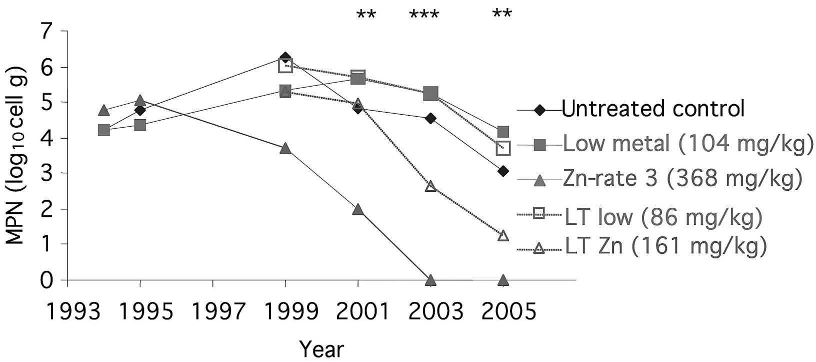
The proposed EU soil framework directive published in September 2006 identified 7 threats to soil namely:- erosion; decline in soil organic matter; contamination; sealing (land development); compaction; salinization; floods and landslides. The relative importance and severity of these threats in a Scottish context will depend on local socio-economic drivers and the nature and geography of our soil resource. Towers *et al*. (2006) reviewed these potential threats to our soil resource. Threats from **erosion**, **compaction** and **contamination** (other than acidification) were judged to be of localised significance, although they can lead to loss of important functions. They were also assessed as being relatively straightforward to rectify. **Sealing** and **acidification** were scored more highly as threats nationally, with sealing affecting almost all soil functions. **Loss of biodiversity** was also considered within this report although it has subsequently been removed from the list of EU threats. **Climate change** and **loss of organic matter** were identified as the most significant threats to soil functioning, although there is much uncertainty in the evidence here. However, for many of the threats, there is a lack or absence of data upon which to make robust conclusions.

CURRENT

Despite the now known risks of past activities, existing information suggests that Scottish soils are generally of good quality (Towers *et al*., 2006). Only a few soils have high levels of contamination and levels in the remainder are generally low. There is little evidence to suggest that serious soil erosion, compaction or other problems related to land management are occurring widely. Some soils are particularly sensitive to acid inputs, but there is some evidence that this problem is less now. The validity of these statements is however reliant on good, recent, soils data and many of the existing national datasets date from the 1970's/1980's.

Where such data become available they demonstrate how we continue to be surprised by soil responses to environmental pressures. For example, recent results from the long term sewage sludge trials (Anonymous, 2007) have highlighted that there are significant knowledge gaps in how soils respond to sewage sludge borne heavy metals. Conventional wisdom at the outset of the experiment in 1994 suggested that sludge cake with enriched metals would have a lower impact on soil rhizobium than metal salts or liquid sludges spiked with metal salts. After 12y, the experiment demonstrated the opposite (Anonymous, 2007). Further, one site at Hartwood, in central Scotland was found to be more sensitive than expected and Rhizobium numbers were reduced to below detection limits in the high Zinc-rich sludge treatments (Figure 1). These studies make a powerful argument for long-term multi-site experiments and also national soil monitoring as the previous evidence from a few studies would not have predicted such effects. It also suggests that biological indicators are essential if we are to integrate the complex responses of different soils.

The Scottish Government is now seeking to develop evidence-based soil policy with research now commissioned to look for evidence of change in our soil resource and examine how we might monitor soils more effectively. A Soil Strategy for Scotland is in preparation and plans are being made for a soil monitoring network. Scotland is fortunate to have soil maps and a soils database, including the National Soils Inventory of Scotland (NSIS) that is one of the best soil databases and archives in Europe. The NSIS is central to defining our present knowledge in respect to current indicators for important land use/soil type combinations; their coverage, and the design of any new sampling. The first NSIS was undertaken from 1978 to 1987 and covers 721 sites sampled at 10 km grid intervals across Scotland.



*/**/***: Significant regressions on dose-response treatments

Figure 1: Most Probable Numbers (MPN) of nodulating Rhizobium over 12 y in untreated soil and soil treated with sludge with either low or high metal concentrations the latter being applied either in first 4 y of Phase 1 (Zn Rate 3) or annually (LT-Zn), (Anonymous, 2007)

We are now updating the NSIS by re-sampling a systematic subset of the 10 km NSIS points at a 20 km interval. Phase 1 of the sampling took place in March-May 2007 with subsequent sampling planned in spring of 2008 and 2009. Sampling has been constrained to spring time when inter-year variation is lowest in part to accommodate biological analysis. This re-sampling exercise has a number of complementary objectives:

1. To detect temporal change over the last 25 years in key properties, e.g. C, N and pH.

2. To provide new data, especially physical and biological, which are missing from the existing database such as bulk density and moisture release characteristics.

3. To enhance our capability to model and predict the risks associated with climate change and land management.

4. To test different field sampling techniques viz i) pedological (horizon–based sampling) ii) composite, 15 cm depth augered samples (25) from a 20 x 20 m grid as used in the NSI of England and Wales and iii) a single 15 cm deep core as used in Countryside Survey.

5. To understand the spatio-temporal variability and uncertainty in soil properties. Additional samples of the uppermost horizons are being taken at random distances and orientation around the central sampling point so variability in key properties can be estimated for the first time.

6. To provide representative soils to test a range of new analytical techniques such as rapid mineralogical (XRPD) and organic matter fingerprinting using Fourier transform infra red spectroscopy (FTIR) and alkane profiles. In addition molecular biology techniques for high throughput taxonomic identifications of fauna and microflora and also functional genes are being deployed.

In parallel, experimental approaches linked to predictive modeling are evaluating how changes in land use or management practices alter the soils capacity to retain or release carbon, phosphorus and nitrogen and the ultimate consequences for different functions including; crop production, habitat restoration, greenhouse gas emissions, phosphorus transfers to water and the application of sludge and composting materials to land. In particular, this information will greatly enhance our capacity to predict risks from climate change and the consequence of changing land management for the range of benefits derived from soils.

A fundamental component of this research is determining the role of soil biodiversity in delivering the range of soil functions. Without soil organisms, a soil cannot function. Thus soil biodiversity is pivotal if soil management and conservation are to be effective. However, we are still relatively ignorant of how soil organisms interact with soil chemical and physical properties. Our first hurdle has been to develop tools that can let us identify, quantify and track what are primarily microscopic organisms. Foremost are a range of DNA-based markers to identify fungal species, including those of high conservation significance, along with taxonomic and functional aspects of bacteria and nematodes. New approaches for estimating the functional significance of soil microorganisms include rapid physiological methods which can be used to assess soil resilience to a range of stressors. Such approaches offer promise as new functional indicators of soil health. Stable isotope are proving valuable in unraveling how carbon is taken up by soils, the mechanisms for soil carbon retention and how management or environmental change alters these mechanisms. We have developed the capacity to track and fully quantify the movement of carbon from the atmosphere through the plant, into the soil and its soil organisms and ultimately back to the atmosphere or into different soil organic matter fractions. This technique is being used to determine how the application of compost materials in arable crops alters soil carbon content and greenhouse gas emissions.

FUTURE NEEDS

It is useful to scan current and future policy targets to predict what future demands we might place on our soils. For example, as a Nation we aim to increase the area under woodland from 17 to 25% by 2050 (Forestry Commission Scotland, 2006);

achieve 50% renewable electricity generation by 2020 (Scottish Government web site); and reduce carbon emissions by 80% by 2050 (www.scotland.gov.uk/news). We are also increasing year on year the land used for development at a time when we are also asking for more locally produced, healthier food from within our existing soil capacity. These pressures raise questions about which soils are best suited for specific uses, e.g. for future woodlands if one of the primary aims is to increase C sequestration. How do we mange our agricultural soils to sustain and enhance food quantity and quality without damaging our soil, water and atmosphere? How can soils contribute to renewable energy targets and reduce greenhouse gas emissions? Which soils are suitable for supporting wind turbines, growing biomass crops? When we develop land what soils and therefore what capacity and functions are we losing?

So there is increasing competitive pressure on soils to contribute to a range of ecosystem services (Millennium Ecosystem Assessment, 2005). Possibly one solution is to find a way of bringing soils into the spatial planning arena. This also argues for monitoring soils not just at a National level but at an appropriate local government level so planning can achieve a more holistic and joined up way of taking soil into account. To achieve this efficiently we need to develop our databases into soil information systems that a number of countries already have. A web-based soils information system for Scotland could deliver data and information in a variety of different ways (languages) that are suitable for a wider range of users.

One might also speculate about future pollutants, e.g. nanoparticles and emergent new pathogens. We cannot predict exactly how these pressures will be manifest on our soils but we can plan a system of monitoring that can alert us to the dangers and by research identify solutions and approaches to manage the risks. New monitoring schemes however designed and implemented are needed to identify the effect of these future pressures on our soil if it is to stay in good health and support our way of life. Soil science and especially soil biological science is discovering soil is a new scientific frontier with levels of complexity not previously imagined (Fitter *et al*., 2005). It is a little ironic that though we accept soil is complex we still use relatively simple bulk soil properties to measure and monitor it, e.g. pH, %C and yet we know from several studies that generalisations from one field site are often difficult to extrapolate to another. We suggest that we are now entering a new era of characterising soil using a variety of physical, chemical and biological methods often as complex fingerprints that better reflect soil's inherent complexity and may be better suited to predict future responses.

Scotland is also fortunate to have its own National soil archive of 40,000 soils collected over 60 years from a range of systematic and long term experiments that complements the soil database. It is an essential resource for validating new analytical techniques and following changes in new indicators. As part of the re-sampling programme of the NSIS the new sample will also be archived but also soil DNA is being extracted and measured. Soil DNA will also be archived for future scientists to use the techniques of tomorrow to look back at the present and so have the opportunity to understand what and how changes have occurred. This is an important part of our future national capacity and planning for the future.

The soil resource of Scotland has developed over many millennia and continues to do so under both natural and human influences. Our soils are generally in good health and this, in large part, has resulted from the sustainable management systems employed by land managers over a prolonged period. Previous analysis of the significance of the threats to Scottish soils has identified two linked threats, climate change and loss of organic matter, as the most significant but there is also the greatest uncertainty associated with these. Notwithstanding the uncertainties associated with this judgment, there is increasing evidence of the need to safeguard our soil resource for future generations and put in place flexible and future proof monitoring systems.

ACKNOWLEDGEMENTS

The authors and work they describe has been funded by the Scottish Government Research, Environment and Rural Affairs Department. We also acknowledge our many colleagues in The Macaulay, SAC and SCRI working within the SG-RERAD sponsored Workpackages 3.2 and 3.3.

REFERENCES

Anonymous (2007). Effects of sewage sludge applications to agricultural soils on soil microbial activity and the implications for agricultural productivity and long-term soil fertility: Phase III. Report Ref: SP0130;CSA 6222 October 2007, UK Water Industry Research Ltd.

Fitter AH, Gilligan CA, Hollingworth K, Kleczkowski A, Twyman RM, Pitchford JW and members of the NERC Soil Biodiversity Programme (2005). Biodiversity and ecosystem function in soil. Functional Ecology 19, 369-377.

Forestry Commission Scotland (2006). The Scottish Forestry Strategy The Scottish Executive, Edinburgh.

Mitchell RJ, Campbell CD, Chapman SJ, Osler GHR, Vanbergen AJ, Ross LC, Cameron CM and Cole L (2007). The cascading effects of birch on heather moorland: a test for the top-down control of an ecosystem engineer. Journal of Ecology 95 (3), 540-554.

Millennium Ecosystem Assessment (2005). Ecosystems and human well-being. Island Press.

Milne R and Brown TA (1997). Carbon in the vegetation and soils of Great Britain. Journal of Environmental Management 49, 413-433.

Towers W, Grieve IC, Hudson G, Campbell CD, Lilly A, Davidson DA, Bacon JR, Langan SJ and Hopkins DA (2006). Scotland's soil resource – Current state and threats.
http://www.scotland.gov.uk/Publications/2006/09/21115639/0

http://www.scotland.gov.uk/News/This-Week/Speeches/Greener/climatechangejun21 (Accessed December 14 2007)

http://www.scotland.gov.uk/News/Releases/2007/11/27095600 (Accessed December 14, 2007)

Soil Strategy for Scotland. http://www.scotland.gov.uk/
Publications/2007/07/30113343/0.

Soil Survey of Scotland (1984). Organization and methods of the 1:250 000 Soil Survey of Scotland. The Macaulay Institute for Soil Research, Aberdeen.

DEVELOPING SOIL POLICY TO MEET SCOTLAND'S FUTURE NEEDS

Mark Aitken

Scottish Environment Protection Agency, Castle Business Park, Stirling, Scotland, FK9 4TR, UK, E-mail Mark.Aitken@sepa.org.uk

SUMMARY

Soil is an integral part of our environmental, social and economic systems, underpinning food production, environmental quality and biodiversity. At present, there is no specific European or Scottish legislation on soil protection. Policies that cover soils, tend to offer protection to a specific function and/or activity, for example, land used for waste application or groundwater protection from nitrate pollution. The profile of soil protection is increasing and various policy developments are either underway or are being implemented at a Scottish, UK and EU level. There is a need to ensure that future soil policy developments are founded on sound scientific evidence and strikes the right balance between EU-level action and the application of the principles of subsidiarity, proportionality and better regulation.

INTRODUCTION

Soil is a key part of our environment and has major implications for air and water quality as well as our climate, biodiversity and economy. In Scotland, as with the rest of the UK, soil has not been afforded the same level of protection compared to the water and air environments. This is partly due to the normally long timescales over which soils respond to pressures and also to difficulties associated with regulating a resource which is primarily in private ownership.

There is a clear need for soils and their functions to be afforded the same level of environmental protection as the water and air environment through explicit legislation. There are many existing EU provisions which have some elements of soil protection although existing directives are generally restricted to specific land uses or management and do not cover the whole land and soil cover. Soils are partly protected through other environmental media legislation, e.g. emission controls of air pollutants will control soil acidification. However the importance of soil as a non-renewable resource, alongside soil functions being essential to a sustainable environment, needs to be fully recognised by soil policy measures. Soil functions which may require protection (Blum, 1993) include:

- Providing the basis for food and biomass production for our agricultural and forestry industries;
- Regulating our water supply and protecting it from contamination;
- Providing nationally and internationally valued habitats;
- Storing carbon and maintaining the balance of gases in the air;
- Sustaining biodiversity;
- Preserving of our cultural and archaeological heritage;
- Providing raw material;

- Providing a foundation for buildings and roads.

SEPA (2001) gave the following recommendations following its review of the state of Scottish soils :

- a Soil Protection Strategy for Scotland was needed;
- a quantitative assessment of the Scottish soil resource should be undertaken;
- a long-term soil monitoring strategy should be developed and implemented;
- existing legislation relevant to soil protection should be integrated.

Towers *et al*. (2006) reported on the state and threats to soil and concluded that climate change and loss of organic matter was the most significant threat to soil functioning. Sealing, loss of biodiversity and acidification were also scored high while threats from erosion, compaction and contamination (other than acidification) were judged to generally be of localised significance.

SEPA (2006) concluded that the main threats to soils in Scotland were as follows:

- inappropriate agricultural and forestry practices that result in an increased risk of loss of soil organic matter and carbon (linked to climate change), soil erosion and nutrient loss (linked to diffuse pollution);
- sealing or replacement of soil with hard surfaces such as roads, car parks and buildings reducing water held in soil which increases run-off and flood risk;
- compaction of soil which affects soil structure and therefore its ability to hold water and nutrients;
- inappropriate use or disposal of chemicals and waste causing contamination;
- land use changes resulting in alterations in natural land cover decreasing biodiversity;
- deposition of substances emitted to air altering soil chemistry.

Soil Organic Matter and Carbon Loss

SEPA (2001) concluded that soil organic matter is vitally important in terms of soil quality because it:

- increases the ability of soil to hold and supply both nutrients and water;
- promotes and enhances soil structure;
- physically binds pollutants to soil particles, so that they are immobilised;
- supports biodiversity in soil.

Organic matter also has major role to play in mediating climatic warming as it is a significant carbon store. Loss of soil organic matter increases carbon dioxide emissions and levels of organic carbon in water, as well as negative effects on other soil functions. Milne and Brown (1997) estimated Scotland's soil carbon stocks accounted for 70% of the terrestrial storage of carbon in GB. Bradley *et al*. (2005) stated that Scotland's soils contain an estimated 2196 million tonnes of soil carbon, to a depth of 100 cm, compared to a total of 4566 million tonnes for the whole of the UK. Small changes in terrestrial carbon stocks will result in large emissions. Soil Association (2005) estimate that only 0.1% of these carbon stocks needs be released to the atmosphere for Scotland's current man-made carbon dioxide emissions to double.

Tackling soil risks and threats is urgent, particularly with regard to climate change. Organic matter loss from soils and therefore increased CO_2 emissions is an especially serious issue, due to the feedback into climate change. Studies in England and Wales have shown that peat soils are losing soil organic matter at a mean rate of 0.6% per year relative to the existing soil carbon content (Bellamy *et al*., 2005). A similar loss of soil organic matter from peat soil in Scotland would be very serious to the atmosphere and local watercourses. There is evidence in Scotland of increased losses of dissolved organic carbon to waters as a result of degradation of upland soils (Worrall *et al*., 2003).

Diffuse Pollution and Soil Management

Morris *et al*. (2006) reported that a total of 488 (24% of the total) river bodies in Scotland, equivalent to a total river length of 5775 km, and 21 (20% of the total) groundwater bodies are at risk of not meeting the WFD environmental objectives due to diffuse pollution. Morris *et al*. (2006) reported on a screening tool to assess sources of diffuse pollution of water and demonstrated that soil management is a key factor in influencing the water environment. Agriculture clearly dominated the losses of nitrogen, phosphorus and soils, contributing 74%, 52% and 88% of the total load respectively from all sources. The modelled total annual losses (tonnes per year) were 46,000 tonnes of nitrogen, 2,800 tonnes of phosphorus and 774,000 tonnes of soils. The magnitude of these losses for Scotland is far higher than can be considered sustainable and improved soil management is required.

Soil Contamination

Use of organic materials applied to soil can convey positive benefits but if poorly or inappropriately applied, can pose a risk to soil, air, water, plants, animals and humans (Aitken *et al*., 2003). SEPA's soil monitoring focuses on assessing impacts on soil from waste recovery or disposals made to land and covered by the Waste Management Licensing Regulations 1994 and the Sludge (Use in Agriculture) Regulations 1989. Emerging research (Towers *et al*., 2006) on the impact of sewage sludge derived heavy metals to soils indicate that UK Sludge Regulations need re-assessed.

There is UK legislation in place for the identification and remediation of contaminated land. The Pollution Prevention and Control (Scotland) Regulations prevents future pollution to land and Part IIA of the Environmental Protection Act 1990 regulates land contaminated from historical pollution and seen to be causing significant harm.

DEVELOPMENTS IN SOIL POLICY

The importance and vulnerability of soils was for a long time not widely appreciated and there are thus claims that soils have been overlooked in environmental policies both nationally and internationally. More recently however there has been growing recognition that soils are a vital resource that should be protected at all levels. The following section highlights some of the milestones in soil policy.

The European Soil Charter in 1972 recognised that soil deterioration was occurring in many parts of Europe and recommended that soil be protected from a range of pressures including urban development, pollution and erosion and recommended the integration of soil assessment approaches and soil protection policies.

In 1979, the United Nations Economic Commission for Europe (UNECE) implemented the Convention on Long-Range Transboundary Pollution. The UNECE Sulphur Protocol was a soil based policy to tackle transboundary air pollution and was set up because of soil acidification and based on acceptable limits of acidifying input to soils.

The Royal Commission on Environmental Pollution report on Sustainable Use of Soils (1996) concluded that environmental policies have 'largely taken soils for granted' and recommended that soils should have equal status with air and water in environmental legislation.

The Water Framework Directive (WFD,) which came in force in 2000, requires regulatory controls be established to prevent or control the input of pollutants to surface or groundwater. There are clear links between diffuse pollution and soil management and land-use. Protecting soil will have additional benefits for assisting attainment of targets set out in the WFD by reducing sediment and phosphorus loads in watercourses, a key issue in Scotland.

In 2002 there was a major step in the development of an encompassing EU policy to protect soils when the EC published a Communication Towards a Thematic Strategy for Soil Protection (Europa, 2006).

Common Agricultural Policy reform in 2005 resulted in cross-compliance and the need for farmers to keep their land in "Good Agricultural and Environmental Condition" (GAEC). In total, 14 specific measures have been agreed in relation to soil erosion, soil organic matter, soil structure and minimum level of maintenance. Where such measures are developed and adequately implemented and enforced, environmental benefits to the soil can potentially be achieved.

In September 2006 the next major European milestone was the publication of the EU Soil Thematic Strategy and associated draft Soil Framework Directive. The overall objective of this Strategy is protection and sustainable use of soils, based on the principals of preventing further soil degradation and restoring degraded soils (Europa, 2006). The proposal for a Soil Framework Directive (SFD) is currently being considered by the Council of Ministers and the European Parliament. The Directive as currently proposed (Europa, 2006) may be significantly revised if and when it is formally adopted by the European Commission and therefore the following comments refer to the current situation (December 2007) which may change.

The proposed SFD, as currently drafted, lays down a framework for the protection and sustainable use of soil based on the principles of integration of soil issues into other policies, preservation of soil functions within the context of sustainable use, prevention of threats to soil and mitigation of their effects, as well as restoration of degraded soils to a level of functionality consistent at least with the current and approved future use of the land. The key elements of the Directive as currently proposed by the Commission (Europa, 2006) are:

- A requirement for central and local Government to consider the impacts that new policies will have on soils whilst they are being developed (Article 3);
- A duty on all land-users to prevent or minimise harm to soils (Article 4);
- A requirement to limit or mitigate the effects of soil sealing (the covering of the soil surface with an impermeable material such as concrete) (Article 5);
- A requirement to reduce the risks relating to soil erosion, organic matter decline, compaction, salinisation, and landslides, by identifying risk areas, and deciding on a programme of measures to address these risks (Articles 6-8);
- A requirement to prevent soil contamination, compile an inventory of contaminated sites and remediate those sites listed on the inventory (Articles 9-14); and
- A requirement to raise awareness of soils issues, report to the Commission, and exchange information (Articles 15-17).

The cross-cutting nature of soils will mean that the EC proposals for soil protection are likely to raised across a number of policy initiatives and delivery mechanisms. It is intended to use a variety of existing policies (e.g. the Programmes of Measures under the WFD, Natura 2000 Management Plans, EC Nitrates Directive Action Programmes, Rural Development Regulation and CAP reform policies, etc.) as vehicles to achieve better soil protection. The Rural Development Regulation is viewed as a particularly useful instrument in this regard.

A very major milestone in soil policy was the commitment made by the Scottish Government (2006) to develop a Scottish Soil Strategy. This will provide a framework for soil protection in Scotland and work on this by a broad range of stakeholders has now commenced. One key aspect of the strategy will be the preservation of carbon stocks in soils.

CONCLUSIONS

Scottish soils are distinctly different to soils elsewhere in the UK and they require specific management guidance and protection strategies. This is of particular significance in the context of climate change. The way in which soil is managed can play an important role in ensuring that soils act as a “sink”, and not a source, of greenhouse gases. There are also clear links between diffuse pollution and soil management.

Industrial, waste disposal, agricultural, forestry and development activities can all impact on quality, either directly or indirectly. Environmental events such as flooding and climate change also have implications for soil quality, although land management plays an important role in mitigating change and has an important role in reducing run-off and alleviating flood risks. Soil, air and water environments therefore are linked and interact. It is vital that future policy developments in Scotland take this into account - i.e. that the soil, air and water environments are viewed as a whole and that each component is given equal importance.

Not all activities that may be damaging to soil are fully subject to regulatory control. Some activities are subject to guidelines and codes of practice that have varying degrees of statutory status. Promoting the adoption of good practice through awareness raising, dialogue and published guidance continues to be a key mechanism to protect soils.

The on-going development of the Scottish Soil Strategy by the Scottish Government has brought together policy makers, regulators, soil scientists and land managers such that there is now considerable consensus and extensive partnership working in Scotland. The need for sound scientific evidence when developing a more strategic approach towards soil protection policy continues to be essential. There is a recognised need to develop a soil monitoring programme that allows for the state of soils in Scotland to be better understood, and to determine whether policies result is an improvement or deterioration in soil quality with passage of time.

The developing Soil Framework Directive (SFD) presents opportunities to ensure soils and their functions are given the same level of environmental protection as the water and air environment through explicit legislation. The policy challenge is to ensure that the SFD strikes the right balance between European-level action and the application of the principles of subsidiarity, proportionality and better regulation.

REFERENCES

Aitken M, Sym G, Douglas J, Campbell C and Burgess S (2003). Impact of industrial waste and sheep dip chemicals applied to agricultural land on soil quality. http://www.sepa.org.uk/pdf/publications/reports4sepa/industrial_waste_sheep_dip_chemicals.pdf

Bellamy PH, Loveland PJ, Bradley RI, Lark RM and Kirk GJD (2005). Carbon losses from all soils across England and Wales 1978-2003. Nature 437, 245–248.

Blum WEH (1993). Soil Protection Concept of the Council of Europe and Integrated Soil Research. In: HJP Eijsackers and T Hamers (Eds.). Kluwer Academic Publisher, Dordrecht, 37 -47.

Bradley RI *et al.* (2005). A soil carbon and land use database for the United Kingdom. Soil Use and Management 21, 363-369.

Europa (2006). A strategy to keep Europe's soils robust and healthy. http://ec.europa.eu/environment/soil/index.htm

Milne R and Brown TA (1997). Carbon in the vegetation and soils of Great Britain. Journal of Environmental Management 49, 413-433.

Morris R, Macdonald J, Greig S and Preston C (2006). Matching water framework directive controls with diffuse pollution challenges in Scotland. Managing Rural Diffuse pollution. Proceedings of SAC and SEPA Biennial Conference, pp. 13-22.

Royal Commission on Environmental Pollution (1996) Report on Sustainable Use of Soils.

Scottish Government (2006).
http://www.scotland.gov.uk/News/Releases/2006/09/25102611

SEPA (2001). State of the Environment: Soil Quality Report.
http://www.sepa.org.uk/pdf/publications/state_of/soil/soil_report.pdf

SEPA (2006). State of Scotland's Environment Report.
http://www.sepa.org.uk/changetomorrowtoday/report/index.html

Soil Association (2005). Climate Change, Land Use and Agriculture: A Scottish Perspective, http://www.soilassociation.org

Towers W, Grieve IC, Hudson G, Campbell CD, Lilly A, Davidson DA, Bacon JR, Langan SJ and Hopkins DA (2006). Scotland's soil resource – Current state and threats to Scotland's soil resource. Report for Scottish Government. http://www.scotland.gov.uk/Publications/2006/09/21115639/0

Worrall F, Burt TP and Shedden R (2003). Long terms records of riverine carbon flux. Biogeochemistry 64, 165-178.

A QUALITATIVE MULTI-ATTRIBUTE MODEL FOR ASSESSING THE IMPACT OF MANAGEMENT OPTIONS ON SOIL SUSTAINABILITY

BS Griffiths[1], PD Hallett[1], M Bohanec[2], G Osler[3] and BC Ball[4]

[1]Scottish Crop Research Institute, Invergowrie, Dundee, DD2 5DA, UK, E-mail: bryan.griffiths@scri.ac.uk; [2]Jožef Stefan Institute, Jamova 39, Ljubljana, SI-1000, Slovenia; [3]Macaulay Institute, Craigiebuckler, Aberdeen, AB15 8QH, UK; [4]SAC, King's Buildings, West Mains Road, Edinburgh, EH9 3JG, UK

SUMMARY

We describe a flexible model to evaluate soil status from a wide range of physical, biological and chemical attributes. The model was used to evaluate the effects of adding compost or slurry to arable systems and the effect of establishing the ley phase after arable in an organic system.

INTRODUCTION

One of the challenges of farming is to maintain or increase soil sustainability. An assessment of the impact of changing soil management should ideally include some measure of soil health or quality. We have taken the definition of soil quality proposed by Eijsackers (1998), 'degree of excellence with a relative nature', as appropriate. Thus, while striving for the best possible quality, the definition is also dependent on soil-type and land-use context. Methodologically, we have addressed this goal using a model-based decision support system, taking the approach of qualitative multi-attribute modeling (Bohanec, 2003). Following this methodology, we developed a hierarchical model, consisting of qualitative attributes and utility (aggregation) functions represented by decision rules, and used it to evaluate and analyse decision alternatives. Several models of this type have been developed for land use, e.g. a model for the assessment of cropping systems (Bohanec *et al*., 2005; 2007). It is a qualitative model in that the outcomes are relative to a particular or control situation and is holistic, taking into account the physical, chemical and biological properties of the soil.

To develop the model a set of measurements was first made in the field and then assembled by the experts involved to arrive at a description of soil quality. We address the methodological aspects of the model and its development, describe the components of the model (attributes and decision rules) and present the initial results of its application to the assessment of amending soil with slurry or compost, and to the benefits of a ley phase for the organic production of arable crops in N.E. Scotland. The aim being to combine very different classes of information to give an overview of effects on the soil system and an understanding of the mechanisms behind those changes.

MATERIALS AND METHODS

Field Sites

Soil samples were collected from two field experiments. In an experiment on a sandy clay loam soil at SCRI soil was sampled from spring barley treated with: cattle slurry (40 t/ha), compost (200 t/ha) or no amendment (control against which the amendments were compared); while in the second on a sandy-loam soil at SAC (Craibstone) samples were taken after three years grass/clover phase and three years arable phase (control against which the grass/clover was compared) of an organic rotation. A variety of physical, chemical and biological parameters (see Table 2) were measured in October 2006 and used to generate the model.

Model Structure

The hierarchical structure of attributes (Figure 1) represents the generation of the overall soil quality indicator from more and more specific indicators, represented by lower-level attributes. Basic attributes that were measured in the field appear as terminal nodes of the hierarchy and represent the input variables of the model. Attributes and values are referred to in italics in the text. Aggregate attributes appear as internal nodes and are built up from lower-level basic and aggregate attributes according to the decision rules outlined below. The attributes can take only one of the three values: *low*, *medium*, or *high*. By convention, the value *medium* is assigned to all the control attributes and to the treatment attributes that lie within the ±15% of the control value. Otherwise the value *low* or *high* is assigned to the attribute, with the value *low* being interpreted as 'bad' and *high* as 'good' but the judgment of 'good', 'bad' or 'neutral' can be altered depending on the land use in question.

Decision Rules

For each aggregate attribute in the model, there is a set of *if-then* rules that define the value of that attribute depending on the values of its immediate descendants in the model. The decision rules were defined by the experts involved in the project using the DEXi software. Table 1 shows decision rules that correspond to the *Faunal Pores* attribute for all combinations of the values of *Earthworms* and *Enchytraeids*, taking the expert view that earthworms have a greater effect on soil pore volume than enchytraeids. Thus, a high value of enchytraeids with a medium value for earthworms will elevate the *Faunal Pores* value to high. There is a decision table for each aggregate attribute. The overall model and decision rules were then placed in the DEXi software package, which facilitates the input and output of data and allows hypothetical scenarios to be investigated.

RESULTS

We tesed the model on data collected in September 2006 and the outcomes (Table 2) reflect the status of the dataset in November 2007 (i.e. not all samples have been analysed). The outcomes presented for the meeting will be changed to reflect the latest dataset.

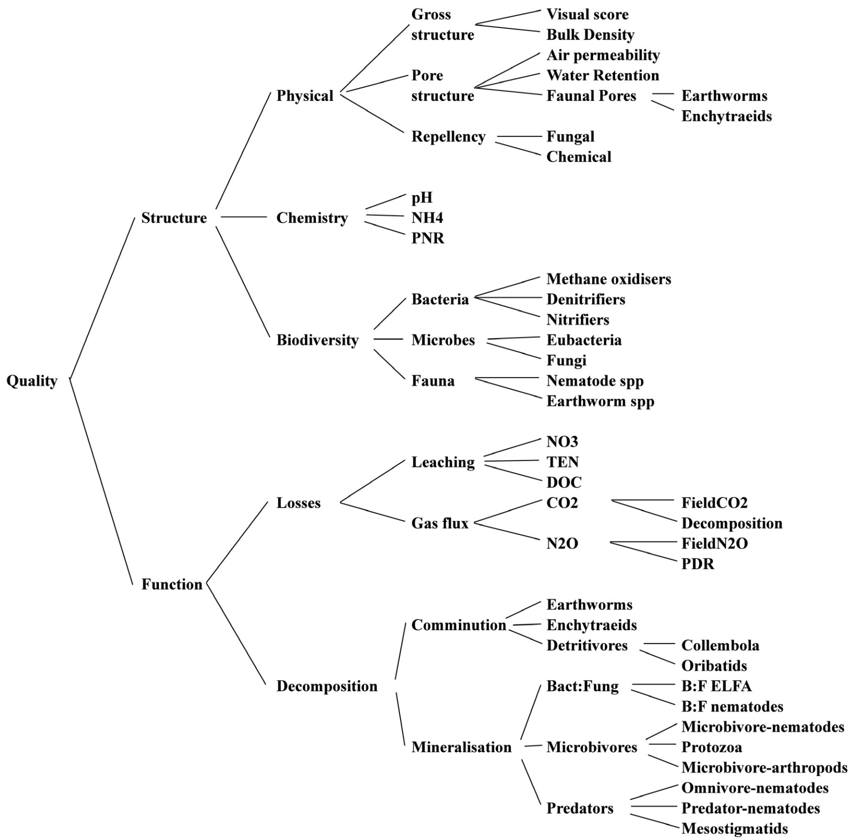


Figure 1: Dendogram showing the combinations of attributes contributing to soil quality. The terminal attributes (apart from Decomposition) were measured in the field

Table 1: Decision rules for the calculation of Faunal Pores from Earthworms and Enchytraeids. Values are: low (L); medium (M) or high (H)

Attribute	Attribute values								
Earthworms	L	L	L	M	M	M	H	H	H
+ *Enchytraeids*	L	M	H	L	M	H	L	M	H
= *Faunal Pores*	L	L	L	M	M	H	M	H	H

Table 2: Outcomes of the model for the attributes detailed in Figure 1, applied to Pilmore with no addition (P), with compost (Pc) or slurry (Ps) and Tulloch arable (T) or after three years in grass (Tg). Values as Table 1, * represents a missing value. The model can return a range of values if there are some missing values

	P	Pc	Ps	T	Tg		P	Pc	Ps	T	Tg
Soil Quality	M	M-H	M-H	M	H	Function	M	M	M-H	M	M
Structure	M	*	*	M	H	Losses	M	L	L-M	M	L
Physical	M	*	*	M	H	Leaching	M	L	L	M	L
Gross structure	M	L-M	L-M	M	H	NO_3	M	L	L	M	H
Visual score	M	L	L	M	H	TEN	M	L	L	M	L
Bulk density	M	*	*	M	M	DOC	M	M	L	M	L
Pore structure	M	M-H	M-H	M	H	Gas flux	M	L-M	*	M	L
Air permeability	M	M	M	M	H	CO_2	M	L-M	L	M	L
Water retention	M	*	*	M	L	Measured	M	*	*	M	L
Faunal pores	M	M	M	M	H	Decomposition	M	L	L	M	L
Earthworms	M	H	H	M	H	N_2O	M	L-M	L-M	M	L
Enchytraeids	M	L	L	M	H	Measured	M	*	*	M	L
Repellency	M	M	L-M	M	M-H	PDR	M	L	M	M	L
Fungal	M	M	L	M	H	Decomposition	M	H	H	M	H
Chemical	M	*	*	M	*	Comminution	M	H	H	M	H
Chemistry	M	H	H	M	H	Earthworms	M	H	H	M	H
pH	M	H	M	M	M	Enchytraeids	M	L	L	M	H
NH_4	M	H	H	M	H	Detritivores	M	H	H	M	H
PNR	M	L	L	M	*	Collembola	M	H	H	M	H
Biodiversity	M	*	*	M	*	Oribatids	M	M	H	M	H
Methane oxidizer	M	*	*	M	*	Mineralization	M	H	H	M	H
Denitrifiers	M	*	*	M	*	Bact:fung	M	L	H	M	M
Nitrifiers	M	*	*	M	*	B:F ELFA	M	L	H	M	M
Microbes	M	*	*	M	*	B:F nematodes	M	M	H	M	M
Eubacteria	M	*	*	M	*	Microbivores	M	H	H	M	H
Fungi	M	*	*	M	*	Micro-nematode	M	H	H	M	H
Fauna	M	H	H	M	H	Protozoa	M	M	M	M	M
Nematode spp.	M	M	M	M	M	Micro-arthropod	M	H	H	M	H
Earthworm spp.	M	H	H	M	H	Predators	M	H	H	M	H
						Omni-nematode	M	H	H	M	H
						Pred-nematodes	M	L	L	M	H
						Mesostigmatids	M	H	H	M	H

The model (Figure 1) has two main branches, *Structure* and *Function.* Within the Function branch, *Decomposition* was one of the attributes for which the dataset was most complete and showed an overall increase in all the treatments. This was most intuitive for the Tulloch-grass, in which all the attributes that differed from the control (Tulloch-arable) were *high*. For the Pilmore slurry and compost treatments there were some attributes that were lower than the control (Pilmore – no amendment): predatory nematodes (*Pred-nematodes*), *Enchytraeids* and bacterial:fungal ratio (*Bact:fung*), but the balance of the effects was *high* leading to increased *Mineralisation* and *Comminution* and so an increase in *Decomposition*. The other attributes contributing to *Function* related to potential losses from the system, either gaseous or leached. Actual gas fluxes were not measured at Pilmore in 2006 but the value of *Gas flux* was *low* in the Tulloch-grass compared to the Tulloch-arable (note that the actual gas flux was increased under grass but as this is deemed 'bad' the value is taken as *low* rather than *high*). The value of the potential denitrification rate (*PDR*), which indicates potential losses, was reduced in the Pilmore-slurry and Tulloch-grass treatments. From this information we would infer a *low* score for *Losses*. The potential leaching losses were increased at Pilmore following amendment with slurry and compost, as a result of increased soil concentrations of nitrate (NO_3), total extractable nitrogen (*TEN*) and dissolved organic carbon (*DOC*). Although the Tulloch-grass treatment had a low concentration of NO_3, the increases in *TEN* and *DOC* gave a *high* value for *Leaching*. We thus had a situation where there were *high* values of *Decomposition* and *low* values of *Losses* which, because they have equal importance in the model, gave a *medium* (i.e. no change) value for *Function*.

Within the *Structure* branch of the model, data on *Biodiversity* is largely lacking (i.e. had not been analysed by November 2007). However, if we extrapolate from the *Earthworm spp* data then the treatments would all have a *high* value for *Biodiversity*. The *Chemistry* branch of the model relates to conditions conducive for plant growth and again all the treatments gave a *high* value. The *Physical* branch impinges on plant growth and microbial habitat, but also relates to wider aspects of soil physical structure. There were differences here in how the soils responded to carbon addition at Pilmore and conversion to grass at Tulloch. While the addition of slurry gave a *low* value for *Repellency*, conversion to grass gave a *high* value. Calculation of the other attributes (*Gross Structure* and *Pore Structure*) at Pilmore was hampered by missing data, but for the purpose of this paper we have assumed a *low* value at Pilmore, thus giving *low* values for *Physical* with slurry addition and a *high* value for conversion to grass. Overall, however, the outcome on *Structure* was *high* for all treatments. This gave an overall *high* value for *Soil Quality* at all sites.

DISCUSSION

There are several advantages in using such a multi-attribute model in helping experimental and field based scientists to understand a complex, multi-component system. It takes the pragmatic approach of making use of the data that is available, whereas implementing a mechanistic model may require the deployment of techniques that the scientific team does not have access to. It requires considerable thinking about the processes occurring in the system which leads to a better understanding of their interaction. It can handle missing data to some extent, for example at Pilmore where there were no values for *Bulk Density* the software calculates a range of outcomes given all possible values of *Bulk Density*. In this case the likely outcomes were *low* or *medium* as shown in Table 2. 'What if' questions can be asked by inputting hypothetical values for attributes. Because the model readily identifies the points where changes occur (e.g. *low* values for *Structure* at Pilmore, *high* values for *Decomposition*) it is relatively easy to pick out what is happening in the system.

Of course there are also disadvantages, the model is relative to a control situation and so is specific for particular circumstances. It could be made generally applicable but that would require parameterization with literature values from a wide range of land uses and soil types. It does rely on ‘expert’ opinion for the weightings, but those should be supportable by reference to quoted studies. A judgment has to be made about the value scales, for example, we have given equal weighting to *Decomposition* and *Losses* under *Function*, but it may be that for a particular system reducing losses is more important and so could be weighted differently.

The output from these examples highlights some interesting results which indicate areas where further decisions and perhaps more detailed research needs to take place. The trade-off between enhanced decomposition and potential losses would suggest that actual losses need to be measured in these systems to ascertain the scale of any problems. Also, the potential compaction problem in the soils given additional carbon might suggest a need to modify soil management, for example.

ACKNOWLEDGEMENTS

We acknowledge the support of the Scottish Government Rural and Environment Research and Analysis Directorate. The experiments form part of the Scottish Government RERAD programme 3, details of which can be found at http://www.programme3.net/index.php.The work of Jožef Stefan Institute is supported by the Slovenian Research Agency programme *Knowledge Technologies* (2004-2008).

REFERENCES

Bohanec M (2003). Decision support. In: Mladenić D, Lavrač N, Bohanec M and Moyle S (eds.): Data mining and decision support: Integration and collaboration. Kluwer Academic Publishers, pp. 23–35.

Bohanec M, Messéan A, Scatasta S, Džeroski S and Žnidaršič M (2005). A qualitative multi-attribute model for economic and ecological evaluation of genetically modified crops. EnviroInfo Brno 2005 Networking Environmental Information. Hřebiček J and Ráček J (Eds.), Masaryk University in Brno, pp. 661–668.

Bohanec M, Cortet J, Griffiths BS, Žnidaršič M, Debeljak M, Caul SC, Thompson J and Krogh PH (2007). A qualitative multi-attribute model for assessing the impact of cropping systems on soil quality. Pedobiologia 51, 239-250.

Eijsackers H (1998). Soil quality assessment in an international perspective: Generic and land-use based quality standards. Ambio 27, 70–77.

A COMPARISON OF THE EFFECTS OF CONVENTIONAL AND ORGANIC FARMING PRACTICES ON SOIL PROPERTIES

LJ Hathaway-Jenkins[1], M Dresser[2], RJ Godwin[1], RC Palmer[3] and R Sakrabani[1]

[1]Cranfield University, College Road, Cranfield, Bedfordshire, MK43 0AL, UK, E-mail: l.j.hathaway-jenkins@cranfield.ac.uk; [2]Landcare Research-Manaaki Whenua, PB 3127, University of Waikato, Hamilton, New Zealand; [3]Soil Assessment Specialist, Fircroft, Sandy Lane, Stockton-on-the-Forest, York, YO32 9UU, UK

SUMMARY

Soil samples were collected from sixteen pairs of farms, throughout England, having both arable and grass fields within each pair on similar soil type. The farms were divided into clusters which are used as replicates in this paper. Chemical (nutrients, pesticides) and physical (aggregate stability, field capacity, shear strength, soil organic matter) soil properties were measured over four main soil textures classes (clayey, coarse, medium, silty) and two land uses (arable and grassland) in organic and conventional fields. The physical soil properties varied significantly between the different texture and land use. However, there are no significant differences between organic and non organic treatments for any of the soil chemical and physical properties measured.

INTRODUCTION

Modifications to policies and farming practices, due to both consumer and governmental pressures, have fuelled the debate over the merits of organic and conventional management methods especially regarding the issues of sustainability, leaching and agricultural pollution (Merrington *et al*., 2002). ‘Soil plays a crucial role in defining sustainable management’ because the maintenance of soil structure and organic matter levels are important if the continued availability of water and nutrients and standards of soil workability are to be sustained (Pulleman *et al*., 2003). There is an abundance of recent literature comparing organic and conventional farms with respect to soil properties, microbiology and nutrient analysis (Marinari *et al*., 2006; Pulleman *et al*., 2003; Parfitt *et al*., 2005). Pulleman *et al*. (2003) compared soil structure and organic matter dynamics on conventional (non-organic) and organic arable farms. However, Stolze *et al*. (2000) emphasised the need for consistent collection of data on soil properties when comparing organic and conventional (non-organic) fields.

Changing land management from conventional to organic farming practices can have significant impacts on environmental factors such as wildlife and soil and water quality (RELU, 2007). This study forms part of an ongoing RELU project which intends to explore the environmental and socio-economic causes of ‘clusters’ of organic farms and to assess whether these clusters are beneficial to wildlife and soil and water quality. The aim of this study is to assess the effects on soil physical and chemical properties of soil surface management under organic and non organic farming systems. The period since conversion for the organic farms studied varies from 3 years to 58 years (RELU, 2007). This paper uses the term non-organic to describe fields that have not been certified organic rather than use conventional because tillage regimes and crop rotations can be the same under the two management systems.

MATERIALS AND METHODS

Site Location

This study investigates sixteen pairs of organic and non-organic farms, in England, with both arable (winter wheat) and grass fields (grass/clover composition) with each pair on the same or similar soil type. The paired farms were chosen in two groups (see Figure 1).

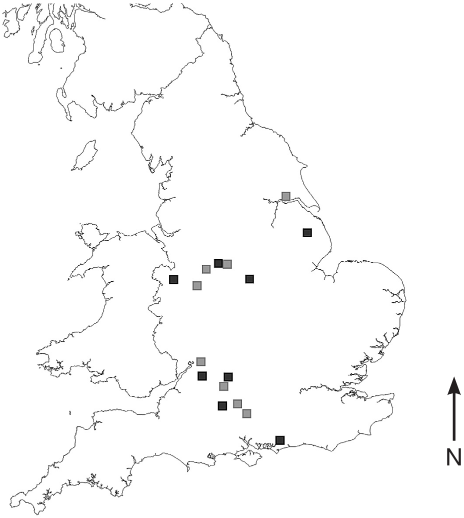


Figure 1: Site locations (RELU, 2007). The dark squares represent cold spots (less than 2% organic land use) and the lighter squares represent hotspots (more than 12% organic land use). These were used as replicates

Field Sampling and Analysis

Soil sampling and within field assessment was carried out in March and April 2007, when soils were at or near to field capacity moisture content as soil structural condition is most clearly assessed at this time. The seasonal effects of variations in soil moisture content were therefore minimised. At each site a soil assessment was conducted and samples were collected to measure a suite of physical (including shear strength) and chemical parameters (nutrients, pesticides, organic matter). To obtain a representative sample of soil, a 'W' shaped path sampling strategy was observed, avoiding untypical areas, taking 10 samples; which were bulked (MAFF, 2000). Samples were obtained from 0-200 mm depth. One or more small pits were excavated at each site to determine the soil structure and physical conditions of the soil. A shear vane was used to measure shear strength *in situ* based on a grid sampling technique using 30 samples to cover the field.

Laboratory and Statistical Analysis

The soil samples were prepared through air drying and homogenisation by grinding and sieving (Allen, 1989). The samples were sieved to either 2 mm diameter (SOM and texture) or passed through a 5 mm sieve and retained on 3.35 mm sieve (aggregate stability). Soil texture was determined using the pipette method (BS 7755). SOM was established by dichromate digestion (BS 1377-3). Aggregate stability was determined through the wet sieving method outlined in Haynes and Swift (1990). Gravimetric moisture content was measured through oven drying at 105°C. Soil water sub-samples were sent to NRM laboratory to be analysed for a suite of pesticides and nutrients using centrifugation.

Data analyses were calculated using Statistica (8.0), under the assumption that data was normally distributed. Factorial analysis was used to determine whether there was a significant difference in soil properties between the two treatments (organic and non-organic); two land uses (arable and grass) and the four textural classes.

RESULTS AND DISCUSSION

Soil Water Nutrients and Pesticides

Soil water samples were tested for a range of pesticides. Pesticides were only discovered within five of the clusters. There were only two organic fields which tested positive for pesticides, but these were only trace levels. There were fifteen non-organic fields which showed traced of pesticides. The pesticides detected in the organic field were compounds of organochlorine and organonitrogen with concentration of 0.3 and 0.02 mg/kg respectively. These pesticides have remained since the farm converted to organic practice in 2000. These pesticides when present in soil are degraded by the microbial community to form metabolites and its half life determines its persistence (Andreu and Pico, 2004). The low concentrations of pesticides detected can be associated with historical application and does not pose a threat of leaching. In addition no differences in pesticide levels were found between organic and non-organic treatments.

There were no significant differences ($p<0.05$) in levels of total inorganic nitrogen (ammonium and nitrate), total phosphorous and total potassium according to treatment, organic or non organic. There was also no influence due to textural class or land use.

Soil Organic Matter (SOM)

There was no significant difference ($p<0.05$) between organic and non organic treatments for SOM content as was reported by Gosling and Shepherd (2005). There were significant differences related to land use, where grass had a significantly higher level of SOM compared to arable ($p<0.05$); and soil textural class where the clayey and silty soils had an improved level of SOM in relation to coarse and medium soils ($p<0.05$). This is due to the protective nature of the clayey soils which reduces the amount of decomposition (Loveland and Webb, 2003).

An argument for the overall lack of difference between treatments is that a reduction in yields for organic compared to non-organic fields could reduce the amount of crop residue available. Yield can be offset against inputs (ley and manure) and could compensate; hence have no significant difference (Gosling and Shepherd, 2002). SchjØnning *et al*. (2007) have recently shown that management can have an effect on SOM level after 5-6 years; however, this research does not support this.

Aggregate Stability

There was no significant difference ($p<0.05$) between organic and non organic treatments for aggregate stability (Figure 2). There were significant differences related to land use, where grass had a significantly higher proportion of stable aggregates compared to arable; and soil textural class where the clayey and silty soils were more stable in relation to coarse and medium soils. Clayey texture soil has the highest amount of SOM and clay content which helps to bind the soil improving the stability of the aggregates.

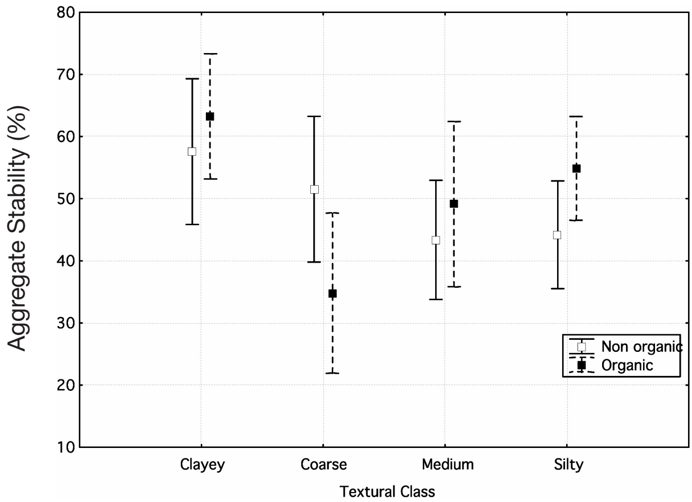


Figure 2: Box and whisker plot showing how aggregate stability varies according to textural class. The vertical bars indicate 0.95 confidence levels for organic and non organic and do not show significant difference between treatments

The management style of grassland such as the removal of grass as silage can remove roots, SOM and binding ingredients (such as calcium ions) which reduces aggregate stability. Over all the fields, a mixture of practices were occurring which could be masking any overall effect of organic or non organic treatments. However, the lack of significant difference between treatments agrees with a number of European studies which found no difference between conventional and organic land uses (Stolze *et al.*, 2000).

Shear Strength and Field Capacity

There was no significant difference between organic and non organic treatments for shear strength (Figure 3) or field capacity. There were significant differences related to land use, where grass had a significantly higher shear strength compared to arable; and soil textural class where the clayey soils had greater shear strength in relation to the other textures.

The amount of SOM present and the moisture content affect the shear strength of the soil. The soils were all sampled at field capacity and the higher amounts of SOM shown in the clayey soils mean that these have higher shear strength. It is important to note that the fields with the higher field capacity (higher moisture content) had lower shear strength as this decreases with moisture content (Smith and Mullins, 1991). The grass fields generally have higher shear strength due to the formation of a strong root mat binding the soil together. The arable fields were more affected by the date of primary tillage, with a few fields having just been tilled and hence the shear strength was much lower than the untilled fields.

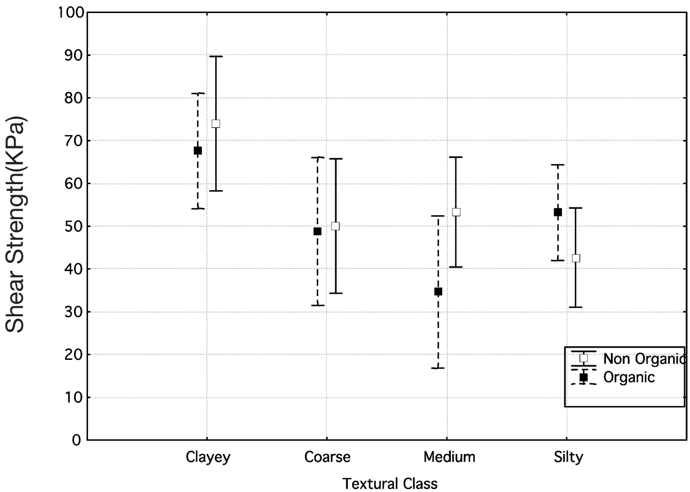


Figure 3: Box and whisker plot showing how shear strength varies according to textural class. The vertical bars indicate 0.95 confidence levels for organic and non organic and do not show significant difference between treatments

This paper only considers some preliminary findings. Further work is currently being undertaken to investigate the effects of soil hydrology and physical properties due to soil surface management in organic and non organic agricultural systems.

CONCLUSIONS

The main conclusions for this paper are as follows:

1. There are no significant differences between organic and non organic treatments for any of the soil properties measured.
2. There are significant differences between grass and arable land use in the following:
   - Aggregate stability, field capacity, shear strength, SOM and SOC are higher in grass land use.

3. These differences are related to the complex interactions between previous land use, current cropping cycle and tillage regime.
4. There are significant differences between the four soil textural groups for all of the soil properties measured. Soil texture plays a key role in determining physical properties which is greater than the current and past land use or treatment.
5. Differences in pesticide and nutrient levels, whilst not undertaken in a statistically rigorous manner revealed no clear differences which could be attributed to treatment.

ACKNOWLEDGEMENTS

We acknowledge the support of RELU and EPSRC for funding this research. We credit Dr Monica Rivas-Casado and Dr Doreen Gabriel for aiding statistical analysis and graphic production.

REFERENCES

Allen SE ed (1989). Chemical Analysis of Ecological materials, 2nd Ed, Blackwell, Oxford.

Andreu V and Pico Y (2004). Determination of pesticides and their degradation products in soil: critical review and comparison of methods. Trends in Analytical Chemistry 23 (10-11), 772-789.

British Standard BS 1377-3:1990 and BS 7755-5.4:1998. Soils for civil engineering purposes – part 3: chemical and electro-chemical tests and method 56 of the MAFF Reference Book RB427 (1986) Analysis of Agricultural Materials.

Haynes RJ and Swift RS (1990). Stability of soil aggregates. Journal of Soil Science 41, 73-83.

Gosling D and Shepherd M (2002). Theory and reality of organic soil fertility – organic matter. In Powell (Eds.) UK Organic Research 2002: Proceedings of the COR Conferences, 26-28 March 2002, Aberystwyth, pp.137-138.

Gosling P and Shepherd M (2005). Long-term changes in soil fertility in organic arable farming systems in England, with particular reference to phosphorus and potassium. Agriculture, Ecosystems and Environment 105, 425-432.

Loveland P and Webb J (2003). Is there a Critical Level of Organic Matter in the Agricultural Soils of Temperate Regions: A Review. Soil and Tillage Research 70, 1-18.

MAFF (2000). Fertiliser Recommendations for Agricultural and Horticultural Crops: RB209, 7th Edition. HM Stationary Office, London.
http://www.defra.gov.uk/farm/environment/land-manage/nutrient/fert/rb209/index.htm
(Accessed 30/10/2007)

Marinari S, Mancinelli R, Campiglia E, and Grego S (2006). Chemical and biological indicators of soil quality in organic and conventional farming systems in Central Italy. Ecological Indicators 6 (4), 701-711.

Merrington G, Winder L, Parkinson R, Redman M (2002). Agricultural Pollution: Environmental problems and Practical solutions, Spon Press, London.

Parfitt RL, Yeates GW, Ross DJ, Mackay AD, Budding PJ (2005). Relationships between soil biota, nitrogen and phosphorous availability, and pasture growth under organic and conventional management. Applied Soil Ecology 28**,** 1-13.

Pulleman M, Jongmans T, Marinissen J and Bouma J (2003). Effects of organic versus conventional arable farming on soil structure and organic matter dynamics in marine loam in the Netherlands. Soil Use and Management 19, 157-165.

RELU (2007). The Effects of Scale in Organic Agriculture.
http://www.relu.ac.uk/
(Accessed 12/07/07)

SchjØnning P, Munkholm LJ, Elmholt S, Olesen JE (2007). Organic matter and soil tilth in arable farming: Management makes a difference within 5-6 years. Agriculture, Ecosystems and Environment 122, 157-172.

Smith KA and Mullins CE (1991). Soil Analysis: Physical Methods, Marcel Dekker, NY.

Stolze M, Piorr A, Häring A, Dabbert S. (2000). The Environmental Impacts of organic farming in Europe. In Organic Farming in Europe: Economics and Policy Vol. 6, Hago Druck and Medien, Germany.

MANAGING CARBON IN A SCOTTISH FARMLAND

RM Rees[1], CFE Topp[1], R McGovern[1], J McP Dick[2], R Smith[2] and AG Coulter[2]

[1]SAC, West Mains Road, Edinburgh, EH9 3JG, UK, E-mail: Bob.Rees@sac.ac.uk; [2]SEE360, North Deanhead, Dunsyre, Carnwath, Lanarkshire, ML11 8NG, UK

SUMMARY

Modelling approaches were used to determine the carbon equivalent (Ceq) footprint of a livestock farm in the north-east of Scotland. Environmental and management data were obtained from a mixed arable and livestock farm rearing cattle and sheep. Two modelling approaches were used to estimate the within farm Ceq footprint. Both included predictions of the emissions of the three greenhouse gases carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). The first approach followed the IPCC recommended procedures for reporting greenhouse gas emissions implemented using the CPLAN model. The second used a process based model (DNDC) to estimate emissions across typical farm rotations. The overall C budget for the farm in 2006 estimated by CPLAN was negative, with a removal from the atmosphere of 11 t Ceq y^{-1} or 44 kg Ceq ha^{-1} y^{-1}. This negative value resulted from a large amount of C uptake by the vegetation offsetting CO_2, N_2O and CH_4 releases. The models estimated total uptake by vegetation of Ceq for the farm of 233 t Ceq y^{-1} or 932 kg Ceq ha^{-1} y^{-1} (CPLAN) and 290 t Ceq y^{-1} or 1161 kg Ceq ha^{-1} y^{-1} (DNDC). Less than 10% of emissions were associated with on farm use of fuel, with land use and animals contributing to the remainder. Improved management of farm animals and fertiliser N use are likely to be the best options for lowering still further the Ceq emissions from the farm.

INTRODUCTION

British agriculture's contribution to the release of greenhouse gases (Ceq) in the UK fell to 7% in 2005, a drop of 1% since 1999 (http://www.statistics.gov.uk/). Despite this relatively low contribution to the national greenhouse gas budget, there is still a recognised need to reduce the carbon equivalent foot print of British farmers because of the perceived market advantage associated with low carbon products. New Zealand, for example, have adopted an emissions trading scheme (ETS) which will include agriculture as they foresee new economic opportunities for agriculture and forestry if they can position themselves at the forefront of the development of new carbon-friendly technologies (Anon, 2007a).

Much of the contribution that agriculture makes to the UK national GHG budget is from N_2O and CH_4. Agriculture contributes 67% of the total N_2O released in the UK and 37% of the CH_4 (http://www.statistics.gov.uk/). The net contribution of agriculture to the total CO_2 emissions of the UK is 1%. Carbon dioxide is both released to the atmosphere in large quantities from agricultural practices while being simultaneously removed and stored in plant vegetation and in soil. The amount of C sequestered can sometimes be sufficient to cancel out the global warming effects of the other greenhouse gases. Measurement of changes in C storage in soils, and C exchange by grassland systems have shown a sequestration rate of about 1700 kg C ha^{-1} y^{-1} at a site near Edinburgh (this was a net effect calculated after taking into account the losses of N_2O and CH_4 (Soussana *et al*., 2007). However, these measurements are technically demanding and difficult to repeat at a wide range of sites. Leading UK farming organisations (NFU, CLA and AIC) have called for the development and deployment of simple tools in order to measure carbon footprints from land-based industries (Anon, 2007b).

In this study, we report on how two contrasting models can be used to assess greenhouse gas emissions, as Ceq, on a mixed farm in Scotland. We restrict our attention to greenhouse gas emissions within the boundaries of the farming enterprise. Such ‘farm gate’ analysis is justifiable as the farmer only has direct control of emissions and sequestration of Ceq within the farm boundaries.

The CPLAN model provides a tool for users to enter data online (www.cplan.org.uk) to a server which holds equations and default emission factors to calculate the Ceq footprint of land-based industries. CPLAN uses the *2006 IPCC Guidelines for National Greenhouse Gas Inventories* (IPCC, 2006) *which* provide methodologies for estimating national inventories of human induced emissions from sources, and removals by sinks, of greenhouse gases. The development of IPCC 2006 guidelines for compiling national greenhouse gas inventories depended on the expertise, knowledge and contribution of over 250 experts worldwide. This was compared with the nutrient cycling model DNDC (DeNitrificationDeComposition) (Li *et al.*, 1992) in order to characterise the exchange of CO_2, N_2O and CH_4. This model calculates daily changes in pool sizes and exchange of carbon (C) and nitrogen (N) from the plant soil system. The relative value of these different approaches to C budgeting on a farm is discussed.

MATERIALS AND METHODS

Two models (CPLAN and DNDC) were used to assess the C budget of a livestock farm in NE Scotland with an area of 249 ha. The farm was stocked with approximately 300 cattle, and 350 overwintering sheep. There was an area of 112 ha of grass-woodland that was used for grazing, and the remainder of the farm operated a six or seven year rotation with cut and grazed grassland being alternated with arable crops (mainly barley). Carbon emissions to the atmosphere (positive values) and removals from the atmosphere (negative values) were expressed as C equivalents (Ceq; incorporating relevant global warming potential multipliers for N_2O (310) and CH_4 (21) over a 100 year time frame). The two approaches tested have different input values and report the emissions in different ways.

CPLAN (ver. 1.0) calculates emissions and sequestration of greenhouse gasses from agricultural systems in accordance with IPCC’s 2006 guidelines (IPCC, 2006) and uses relevant UK emission coefficients where appropriate. Gas fluxes (CO_2, N_2O and CH_4) are estimated in two ways: 1) as net changes in carbon stocks over time (used for most CO_2 fluxes) and 2) directly as gas flux rates to and from the atmosphere (used for estimating non-CO_2 emissions and some CO_2 emissions and removals). The uncertainties in the estimated emissions (upper and lower bounds for an approximate 95% confidence interval) are also calculated as recommended by IPCC 2006 guidelines. The input values used for the CPLAN are based on farm management data (Table 1).

Table 1: Input data and resulting estimated Ceq values obtained from CPLAN (ver 1.0) for a mixed arable and livestock farm in NE Scotland in 2006

	Unit	Value	Total carbon equivalent emissions (tonnes)		
Energy			lower estimate	mean	upper estimate
Grid electricity	kWh	2572	0.3	0.3	0.3
Diesel	litres	25853	16.7	18.5	20.4
Livestock					
Cows	head	122	11.9	78.1	271.2
Cattle >2 years	head	63	6.6	41.5	141.3
Cattle <1years	head	122	13.3	33.6	70.9
Breeding sheep	head	50 a	0.8	5.4	19.1
Sheep < 1 year	head	300	2.3	16.1	57.2
Crop Residue					
Rye, Mixed corn, Triticale	tonne	63	0.1	2.2	10.9
Barley	tonne	276	0.4	7.1	34.5
Inorganic fertilizer					
46% Urea	tonne	30	0.4	8.7	43.5
25:5:5+Na+Se	tonne	38	0.3	6.0	29.9
22:04:14	tonne	27	0.2	3.7	18.7
Forestry					
Grazed woodland	ha	58 b	-174.4	-232.6	-290.8

[a] 100 sheep only on farm for 6 months therefore value of 50 used in calculations

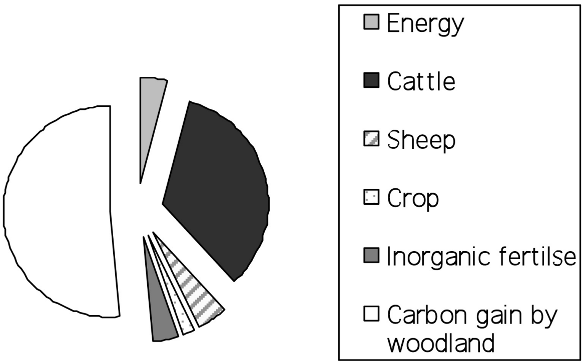
[b] tree growth/yield data was not available therefore birch yield class 8 was assumed and the area of woodland reduced to reflect the non-managed open spaced status of the woodland

The DNDC model is a systems based model that simulates C and N flows in agricultural ecosystems. The model has been applied extensively to agro ecosystems around the world and is widely acknowledged as a state-of-the-art model for use in assessing nutrient fluxes in arable farming systems (Li *et al*., 1992; Saggar *et al*., 2004; Li *et al*., 2006). In this study, we used DNDC (ver 9.1) to model nutrient fluxes within differently managed components of the farm. Farm management data was provided by the farmer (Table 1) and climate data for a period between 1992-2006 were obtained from the weather generator "Earwig" (Kilsby *et al*., 2007). The DNDC model was used to simulate greenhouse gas emissions over a 14 year period, and average annual data for the last seven years of this simulation (2000-2006) are reported in this paper.

RESULTS

The CPLAN model estimated total emissions for the farm in 2006 of 221 t Ceq y^{-1} and 232 t Ceq y^{-1} sequestration of C (Table 1). Emissions from livestock contributed 79% of the total emissions (Figure 1). On-farm energy consumption represented only 8.5% of emissions. All of the emissions from the farm were offset by C uptake by the vegetation resulting in a net uptake of 11 t Ceq y^{-1} or 44 kg C ha^{-1} y^{-1}.

1 (a)



1 (b)

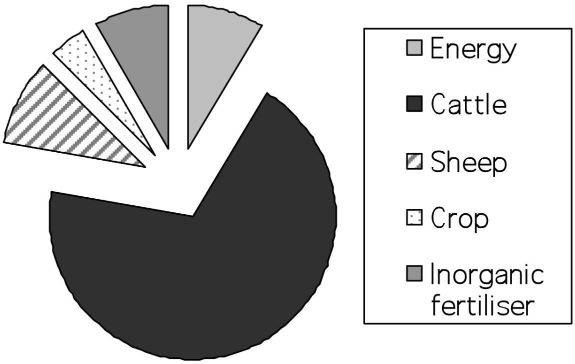


Figure 1: Farm C budget using CPLAN, based on data from 2006. 1 (a) total emissions and uptake; 1(b) emissions only divided by source

The DNDC model does not calculate CH_4 emissions from animals (including housed livestock and slurry stores) or C emissions from fuel use. Comparisons with CPLAN were therefore restricted to the emissions and uptake of C that excluded these categories. Both CPLAN and DNDC predicted a significant C uptake by vegetation of 233 and 290 t C (eq) y^{-1}, respectively, within the farm. Emissions of N_2O estimated by DNDC were larger than those by CPLAN (Figure 2), resulting in a lower net C uptake of 173 t C (eq) y^{-1} or 694 kg C ha^{-1} y^{-1} by DNDC, and 205 t C (eq) y^{-1} or 823 kg C ha^{-1} y^{-1} by CPLAN.

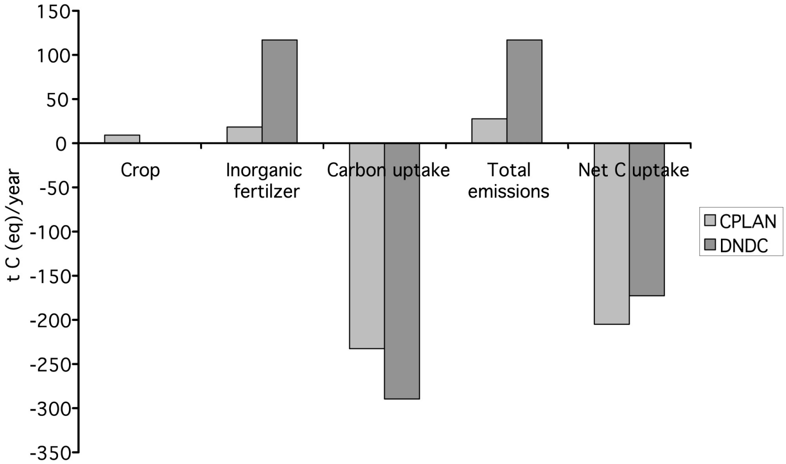


Figure 2: A comparison of uptake and release of greenhouse gases (excluding fuel and CH_4 emissions from animals) using DNDC and CPLAN

The DNDC C budget was calculated from the average C flux over the length of the farm rotation (in this case seven years). Within that period large variations in greenhouse gas exchange occurred as a consequence of the transition between grassland and arable phases (Figure 3). As land moved into the arable phase there was a loss of C resulting from mineralisation of soil organic matter. However, there was a gain of C (sequestration) during the grassland phase as soil organic-C increased. Nitrous oxide emissions resulted in a net loss of C equivalent emissions throughout the rotation, although losses were higher during the grassland phase.

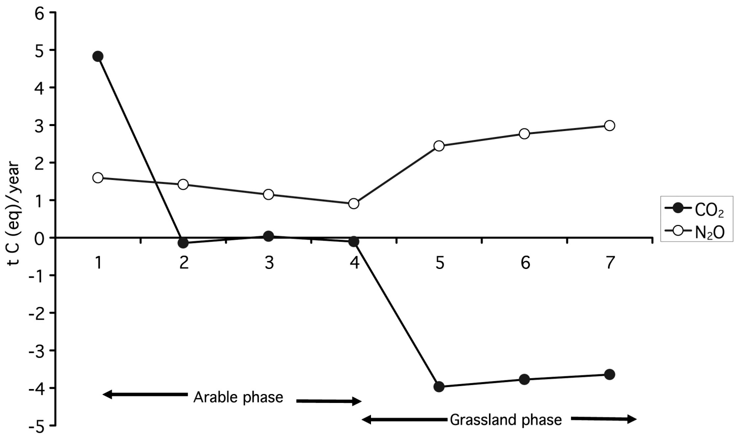


Figure 3: Emissions of N_2O and CO_2 emissions throughout the seven year rotation of the farm. Year 1 ploughing from a grass field after four years in grass. Three years of arable cropping were followed by four years of permanent pasture

DISCUSSION

The results of both modelling approaches highlight the dominance of biological processes in contributing to the C budget of a farming system. On-farm fuel use, estimated by CPLAN, made up less than 10% of the total Ceq emissions on the test farm. Similar results have been found for average Irish dairy units (Casey and Holden, 2005). In order to reduce the C footprint of the farm investigated in this study, mitigation options that target biological processes are therefore likely to be most effective. Three processes were particularly important; CH_4 emissions from animals, N_2O emissions from fertilisers and C uptake by vegetation. It is possible to reduce CH_4 emissions from livestock by manipulation of diet and husbandry (Misselbrook *et al*., 1998; Hindrichsen *et al*., 2005). Because of the large emissions of CH_4 from the farm, mitigation of this source could contribute to significant overall reductions in the Ceq footprint. Similarly there is a growing consensus on options for reducing N_2O emissions from farms by better management of fertiliser N (Ball *et al*., 2008).

Uptake of C by the vegetation was estimated at between 233-290 t C (eq) y^{-1}, but the process of uptake differed in the two modelling approaches. The entire C uptake predicted by CPLAN was a consequence of sequestration by the woodland. CPLAN model does not yet estimate changes in soil or biomass carbon stock for land while under grass or arable management and therefore predicts no overall change in carbon stock over the complete rotation. In the DNDC modelling approach, a more complex pattern of C uptake during the grassland phase and release during arable cropping was simulated and resulted, as indicated in Figure 2, in predictions of a small C uptake, as has been observed under field conditions (Jones *et al*., 2006). In DNDC, it was difficult to adequately simulate the grazed woodland, and the C flows may represent an underestimate of the true C sink. Further model development and validation would be desirable in order to provide improved estimates of C exchange by the system. The results do nevertheless offer compelling evidence of the importance of altering the balance of land use and sequence of land use change as a means of influencing the farm's C footprint.

The two modelling approaches used in this study offer different opportunities to help modify a farm C footprint through management. Both have strengths and weaknesses. The method used in CPLAN follows a widely accepted methodology and could be considered as a benchmark assessment of the C budget. Input parameters are easily available and the approach used by CPLAN has the clear advantage of encompassing all exchanges of C between the farm and its environment (including fuel consumption, methane emissions from animals and land management components). DNDC by comparison is restricted to analysing effects of land management and land use on C and N_2O emissions. It is also relatively demanding in terms of input parameters. However, the DNDC model has been shown to provide realistic estimates of greenhouse gas exchange, and the values of N_2O and CO_2 exchange predicted in this farm are consistent with studies of grassland farming systems elsewhere in Scotland (Smith *et al.*, 1998; Jones *et al*., 2005; Jones *et al.*, 2006; Jones *et al*., 2007). The main advantages of the DNDC approach is that it allows a sophisticated analysis of the affects of changes in management on greenhouse gas fluxes, and it is possible to study this in response to subtle differences in soil and climate. DNDC is therefore particularly valuable in terms of its ability to predict how subtle changes in the timing and form of N fertiliser might influence a C footprint for an individual farm, and how that system might respond to predicted climate change. An improved understanding of these processes is vital to inform IPCC and national government guidelines. In contrast, the CPLAN approach does not operate at the process level but provides higher level approximations which can be tuned to specific country or regional situations and is easily modified via the emission coefficients as new scientific data becomes available.

ACKNOWLEDGEMENTS

We are grateful to Mr A Adams for the provision of farm data and to QMS for funding SAC staff to collate the farm data and produce the DNDC model output reported here. The LEADER+ programme of South Lanarkshire is gratefully acknowledged for providing funding to test the feasibility of the free web based calculator for farmers (www.cplan.org.uk).

REFERENCES

Anon (2007a). Framework for New Zealand Emission Trading Scheme. Ministry for the Environment and Treasury, 147.

Anon (2007b). Part of the solution: climate change, agriculture and land management. Seacourt, 43.

Ball BC, Rees RM and Sinclair AH (2008). Mitigation of nitrous oxide and methane emissions from agricultural soils - a summary of Scottish experience. In: Proceedings of the SAC/SEPA Conference.

Casey J and Holden NM (2005). Analysis of greenhouse gas emissions from the average Irish milk production system. Agricultural Systems 86, 97-114.

Hindrichsen IK, Wettstein, HR, Machmuller A, Jorg B. and Kreuzer M. (2005). Effect of the carbohydrate composition of feed concentratates on methane emission from dairy cows and their slurry. Environmental Monitoring and Assessment 107, 329-350.

IPCC (2006). *2006* IPCC Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories Programme, Eggleston HS, Buendia L, Miwa K, Ngara T and Tanabe K (eds). Published: IGES, Japan.

Jones SK, Rees RM, Kosmas D, Ball BC and Skiba UM (2006). Carbon sequestration in a temperate grassland; management and climatic controls. Soil Use and Management 22, 132-142.

Jones SK, Rees RM, Skiba UM and Ball BC (2005). Greenhouse gas emissions from a managed grassland. Global and Planetary Change 47, 201-211.

Jones SK, Rees RM, Skiba UM and Ball BC (2007). Influence of organic and mineral N fertiliser on N_2O fluxes from a temperate grassland. Agriculture Ecosystems and Environment 121, 74-83.

Kilsby C, Jones P, Burton A, Ford A, Fowler H, Harpham C, James P, Smith A and Wilby R (2007). A daily weather generator for use in climate change studies. Environmental Modelling and Software 22, 1705-1719.

Li CS, Farahbakhshazad N, Jaynes DB, Dinnes DL, Salas W and McLaughlin D (2006). Modeling nitrate leaching with a biogeochemical model modified based on observations in a row-crop field in Iowa. Ecological Modelling 196, 116-130.

Li CS, Frolking S and Frolkin TA (1992). A model of nitrous-oxide evolution from soil driven by rainfall events.1. model structure and sensitivity. Journal of Geophysical Research-Atmospheres 97, 9759-9776.

Misselbrook TH, Chadwick DR, Pain BF and Headon DM (1998). Dietary manipulation as a means of decreasing N losses and methane emissions and improving herbage N uptake following application of pig slurry to grassland. Journal of Agricultural Science 130, 183-191.

Saggar S, Andrew RM, Tate KR, Hedley CB, Rodda NJ and Townsend JA (2004). Modelling nitrous oxide emissions from dairy-grazed pastures. Nutrient Cycling in Agroecosystems 68, 243-255.

Smith KA, McTaggart IP, Dobbie KE and Conen F (1998). Emissions of N_2O from Scottish agricultural soils, as a function of fertilizer N. Nutrient Cycling in Agroecosystems 52, 123-130.

Soussana JF, Allard V, Pilegaard K, Ambus P, Amman C, Campbell C, Ceschia E, Clifton-Brown J, Czobel S, Domingues R, Flechard C, Fuhrer J, Hensen A, Horvath L, Jones M, Kasper G, Martin C, Nagy Z, Neftel A, Raschi A, Baronti S, Rees RM, Skiba U, Stefani P, Manca G, Sutton M, Tubaf Z and Valentini R (2007). Full accounting of the greenhouse gas (CO_2, N_2O, CH_4) budget of nine European grassland sites. Agriculture Ecosystems and Environment 121, 121-134.

USING WOODLAND FOR SOIL PROTECTION AND SEDIMENT CONTROL

TR Nisbet[1], H Orr[2] and SB Broadmeadow[1]

[1]Forest Research, Alice Holt Lodge, Farnham, Surrey, GU10 4LH, UK, E-mail: tom.nisbet@forestry.gsi.gov.uk; [2]Geography Department, Lancaster University, Lancaster, LA1 4YQ, UK

SUMMARY

Woodland planting offers an effective measure for reducing soil erosion and sediment delivery to watercourses by providing physical shelter, improving soil strength and stability, increasing rainwater infiltration, and reducing water run-off. A case study was undertaken in 2004 in the catchment of Bassenthwaite Lake in northwest England to explore how woodland could aid sediment control. This involved a catchment audit to identify sediment sources followed by an evaluation of opportunities for woodland planting. A range of woodland options were considered, some directed at protecting sediment sources, while others focused on interrupting sediment transport to rivers or fixing deposited material. The Forestry Commission is now working with the Bassenthwaite Partnership to realise these opportunities through better targeting of grants and advisory services. There is significant scope for using woodland to help reduce diffuse sediment pollution elsewhere in the UK as part of a whole-catchment approach to sustainable water management.

INTRODUCTION

Forestry as a land use in the UK is generally viewed as posing a potential threat to the freshwater environment. This stems from the adverse impacts on water quality and quantity associated with the large-scale expansion of upland conifer forests during the second half of the 20th century (Farmer and Nisbet, 2004). A particular issue was the soil disturbance, erosion and siltation that could result from ploughing, drainage, road construction and harvesting operations. Several cases arose during the 1980's where forestry was responsible for polluting streams with sediment and disrupting water supplies (Nisbet, 2001).

The Forestry Commission introduced their Forests and Water Guidelines in 1988 to address these problems through improved management practice. Since their introduction, there have been three revisions to ensure that the guidance continues to reflect the most recent research and experience (Forestry Commission, 2003). Forestry is now recognised by many as leading the way in developing best planning and management practice to protect good water status. The success of the guidelines in addressing forest-water issues is gradually resulting in forestry being viewed as less of a threat and more as a potential benefit for water quality.

The benefits are considered to be greatest for native woodland due to its benign management (Calder *et al.,* 2008). It is widely acknowledged that soils under woodland are generally well protected and are often improved. Woodland has been shown to benefit sediment control by providing physical shelter from the wind, improving soil strength and stability, increasing the entry of rainwater into the soil, and by reducing water run-off.

Sediment delivery to watercourses is a major diffuse pollution pressure placing a large number of water bodies at risk of failing to achieve good water status. This has led to a major drive to identify and address the main causes and sources of sediment pollution. It is increasingly recognised that solving the sediment problem will require integrated action to improve overall land use planning and management. Woodland is one land use option that has the potential to reduce soil erosion at source, to limit the delivery of sediment to watercourses, to protect river banks from erosion, and to encourage sediment deposition within the floodplain. It therefore presents an effective means of tackling diffuse sediment pollution, in addition to providing a wide range of other environmental, social and economic benefits.

A case study was undertaken in 2004 in the catchment of Bassenthwaite Lake in northwest England to identify opportunities for woodland planting to reduce soil erosion and sediment pressures. Bassenthwaite Lake lies in the Lake District National Park and is designated a National Nature Reserve (NNR), Special Site of Scientific Interest (SSSI) and candidate Special Area of Conservation (cSAC). Its conservation status is at risk from a high level of soil erosion in the catchment caused by a range of pressures, including overgrazing, land cultivation, drainage, and human trampling. This paper describes the development of a catchment-based approach involving a sediment audit to identify sediment sources followed by an evaluation of where new woodland could best aid sediment control.

IDENTIFICATION OF SEDIMENT SOURCES

The main sources of sediment within the catchment of Bassenthwaite Lake were identified using digital aerial photography and a fluvial sediment audit. Recent digital aerial photography in 1 km squares was available for the whole catchment (35,534 ha) showing the occurrence of erosion scars, scree slopes and sediment deposits. A polygon was drawn around each patch of bare ground and the percentage of exposed soil or rock estimated. A distinction was made between bare rock such as exposed crags that would normally be devoid of vegetation and bare soil where the vegetation appeared to have been lost due to erosion. Polygons occupying a total area of 3168 ha (8.9% of catchment) were identified as containing 1252 ha (3.5 % of catchment) of actual bare ground. Most of this was concentrated in the uplands, with 8.2% of the land above 450 m estimated to be bare. The main causes of erosion were considered to be overgrazing and human trampling. Notably, <1% of the recorded bare ground was associated with woodland, demonstrating the benefit of woodland for soil protection.

A fluvial sediment audit recorded the length of the designated main river channels and a number of minor watercourses that were subject to erosion or bank collapse. A total channel length of 110 km was surveyed, representing 17% of the whole open channel network. Details of the approach are described in Orr *et al.* (2004). Some 20.7 km of stream/river channel exhibited evidence of significant erosion on one or both banks. A combination of lack of channel and bank maintenance, together with livestock trampling, overgrazing and subsequent loss of protective vegetation, were thought to be the main factors contributing to the observed high levels of bank erosion.

The next stage was to relate the observed sources of erosion to soil type as a way of identifying areas at greatest risk of future erosion. Soil information was obtained from the National Soil Map at a scale of 1:250,000, which combines soil series into distinctive map units (soil associations). There were 15 soil associations in the catchment, ranging from humic rankers and peats on the hill tops to a mixture of fine loamy, silty and permeable soils in the valley bottoms. Some 66% of the bare ground was concentrated within two soil associations (311b and 311e: humic rankers), while most of the remainder (32%) was spread between another three (611a and 611c: typical brown podzolic soils; and 1011b: peat). The soil associations are ranked according to the percentage of each covered by bare ground in Figure 1 and this was used to assign soils to high (>5% bare ground), medium (1-5% bare) and low (<1% bare) risk classes.

A total of 23% of the catchment, mainly comprising the hill tops and upper slopes, was covered by soils in the high risk class. Information on the Hydrology of Soil Types (HOST) showed that these soils were subject to prolonged waterlogging and at an extreme risk of structural damage by poaching from livestock (Boorman *et al*., 1995; Harrod, 1998), resulting in a high risk of eroded soil moving to streams. A further 47% of the catchment had soils that are at a medium risk of erosion and poaching.

A similar approach was applied to the bank erosion data, the distribution of which also appeared to be linked to certain soil associations. The soil associations are ranked according to the percentage of the total length of river bank traversing each exhibiting significant bank or channel erosion in Figure 1. Once again, it was possible to identify three distinct soil groupings, which were classed as having high (>6%), medium (1-6%) and low (<1%) risk of bank erosion. A total length of 61 km and 138 km of stream/river in the catchment were considered to be at high (comprising soil associations 541u: typical brown earths; and 811b: typical alluvial gleys) and medium (1011a: peat; 713f: cambic stagnogleys; and 721c: cambic stagnohumic gleys) risk of damage, respectively. The majority of the high risk soil types were also classed as having a very high risk of structural damage by poaching.

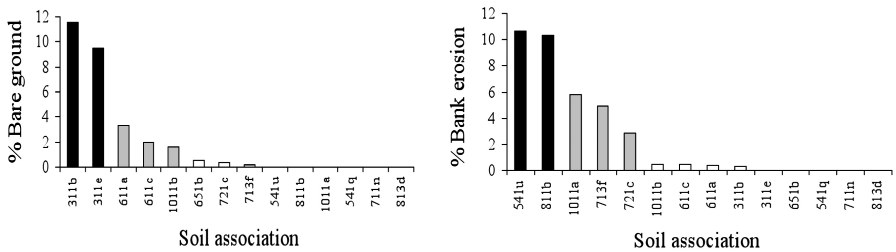


Figure 1: Relationship between observed bare ground, bank erosion and soil association

WOODLAND OPTIONS FOR SEDIMENT CONTROL

Figure 2 displays a number of ways that woodland can be used in the landscape to help reduce sediment delivery to watercourses. Some of these are directed at protecting sediment sources, while others are aimed at interrupting the transport of sediment to rivers (e.g. by planting woodland on downslope field boundaries, runoff source areas, infiltration basins, wetlands and riparian zones) or fixing deposited material. The main opportunities within the Bassenthwaite Lake catchment were considered to be: large-scale woodland planting on soils classed as having a high or medium vulnerability to erosion; targeted planting of woodland on and immediately around areas of bare ground (particularly downslope to retain mobilised sediment); targeted planting of riparian woodland along river reaches with a medium or high risk of bank erosion; and medium-scale planting and restoration of floodplain woodland around the confluence of major tributaries and the main inflows into Bassenthwaite Lake.

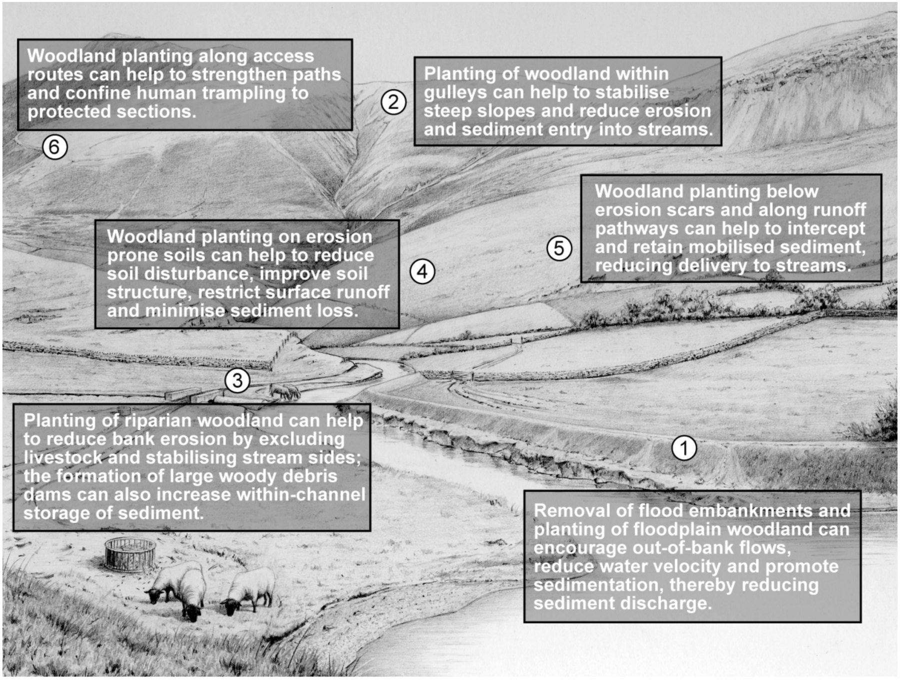


Figure 2: Different ways that woodland can aid sediment control

CONSTRAINTS ON WOODLAND PLANTING

Although woodland planting offers an effective measure for reducing sediment delivery to watercourses, there are many constraints to land use change. This was especially the case in the Bassenthwaite Lake catchment, all of which lay within the Lake District National Park. The National Park places restrictions on the location, scale and type of new woodland in order to preserve the special character of the landscape. Another key constraint was the designation of a large part of the catchment as a candidate Special Area of Conservation (Lake District High Fells). This reflected the high ecological value of the predominantly open upland grassland and heath vegetation and encompassed all of the high fells. Some 66% of the area classed as being at high risk of soil erosion lay within this designation. Nevertheless, although large scale woodland planting was unlikely to be permitted on the open fells, there could be scope for localised planting, such as in the steep upland valleys or around areas of eroding ground.

High ground represented another significant constraint, with 29% of the catchment occurring above the natural tree line (>450 m), although this line may rise in the future due to climate change. Other important constraints included SSSI's, Common or unenclosed land, sites of special archaeological interest, and existing woodland cover. Only 12% of the catchment was under woodland, comprising <1% of the land in the high erosion risk class and a relatively small proportion (12%) of the river length identified as at high risk of bank erosion.

IDENTIFYING OPPORTUNITIES FOR NEW WOODLAND TO AID SEDIMENT CONTROL

The final stage was to overlay the erosion vulnerability and constraint data sets to reveal the areas suitable for using woodland to aid sediment control. Opportunities for woodland planting are presented on a map of the Bassenthwaite Lake catchment in Figure 3. Unfortunately, most of the highly erodible land was affected by conservation or landscape designations resulting in only a small area (95 ha) being available for land use change. However, although these designations rule out the development of high forest, there may still be a role for a dwarf tree cover such as juniper to provide soil protection.

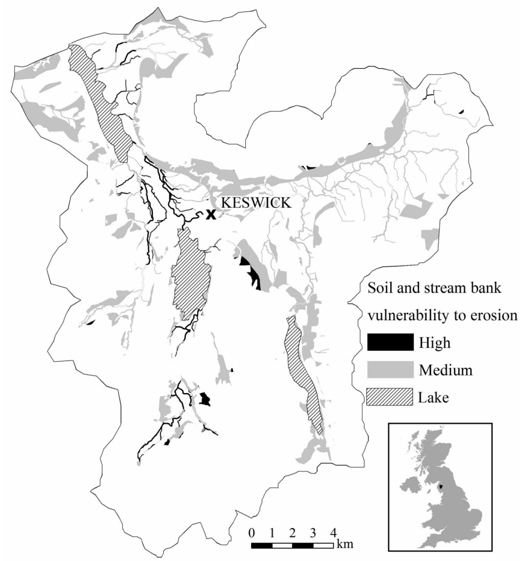


Figure 3: Opportunities for woodland planting for sediment control in Bassenthwaite Lake catchment

There was more scope for using woodland to help reduce river bank erosion with 37 km of river length in the high risk class suitable for planting. Assuming that buffer zones of 10, 20 and 40 m width were appropriate for protecting the banks of first, second and third order stream channels, respectively, this equated to a total area of 223 ha. Riparian planting would have the added advantage of contributing to the Habitat Action Plan target for wet woodland.

A much larger area of ground (3184 ha) and length of river (89 km) were available in the medium vulnerability class, although the case for targeting these for woodland planting to aid sediment control would not be as strong as for the high vulnerability land. Nevertheless, soil protection would make an important contribution to the range of potential benefits provided by woodland and help strengthen the case for land use change. Significant core areas of woodland were already present in these areas, providing good potential for woodland extension to create forest habitat networks.

The Forestry Commission is now working with the Bassenthwaite Lake Partnership to realise these opportunities through better targeting of grants and advisory services. A guide 'Using Woodland for Sediment Control' has been produced which shows how integrated land use planning and management can help resolve sediment problems in at-risk water bodies (Nisbet *et al.,* 2004). The establishment of demonstration woodlands is being considered as a way of communicating the advantages for sediment control to local landowners. It is hoped that a programme of monitoring can be put in place to allow the benefits to be quantified. This information could then be used to justify increasing grant levels for new planting in the most effective locations.

CONCLUSIONS

The Bassenthwaite Lake case study demonstrates how woodland planting can be used to reduce diffuse sediment pollution as part of a whole-catchment approach to sustainable water management. Woodland provides a number of options for sediment control, including reducing soil erosion at source, limiting the delivery of sediment to watercourses, protecting river banks from erosion, and encouraging sediment deposition within the floodplain. A guide 'Using Woodland for Sediment Control' describes how targeted planting of woodland can help tackle sediment problems. Successful implementation of woodland measures will depend on improving the synergy between woodland and agricultural grants, as well as advisory services.

ACKNOWLEDGEMENTS

We acknowledge the support of the Forestry Commission in funding this work and of the Environment Agency for supplying relevant data.

REFERENCES

Boorman DB, Hollis JM and Lilly A (1995). Hydrology of soil types: a hydrologically based classification of UK soils. Institute of Hydrology Report No.126. IOH, Wallingford.

Calder IR, Harrison J, Nisbet TR and Smithers RJ (2008). The role of native woodland in water management. Woodland Trust, Grantham (in press).

Farmer RA and Nisbet TR (2004). An overview of forest management and change with respect to environmental protection. Hydrology and Earth System Sciences 8 (3), 279-285.

Forestry Commission (2003). Forests and Water Guidelines (4th Edition). Forestry Commission, Edinburgh.

Harrod TR (1998). A systematic approach to national budgets of phosphorus loss through soil erosion and surface run-off at National Soil Inventory (NSI) nodes. Final Report to MAFF. MAFF Project NT1014, SSLRC Project JF3818.

Nisbet TR (2001). The role of forest management in controlling diffuse pollution in UK forestry. Forest Ecology and Management 143, 215-226.

Nisbet TR, Orr H and Broadmeadow S (2004). A Guide to Using Woodland for Sediment Control. Forest Research, Farnham, Surrey.

Orr HG, Davies G, Quinton J and Newson MD (2004). Bassenthwaite Lake Geomorphological Assessment: Phase 2. Environment Agency, Penrith.

LAND MANAGEMENT AND ACHIEVING GOOD WATER QUALITY

CJ Spray

Scottish Environment Protection Agency, Corporate Office, Erskine Court, Castle Business Park, Stirling, FK9 4TR, UK, E-mail Chris.Spray@sepa.org.uk

INTRODUCTION

Land management and water quality are inextricably linked. This relationship needs to be recognised and understood if water resources are to be adequately protected and, where necessary improved in quality. The nature of the relationships, together with management options to achieve water quality benefits is a complex one, involving variations in geography, scale and time, but clearly offers opportunities for achievement of multiple environmental and socio-economic objectives. The challenge will be to identify, quantify and deliver these benefits on the ground.

THE IMPACTS OF LAND MANAGEMENT ON WATER QUALITY

In broad terms, the Scottish water environment is of good, indeed often excellent quality across much of the country, with many kilometres of good quality rivers, streams and coastal waters, as well as lochs and groundwater (SEPA, 2006, State of the Environment Report). But there are still problems of course. Table 1, from the Significant Water Management Issues Report (SWMI) for the Scottish River Basin District (SEPA, 2007), summarises the length/area and numbers of water bodies at risk of failing good status in 2007 (The Water Framework Directive (WFD) defines good ecological status as being the desired state individual water bodies should meet to achieve the requirements of the Directive through river basin management plans).

Table 1: Summary of length/area and numbers of water bodies at risk of failing good status in 2007 (SEPA, 2007)

Water category	Length/area at risk of failing good status in 2007 (% of total)	Total length/area of all water bodies	Number of water bodies at risk of failing good status in 2007 (% of total)	Total number of water bodies
River	9,083 km (44%)	20,819 km	828 (41%)	2,008
Loch	633 km^2 (66%)	961 km^2	162 (52%)	309
Transitional	425 km^2 (70%)	605 km^2	21 (53%)	40
Coastal	3,025 km^2 (6.6%)	45,796 km^2	53 (12%)	449
Groundwater	20,805 km^2 (31%)	66,567 km^2	142 (52%)	275
Total	-	-	1,206 (39%)	3,081

(More information about the classification categories and numbers involved is on the SEPA website: www.sepa.org.uk/wfd/rbmp)

What are the underlying causes of that at risk status? It is important to note that many kilometres of rivers for example are at risk of failing to meet good quality due to modifications to the physical or hydrological regime of the river, rather than water quality (pollution) issues. But often issues are linked, with intensively managed landscapes, such as towns and productive farmland, often having heavily or at least somewhat modified river channels, and sometimes abstraction issues too. For the Water Framework Directive, pollution issues are considered in two categories:

- Diffuse sources; encompassing the multiplicity of non-point and minor point sources, such as steading and urban drainage, field drains and surface runoff, and infiltration from intensively managed rural and urban landscapes into groundwater. Pollution occurs most readily when rainfall mobilises dispersed pollutants.
- Point sources; individual sources or locations, such as outfall pipes discharging sewage or industrial/commercial process effluents. These are classically directly regulated by SEPA and individually monitored and controlled.

Table 2 sets out the significant diffuse and point source pollution issues in the Scotland River Basin District, (comparable information for the adjacent Solway/ Tweed river basin district is given in SEPA, 2007b). Overall, for 2007 data, 40% of Scotland's waters are at risk of failing "good status". Approximately half of the water bodies at risk of failing are affected by agricultural diffuse pollution, which is now the single most widespread cause of poor water quality for groundwater, lochs and rivers in Scotland. Although sewage effluents remain the most significant problem in transitional and coastal waters, it is only second for rivers, behind diffuse agriculture, and the third and fourth most extensive sources for rivers are also diffuse source problems: from forestry and urban drainage.

Table 2: Polluting sectors implicated in risks of Scotland's waters failing "good status" in 2007

Water category	River	Loch	Transitional	Coastal	Groundwater
Diffuse pollution					
Number of water bodies at risk of failing good status in 2007 (% of total)	828 (41%)	162 (52%)	21 (53%)	53 (12%)	142 (52%)
Agriculture	**313**	**27**	10	16	**129**
Forestry	**53**	**21**	-	1	-
Urban development	**88**	2	4	2	-
Sea and coastal water transport	-	-	7	17	-
Point source pollution					
Collection and treatment of sewage	**230**	15	14	34	-
Refuse disposal	16	-	3	2	14
Aquaculture	15	23	-	3	-
Manufacturing	32	1	8	14	7
Mining and quarrying	36	-	-	-	14

LAND MANAGEMENT AND QUALITY

Before considering these issues, what is the state of the land environment of Scotland, since land quality is so strongly related to water quality? Scotland has a land area of some 78,000 km^2 and a coastline of approximately 10,000 km, with about 100 inhabited islands. The landscape comprises mountains, glens, moorland, forestry plantations and farmland, as well as urban settlements. The biodiversity of the Scottish landscape is an indication of ecological status, and much of the rural landscape is of high environmental quality; over 20% of the land area is protected by a variety of natural heritage designations (see www.snh.org.uk). Urban and industrial areas, whilst only a small proportion of Scotland's land area of course, are where most of the population live and work, and include some problem areas where contaminated land is an issue SEPA (2006) reported that land quality in Scotland is generally considered to be reasonable, but noted significant gaps in knowledge, especially in relation to soils.

Finally, in considering the state of the Scottish land and water environment, it is important to note that these qualities have tremendous economic and social importance for the country. Recreational uses of the landscape have a long history in Scotland and contribute significantly to the Scottish economy. Many of these recreational uses shape the landscape, which in turn determines water quality.

The clearest relation between land management and water quality is evident in the impact of diffuse sources of pollution, noted above and summarised in Table 2. Some pollution is caused by bad practice - for example the misapplication of slurry on steeply sloping land draining to a watercourse or the application of some pesticides within the 6m prohibition buffer zone specified for them by the pesticide approval scheme for the UK. Codes of good agricultural practice were designed to advise land managers how to avoid pollution in such ways. However it would be wrong to suggest that bad practice is restricted solely to agricultural operations; forestry and the urban environments receive their (un) fair share as well.

Whilst such diffuse pollution hotspots do occur, diffuse pollution is often simply contamination of water that is incidental to normal activities; a sort of anthropogenic level on top of natural processes. The relationship with natural processes is illustrated by the way pollutant concentration follows river level (and hence is associated with rainfall) as shown in Figure 1 for the Cessnock (a small river in Ayrshire that drains a catchment that is influenced by livestock farming). At no point are the high flow concentrations of TP or TSS in that example lower than those recorded during low flow conditions.

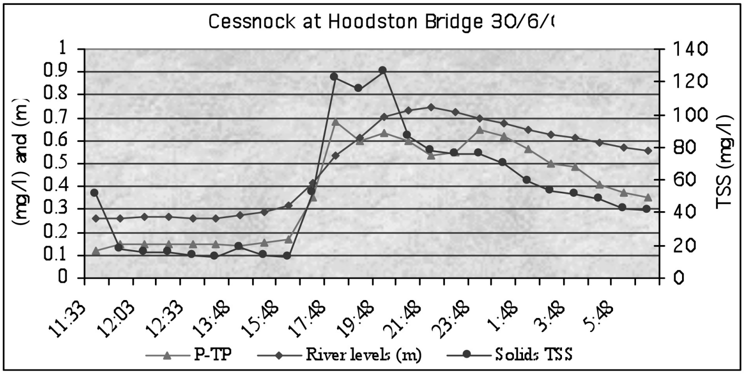


Figure 1: Example relationship between river levels, TP and TSS (30/6/02)

Rainfall can thus mobilise contaminants and different land uses expose contaminants to varying extents; dependant on the nature of the land-use, as well as the fundamental nature of the soils and topography. The underlying local physico-chemical features, which partly determine the type of farming practice, are an important factor in determining impacts on a catchment scale, rather than simply the good or poor practice of individual farmers. This is a principal challenge for managing diffuse sources. Table 3 (below), shows that a simple decision to grow potatoes in the study catchment was estimated to be likely to result in eight times as much potential P loss to the water environment through soil erosion than by keeping the land in permanent pasture (D'Arcy and Frost, 2001).

Table 3: Loss coefficients from studies of arable fields in Windrush catchment Jones PJ 1996 J of Hydrology 183, 323-349 (in Campbell *et al*., 2004)

Permanent grass	0.10 kg/ha/yr P	5% annual N application
Autumn sown cereals	0.65 kg/ha/yr P	12% annual N application
Potatoes	0.80 kg/ha/yr P	20% annual N application
Brassicas	0.65 kg/ha/yr P	20% annual N application
Oil seed rape	0.65 kg/ha/yr P	30% annual N application

It is evident that specific land-uses have characteristic consequences for the local water environment, as well as distant water bodies in some instances, such as bathing waters. They may also result in long term cumulative impacts in relation to groundwater or lochs. There needs to be specific measures that can be put in place to mitigate those impacts. Such measures are called Best Management Practices, or BMPs. The BMPs concept is discussed at length in Campbell *et al.* (2004), as well as being noted in earlier SAC/SEPA conference papers.

SEPA has developed a national screening tool to identify in some detail where diffuse sources of pollution are likely to be significant, as an aid to managing the risks. More detailed modelling tools are under development to be applied in particular problem catchments to allow a focus at field scale on problems and targeting mitigation measures.

MONITORED PRIORITY CATCHMENTS – PILOT FOR DIFFUSE POLLUTION MITIGATION

SEPA has established a project with partner organisations including the Scottish Agricultural College, The Macaulay Institute and the Centre for Ecology and Hydrology to assess the effectiveness of measures to mitigate diffuse pollution in catchments representative of typical land uses in Scotland. These include representative catchments for the various categories of diffuse pollution activities, for example livestock farming in the east, and a couple of urban sites. That programme of work (Monitored Priority Catchments), aims to quantitatively characterise the diffuse pollution loads and risks in the selected catchments, establishing baseline quality for surface waters and groundwater, and then beginning a local programme of roll-out BMP's. Initial efforts for this relatively new initiative have been focussed in the Lunan Water Catchment.

Regulating these disparate and variable sources is a requirement of the Water Framework Directive, but presents new challenges (see below).

A CHANGING ENVIRONMENT

Post war intensification of agriculture driven by meeting the demands for food led to well documented increases in environmental pollution - an increase in nitrate concentrations in rivers and aquifers, eutrophication of lochs and reservoirs, and problems of poor bathing water compliance with EU standards associated with livestock farming. The intensification of farming was also characterised by major modifications to hydrology of the farmland (field drainage) and by substantial changes to river morphology (straightening and loss of flood lands, culverting smaller tributaries, for example). Biodiversity trends have shown a similar impact, with dramatic declines in farmland birds. However, evidence is now indicating a change in some areas and there have been indications of improvements, such as the levelling off of nutrient concentrations in UK. Careful land management modifications are beginning to show that environmental quality can be improved alongside efficient production.

The biggest changes in forestry for many years are now becoming evident, as the majority of the conifer plantations established by the Forestry Commission are now mature and the Scottish Forestry Strategy aims for 25% forest cover. Both felling and planting are high risk activities in terms of losses of diffuse pollutants that, where located in very sensitive catchments can have adverse impacts on nutrient sensitive lochs, salmonid fisheries and pearl mussel rivers for example.

The biggest environmental change – affecting all sectors – is of course climate change. Not least of its impacts are changes in the water environment. Scotland has become much wetter since 1961, with an increase in average winter precipitation of almost 60% in the north and west, and an increase in average annual precipitation of 20% for the whole country. The greater intensity of rainfall means more mobilisation of contaminants from the land, for example soil erosion is likely to be an increasingly serious issue, contributing suspended solids to silt up waterbodies, as well as associated nutrients and other pollutants (SEPA, 2006). More intensive rainfall leads to greater risks of flooding too, with implications for land management.

Economic trends also have implications for land management and hence the quality of the water environment in rural areas. For many years the good quality of the Scottish environment has been a factor in drawing tourists to the country, and water is an iconic aspect of that attraction. SEPA (2006) has estimated that the value to the Scottish economy of ecosystem services in Scotland is worth some £17 billion; equivalent to 27% of gross domestic product (GDP) in 1998. Economic pressures on the farming and forestry sectors in recent decades, as well as policy initiatives by central and regional government, have favoured a diversification of these sectors to embrace tourism by developing amenity and wildlife interests in the rural environment. The emergence of the ecotourism concept offers a means to ensure that such business development does not jeopardise water quality or other environmental considerations.

The Climate change agenda has necessitated a greater focus on energy production and conservation with implications for the rural sector (see Table 4 below). As well as the need for the sector to address efficiency and generation/provision of energy needs itself, the greater national and international requirements for cleaner energy is already having effects on the Scottish landscape, including implications for water quality to varying extents. Wind farms for example have possible water quality issues locally associated with access roads and construction, whilst willow and other cash crops for renewable fuels often need less fertiliser and pesticides, and can form useful landscape measures to stabilise erodable soils at risk of erosion in hitherto arable land. Biofuels such as sugar beet and oil seed rape by contrast may be as intensive in production and agrochemicals needs as any food produce, with implications for the water environment. Farm slurry is an energy resource with only very limited exploitation to date (by contrast with sewage sludge which has decades of off the shelf technology behind it), and it is generated by still producing food crops and livestock.

Table 4: The changing environment for land management: issues and trends

Sector	Change perceived	Water quality implications	Socio- Economic prospects
Forestry	Harvesting and replanting phase of forest development prevalent.	Risk to sensitive water bodies and habitats from nutrients and SS	Employment stabilised or increased, recreational opportunities
Dairy farming	Intensification and larger units expected although numbers of livestock declining	Lack of funds for capital investment jeopardises options to manage pollution risks: FIOs, nutrients, ammonia, BOD	Need radical developments to use resources more profitably?
Arable farming	Intensification and larger units likely. Increase in cereal and fertiliser prices – continue?	Further levelling off of nutrient and pesticide concentrations in aquatic environment	Arable farming still viable and sustainable.
	Increase in matching fertiliser input to soil nutrient status and crop offtake. A profitable sector.		

Table 4 continued

Hill farming	Decline likely to continue. Livestock numbers declining.	SS risks and nutrients Negative impacts on biodiversity	Additional sources of income needed?
Rural Recreational land-uses	Walking and wildlife watching Golf courses? Increase?	Potential for increased sewage pollution, SS associated with path erosion, and localised traffic pollution (oil, metals, PAHs)	Greater numbers of people using the rural landscape, economic support for rural sector economy. Impact on GHG?
Urbanisation	Steading conversions, green belt encroachment	Lower nutrient pollution, higher oil and toxic metals	Significant income generation for a few farmers
Energy	Wind farms (but mainly uplands), biofuel crops, energy from organic waste?	Positive if don't need agrochemicals (e.g. wind farms, willow) but neutral if grain or root crop biofuels.	Stabilise or increase employment and rural economy
Flooding	Land-use for flood storage in rural areas above towns and cities.	Less intensive farming (if convert arable to pasture) should reduce pollution risks.	Habitat creation should result; could bring eco-tourism benefits.
Climate change	The driver for last two points, plus need to address greenhouse gas emissions from livestock, and from CFWs; pollutant trade offs	?	?

Table 4 summarises a number of changes in the rural landscape of which all have implications for the water environment. Some of those links are examined below.

INTER-RELATIONSHIPS ACROSS ISSUES

The report of a government advisory group, set up to look at the future for the rural environment (Agriculture and Environment Working Group, 2002) concluded that Agriculture and the environment are inextricably linked. The report concluded then that the three priority environmental issues for Scottish agriculture for the next 5-10 years were:

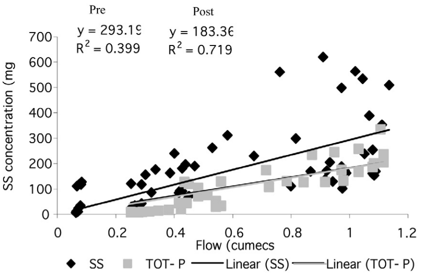
1. Diffuse pollution to water.
2. Biodiversity and habitat protection.
3. landscape change.

To this list, one might now add:

- climate change – as the overriding environmental challenge;
- a growing interest in sustainable flood management; and
- ecosystem services and ecological networks.

Delivery measures to tackle these issues singularly, l*et al*one the multiple costs and benefits continue to be a challenge.

Back in 2002 Buffer strips were recognised in the report of being landscape measures with the potential to address all three of those early priorities How effective are they in practice? A SEPA diffuse pollution monitoring station on the Greens Burn tributary of Loch Leven, has provided data to illustrate the pollution reduction that has been measured there (figure 2, from S Greig, DPI no.21) The storm event data captured at the Greens Burn field site provided flow related responses in pollutant concentration. The results of this analysis highlighted a slope difference of -37% and -42% for suspended solids and phosphorous respectively, suggesting a consistent decline in inputs of these pollutants post buffer strip generation.



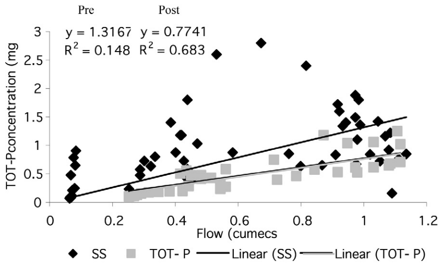


Figure 2: Pre and Post establishment of riparian buffer strip linear regression analysis pollutant concentration and flow. Pre-buffer strips. Post-buffer strips (from Greig *et al*., 2004).

No specific measures to enhance biodiversity value were used when those buffer strips were established, but preliminary comparisons of beetles present in the grass strips as compared with a particular stage in the growth of the adjacent crop, suggested some modest increase in biodiversity. On a wider scale though, the linkage between wetland environments, water quality and wildlife is one of increasing importance, both at the individual farm and catchment scale. More theoretical interests in ecological networks and ecosystem services now need to be converted into information and action on the ground.

Our future approach to flood risk management in Scotland may provide opportunities for land use management change that could have water quality and biodiversity benefits. This idea was championed at the RSPB/Water UK Conference 2002, and is endorsed by various current initiatives in Scotland. There may be some potential in Scotland for land use management techniques to reduce flood risk. However, the relationship between flood risk and land management has not been well demonstrated in a Scottish context. In particular further research is required to demonstrate the influence that land use management can have at the catchment scale for the flood magnitudes that have the greatest impact on those at risk. The need for detailed empirical studies with appropriately measured and monitored pilot catchments is urgent.

The EU Flooding Directive requires member states to assess flood risks and hazards and to map these where a significant risk is identified. Catchment flood management plans must be prepared outlining the measures required to mitigate the impact of flooding. Plans must be integrated as much as possible with the existing river basin management planning process. In Scotland the requirements of the EU Flooding Directive will be transposed through the Scottish Flooding Bill expected before Parliament in June 2008.

And whilst inundation may be one of the most obvious impacts and immediate threats from climate change, another key issue for Scotland is the loss of carbon from upland peat soils. Scotland's soils contain an estimated 2196 million tonnes of soil carbon (48% UK total), and whilst land use and management have a major influence on soil organic matter content, there is increasing evidence of significant losses due to climate change. Long term monitoring is showing a significant increase in dissolved organic carbon levels in rivers, particularly in autumn. Drier summers and wetter winters are promoting oxidation and erosion which, along with overgrazing and muir burn are producing water quality issues further down the catchments.

Finally, eco-tourism as a means of economic diversification and building on a mainstay of the Scottish economy, can assist by putting more emphasis on environmental best practice and minimising environmental impacts. Clean water management systems, minimising, reusing, composting or recycling wastes, enhancing biodiversity at the site, and encouraging public transport/cycling /walking, without undue disturbance of wildlife and the natural environment – all of these will help improve the land management and water quality.

LAND MANAGEMENT AND IMPROVEMENTS IN WATER QUALITY

The mitigation of diffuse pollution requires a combination of regulatory, economic and voluntary mechanisms to deliver the on the ground changes to maintain and improve water quality. A number of recent developments in Scotland, most notably economic incentives through the Scottish Rural Development Programme and new regulations in the form of General Binding Rules, provide a major opportunity to address diffuse pressures at a national rather than project scale for the first time.

The proposed General Binding Rules cover the main activities posing a risk to water quality. They include:

- the storage and application of fertilisers;
- the keeping of livestock;
- the cultivation of land;
- the discharge of water;
- the construction and maintenance of roads;
- the handling and use of pesticides;
- the dipping of sheep.

The rules for each of these activities are based on widely accepted standards of good practice such as the Prevention of Environmental Pollution from Agricultural Activity (PEPFAA) Code, The 4 Point Plan and the Forests and Water Guidelines. The GBRs will provide a statutory baseline of good practice. These proposals are regarded as good examples of 'better regulation' in that they are risk-based and contact with the regulator is low and so costs and bureaucracy are minimal. Key to the success of the proposed measures will be their implementation. Although the activities covered by the GBRs individually represent a low risk to the water environment, cumulatively impacts are highly significant. The GBRs are intended to be a ‘light touch’ approach. However, in order to be effective this ‘light touch’ regulatory approach must be accompanied by effective implementation through, for example, awareness raising campaigns, advice given during routine advisory visits, workshops and road shows for land managers. The delivery of 1 to 1 advice on a catchment management plan type basis is an essential part of diffuse pollution mitigation.

An implementation strategy is urgently required, to link the regulatory, economic and voluntary strands in the context of River Basin Management Planning, signed up to by all partners to enable a clear and consistent message to be delivered to land managers.

It is important to note that successful implementation of the tools now available to mitigate diffuse pollution will have significant add-on benefits for other areas of work including compliance with the new Bathing Water Directive, remediation and restoration of rivers, soil protection and biodiversity.

Sector-led initiatives (e.g. voluntary initiative for pesticides) have had and continue to have an important part to play in improving the environment, especially in conjunction with national issues (often led by industrial/commercial trade associations) or increasingly as welcome inputs to local catchment initiatives. They will be particularly important for WFD implementation (programmes of measures, for example).

A major benefit of a move away from narrow regulation driven actions is the possibility of multiple land management objectives for best socio-economic and environmental outcomes. Partnerships, especially catchment plans, require that the aims and needs of a range of organisations and people have to be considered. The new Area Advisory Groups set up by SEPA for implementation of programmes of measures under WFD, provide an opportunity for stakeholders to develop mutual understanding and promote effective measures to address water quality issues.

The government driven SEARS initiative is aiming to ensure that all the tax-payer funded bodies that employ staff to visit or regulate farms, work together and agree consistent messages for the rural sector from them. More resources and measures, co-ordinated in a closer way than ever before, informed by better science including catchment scale modelling, should enable water quality to be restored, protected and improved by 2020 in the rural sector.

ACKNOWLEDGMENTS

I would like to acknowledge the assistance of Brian D'Arcy, Janette MacDonald, David Faichney and other colleagues in SEPA for their inputs to this paper. The opinions expressed in this paper are those of the author and do not necessarily reflect the view of the Scottish Environment Protection Agency.

REFERENCES

Agriculture and Environment Working Group (2002). Custodians of Change: Report of the Agriculture and Environment Working Group. Scottish Executive, Edinburgh.

Campbell NS, D'Arcy BJ, Frost CA, Novotny V and Sansom AL (2004). Diffuse Pollution – An Introduction to the Problems and the Solutions. IWAP, London. ISBN 1900222531

D'Arcy B J and Frost C A (2001). The role of best management practices in alleviating water quality problems associated with diffuse pollution. The Science of the Total Environment 265, 359-367.

Greig S, Dawson P and Craig W (2004). Trends in diffuse pollution: data report to assist with the design and implementation of effective diffuse pollution monitoring programmes. Unpublished technical report DPI No. 21/SG/Sept 04, Scottish Environment Protection Agency, Stirling.

Jones A, Dawson P, MacDonald J (2003). Interim Report on DPI Diffuse Pollution Monitoring Stations. Unpublished technical report DPI No. 13/A Jones/July 03, Scottish Environment Protection Agency, Stirling.

SEPA (2007a). An introduction to the significant water management issues in the Scotland river basin district. Scottish Environment protection Agency, Stirling.

SEPA (2007b). A summary of significant water management issues in the Solway Tweed river basin district. Scottish Environment protection Agency, Stirling.

SEPA (2006). State of Scotland's Environment 2006. Scottish Environment Protection Agency, Stirling.

THE BRIGHOUSE BAY PROJECT: BATHING WATER PROTECTION THROUGH SUSTAINABLE LAND USE MANAGEMENT

D Kay[1], W Jeffrey[2], JW Dickson[2], J Crowther [2], A Edwards[1], C Davies[3], C Francis[3], C Kay[1], AT McDonald[4], C Stapleton[1], J Watkins[3] and MD Wyer[1]

[1]Catchment and Coastal Research Centre, River Basin Dynamics and Hydrology Research Group, IGES, University of Wales, Aberystwyth, Ceredigion, SY23 3DB, UK, E-mail dave@crehkay.demon.co.uk; [2]SAC, Bush Estate, Penicuik, Midlothian, EH26 0PH, UK; [3]CREH Analytical, Hoyland House, Horsforth, Leeds, LS18 4RS, UK; [4]School of Geography, University of Leeds, Leeds, LS2 9JT, UK

SUMMARY

Brighouse Bay bathing waters have been impacted by diffuse agricultural pollution prompting series of land use interventions and environmental monitoring which have been conducted between 2003 and 2007. This presents one of the very rare catchment-scale case studies of pollution remediation for faecal indicator control. This is of direct relevance to the EU Water Framework Directive and the US Clean Water Act. Preliminary results were published in Environmental Pollution in 2007.

This paper reports the full monitoring data following new data acquisition in summer 2007 on the land management measures implemented and their impacts on sustainable pollutant attenuation. The authors explain the relevance of these data for regulatory community and suggest how these empirical results can strengthen the evidence-base for the design of appropriate farm-support mechanisms designed to ensure the maintenance of 'good' water quality as legally required in EU Member States.

BUILDING INTEGRATED NATURAL AND SOCIAL SCIENCE SOLUTIONS TO ASSESS THE RISK OF FIO LOSS FROM LAND TO WATER: A CROSS-DISCIPLINARY TOOLKIT

DM Oliver[1], RD Fish[2], CJ Hodgson[3], AL Heathwaite[1], DR Chadwick[3] and M Winter[2]

[1]Centre for Sustainable Water Management, Lancaster University, LA1 4AP, UK, E-mail: d.m.oliver@lancaster.ac.uk; [2]Centre for Rural Policy Research, Department of Politics, Exeter, Devon, EX4 4RJ, UK; [3]IGER, North Wyke Research Station, Okehampton, Devon, EX20 2SB, UK

SUMMARY

This paper contends that environmentally and socially appropriate management interventions arise from understanding the interplay of social and natural processes at the farm scale. It makes this argument in the context of conceptual and empirical research undertaken to develop an interdisciplinary risk assessment tool; one that can both represent factors promoting or preventing the accumulation of faecal indicator organisms (FIOs) within the farm environment, and also their subsequent transfer to watercourses. Through identification of governing controls on FIO loss from land to water it explains how we can devise the basis for different styles and forms of mitigation to reduce FIO delivery to watercourses. Such risk tools allow the policy community not only to target high risk areas, but also develop mitigation strategies that are sensitive to the different ways in which risk is produced. Capacity for long-term cross disciplinary research is argued to be the means by which these integrated and more sustainable solutions may emerge.

INTRODUCTION

Farming practices represent one of the most critical sources of diffuse water pollution in the UK (Defra, 2007). Microbial contamination of watercourses from agriculture is gaining increased recognition as a significant contributor to diffuse water pollution, and one that can impact not only on water quality but also human health (Kay *et al*., 2007). This is reinforced by the inclusion of daughter directives such as the Bathing Waters Directive 76/160/EEC within the Water Framework Directive 2000/60/EC and microbial Total Maximum Daily Load (TMDL) assessments of water quality in the U.S., under its Clean Water Act.

Approaches to farm risk assessment in the UK, for example, the Defra manure management plan; Defra soil management plan and SEPA risk assessment for manures and slurries (Defra, 2003; 4 Point Plan, 2004; Defra, 2005a) can provide decision support and guidance to farmers and land managers with respect to limiting contamination of surface waters from a range of contaminants. These strategies are linked to codes of good agricultural practice and environmental stewardship schemes (Defra, 2005b). They raise awareness of manure spreading strategies and other farm activities that could potentially cause pollution of watercourses. In devising risk assessment tools to accommodate microbial contaminants we need to understand that agriculturally-derived faecal indicator organisms (FIOs) can contaminate surface waters through (i) direct routes such as animal access and defecation into streams; and via (ii) rainfall-driven transfer from land to water through surface and subsurface hydrological pathways. Furthermore, once outside the animal gut, FIOs are not well adapted to survive in the farm environment and their numbers reduce over time as a function of environmental variables (Avery *et al.*, 2004; Oliver *et al*., 2006).

The insights we present in this paper form a working framework for a farm-scale relative risk FIO assessment tool that integrates natural and social science risk drivers. This forms a holistic strategy for identifying farm vulnerability of contributing FIOs to water. The project within which the framework is being developed focuses on the Taw catchment in North Devon, UK (Figure 1), and involves: a survey of 77 farmers across the catchment; eliciting attitudes and practices towards manure; land and livestock management and detailed microbial monitoring of water courses on ten of these farms. In refining this framework we aim to construct generic, scale dependant rules that are appropriate for targeting and managing land considered most vulnerable for contributing FIOs to water.

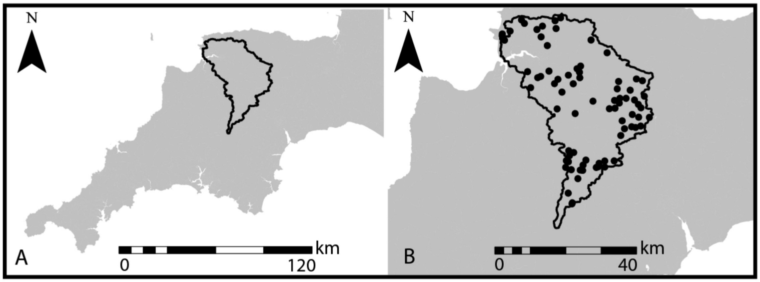


Figure 1: The Taw catchment set within the southwest peninsula, UK (A) and the distribution of farms participating within the study catchment (B)

WHOLE FARM SYSTEM RISK ASSESSMENT

To gauge the likelihood of farm enterprises having an impact on microbial water quality we require holistic frameworks of strategic risk assessment. As with other environmental risks, microbial contamination of water from agriculture is a product of economic, technical and natural processes interacting in distinctive and non-linear ways. There is a need to develop approaches to assess such risks that, while driven by the need for generic application, are context specific by design. Approaches to risk assessment that fail to embrace the social and natural complexions of these processes, nor attune themselves in ways that exemplify real world circumstances, may lead to pathways to environmental protection and enhancement that are disproportionate or inappropriate in conception.

Here we consider the parameters of a "whole farm system" assessment of the relationship between agricultural practices and microbial water course pollution, one in which different social and natural processes can be factored in, weighted, configured, observed, and ultimately, evaluated. Farm environments are diverse. The mechanisms which may inhibit or drive the development of sustainable agricultural systems reflect, in part, a complex political, cultural and financial economy to farming. These processes shape prevailing attitudes and dictate capacities to act upon environmental risk. But they do so in the context of heterogeneous physical landscapes and diverse configurations of infrastructure. Assigning significance to the factors that govern microbial watercourse risk is one version of this complex domain and demands models that are explicitly interdisciplinary in design. We have opted to use *E. coli* as our FIO of choice and to develop a general model based on its traits in the environment. This is because it is a well researched FIO and we can draw on an extensive body of published research in the development of our work.

Assessing Farm Characteristics

A detailed farm survey was conducted to determine key livestock, manure and land management practices across 77 farms in the River Taw catchment (see Figure 1). This also included more open-ended discussion around decision making processes and social science risk drivers linked to farm economics. When analysing the farm survey results, a number of social and natural scientists were enlisted to provide expert weightings with regard to the importance and direction of influence of a number of natural and social science risk drivers identified within the farm survey. These were subsequently accommodated within the risk assessment framework. Four key over-arching risk criteria were identified as primary indicators of FIO risk at the farm scale, within which the subset of associated risk drivers were accommodated. These were:

- *E. coli* burden to land
- Obstacles to [farmers] taking action
- FIO transfer potential
- Farm infrastructure.

Briefly, accumulating *E. coli* burden to land accounts for the fundamental measure of *E. coli* applied and deposited (via faeces or manures) to land per hectare for a given time period. Livestock numbers and manure spreading activity form key information so as to generate a dynamic *E. coli* burden through time. Obstacles to taking action are criteria designed to recognize how risks are shaped by the *capacities*, *responsibilities* and *knowledges* of farmers to manage for them. Transfer factors allow us to appreciate that poor management of a high risk source can be buffered by the landscapes ability to act as a ‘safety net’ in preventing or limiting FIO movement in the farm environment from occurring and *vice versa*. Finally, infrastructure is understood to be the physical capital of the farm landscape and how it is configured and managed. Infrastructure reflects long-term financial investment, but also signifies the cultural practices of farming systems that produce and work within it. Importantly, the framework assigns equal importance to these risk criteria. In doing so it is worth noting from the outset that the approach avoids seeking to determine whether one of these criteria should be given elevated importance over another in making an overall judgment of risk. There is currently no integrated scientific and social scientific evidence base for microbial risk that has allowed us to make discriminations of this kind, and indeed, the sense in which factors are treated even-handedly reflects the prototypical nature of this work. Testing of this model against real world circumstances will validate or allow iterative reworking of this initial proposition.

The comprehensive dataset for the River Taw catchment derived from the farm survey provided much of the data required by each risk criteria to begin framing a risk assessment approach. These data populated a series of questions relating to the extent of influence of each risk factor outlined in Table 1. The series of risk factors listed within the Table are critical to the functioning of the risk tool and exert varying degrees of influence. However, these data were supplemented (on some farms) with data derived from field-by field risk assessments to fulfil a seasonal audit of farm operational areas within each farm enterprise. Furthermore, spatial data generated within a GIS environment was utilised to further support and enhance farm scale risk classification for all participating farms. In agricultural terms, the study area remains predominantly a livestock area. Almost three quarters of agricultural holdings in Devon are beef and sheep or dairy. In North Devon, where the study took place, dairy farms comprised 15% of all farm holding types in 2004. The largest percentage of farms in North Devon were lowland cattle and sheep (44%) and cattle and sheep in Lesser Favored Areas (LFA). Mixed farming accounts for 9% of farms, while horticulture, pigs and poultry, and cereals play rather minor roles in the farm economy of North Devon.

PRIORITISING MITIGATION

Effective and pragmatic approaches to tackling microbial pollution from agriculture at the farm scale require identification and understanding of: (i) critical source areas (CSAs) of FIO loss; and (ii) management strategies to reduce risk. Thus, a strategic risk assessment tool must capture the interplay and importance of different farm scale elements in order to identify where interventions might be made to constrain microbial transfer risks. In other words, it is important to design assessment frameworks that allow for prioritised targeting of mitigation efforts In particular, we suggest that any such prioritised targeting must be sensitive to both direct and observable controls on risk – principally issues of ‘source’, ‘transfer’, ‘infrastructure’ - but also the wider attitudinal and structural factors that shape a farmer’s capacity to take action – which we term ‘obstacles’. Risk frameworks should therefore be designed to allow for prioritised targeting of mitigation efforts within source, transfer, infrastructure or management related areas but ones that are sensitive to the obstacles that may prevent a farmer from taking action. This is an approach that embodies a complex adaptive systems way of thinking whereby research shifts from a command and control mentality to one that deals with unpredictable systems through integrating diverse knowledge inputs into the process (Macleod *et al*., 2007). Developing holistic risk assessment frameworks means recognising, for instance, that changing farmer attitudes to manure and land management is part of this process by which we can make our food and water safer, but that changing attitudes is not always enough. Similarly, it is a framework that acknowledges that microbial source burden alone is in itself meaningless. Not only do we need to take account of the obstacles faced by farmers in managing farm scale risks, but also we need to configure interactions of the farms assets (transfer potential and infrastructure). In other words, appropriate management interventions arise from understanding this dynamic interplay of social and natural processes at the farm scale and understanding this interaction will allow the policy community not only to target high risk areas, but also develop mitigation strategies that are sensitive to the different ways in which risk is produced.

FUTURE DEVELOPMENTS

Refinements to the FIO risk assessment framework are ongoing and when finalised will provide a farm scale tool to highlight relative risk of FIO loss from land to water attributed to a given farm. The simple structure of the tool will ensure a transparent approach to knowledge transfer of the underlying science through to the end-user. Complexity is not justified if all that is required is a general risk assessment framework for more specific, on-farm advice (Strauss *et al.*, 2007) and prioritisation of mitigation. Appropriate mitigation, both in terms of physical landscape management and social learning processes will be suggested in accordance with the farms associated risk categorisation, and all participating farms will receive advice and feedback as a result of the development of the risk tool.

Table 1: Risk factors associated with four over-arching risk criteria of farm scale FIO loss from land to water

Infrastructure	Obstacles to taking action
• Farmyard drainage • Steading/yard area • Slurry storage capacity • FYM storage facilities • Domestic wastewater treatment • Gateway location as promoter/ preventer of field-to-field connectivity to watercourse • Farm track co-efficient (extent of farm tracks across farm area) • Livestock watercourse access for drinking • Livestock watercourse access for fording/ crossing	• Receipt of technical grants • Influence of debt • Sufficiency of farm labour force • Participation in training, accreditation and learning networks • Understanding and awareness of microbial risk discourses • Presence of a regulatory environment • Participation in agri-environmental schemes • Organic status • Membership of quality assurance schemes
Transfer potential	*E. coli* burden to land (ha^{-1})
• Averaged farm slope • Typical slope shape (convex/ concave) • Soil type • Extent of soil compaction • Artificial drainage	• Livestock type • Faecal inputs • Grazing seasons and frequency of application • Die-off rates • Farm area

Embracing a cross-disciplinary approach to farm FIO risk assessment is a necessary part of this research process for it allows a more holistic evaluation of both landscape features and FIO sources in relation to land-owners capacities, knowledges and responsibilities for protecting watercourses. This is an approach that does not pretend to offer overnight solutions. Indeed, our experience suggests that cross-disciplinary research tends to magnify uncertainties regarding the functional definition of a risk system and how we might begin act upon it. Yet, if the initial stakes seems high, the long term rewards in terms of designing more sustainable and appropriate solutions may be greater still. The inclusion of both physical and socio-economic risk factors extend the range of mitigation strategies at our disposal and reinforce the advantages of coupling both natural and social sciences in farm-scale risk assessments

ACKNOWLEDGEMENTS

This work was funded as part of project RES-224-25-0086 by the RELU programme (supported by BBSRC, NERC, ESRC, Defra and the Scottish Office).

REFERENCES

Avery SM, Moore A and Hutchison ML (2004). Fate of *Escherichia coli* originating from livestock faeces deposited directly onto pasture. Letters in Applied Microbiology 38, 355-359.

Defra (2003). Manure management plan: a step by step guide for farmers. Viewed on 10 November 2007. Available online at: http://www.defra.gov.uk/corporate/regulat/forms/agri_env/nvz/manureplan.pdf

Defra (2005a). Producing a soil management plan for environmental stewardship. Viewed on 10 November 2007. Available online at: http://www.defra.gov.uk/erdp/pdfs/es/guidance/es-soil-management-plan.pdf

Defra. (2005b). Entry Level Stewardship Handbook. Viewed on 10 November 2007. Available online at: http://www.defra.gov.uk/erdp/pdfs/es/els-handbook.pdf

Defra (2007). Catchment Sensitive Farming. Viewed on 10 November 2007. http://www.defra.gov.uk/farm/environment/water/csf/index.htm.

Kay D, Edwards AC, Ferrier RC, Francis C, Kay C, Rushby L, Watkins J, McDonald AT, Wyer M, Crowther J and Wilkinson J (2007). Catchment microbial dynamics: the emergence of a research agenda. Progress in Physical Geography 31, 59-76.

Macleod CJA, Scholefield D and Haygarth PM (2007). Integration for sustainable catchment management. Science of the Total Environment 373, 591-602.

Oliver DM, Haygarth PM, Clegg CD and Heathwaite AL (2006). Differential *E. coli* die-off patterns associated with agricultural matrices. Environmental Science and Technology 40, 5710-5716.

Scottish Executive (2004). The 4 point plan for improved farm waste management. 2nd edition, Scottish Environment Protection Agency, Scotland.

Strauss P, Leone A, Ripa MN, Turpin N, Lescot JM and Laplana R (2007). Using critical source areas for targeting cost-effective best management practices to mitigate phosphorus and sediment transfer at the watershed scale. Soil Use and Management. 23S1, 144-153.

SOURCES OF NUTRIENTS IN RURAL LOWLAND HEADWATER CATCHMENTS AND THEIR ECOLOGICAL SIGNIFICANCE

PJA Withers[1], RH Hodgkinson[1], HP Jarvie[2], EJ Sutton[2], D Harper[3], K Kwiatkowska[3], P Wasiak[3], C Stoate[4] and RH Foy[5]

[1]Catchment Management Group, ADAS Gleadthorpe, Mansfield, Notts, NG20 9PF, UK, E-mail: paul.withers@adas.co.uk; [2]Centre for Ecology and Hydrology, Wallingford, Oxfordshire, OX10 8BB, UK; [3]Department of Biology, Leicester University, Leicester, LE1 7RH, UK; [4]The Game and Wildlife Conservation Trust, Loddington, Leicestershire, LE7 9XE, UK; [5]Queen's University, Belfast, BT9 5PX, Northern Ireland

SUMMARY

Monitoring of storm run-off and streams in seven small (<10 km^2) rural catchments in England and Wales suggests that surface and sub-surface run-off from agricultural land is not the only source of P contributing to the risk of eutrophication. Farmyard run-off, road run-off and discharges from septic tanks were all more concentrated sources of phosphorus (P) than run-off from farmed land. The more continuous delivery of P in run-off from impervious surfaces and/or in direct piped discharge of wastewater to the streams during the ecologically sensitive summer period suggests these sources need to be urgently controlled in order to achieve the water quality goals required by the EU Water Framework Directive. Catchment management strategies to control P transfer require collective actions and not simply those associated with control of major point sources and farming.

INTRODUCTION

Phosphorus (P) in catchment run-off is not directly toxic to aquatic biota but nevertheless causes a range of environmental, social and economic problems at the regional level that require holistic, catchment-based control strategies that address the sources of pollutants and not just the symptoms. Nutrient criteria with regulatory applications have been established to define what constitutes water quality impairment and cost-effective targeting of measures to reduce concentrations and loads of P in catchments is required to achieve these criteria. Within Europe, there is a statutory requirement for programmes of measures to be implemented in catchments under the Water Framework Directive (WFD) by 2012 so there is some urgency. Measures are required to control both point sources and diffuse sources within the catchment but targeting all sub-catchment areas equally has been shown to be neither cost-effective nor likely to reduce pollutant discharge.

Correct apportionment of sources, knowledge of their distribution and understanding of their mode of delivery to the watercourse is an essential part of catchment P management and achievement of water quality goals. Point sources are usually flowing continuously from single points, require some form of consent, or permit, and are therefore routinely monitored. Diffuse sources have a more rural origin and originate from a number of different areas within the upstream catchment and are more episodic in nature and therefore temporally and spatially more variable. In reality, the large variety of different P sources entering surface waters have different hydrological and compositional characteristics which often makes their simple grouping into point or diffuse difficult (Edwards and Withers, 2007). As part of the PARIS (Phosphorus from Agriculture: Riverine Impacts Study) project, we examined a range of nutrient sources in rural headwater catchments of three major rivers: the Welland, the Hampshire Avon and the Wye.

CATCHMENT SITES AND SAMPLING DETAILS

For each river, one catchment had low intensity agriculture and one (or two) had higher intensity agriculture in terms of either the proportion of the catchment area under cultivation, with field underdrainage systems or in terms of P inputs or soil P fertility (Table 1). One catchment also contained a village sewage treatment works (STW).

Table 1: General characteristics of the catchment sites

Headwater	Low intensity catchment	Higher intensity catchment
Welland	Digby Farm (DF)	Belton Bridge (BB)
	Beef/sheep farming on iron-rich loamy over clayey grassland soils. Low P inputs and low soil P	Cereal based arable rotations on sloping, intensively drained heavy boulder clay soils. Low P inputs and low soil P
Avon	Cools Cottage (CC)	Priors Farm (PF)
	Summer beef/sheep grazing on drained heavy clay soils. Low P inputs and medium soil P	Beef and dairy farming with forage maize on drained heavy clay grassland soils. Variable P inputs and medium soil P
Wye	Whitchurch (WC)	Dinedor (DD)
	Beef/sheep farming on sloping silty soils with ley/arable farming on plateau land. Low P inputs and medium soil P	Mixed beef/sheep and cereal/potato farms silty and clayey soils with variable slopes. Variable P inputs and medium soil P
		Kivernoll (KN)
		Intensive cereal/potato rotations on drained clayey soils. High P inputs and high soil P. Village sewage works

In each catchment, stream chemistry was monitored on a weekly basis (with additional intensive storm-event sampling) over two years to examine suspended sediment (SS), P forms, nitrogen (N) forms and boron (B), which was used as a marker for detergent inputs (Neal *et al*., 2005). Grab samples of run-off generated during storm events were also taken from a range of catchment sources to characterise their nutrient composition. These sources included field surface and sub-surface run-off, road and track run-off, farmyard run-off and field ditches or run-off receiving septic tank discharges. Surface fine (<2 mm) stream bed sediments were also sampled in spring and late summer to determine nutrient status and P release properties. Measurements of ecological structure and function were carried out in spring, summer and autumn to assess the relationship between stream nutrient status and ecology. Some preliminary data are presented here on the P chemistry of streams and run-off sources and potential ecological impacts.

STREAM PHOSPHORUS

Weekly stream monitoring showed a large gradient in mean soluble reactive P (SRP) and total P (TP) concentrations between sites, with consistently greater concentrations in those catchments with more intensive agriculture (Figure 1). The majority of the increased P was SRP for the Avon and Wye catchments, but particulate P (PP) in the Welland. The largest increase in SRP was obtained in the Kivernoll catchment which contained a village sewage treatement works (STW). Stream SS and stream bed sediment P concentrations were also greatest in the agriculturally intensive catchments. Preliminary evaluation of ecological measurements suggest that stream ecology is limited by P at concentrations below 100 µg L^{-1} and by other factors above 100 µg L^{-1}.

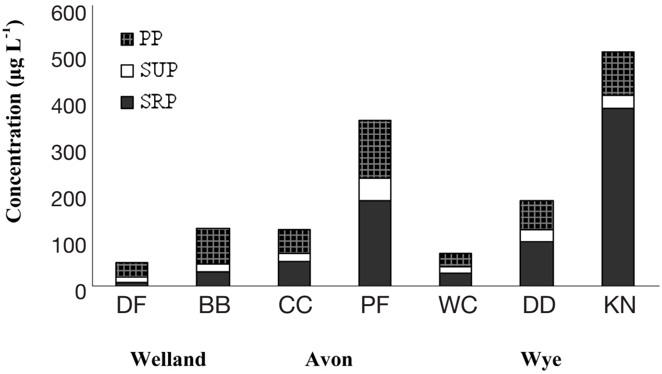


Figure 1: Weekly mean concentrations of soluble reactive P (SRP), soluble unreactive P (SUP) and particulate P (PP) in the catchment streams

PHOSPHORUS IN STORM RUN-OFF

Over 200 storm run-off samples were collected over a two year period with concentrations of TP varying by up to two orders of magnitude across the different sampling sites and occasions (Table 2). Concentrations of SRP and TP in run-off from field drains tended to be lower than those in field surface run-off, although the variation was still large. However, run-off from tracks, roads and farmyards, or contaminated by septic tank effluent, contained larger range and average TP concentrations than run-off from farmed land. Run-off from farmyards and associated with septic tanks, had particularly large SRP concentrations, whilst run-off from roads also contained greater SRP concentrations than might be expected in rural areas dominated by agricultural land (Table 2). There were also large differences in average P concentrations between sites within the same run-off type. For example, within the Wye catchments, some field drain, road run-off and farmyard run-off sites showed elevated concentrations of SRP and B compared to other sites. The relationship between these two nutrients at these sites was similar to that obtained in field ditch samples contaminated by septic tank discharges, suggesting a common detergent source (e.g. Figure 2). At other sites there was no relationship between SRP and B.

Table 2: Range, mean and median concentrations (mg L-1) of soluble reactive P (SRP) and total P (TP) in different storm run-off sources

Run-off type	n	SRP			TP		
		Range	Mean (s.e.)	Median	Range	Mean (s.e.)	Median
Field surface	35	0.01-1.6	0.17 (0.05)	0.09	0.17-6.8	1.29 (0.22)	0.94
Field drain	56	0.01-0.5	0.10 (0.02)	0.05	0.02-6.2	0.76 (0.14)	0.39
Tracks	13	0.02-0.9	0.16 (0.07)	0.08	0.24-7.3	2.69 (0.74)	1.46
Roads	51	0.01-1.3	0.29 (0.05)	0.16	0.11-16	2.31 (0.35)	1.70
Farmyard	26	0.01-5.7	1.04 (0.27)	0.51	0.08-15	2.74 (0.68)	1.47
Septic tank	22	0.01-1.4	0.55 (0.09)	0.48	0.17-6.5	1.62 (0.32)	1.23

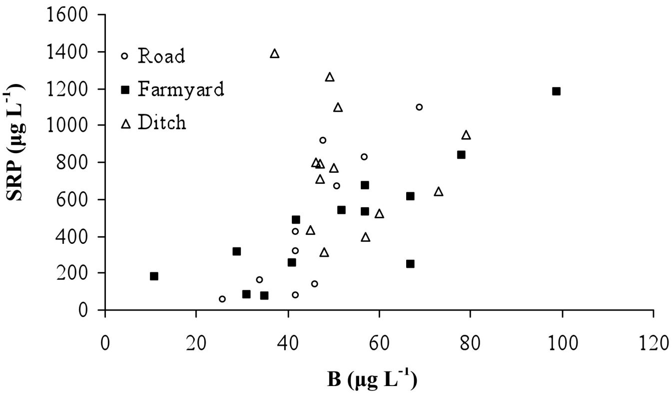


Figure 2: Relationship between soluble reactive phosphorus (SRP) and boron (B) for three storm run-off sampling sites in the Wye catchment

DISCUSSION

The farming community within the UK is under increasing government pressure to reduce P emissions from agricultural land to water because of the risk of eutrophication, which is becoming a major environmental issue for the developed world. The focus on P is because of the central role this nutrient plays in biological productivity and the current emphasis on the farming community relates both to the established link between agricultural intensification and increased P loss in land run-off and the need to control both point and diffuse sources of P in catchments to meet ecological water quality targets. The catchments studied in the PARIS project are typical of rural communities in England and Wales but of a sufficiently small size to suggest that 'agriculture' would be the main source of P entering streams during storm events.

The weekly stream sampling showed large differences in P export between the catchments, suggesting an effect of farming intensity. However, it is likely that differences in stream P chemistry were also due to the distribution of other sources termed 'intermediate' by Edwards and Withers (2007). In some instances, there was a multitude of drains from different sources discharging into the same headwater stream. There were other instances where multiple sources were being diverted into a single field drain and there were other instances where it was uncertain where run-off originated from. Road run-off was an important source in all catchments with clear evidence of SRP contamination where roads passed domestic dwellings. As expected, P was particularly concentrated in farmyard run-off due to the presence of livestock excreta and use of detergents used for washing down (Edwards *et al*., 2007). The majority of this P was highly bioavailable. Septic tank discharges also appear to be a major issue in these rural catchments, particularly in the more agriculturally intensive catchments.

ACKNOWLEDGEMENTS

The PARIS project is funded by Defra and the Environment Agency. The co-operation of participating farmers in the catchment, help with storm sampling by Robert Howells and Adam Bates of ADAS UK Limited and sample analysis by Margaret Neal and Heather Whickham of the Centre for Ecology and Hydrology is also gratefully acknowledged.

REFERENCES

Edwards AC, Kay D, McDonald A, Francis C, Watkins J, Wilkinson RJ and Wyer MD (2007). Farmyards, an overlooked source for highly contaminated run-off. Journal of Environmental. Management (in press).

Edwards AC and Withers PA (2007). Linking phosphorus sources to impacts in different types of waterbody. Soil Use and Management 23 (Supplement 1), 133-143.

Neal C, Jarvie HP, Neal M, Love AJ, Hill L and Wickham H (2005). Water quality of treated sewage effluent in a rural area of the upper Thames Basin, southern England, and the impacts of such effluents on riverine phosphorus concentrations. Journal of Hydrology 304, 103-117.

MITIGATION OPTIONS FOR PHOPSHORUS AND SEDIMENT (MOPS): REDUCING POLLUTION IN SURFACE RUN-OFF FROM ARABLE FIELDS

C Deasy[1], JN Quinton[1], M Silgram[2], CJ Stevens[1], R Jackson[2] and A Bailey[3]

[1]Lancaster Environment Centre, Lancaster University, Lancaster, LA1 4YQ, UK, E-mail: c.deasy@lancaster.ac.uk; [2]ADAS Wolverhampton, Wolverhampton, WV6 8TQ, UK; [3]Department of Agriculture, University of Reading, Reading, RG6 6AR, UK

SUMMARY

The Mitigation Options for Phosphorus and Sediment (MOPS) project is investigating a range of mitigation treatments with potential for reducing phosphorus (P) and sediment losses from arable land with combinable crops. Research undertaken at three field sites in the UK over a range of soil types has indicated that the majority of P lost in run-off is transported in association with sediment, principally down in-field tramlines. Disruption of tramlines using tines reduces run-off and hence transport of sediment and associated pollutants. Other treatments which reduce run-off and/or trap sediment on the hillslope, including minimum tillage, contour cultivation and vegetative barriers, are also effective in reducing P losses from arable land.

INTRODUCTION

Diffuse P pollution presents a serious problem in the UK, contributing to the eutrophication of surface waters. Losses of P from agriculture are of particular concern, as agricultural systems traditionally have high inputs of P applied in fertilisers and manures to enhance productivity. In the UK, the agricultural P surplus has been estimated to average around 16 kg ha^{-1} per year (Withers *et al*., 2001). Although there has been extensive research into effective treatments for reducing soil erosion from arable land (e.g. Quinton and Catt, 2004), less is known about the effectiveness of mitigation options for reducing P losses. To address this research gap, the Defra funded MOPS (Mitigation Options for Phosphorus and Sediment) project is investigating a range of tillage treatments with potential for mitigating P losses from arable land associated with combinable crops.

METHODS

Experimental Design

Field monitoring is being carried out at three field sites in the UK, each with contrasting soil types for which appropriate mitigation treatments have been selected and trialled (Figure 1). Fifty-two unbounded hillslope length plots are being monitored over three winters across the three sites, which allows replication of different treatments and combinations of treatments. Surface run-off is intercepted at the base of each hillslope plot by a 3 m collection trough, from which run-off is piped into a flow splitter and collection tank for sampling. After run-off events, samples are collected from each tank and refrigerated at 4°C prior to analysis.

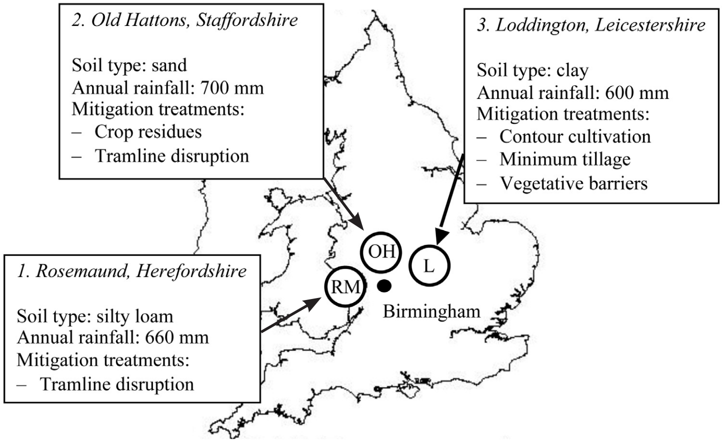


Figure 1: Location and characteristics of the three MOPS field sites

Laboratory Analysis

Samples were analysed for suspended sediment (SS), total P (TP) and total dissolved P (TDP). Suspended sediment was analysed by determining the mass of sediment evaporated from a 200 ml run-off sample at 105°C. Samples for TDP were filtered through 0.45 μm cellulose nitrate filters within 24 hours of collection, and P concentrations were determined colorimetrically (Murphy and Riley, 1962) using flow injection analysis after persulphate digestion. All samples were analysed within one week of collection. Particulate P (PP) was determined by difference (PP = TP - TDP).

RESULTS

The results from the first two field seasons show that P losses at all three sites are principally particulate (>76%). Results from two of the three sites (RM and OH) indicate that tramlines (unseeded lines compacted by tractor wheels) are the main route of P and sediment transfer from arable fields, with losses of run-off, SS and P from plots containing tramlines (T) at least an order of magnitude higher than losses from plots without tramlines (NT) (Table 1). However, the results also showed that it was possible to reduce run-off by disrupting tramlines using a ducksfoot tine (DT). Run-off, SS and P losses from disrupted tramline plots were reduced to levels comparable to non-tramline areas at the silty loam site (RM).

Table 1: Results of mitigation trials at Old Hattons (OH) and at Rosemaund (RM) for the first winter of monitoring. Data are aggregate over-winter yields. NT = no tramline, T = tramline, DT = disrupted tramline, B = straw baled and removed, C = straw chopped and spread

Site	Treatment	Run-off (m^3 ha^{-1})	SS (kg ha^{-1})	TP (kg ha^{-1})
RM	NT	3	3	0.01
	T	58	357	1.32
	DT	3	6	0.02
OH	B, NT	4	21	0.06
	B, T	84	499	1.52
	C, NT	2	12	0.03
	C, T	64	298	0.99

Results from the sandy site (OH) indicated that crop residue treatments can reduce run-off, SS and P losses from arable land. Chopping and spreading straw (C), instead of baling and removing it (B) significantly reduced losses from arable land, typically by 30-60 % (Table 1). However, the losses from non-tramline areas were small compared to losses from tramline areas. In the second winter, tramline treatments were trialled at both the sandy and silty loam sites. The results indicate that at both sites, disrupting tramlines could reduce run-off, SS and P losses to levels from non-tramline areas (Table 2).

Table 2: Results of mitigation trials at Old Hattons (OH) and at Rosemaund (RM) for the second winter of monitoring. Data are aggregate over-winter yields. NT = no tramline, T = tramline, DT = disrupted tramline, OT = offset tramline

Treatment	Run-off (m^3 ha^{-1})		SS (kg ha^{-1})		TP (kg ha^{-1})	
	OH	RM	OH	RM	OH	RM
NT	27	20	24	21	0.1	0.0
T	153	778	275	4776	0.8	2.9
DT	50	27	72	40	0.2	0.0

Results of the mitigation treatments trialled on clay soils at Loddington (L) in 2005-2006 and 2006-2007 are presented in Figure 2. Both minimum tillage (MT) and cultivation on the contour (C) reduced run-off, SS and P losses compared to conventional tillage and up-and-down slope cultivation (P) at this site, although with high variability between replicate treatments. On average, minimum tillage reduced TP losses by 0.02 kg ha^{-1} compared to the plough treatments. Total P losses were lower under minimum tillage compared with the plough treatment in 2006-2007, but losses were not lower under minimum tillage in 2005-2006, when TP values were below 0.02 kg ha^{-1} in both treatments. On average, losses of TP were reduced by 0.03 kg ha^{-1} for contour cultivation compared to up-and-down slope cultivation, but the reductions observed in 2006-2007 were not observed in 2005-2006. The vegetative barrier (BB) was effective in reducing TP loads, with losses reduced by a further 0.03 kg ha^{-1} for contour treatments containing the vegetative barrier.

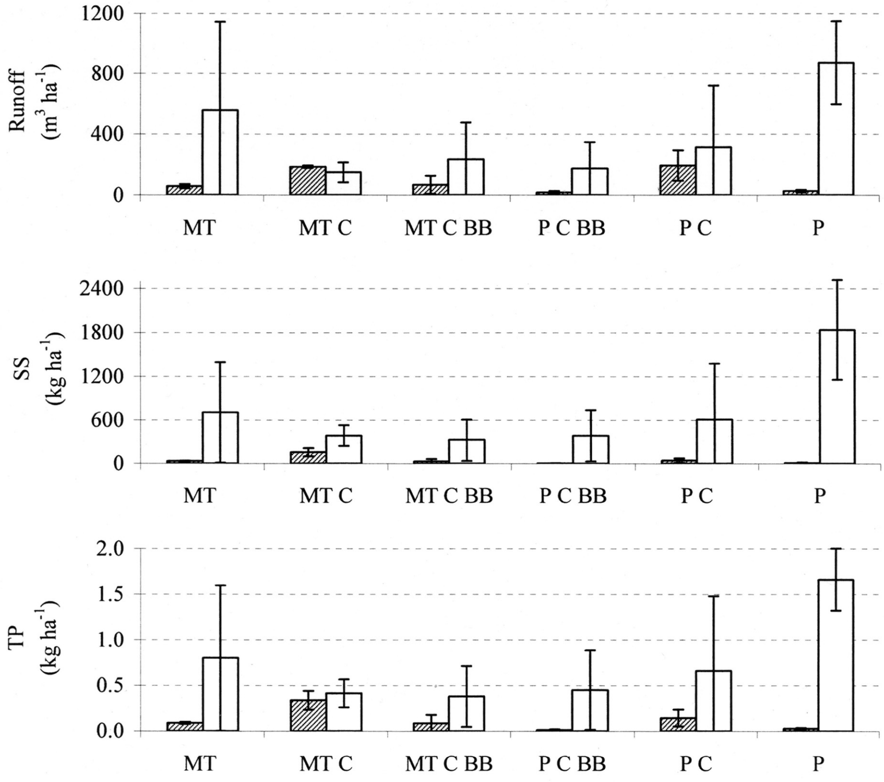


Figure 2: Results of mitigation options trialled on clay soils at Loddington (L) in Leicestershire for (a) run-off, (b) SS and (c) P. Data are aggregate over-winter yields for events monitored in 2005-2006 and 2006-2007, for hillslope lengths under different treatments. MT = minimum tillage, C = contour cultivation, BB = beetle bank, P = plough

DISCUSSION

Results from three field sites on different soil types have indicated that losses of P from arable land are principally particulate, therefore treatments which reduce erosion, either by reducing run-off and particulate carrying capacity, or by trapping particulate material on the hillslope, have potential for reducing P losses. Tramlines are a principal pathway for run-off and transfer of eroded material on arable hillslopes, and hence tramline disruption treatments, which allow water to infiltrate, have been shown to be effective at reducing SS and P losses. However, as tramline disruption is associated with decreased operating margins (-£11 ha^{-1}) (see Bailey *et al.* 2007), it is unlikely to be widely adopted as a mitigation treatment until the disruption mechanism can be incorporated into standard field management activities. Incorporation of crop residues, which is an effective treatment in non-tramline areas, is also associated with decreased operating margins (-£19 ha^{-1}). Treatments such as minimum tillage and contour cultivation, which are likely to involve cost savings, are more likely to be adopted. Bailey *et al.* (2007) suggest that minimum tillage could involve considerable cost savings (+£48 ha^{-1}) because of reduced mechanisation and labour costs, although this treatment may be less effective at reducing SS and P losses than alternative treatments. Cultivation on the contour rather than up-and-down slope may have no effect on operating margins, although this is dependent on the field size and slope characteristics. In-field vegetative barriers, although effective, are associated with small cost increases (-£2 ha^{-1}), and are difficult to cultivate around. In addition, as they need to be applied on the contour, substantial benefits may be achieved almost as easily and at lower cost by converting to contour cultivation.

The treatment which has the greatest impact on SS and P losses, tramline disruption, is successful because it breaks up compacted soil surfaces and allows rainfall to infiltrate where it would previously have run off. Surface erosion and transport processes are therefore reduced. It is unclear whether the water which would have flowed down the tramlines is stored in the soil, or whether it is transferred to the stream by an alternative subsurface pathway, but as concentrations of SS and P in subsurface run-off are usually lower than in surface run-off (e.g. Dils and Heathwaite 1996), treatments which displace run-off from surface to subsurface pathways can still be considered beneficial mitigation options. Of more concern is the effect of controlling sediment and P losses on other pollutants, which may be positive or negative. In a review of pollution swapping in agricultural systems, Stevens *et al.* (in press) report that the use of crop-residue treatments may increase leaching losses of N, P and C, and increase emissions of N_2O and CO_2. Evidence discussed in the same review suggests that minimum tillage is likely to be a beneficial treatment not only for surface losses of SS and P, but also for N, C and pesticides, with little effect on gaseous emissions.

The MOPS project is currently in its third year, and the results from the 2007-2008 field season are expected to provide further evidence to support the effectiveness of tramline treatments, minimum tillage and contour cultivation in reducing P losses from arable land.

ACKNOWLEDGEMENTS

The MOPS project is funded by Defra under contract PE0206. We thank Severn Trent water, ADAS and The Allerton Project for access to field sites, the Environmental Science department at Lancaster and ADAS for field assistance, and the Lancaster Environment Centre for undertaking the analyses.

REFERENCES

Bailey A, Quinton JN, Silgram M, Stevens CJ and Jackson B (2007). Determining the cost effectiveness of solutions to diffuse pollution: The case of in-field mitigation options for phosphorous and sediment loss. In A vibrant rural economy – the challenge for balance, proceedings of the 16th International Farm Management Association Congress, University College, Cork, Ireland, 15-20 July 2007, pp. 657-664.

Dils RM and Heathwaite AL (2000). Tracing phosphorus movement in agricultural soils. In: Foster I D L (Ed.), Tracers in Geomorphology. Wiley, Chichester, pp. 259-276.

Murphy J and Riley JP (1962). A modified single solution method for the determination of phosphate in natural waters. Analytica Chimica Acta 27, 31-36.

Quinton JN and Catt JA (2004). The effects of minimal tillage and contour cultivation on surface run-off, soil loss and crop yield in the long-term Woburn Erosion Reference Experiment on sandy soil at Woburn, England. Soil Use and Management 20, 343-349.

Stevens CJ and Quinton JN (in press). Diffuse Pollution Swapping in Arable Agricultural Systems. Critical Reviews in Environmental Science and Technology.

Withers PJA, Edwards AC and Foy RH (2001). Phosphorus cycling in UK agriculture and implications for phosphorus loss from soil. Soil Use and Management 17, 139-149.

A COMBINED MONITORING AND MODELLING APPROACH TO ASSESS THE EFFECTIVENESS OF MEASURES TO CONTROL NITRATE POLLUTION

MA Shepherd, EI Lord, RA Hodgkinson, M Silgram, G Goodlass, RD Gooday and SG Anthony

ADAS UK Ltd., Woodthorne, Wergs Road, Wolverhampton, WV6 8TQ, UK, E-mail: Eunice.Lord@adas.co.uk

SUMMARY

Demonstrating the effectiveness of changed land management practices on water quality is a challenge because of several complicating factors, including annual variation in weather and lags in the system. This is highly relevant to the implementation of the Water Framework Directive. We advocate that much can be learnt from the approaches adopted under the Nitrates Directive in England of a combination of field monitoring and modelling. Field monitoring provides evidence of typical nitrate levels in water draining from agriculture, as well as allowing validation of leaching models; modelling, combined with farm practice data, allows an estimate of the likely impacts of proposed mitigation methods.

INTRODUCTION

Approximately 70% of nitrate in surface and groundwaters in the EU is derived from diffuse agricultural sources (EA, 2002). Improved farm management practices can decrease nitrate loss (Shepherd and Chambers, 2007); the challenge is bringing about these changes, with available policy options ranging from voluntary Codes of Practice through to mandatory practices underpinned by regulation. The Nitrates Directive (91/676/EEC) requires the implementation of an Action Programme (AP) by farms within designated Nitrate Vulnerable Zones (NVZs), i.e. it takes a mandatory approach to implementing specific mitigation methods on farms. The Directive specifies a 4-yearly review of progress, including assessing the effectiveness of these mitigation practices in decreasing nitrate loss to water, which can then inform the need for subsequent revisions to the AP measures.

This 4-yearly cyclical approach to diffuse nitrate pollution (identify problems, identify and implement solutions, monitor effectiveness, review and revise if necessary) is in line with the (6-yearly) review process under the Water Framework Directive (WFD). However, there are many challenges associated with assessing the effectiveness of changed practices solely from measurements of water quality at the catchment-scale (i.e. the usual scale of monitoring). These include the difficulty in detecting a small change in N concentration against background variability (especially in surface waters), the long time-scale for response to changes in land management practices (especially in some groundwaters), and the difficulty in distinguishing effects from other factors (e.g. other drivers of farm management change, such as economics).

In its review of the 2002 AP, England (Defra) adopted an evidence-based approach to the assessment of the effectiveness of the AP on nitrate loss from agricultural land. The approach was applicable to both surface and groundwaters and used a combination of monitoring and modelling to determine both the likely long-term trends and the effectiveness of individual mitigation methods. The aim of this paper is to review the approach and to discuss its applicability to the assessment of WFD Programmes of Measures.

APPROACH

The assessment was based on 3 components (Figure 1): field measurements on commercial farms (2004-07); development and use of field-scale and catchment-scale models; use of other data and information to supplement the assessments. More detail can be found in Lord *et al*. (2007).

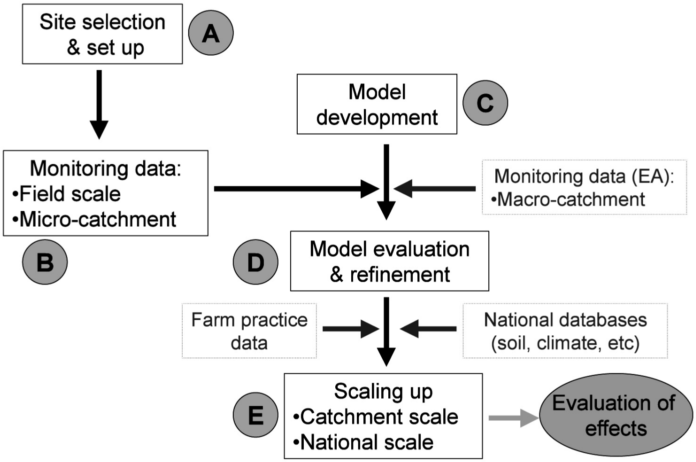


Figure 1: Schematic representation of the project structure

The approach was based around the linkage between field measurements and modelling and is the prescribed approach presented in the EC's draft reporting guidelines. The two strands of field evidence and modelling were complementary. Whereas field measurement data were essential to provide 'typical' nitrate losses under commercial agriculture, it was not feasible to monitor every field within NVZs, and modelling was required to bring in the understanding gained from the scientific knowledge base, and to scale up the results.

Field Measurements

A network of sites was established on a range of commercial farms in both surface and groundwater catchments within NVZs. A total of 203 fields were monitored on groundwater sites in 8 locations and 125 fields on surface water sites in 8 micro-catchments, ensuring measurements across a range of cropping systems and climatic regions. Soil mineral nitrogen (SMN) was measured in late autumn on all fields to 60 or 90 cm depth as an indicator of risk of nitrate loss arising from the management of that particular field. These data were complemented on about 80 of the groundwater sites by annual nitrate leaching measurements using porous ceramic cups (Lord and Shepherd, 1993). The cups were buried at 90 cm, 10 per field, away from the headland and at least 2 m apart. Capillary access tubes were buried prior to cultivations, to allow the area above the pots to be treated exactly as the rest of the field. The N flux was calculated (Lord and Shepherd, 1993) as the integration of measured N concentrations and calculated water balance for the specific cropping and weather conditions, using the IRRIGUIDE program (Bailey and Spackman, 1996). Porous cups were not suitable for quantifying nitrate loss from clay soils, (the dominant soils in many surface water catchments) because runoff and drain water is usually not at equilibrium with the soil solution. Instead, hydrological 'micro-catchments' were identified, and individual drains or groups of fields within the micro-catchment were monitored using flow monitoring and automated water samplers. N flux was calculated by a similar method to that used for porous cups, except that water fluxes were measured directly. Detailed land use, cropping and land management data were also collected as input data for model validation.

Model Development and Evaluation

A field-scale model (NIPPER: Lord *et al*., 2007) was developed and tested within this project with the objective of correctly representing current N loss and the impact of management change. Particular attention was paid to improved representation of nitrate transport modelling for clay soils (Gooday *et al*., 2008) because of growing concern regarding the implications of their hydrological behaviour in relation to impacts of measures such as timing of nutrient and manure inputs. A daily time step was chosen, to allow the model to reflect the rapid variations in nitrate concentration in drainage from clay soils. Model results, and response to management change, were evaluated initially against a wide range of data from controlled experiments, and against the behaviour of other proven models such as MANNER (Chambers *et al*., 1999). The model was then evaluated against the monitoring data to demonstrate it could reproduce nitrate losses under a wide range of management conditions on commercial farms (Lord *et al*., 2007).

ESTIMATION OF EFFECTIVENESS OF MITIGATION METHODS AT CATCHMENT SCALE

The effectiveness of mitigation methods at catchment scale was estimated by linking the NIPPER model with farm management data, developed as statistically representative management scenarios. Crop areas, livestock numbers, soil and climate data were taken from the MAGPIE database (Lord and Anthony, 2000) updated to year 2000 census data. Baseline conditions (representing practice prior to implementation of measures) were taken from recent surveys, especially the British Survey of Fertiliser Practice (e.g. Goodlass and Allin, 2004) The scenarios were then modified to represent implementation of the NVZ AP.

RESULTS

Field Measurements

The results were consistent with findings from previous work, in that nitrate concentrations in leachate from arable cropping in eastern areas of England (i.e. the majority of the NVZ area) typically exceed 50 mg l^{-1}. However both reported winters of the project (04/05 and 05/06) were unusually dry, and in a more typical winter nitrate concentrations would be expected to be somewhat less than the reported values.

For the 171 site years (groundwater sites) monitored with porous ceramic cups, the average flow-weighted mean nitrate concentration was 119 mg l^{-1} (28 kg N ha^{-1} loading) with an average drainage of 100 mm. Only 30% of site years had average flow-weighted nitrate concentrations of <50 mg l^{-1}. Table 1 summarises data for winter 2005/06, as an example.

Table 1: Autumn soil mineral N (SMN) and subsequent nitrate leaching from groundwater sites, winter 2005/6. No. of measurements in brackets

Previous crop	SMN (kg ha^{-1})		N leached (kg ha^{-1})	Water flux (mm)	NO_3 (mg l^{-1})	
Wheat (winter)	89	*(48)*	38	103	129	*(18)*
Barley (winter)	68	*(26)*	16	74	96	*(14)*
Sugar beet	50	*(14)*	6	78	40	*(4)*
Peas	80	*(10)*	38	66	207	*(3)*
Potatoes	93	*(5)*	31	118	147	*(3)*
Set-aside	201	*(6)*	43	66	406	*(3)*
Grass ley > 3 years	96	*(21)*	26	94	84	*(14)*
Grass/clover	97	*(5)*	13	148	37	*(4)*

In particular, nitrate concentrations after rotational set-aside were extremely variable, reflecting variable management practices. For example, in winter 2005/06, three set-aside fields were monitored using porous ceramic cups, with losses ranging from 13-78 kg N ha^{-1} (drainage varied from 39-240 mm). The data demonstrate that losses can be extremely large, especially where the crop is destroyed early and/or manures have been applied.

Examples of water quality measurements collected from surface water sampling points are summarised in Table 2. The areas draining to these sampling points ranged from *c.* 2 ha of a single crop through to 138 ha parcels of land with mixed cropping, more akin to micro-catchments. A total of 37 sampling points were monitored in the two years, with 10 of the 37 site years of data (27%) showing average flow-weighted nitrate concentrations of <50 mg l^{-1}. Seven of these 10 sites were in grass, mostly in the South West. The larger concentrations tended to be at sites that were predominantly arable, or grass with large manure loadings.

Table 2: Examples of measured N losses from sampling points draining micro-catchments, winter 2005/6

Cropping	Area (ha)	N leached (kg ha^{-1})	Drainage (mm)	NO_3 (mg l^{-1})
Permanent grassland	57	7	230	14
Grass/maize/cereal	81	58	503	51
Permanent grassland	70	10	425	10
Grass/wheat/set-aside	128	13	159	35
Wheat/Oilseed rape	138	5	88	23
Grass/wheat	66	5	47	42
Wheat/Oilseed rape	55	21	78	118

Estimation of Effectiveness of Mitigation Methods at Catchment Scale

The NIPPER model was used to explore the impacts of mitigation practices in case study catchments to take account of applicability of the measure within local agriculture and effects of soils and climate. The Meden catchment (Nottinghamshire) serves as an example of the type of analysis undertaken in the project. The Meden is typical of the local NVZ area in terms of climate and cropping. It is dominated by sandy soils, and has almost double the average quantity of manure per hectare (74 kg N ha^{-1} of agricultural land) largely in pig and poultry units.

Preliminary modelling using data on current practice indicated that two measures represented the main impact of the (2002) NVZ Action Programme in England. These were: apply no more N than the crop requires, taking account of all other sources of N; and, do not apply slurries or poultry manures during autumn on sandy or shallow soils. In order to explore the potential for minimising nitrate reduction at catchment scale, two far more severe indicative measures (not part of the Action Programme) were explored: a 10% reduction in chemical fertiliser inputs; and removal of all managed manures from the catchment.

The model calculated the average amount of nitrogen (N) leached to be 58 kg N ha^{-1} of agricultural land (Table 3). The considerable variation between subcatchments in both current N loss and impact of mitigation measures was mainly associated with variation in livestock numbers. Calculated nitrate concentrations in leachate from agricultural land were high, consistent with measured data, and even the most severe measures could not reduce the estimates to less than 50 mg/l.

Table 3: Modelled N loss from agricultural land, and effectiveness of selected mitigation methods in reducing nitrate losses and concentrations in leachate in the Meden catchment, Nottinghamshire. % reductions are shown as average and range between sub-catchments

Mitigation method	N Leached (kg ha^{-1})	NO_3 Conc. (mg l^{-1})	Reduction (%)	
			Average	Range
Baseline	58	147	-	-
Do not exceed crop N requirement	55	138	6.5	1.7 - 20
Closed period PLUS do not exceed crop N requirement	50	125	15.0	1.9 - 23
10% Fertiliser Reduction	54	137	7.4	4.0 - 11.7
Remove all Manures	43	109	25.9	5.8 - 37.4

DISCUSSION

The measurement data showed that nitrate concentrations in leachate from typical agricultural land within NVZs often exceed 50 mg l^{-1}, especially under arable cropping or intensive dairying. This finding has been noted in many experiments (e.g. Anon, 2000). However, the strength of these data is that they are from current commercial farming, not from historic and/or experimental sites. These results both underpin modelled estimates of N loss, and are important for stakeholder engagement. Modelling allows results to be extended beyond the monitoring sites, and allows a wider range of scenarios to be explored. The catchment-scale modelling allows proper account to be taken of the proportion of land in the catchment to which each measure applies, and the associated impact under local soil and climatic conditions. In all, the approach provides an 'early warning' of the likely effects of measures on water quality, which is essential given the delays and other factors that affect the link between changed land management and changes to water quality at the river or groundwater abstraction point. The development of a model to scale up and extend field measurements is essential for providing quantitative assessments of the current AP measures and testing the likely impacts of both new measures, and background changes to farming practice.

This project therefore provided the following information relevant to mitigation of nitrate from agricultural land:

- Underpinning data on current field-scale nitrate losses from agricultural land under a range of systems that complement river and groundwater data reported by the Environment Agency. Such a micro-catchment monitoring network and associated data is recommended by the EC in their guidance for implementing the Nitrates Directive.
- A method of estimating present and future nitrate loss based on current and predicted farming practice.
- A method by which the effectiveness of individual measures can be assessed both at field scale, and in the catchment context.
- A method by which the impact of other measures, or externally driven changes in agriculture or climate, can be predicted.

The challenges faced by the WFD in assessing the effectiveness of implementation and demonstrating improvements (or likely improvements) in water quality have much in common with the Nitrates Directive. The approach taken for this project could easily be widened to follow other potential contaminants. The surface water sites would be well suited to extending data collection to P and sediment transfers and exploring relationships with N loss. There is also a need to consider other N species, especially NH_4^+ and organic forms of N.

ACKNOWLEDGEMENTS

Funding for this work from the Department for Food, Environment and Rural Affairs (Defra), and the cooperation of a large number of commercial farmers, is gratefully acknowledged.

REFERENCES

Anon (2002). Tackling nitrate from agriculture. Soil Use and Management 16 (supplement), pp. 89.

EA (2002). Agriculture and natural resources: benefits, costs and potential solutions. Environment Agency, Bristol, pp. 102.

Bailey R and Spackman E (1996). A model for estimating soil moisture changes as an aid to irrigation scheduling and crop water-use studies. 1. Operational details and description. Soil Use and Management 12 (3),122-129.

Chambers BJ, Lord EI, Nicholson FA and Smith KA (1999). Predicting nitrogen availability and losses following application of organic manures to arable land: MANNER. Soil Use and Management 15, 137-143.

Gooday RD, Anthony SG and Fawcett LE (2008). A model of soil drainage and nitrate leaching for application in nitrate vulnerable zones. Environmental Modelling and Software. In press.

Goodlass G and Allin R (2004). British Survey of Fertiliser Practice: fertiliser use on farm crops for crop year 2003. The BSFP Authority, London.

Lord EI and Anthony SG (2000). MAGPIE: A modelling framework for evaluating nitrate losses at national and catchment scales. Soil Use and Management 16, 167-174

Lord EI and Shepherd MA (1993) Developments in the use of porous ceramic cups for measuring nitrate leaching. Journal of Soil Science 44, 435-449.

Lord EI, Shepherd MA, Silgram M, Goodlass G, Gooday, RD, Anthony SG, Davison P and Hodgkinson RA (2007). Investigating the effectiveness of NVZ Action Programme measures: Development of a strategy for England. Report for Defra project WT03017.

Shepherd MA and Chambers BJ (2007). Managing nitrogen on the farm: the devil is in the detail. Journal of the Science of Food and Agriculture 87, 558-568.

A POTENTIAL FRAMEWORK FOR NATIONAL AND LOCAL SCALE COST:EFFECTIVENESS AND COST:BENEFIT ANALYSIS OF PHOSPHORUS POLLUTION MITIGATION OF SURFACE FRESHWATER BODIES

AJA Vinten[1], P Booth[1], M MacLeod[2], K Urama[1], M Lago[1], D Moran[2] and M Jones[1]

[1]Macaulay Land Use Research Institute, Craigiebuckler, Aberdeen, AB15 8QH, UK, E-mail: a.vinten@macaulay.ac.uk; [2]SAC, West Mains Road, Edinburgh, EH6 5AT, UK

SUMMARY

We explore the development of a framework for prioritising water bodies for pollution mitigation, under the Water Framework Directive. This uses (1) data from a national scale screening tool, (2) estimates of cost:efficacy of P pollution mitigation, and (3) loss of value to water bodies caused by P pollution. The analysis shows the importance of treating sewage treatment work sources of P, and identifies Scottish river catchments and lochs which have a high benefit:cost ratio for mitigation. Other water bodies have mitigation costs which may be disproportionate.

INTRODUCTION

Scottish catchments are expected to provide a range of 'ecosystem goods and services' including clean drinking water, diverse habitats, recreational opportunity (e.g. fishing), visual beauty, and a resource for a range of industries. They face many pressures, including point and diffuse pollution. The EU Water Framework Directive (WFD) requires member states to mitigate this pollution to achieve Good Ecological Status (GES) by 2015. Phosphorus is often the pollutant that limits water quality status (SNIFFER, 2006). The advisory body charged with providing technical advice on the WFD to regulatory agencies defines GES for P in streams as an annual mean soluble P concentration of 40 μg/l (siliceous streams) or 100 μg/l (calcareous streams) (UKTAG, 2006). In lochs, reference conditions for GES depend on the depth, alkalinity, geological status and elevation. WFD allows that if costs are disproportionate, pollution mitigation can be derogated, at least temporarily. The exploration of disproportionality requires Cost Effectiveness Analysis (CEA), Cost Benefit Analysis (CBA), and other qualitative and participatory techniques. Neo-classical economic theory says that resources should be deployed such that the marginal costs of pollution abatement equate marginal benefits of improvement at a relevant scale. These relationships are often not well defined with respect to diffuse pollution. We explore the development of a potential framework to do this for phosphorus in Scottish freshwater catchments.

MATERIALS AND METHODS

Screening Tool Description of P Loads and Concentrations

A recent national scale exercise has led to the development of a screening tool to identify water bodies vulnerable to specific pressures (SNIFFER, 2006). Estimates of the total loss of phosphorus were made using the Phosphorus and Sediment Yield Characterisation In Catchments (PSYCHIC) model (Defra Project PE0202). In addition to diffuse sources of particulate P, the Screening Tool provides estimates of soluble P transport, incidental transport from manures, and point source contributions from septic tanks and sewage treatment works. The modelling framework estimates the total P loads from land and consequent mean stream soluble P concentrations with 80% risk of exceedance for both diffuse and point sources to local catchments (LC), i.e. land area contributing directly to each ≈10 km reach of stream) and total catchments (TC), i.e. the total contributory area to a receiving water body (stream or loch). These figures are corrected for retention by the watercourse, which depends on catchment hydraulic load. From the load calculation, a "Perfect Mixer Average P concentration" in streams is calculated using LC data:

$$\text{PMAC (mg/L)} = 100 * \text{P load (kg/ha)} / \text{HER (mm)} \tag{1}$$

Where HER = hydrologically effective rainfall (mm). These predictions were compared with observed data provided by SEPA to obtain a regression equation which both predicts observed soluble P concentrations in stream, and gives a measure of the uncertainty of predictions. This allows a likelihood of exceedance of a given concentration to be determined (SNIFFER, 2006):

$$\text{Ln [stream P]} = 0.714 * \ln \text{PMAC} - 1.0478 \quad (n=597,\ r^2 = 0.49) \tag{2}$$

where [stream P] = observed annual mean soluble P concentrations in stream (µg/L).

Predicted *loch* [P] is determined from TC data using Vollenweider's estimate of phosphorus retention in lakes (OECD 1982, see SNIFFER, 2006) :

$$[lochP] = \frac{1.118 \cdot \dfrac{10^5 \cdot L}{H}}{\left(1 + \sqrt{t}\right)^{1.135}} \tag{3}$$

Where [*loch* P] is the predicted mean total phosphorus concentration (µg l^{-1}), *L* is the catchment average load (kg P ha^{-1}) *H* is the modelled average catchment drainage (mm), and *t* is the average hydraulic residence time of the lake (y). Note that outputs from PSYCHIC may not be reliable for the upland areas of Scotland where peat dominates, and lochs designated as having a peat dominated catchment have been excluded from the analysis.

Estimation of Reduction in Loading Required and Mitigation Cost/ Effectiveness of BMPs

Equations 1-3 allow us to obtain an estimate of how much P load reduction is needed to achieve a given water quality status. An equation of the form below has been fitted to output from the Screening Tool:

$$[P] = a(1-\exp(-b.\text{PMAC})) \tag{4}$$

For STW P load only, a=0.216 and b=-1.101. For diffuse P load only, a=0.0657 and b=-6.999. For Total P load a=0.235 and k=-1.377. Based on comparison with observed data, to achieve an 80% likelihood of 40 µg/L a target [P] of 0.019 µg/L is required. To achieve an 80% likelihood of 0.10 mg/L a target [P] of 48 µg/L is required. We used a fixed cost for P removal from sewage inputs of £6/kg P from sewage treatment works (STWs) and £20/kg P for septic tanks, in line with costs for small STWs. (Hutchinson *et al*., 2005). For estimation of costs of reduction of P inputs from farming, three cost curves have been used, one for arable area, one for improved grassland area and one for rough grazing using information from Haygarth *et al*. (2003), who estimate the cost of mitigation and the amount of P mitigation per ha for a series of measures that contribute to P loading from farmland. A fixed proportion of the landscape is affected by each measure. We have selected appropriate measures and ranked them in order of cost-effectiveness. A sigmoid curve has then been fitted to this data, to give a function estimating cumulative costs of mitigation, C(P) (in £/ha) vs cumulative effect $P_{s,r}$:

$$C_s(P_r) = \frac{k_1}{1 + k_2 e^{-k_3}} \tag{5}$$

Where, $C(P)_{s,r}$ denotes the cost function for each sector s in the catchment *r (in £/year); $P_{s,r}$* is the phosphorus emission reduction by each sector s in catchment *r (kg P).* Values of constants are as follows: arable: k_1=400, k_2=1.81, k_3=2.60; grassland: k_1=400, k_2=7.02, k_3=1.20; rough grass: k_1=400, k_2=8.08, k_3=0.97. Mean maximum potential P mitigation per ha has also been set to: 1.94, 1.02 and 0.38 kg P/ha, for arable, grassland and rough grazing respectively. Any further residual P loss from urban, road or forestry are not yet considered. The objective function is to minimize the costs to reduce *P* loads at some specific water body/catchment:

$$Min(C(P)) = \underset{P}{Min} \sum_{s,r} C_{s,r}(P_{s,r}) \tag{6}$$

The least-cost combination of measures, *Min(C(P),* is calculated using the solver in Microsoft Excel, for the combination of P pollutant sources present in each water body.

Estimation of Change of Value of Water Bodies in Response to Mitigation

Rivers

In low retention time streams, Hilton *et al.* (2006) consider nutrients, stream velocity, stream substrate and shading as co-determinant in the development of eutrophic plant growth. Under summer low flows and well lit conditions we can assume that nutrients are the main limitation. Hilton *et al.* (2006) consider that the median growing season soluble [P] may be a better indication of nuisance eutrophication than total P load. Westlake (1981) suggests nutrients will not be limiting in waters with P concentrations >30 µg/l and N concentrations > 1 mg/l. Since the prospect of getting N concentrations below 1 mg/l in agricultural areas is very low, it seems appropriate to focus on P as a potential limiting nutrient. For water resources management, and indeed managing the enrichment of streams, it is the periods of peak biomass that are most important (Biggs *et al*., 2000). Dodds *et al*. (1997) provide a relationship between maximum chlorphyll a in streams and TP and TN:

$$\text{Log (max chl a)} = 0.00652 + 1.100671\log(TP) - 0.1929\log(TP)_2 + 0.3129 \log TN \ (r_2 = 0.370) \qquad (7)$$

For New Zealand streams, Biggs *et al.* (2000) give a relationship between chl a and Ash Free Dry Weight (AFDM):

$$\text{Ln chl a } (mg/m_2) = 0.338 + 1.396 \text{ X Ln AFDM } (g/m_2) \ (r_2 = 0.790) \qquad (8)$$

Biggs *et al*. (2000, fig 29) also provide a relationship between %clean water species in invertebrate samples and AFDM. Combining these three functions gives a relationship between [stream P] and relative fishery value. For absolute pristine value, we use an estimate for the River Dee, whose fishery value has been estimated at £6m per year (SEERAD, 2005) over a catchment area of 210,000 ha or £29 per ha of catchment.

Lochs

The relative value of lochs as a function of trophic condition has been estimated by assuming the value declines on a sigmoidal curve, fitted to the UKTAG (2006) class boundaries, with 80%, 60%, 40% and 20% of the value of the Loch retained at the excellent/good, good/moderate, moderate/poor and poor/bad class boundaries respectively, and a residual value of 10% of the pristine value at very high loadings (see Figure 1).

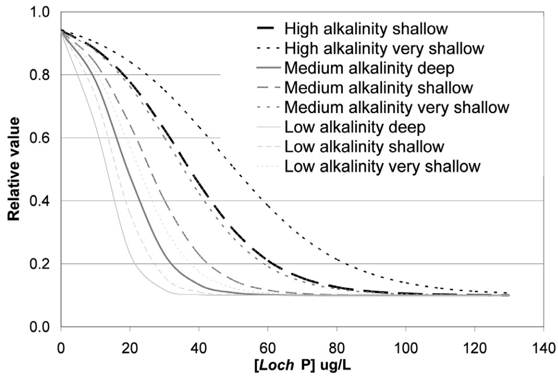


Figure 1: Curves showing assumed relative values of Scottish lochs, as a function of [loch P]

Pending further valuation work, we assume that loch value is proportional to surface area, and that all lochs have a pristine value in line with that for Loch Leven, based on the work of Frost and McTernan (1997). This work gave a total (use and non-use) value of the Loch of £522 pa, or £1.4k/ha of loch in pristine condition. Figure 2 illustrates these relationships for a range of Loch types. For both rivers and lochs, the calculations described above have been combined with spreadsheet output from the Screening Tool (2006) to give estimated requirements for P mitigation, costs of this mitigation, and estimated benefit:cost ratios.

RESULTS

Figures 2a and 2b shows the estimated load reductions per ha of catchment required to achieve an 80% likelihood of good status for (a) all sources (b) after removal of sewage treatment work sources with good status defined as 40 µg/L annual mean soluble P in streams. Figures 2c and 2d shows the estimated cost of achieving either a 40 µg/L compliance level and the ratio of estimated benefit to cost of mitigation, based on loss of value as a fishery only. Figures 2e and 2f shows the estimated cost of achieving Good Ecological Status as defined by UKTAG (2006) in standing waters and the ratio of estimated benefit to cost of mitigation, using a pristine value of £1.4k/ha of standing water. The estimated mitigation costs over 0.6 m ha of Loch catchments to achieve Good status (defined as the mid-point of the Good [*Loch* P] range), excluding Lochs with peat catchments (2.2 m ha of Loch catchment), or catchments which require more P mitigation than can be achieved according to the benchmark costings described above (2.1 m ha). These costs total £235,000 per year, with an estimated value for Loch mitigation of £2.9 m/year.

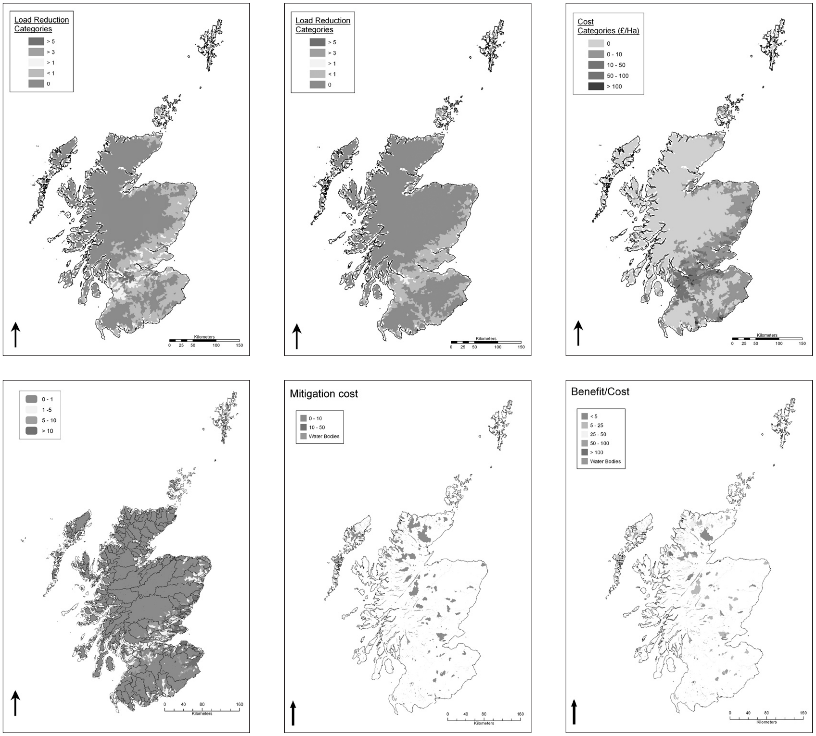


Figure 2: Reduction in (a) P loads required from all sources, (b) after removal of STW P for rivers across Scotland (kg P/ha LC); (c) Estimated costs and (d) benefit to cost ratio classes for P mitigation to a 40 µg/L standard for rivers across Scotland; (e) Estimated costs and (f) benefit to cost ratio classes for P mitigation to UKTAG (2006) GES standard for Lochs across Scotland, using Total Catchment data

DISCUSSION AND CONCLUSIONS

The framework is aimed at identifying priorities and assessing the scale of problems with achieving compliance due to disproportionality. Uncertainty and sensitivity analysis is still to be done, but the results suggest that:

- Sewage P removal will mitigate a large part of the problem, but diffuse P mitigation is needed to achieve the 40 µg/L standard. River water bodies with high sewage inputs tend to have a low benefit to cost ratio of mitigation.
- Based on the criterion that a high benefit to cost ratio indicates priorities for action, catchment areas in E and SE Scotland (e.g. Lunan Water, the Angus Esk rivers, the lower Tweed, the River Tyne and the River Eden), give the highest priority for action for [stream P]. However the absolute value of the river water bodies is open to question, as Biggs *et al*. (2000) note that several other factors control the occurrence of filamentous algae in streams. These include riparian shading, artificial flushing events in regulated rivers, optimising benthic invertebrate habitat to increase losses through grazing activity. Moreover, fishery value is not simply related to algae. Hence effective restoration of ecological status may not necessarily need mitigation of P loads.
- Mitigating of [*loch* P] appears cost:effective (benefit>>costs), although marginal analysis is required for each Loch catchment to consider how much mitigation is appropriate without claiming disproportionality. Figure 3 gives an example of marginal analysis for Loch Leven. The Screening Tool predicted and UKTAG reference [*loch* P] values are shown along with marginal abatement cost and marginal value curves. The marginal value curve exhibits a maximum, because of the sigmoidal shape of the original function (Figure 2), but there is only one point where marginal costs and marginal benefits are equal. This point is close to the GES standard, suggesting that mitigating to the WFD standard should not be disproporionate, in this case. Lochs where the mitigation is disproportionate include Loch Forfar and Loch Gelly.

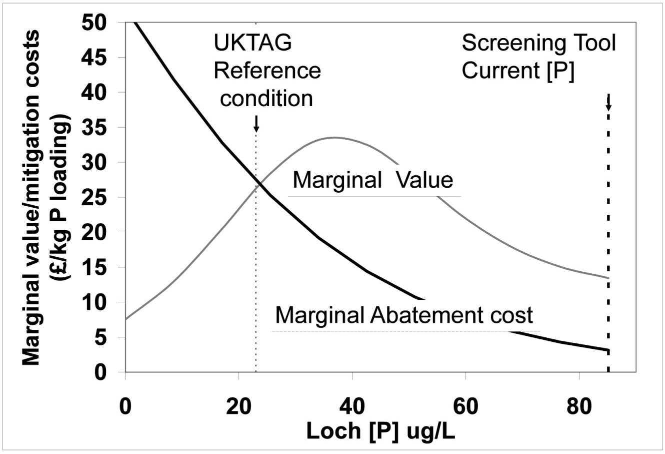


Figure 3: Marginal abatement and marginal value curves for Loch Leven

REFERENCES

Biggs BJF and Price GM (1987). A survey of filamentous algal proliferations in New Zealand rivers. New Zealand Journal of Marine and Freshwater Research 21, 175-191.

Biggs B (2000). New Zealand periphyton guide. http://www.mfe.govt.nz/publications/water/nz-periphyton-guide-jun00.pdf

Defra (2002).The Government's Strategic Review of diffuse water pollution from agriculture in England: Agriculture and Water: A Diffuse Pollution Review.

Dodds WK, Smith VH and Zander B (1997). Water Research 31, 1738-1750.

Frost A and McTernan J (1998). Cost Benefit Analysis of Proposed Erosion Control at Loch Leven. In Diffuse Pollution and Agriculture II. SAC/SEPA Conference. A Petchey, B D'arcy, A Frost (Eds.).

Haygarth P (2003) Cost curve assessment of phosphorus mitigation options relevant to UK agriculture. Report for Defra Project PE0203. http://www.defra.gov.uk/science/default.htm

Hilton J, O'Hare M, Bowes MJ and Jones JI (2006). How green is my river? A new paradigm of eutrophication in rivers. Science of the Total Environment 365, 66-83.

Hutchison WG, Smith RV, Glass C and Annett JA (2005). Cost-Benefit and Cost-Effectiveness Analysis of Mechanisms for Reducing Phophorus Inputs to Inland Waters in Northern Ireland: A Report to Environment and Heritage Service, Queen's University, Belfast.

SEERAD (2005). http://www.scotland.gov.uk/library2/doc14/saff-05.asp

SNIFFER (2006). Provision of a screening tool to identify and characterise diffuse pollution pressures. Phase II and III reports. Project WFD19.

UKTAG (2006).
http://www.wfduk.org/UK_Environmental_Standards/ES_Phase1_final_report/Library/PublicDocs/UKTAG%20ReportAug%20 2006UKEnvironmentalStandardsandConditionsFinalReport (last access 3 July 2007)

GREENHOUSE GAS EMISSIONS, INVENTORIES AND VALIDATION

D Fowler[1], U Skiba[1], JB Moncrieff[2] and D Polson[1,2]

[1]Centre for Ecology and Hydrology, Bush Estate, Penicuik, Midlothian, EH26 0QB, UK; E-mail: dfo@ceh.ac.uk; [2]Institute of Atmospheric and Environmental Science, School of Geosciences, Edinburgh University, Crew Building, The King's Buildings, West Mains Road, Edinburgh, EH9 3JN, UK

SUMMARY

The emission of greenhouse gases has become a very high priority research and environmental policy issue due to their effects on global climate. The knowledge of changes in global atmospheric concentrations of greenhouse gases since the industrial revolution is well documented, and the global budgets are reasonably well known. However, even at this scale there are important uncertainties in the budgets, for example, in the case of methane while the main sources and sinks have been identified, temporal changes in the global average concentrations since the early 1990s are not understood. In the absence of a quantitative explanation with appropriate experimental support, it is clear that current knowledge of the causes of changes in the global methane budget is inadequate to predict the effect of changes in specific emission sectors.

In developing control strategies to reduce emissions it is necessary to validate national emissions and their spatial disaggregation. The methodology to underpin such a process is at an early stage of development and is not fully implemented in any country, even though target emission reductions have already been announced. Furthermore, the scale of the emission reductions is large (e.g. 60% reductions by 2050 relative to 1990 baseline). There is therefore an urgent requirement for measurement based verification processes to support such challenging emission reductions.

In this paper we provide the background in greenhouse gas emissions globally and in the UK followed by examples of approaches to validate emissions at the UK scale and within the regions.

INTRODUCTION

The emission of naturally occurring greenhouse gases to the atmosphere as a consequence of human activities has led to large increases in concentrations since the beginning of the industrial revolution. Clearly the sources of these gases are in excess of the sinks and the rates of change in their concentrations in the atmosphere provide a measure of the degree of imbalance of sources and sinks. Carbon dioxide concentrations (Figure 1) have increased from about 280 ppm in 1750 to 367 ppm in 1999, and the rate of increase continues to accelerate (Houghton, 2004). Methane concentrations grew from around 770 ppb in 1750 to 1891 ppb in 1999, concentrations growing rapidly until the early 1990s, but since then have increased some years and decreased in other years, and the cause of these changes remains unclear. Uncertainties in the cause of changes in methane reveal major gaps in understanding the atmospheric methane cycle. Nitrous oxide concentrations grew from 270 ppb in 1850 to 410 ppb in 2000, and total global emissions are well known as detailed later, but country specific sources are poorly understood due to the very large spatial and temporal variability in emissions. There is a requirement to quantify country specific greenhouse gas emissions as part of international commitments within the UNFCCC. However, there are too few direct measurements to provide these emissions from country specific measurements directly. Instead, protocols to estimate national emissions based on prescriptive emission factors and activity data have been developed as part of international efforts in support of Kyoto and related protocols.

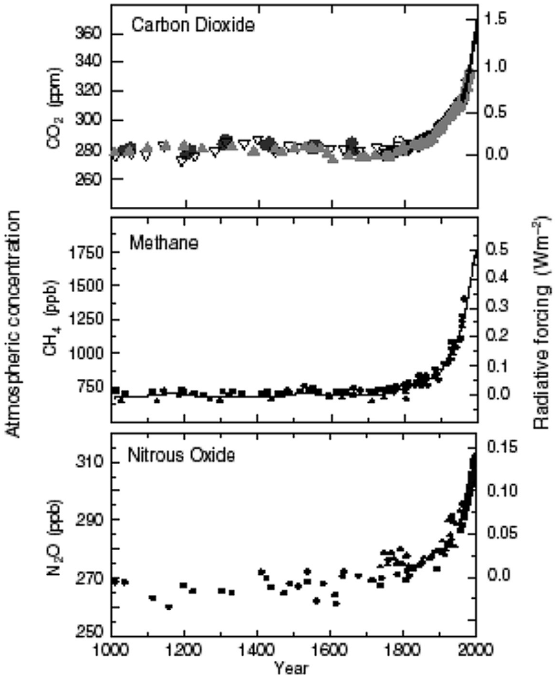


Figure 1: The long term change in greenhouse gas concentrations in the global atmosphere derived from ice cores from Antarctica (IPCC TAR, 2001)

Greenhouse gas emissions inventories are provided for the UK (Baggot *et al*., 2005). The energy sector, i.e. fuel combustion, is the largest contributor to the overall emission, and contributes in excess of 80% of UK greenhouse gas emissions. The second largest source of greenhouse gases (7% annual UK emissions) is agriculture, principally as nitrous oxide emissions from fertilised soils and methane emissions from enteric fermentation in ruminants.

Country specific emissions are calculated according to prescribed guidelines, developed by the IPCC (IPCC, 1996). The approach ensures a methodology that is consistent between countries to underpin the assessment process. However, the estimates of emission are not directly linked to measurements of the individual gases, or their emission fluxes over for the countries specified. Thus uncertainties in the underpinning science or weaknesses in the activity data may lead to errors in estimates of emission within a country.

These introductory comments reveal basic weaknesses in the underpinning science necessary to support policies for reductions in emissions of greenhouse gas emissions. In an ideal world, measurements would test and validate country specific emissions and their spatial disaggregation. The sources and sinks and their response to major controlling variables would all be known at the country and sector level in advance of specific control measures. However, the scale of the global problem of climate change, the very long response time to changes in emissions and the complexity of the interactions have made it necessary to develop policies for large scale emissions reductions in advance of a full understanding of sources and sinks.

In this paper the main sources of greenhouse gases in the UK, Scotland, England, Wales and Northern Ireland, are described. Recent developments in the measurement of greenhouse gas emissions at the landscape and country scale are presented to illustrate possible strategies to validate national and regional emission estimates.

Nitrous Oxide

Soil is the single largest source of nitrous oxide globally (Climate Change, 1994; Prinn *et al*., 1990). For example, in the UK, soils, both agricultural and semi-natural are responsible for more than 70% of the total annual nitrous oxide emission (Figure 2). Other important sources of nitrous oxide are adipic acid and nitric acid production and fuel combustion (Baggott *et al*., 2006). Vehicle emissions of nitrous oxide have risen sharply since the installation of catalytic converters, but are minor contributors to annual emissions, whereas industrial emissions have declined as processes to regulate the release of N_2O have been introduced.

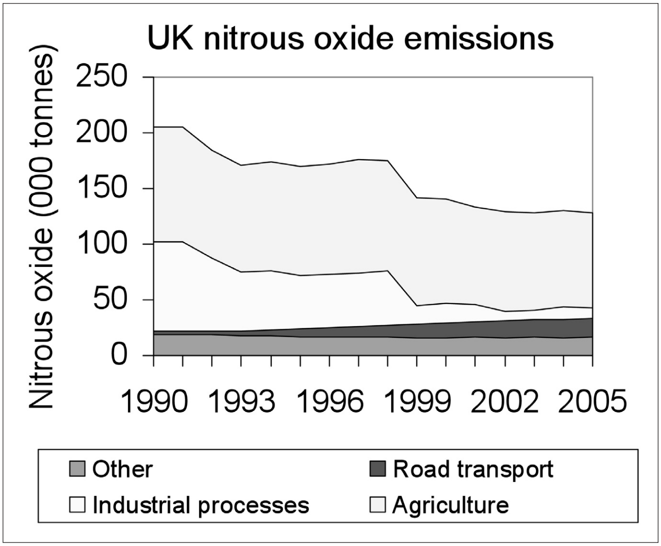


Figure 2: UK emissions of Nitrous oxide between 1990 and 2005 by sector

In soil nitrous oxide is produced by two microbial processes: nitrification and denitrification (Bremner, 1997). Many experiments in the laboratory and field have shown a linear relationship between nitrogen additions and nitrous oxide emissions. This relationship has been adopted by the Intergovernmental Panel of Climate Change (IPCC) to provide a simple methodology to calculate country scale annual nitrous oxide emissions. For example, it is assumed that 1.25% of the mineral nitrogen fertiliser applied to agricultural soils and 1% of the atmospheric nitrogen deposition is emitted as nitrous oxide (IPCC, 1996). This approach provides a simple methodology for widespread application and a broad picture of the contribution of soils to total emissions. The uncertainties in this estimate, however, are very large, partly because other important environmental variables including rainfall, temperature and land management are not taken into account. This leads to large uncertainties in the emission inventory. In the UK, grasslands are the largest soil source of nitrous oxide and together with manure from housed and grazed animals returned to grassland, this source contributes 74 kt nitrous oxide (>80% of total soil emissions) annually (Figure 2). Grassland emissions of N_2O are larger than those from fertilised arable land when expressed per unit area because a) grasslands receive larger rates of mineral fertiliser and manure application, b) grasslands occur in high rainfall regions, promoting the anaerobic soil conditions necessary for N_2O production, c) grazed grasslands have compacted soils. Such conditions are all favourable for nitrous oxide production (Smith *et al*., 1994). The location of intensively managed grasslands is mainly in the wetter, Western parts of Great Britain, these areas are responsible for larger annual rates of nitrous oxide emissions than eastern areas, (Sozanska *et al*., 2002). The contribution of arable land to the total nitrous oxide emission (8%) is of similar importance as nitrogen deposition derived nitrous oxide emission rate (6%). In Scotland, the high nitrogen input grasslands in the west are important N_2O sources while the fertile soils in Fife and East Lothian, capable of producing a wide variety of arable crops, contribute significantly to the local nitrous oxide emission. Forests, heath and moorlands, while large in area, contribute less than 5% to the total soil emissions.

Methane

At the global scale the main sources of methane are natural wetlands in the northern latitudes and the tropics, enteric fermentation, rice paddies, landfills and natural gas and coal mining industries (Prather *et al*., 2001). Natural wetlands, enteric fermentation of farmed livestock and rice paddies contribute to 29%, 15% and 11% of the global annual emission, respectively. The uncertainty in these estimates unfortunately is high, especially for the wetlands. Natural wetlands are estimated to emit 115 to 237 Mt/year and rice paddies 25 to 100 Mt/year.

Figure 3 illustrates global methane sources both natural and anthropogenic.

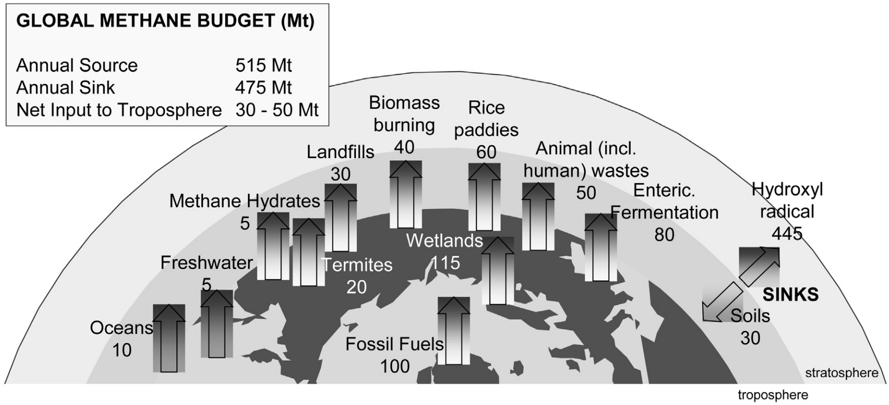


Figure 3: Global sources and sinks of atmospheric methane

Methane is produced by strictly anaerobic microorganisms (methanogens). Production is therefore restricted to micro-habitats where oxygen diffusion is inhibited, for example by high water content in rice paddies or inside the rumen of ruminant livestock. The primary factors controlling the rate of methane production in wetlands and rice paddies are water table height and temperature. Plants play a significant role in the transport of methane from the zones of production to the atmosphere. For example, surface water on a blanket bog in Caithness emitted 10 times more methane when vegetated with *Menyanthes trifoliata* (Bog-bean) compared to adjacent non vegetated parts of the same surface water (MacDonald *et al.*, 1998).

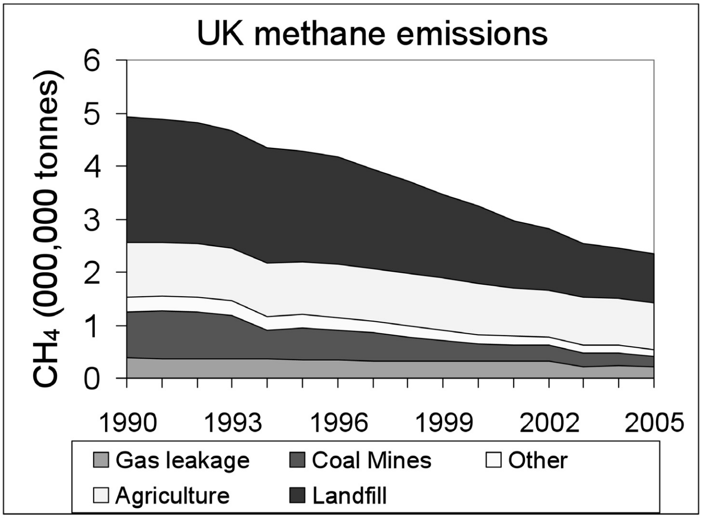


Figure 4: UK sources of methane since 1990, by sector, showing a marked decrease in the coal mining and landfill sectors

Methane and wetlands

The variables controlling biological methane production, transport, oxidation and emissions are reasonably well researched, especially for northern wetlands (MacDonald *et al*., 1998; Hargreaves and Fowler, 1998) and tropical rice cultivation (Wassmann *et al*., 2000). Our understanding of the processes determining methane emissions is sufficient to simulate the temporal variability of methane emissions at small scales. For example, the Denitrification-Decomposition (DNDC) model has been used successfully to predict methane emissions from rice paddies (Pathak *et al*., 2005). However, applying bottom-up, process-based models to the globe is difficult, especially for heterogeneous systems as natural wetlands. Furthermore, recent findings have demonstrated the importance of labile substrate supply (Christensen *et al*., 2003), vegetation and sulphate deposition via acid rain pollution (Gauci *et al*., 2002; 2004).

A number of global-scale methane wetland models have been developed, based on water table height, soil temperature and substrate availability (e.g. Walter *et al*., 2001; Gedney *et al*., 2004). However, there are important discrepancies between model predictions, even at the coarse scale. Observational data with high resolution in space and time are required to discriminate between these alternatives. High quality observational data for CH_4 are available from a network of remote sites distributed throughout the world provide a time series of ground-based concentrations. These data have been used in models to estimate the source strengths and their global distribution using inversion techniques (Dlugokencky, 2001). At best, this resolves the surface flux for coarse latitudinal bands, i.e. northern, tropical and southern latitudes.

In the UK, sources of methane are dominated by land fill and livestock farming activities (Figure 4) Peat bogs are the only significant soil source of methane in the UK and may contribute around 120 kt methane/y. Other ecosystems are only occasional small sources of methane during prolonged wet periods. Soil as a source of methane is not included in the national atmospheric emission inventory. Soils also act as sinks for methane, either from sources located in deeper horizons or from the atmosphere. The soils sink for atmospheric methane is significant at a global scale, but is relatively unimportant in the UK.

Carbon Dioxide

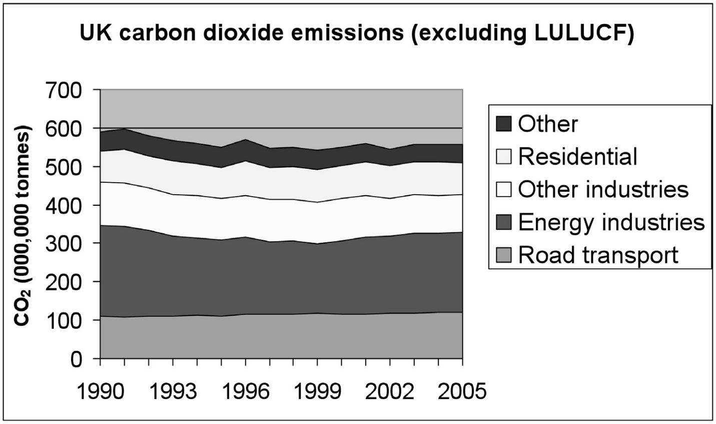


Figure 5: UK emissions of CO_2 since 1990, showing a small decrease in the energy sector and an increase in the transport sector

Global anthropogenic sources of CO_2 total approx 7 Gt C annually in recent years (IPCC 2007), and are dominated by fossil fuel combustion and cement production, however land use change is also associated with net release of CO_2 to the atmosphere, the amounts being approximately 2 Gt C annually (IPCC, 2007). Thus soils are an important reservoir of carbon, and this may accumulate or be depleted depending on land use and climate. Globally, soils hold three times as much carbon as the atmosphere. UK soils hold 4562 Tg of carbon, much of which is held in semi-natural vegetation. Soils act as a sink for carbon dioxide if the input from plant litter exceeds the loss from decomposition or a source if this balance is reversed. In cool, wet climates, where decomposition is slow, carbon accumulates in the form of peat, which comprises 60 % of UK soil carbon. If the climate warms and dries, as predicted, this large store of carbon is vulnerable to being released, with a potential positive feedback loop being created as the additional carbon dioxide adds further to the climatic warming. However, the peatland in the UK is located mainly in the north and west of the country, where climate change may increase annual precipitation, favouring increased sequestration of carbon as peat. It is unclear therefore whether climate change will increase or decrease the net soil to atmosphere exchange of carbon.

Recent work has suggested that carbon is generally being lost from UK soils, possibly at high rates (Bellamy *et al*., 2005), though the amount of climatic warming to date cannot account for the reported losses. Land use change is another possible explanation for this trend. Land use change has a major influence on the soil carbon balance, as the soil structure may be disturbed (e.g. by ploughing) and different vegetation types produce different quantities and qualities of litter. Globally, it is estimated that around 50 Pg C have been emitted to the atmosphere from soils, following conversion of natural, undisturbed land to cultivated, agricultural land (Paustian *et al*., 2000).

Validating Regional and National Emissions

The provision of greenhouse gas emission estimates is generally at the country and annual time scale, and these data are provided both at the UK and at the devolved countries, as illustrated in Figure 6. These data are subject to uncertainty both in magnitude of the annual values and also in spatial variability. It is not very useful to know that the total is incorrect and by how much, it is necessary to know which sector is responsible.

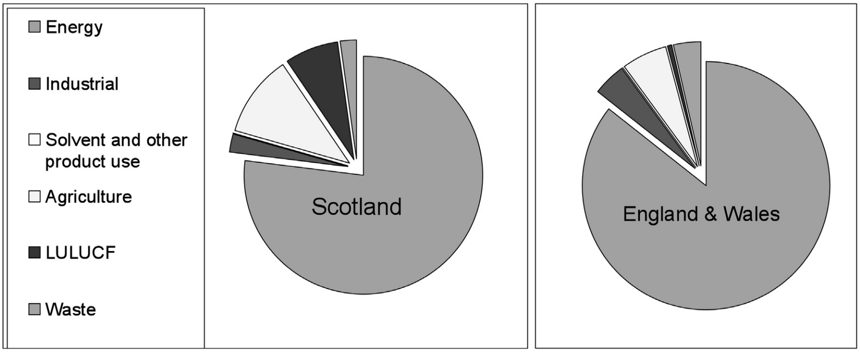


Figure 6: Shows the sector contributions for greenhouse gas emissions, and revealing the dominance of energy related activity overall for Scotland and for England and Wales. LULUCF refers to the contribution of greenhouse gas emissions resulting from land use change

The development of validation methods is at a relatively early stage. This in part results from the processes leading to our current understanding of greenhouse gas emissions and their effects on the global climate. The individual contributions towards the overall knowledge have been mainly in the process science, measuring and modelling individual trace gases. The political developments leading to control strategies for greenhouse gas emissions, all stem from the synthesis provided within the IPCC and the Framework Convention on Climate Change. These processes are now leading towards clear long term targets at the country scale for emission reductions. For example, in the UK, legislation is currently being developed with a target emission reduction in greenhouse gas emissions of 60% (http://www.defra.gov.uk/ENVIRONMENT/climatechange/uk/legislation/index.htm).

Validation from surface measurements using tall towers The 'Keeling Curve' showing 50 years of observations of carbon dioxide concentration measured in Hawaii is something of an icon for this environmentally-aware age. The observations of CO_2 concentration in the air, the increase in its concentration year-on-year and the variability of the annual increases, provide a greater insight to the interactions between anthropogenic emissions and sequestration of CO_2 by the biosphere than any other data series (Conway *et al.*, 1994). Much more recently, networks of 'Tall Towers' have been set up, primarily in Europe and the USA to monitor carbon dioxide and other greenhouse gases on a regular basis and to a very high standard of accuracy (Table 1).

Precision and accuracy goals for greenhouse gas observation in the CHIOTTO project and CarboEurope-IP

Table 1: The precision and accuracy in measurement of trace gases in the CHIOTTO and CARBO EUROPE EU projects. This level of accuracy is required for the validation of inventories using dispersion modeling

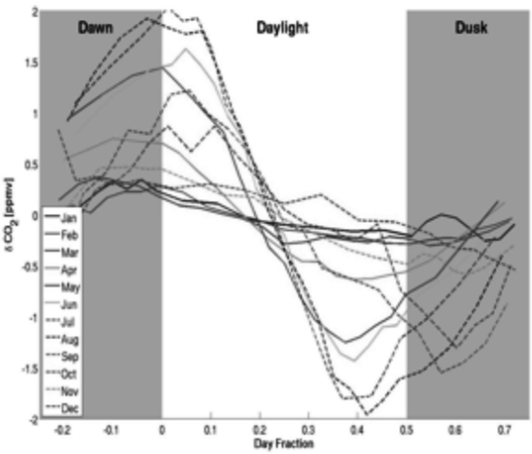
Gas Species	Intra-laboratory instrumental precision	Inter-laboratory calibration scale accuracy	Precision in %
CO_2	0.05 ppm	0.10 ppm	0.01
CH_4	2.0 ppb	3.0 ppb	0.1
CO	1.0 ppb	3.0 ppb	0.3
N_2O	0.1 ppb	0.2 ppb	0.03
SF_6	0.1 ppt	0.2 ppt	1.6
Rn	0.2 Bq m^{-3} or 10%	10%	10

By taking samples of air at a few hundred metres above the ground surface, it is possible to determine the local sources and sinks of these gases in addition to monitoring their rates of change on a global basis. The observations are usually combined with atmospheric transport models to calculate the back trajectories of air parcels that brought any particular air sample to the Tall Tower. By combining observations with atmospheric transport and dispersion, it is possible to calculate budgets for the main greenhouse gases (Bergamaschi *et al*., 2005); it is also possible to model natural and anthropogenic emissions and, in combination with the observations and the transport model, to partition the measured concentrations into the relevant source sectors (Bousquet *et al*., 2006; Vermeulen *et al*., 2006).

For the UK, tall towers have been established very recently and in time will provide the means of validating regional and national emissions. The network is at an early stage will be some time before sufficient data are available for this comparison. However, a long running monitoring station at Mace Head on the west coast of Ireland has provided greenhouse gas measurements as part of an international network since the 1980s (Manning *et al*., 2003; Simmonds *et al*., 1996).

Figure 7: Monthly mean diurnal variations in concentrations of CO_2 revealing the very large diurnal changes during the growing season as a consequence of photosynthesis during the daylight hours and respiration at night

Tall Tower observations can reveal seasonal cycle of photosynthesis/respiration at the country scale. (Observations limited to +/4 hours from noon; δCO_2 is the difference between the 8 am and 4 pm measured concentrations.



The measurements at individual stations may be used in combination with dispersion models to show whether current inventories are consistent with measured concentrations of these long lived gases. The comparison can be achieved in a forward mode, in which the model simply disperses the emissions within a three dimensional field of the airflow and turbulence. In this case, emissions may be increased or decreased to achieve agreement with the model, providing an estimate of the emissions needed to be consistent with surface observations. For the greenhouse gases, this approach using a network of global background stations has been used to evaluate the sources and sinks of methane (Hein *et al*., 1997), and by Kaminski *et al*. (1996) to investigate the annual and seasonal cycles in CO_2 sources and sinks.

Sciamachy and Methane

New satellite data are now available from the satellite-borne SCIAMACHY instrument, which estimate the atmospheric methane concentration at much higher spatial and temporal resolution than previous ground-based air sampling data. For example, SCIAMACHY can clearly detect spatial and temporal variations in methane concentrations and emissions due to rice cultivation, ruminants and wetlands are visible for China and India and the Po valley in Italy (Buchwitz *et al*., 2006). Initial comparisons between SCIAMACHY observations of methane concentrations and those derived from simple emission inventories (Frankenberg *et al*., 2005; Buchwitz *et al.*, 2006) revealed large regional and seasonal differences. Discrepancies were particularly large for the tropical regions, especially the tropical rain forest. These differences can be caused by (1) uncertainty in the interpretation of the satellite data, (2) uncertainty in the models or (3) missing methane sources.

The SCIAMACHY (Scanning Imaging Absorption Spectrometer for Atmospheric Chartography), added to ENVISAT-1 satellite in 2002, is a spectrometer that measures reflected, scattered and transmitted solar radiation and thereby allows the characterisation of the composition of the Earth's atmosphere from the ground to the mesosphere. The data have been analysed to retrieve atmospheric concentrations of CO_2, CO and CH_4 for the year 2003 (Buchwitz *et al*., 2005). Improvements have been made by moving to a different absorption band, located closely to the CO_2 absorption band (Buchwitz *et al*., 2006) and the data have been validated against FTIR concentration measurements (Dils *et al*., 2006).

Validation of regional emissions in the UK using aircraft

The use of aircraft, in combination with atmospheric transport and dispersion models, allows, in suitable conditions, the entire UK emissions over a day to be measured and the measurements compared with the official inventory. The UK is ideally suited to these measurements by virtue of being located at the western boundary of Europe, with the Atlantic ocean providing uniform and reasonably clean upwind atmospheric composition. The aircraft samples this clean 'background' air during westerly airflow off the west coast. This is followed by a transect of the East coast, again within the boundary layer, at heights over the North sea of 300 m to 100 m allows the green house gas additions to the background air to be measured directly, as illustrated in Figure 8.

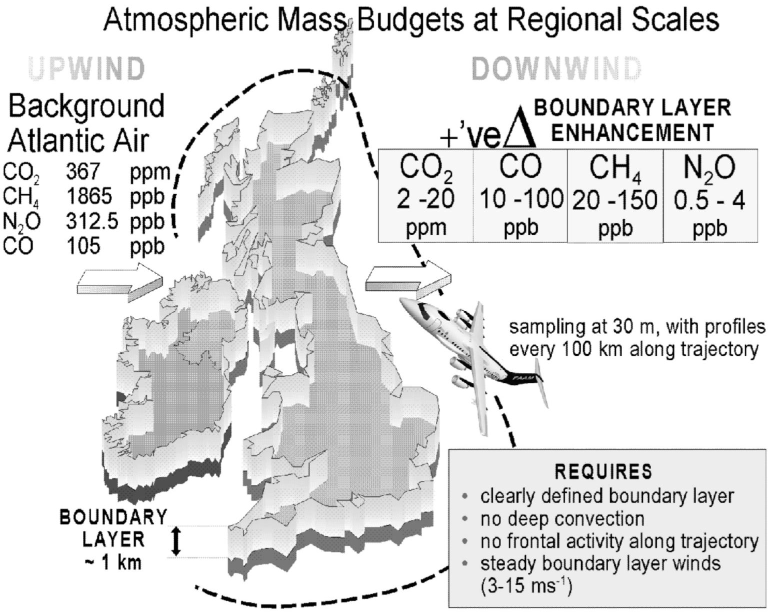


Figure 8: Illustrating the measurement of greenhouse gas emission from the UK using aircraft, the magnitude of the increases in boundary layer concentrations of the greenhouse gases and atmospheric conditions required for the measurements

The atmospheric conditions in which these measurements are possible are provided by an anticyclone to the south of the UK and westerly airflow over the country in the absence of frontal activity or deep convection. Clearly these conditions do not occur every day, and the field measurement programme to develop the method provided 16 days during the six months from May to October 2006.

The boundary layer budget measurements have provided a unique set of measurements of UK scale fluxes of CO, CO_2, CH_4 and N_2O. The measurements of CO were included in the work as this gas has a sufficiently long residence time in the atmosphere to be confident that atmospheric chemistry would not greatly influence the concentrations in the boundary layer over 24 hours. Thus the variability in the measured concentrations observed were due to emissions and dispersion processes.

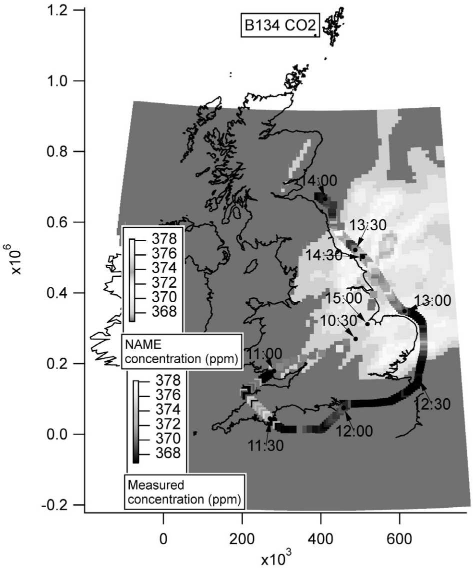


Figure 9: Emissions of CO2 from the UK modelled in south westerly airflow using the NAME model and compared with measurements upwind and down wind of the UK coastline using the NERC aircraft (Polson, 2008)

The measured concentrations may be compared with modelled values using UK spatially disaggregated emission inventories and an atmospheric transport and dispersion model. In this case the model (NAME) developed and provided by the UK Meteorological Office and described by Manning *et al*. (2003) has been used (Figure 9). The model vs. measurement comparison may be used to modify the emissions by sector iteratively to seek the statistically 'best fit' between measurements and model. Using this method Polson (2008) has deduced emissions of the main greenhouse gases emitted by the UK from 16 days of flying in 2006. An alternative to the dispersion model for analysis of the field data is provided by a simple box model. In this method the upwind and downwind edges of the box are used in a simple mass balance study.

The measured emissions of CO and CO_2 agreed well with the UK national inventory, within approximately 8%. For CH_4 and N_2O, the measured values substantially exceeded the inventory, by a factor of two in the case of CH_4, and by a factor of three in the case of N_2O. A key question in this analysis is how representative these 16 days are for the annual emissions of these greenhouse gases. In the case of CO_2 and CH_4, the summer months are not considered to be unrepresentative of the year as a whole, but in the case of N_2O, the temporal variability in emissions is very large and a substantially larger number of days of measurement spread through the seasons and the surface conditions would be needed to reduce the uncertainty in the measured estimate of UK emissions.

However, the data obtained to date demonstrates the applicability of the method.

Table 2: UK emissions of CO, CO_2, CH_4 and N_2O (kt) estimated from aircraft sampling in the boundary layer over the UK combined with either a dispersion model (NAME) or a box model and compared with the UK national inventory of greenhouse gases

	CO	CO_2	N_2O	CH_4
UK Inventory	2,757	572,196	130	1,933
Ireland Inventory	239	43,469	31	607
NAME	2,400 ± 226	560,000 ±	350 ± 208	4,000 ±
Box Approach	2,700 ± 898	670,000 ± 94,000	310 ± 217	4,200 ± 2,130

Inverse modelling

Simulating regional concentration fields and budgets of greenhouse gas emissions using a dispersion model has so far been described using a forwards modelling approach, working from given emissions to the concentrations. An alternative procedure is to work backwards with the model from the measured concentrations to calculate the likely emission footprint on the surface responsible for the observed concentrations. Inverse modelling at a regional scale has been used as a part of the validation process, but as yet is not widely applied. Taking the measurements from the aircraft described above, Polson (2008) has used an inverse modelling approach to estimate the regional distribution of UK sources of the gases measured.

The example presented is for carbon monoxide (CO) simply because emissions of CO and their spatial distribution in the UK are well known. The high quality aircraft measurements allow the technique to be tested before applying it to the other, more challenging trace gases.

An example of inverse modelling is presented in Figure 10, showing the footprint for the enhanced concentration measured in westerly airflow at a point over the North Sea a few km from the Norfolk coastline. The method builds a picture of the footprint of all samples along the East coast. This potentially powerful technique allows the geographical location of discrepancies in the inventory to be located.

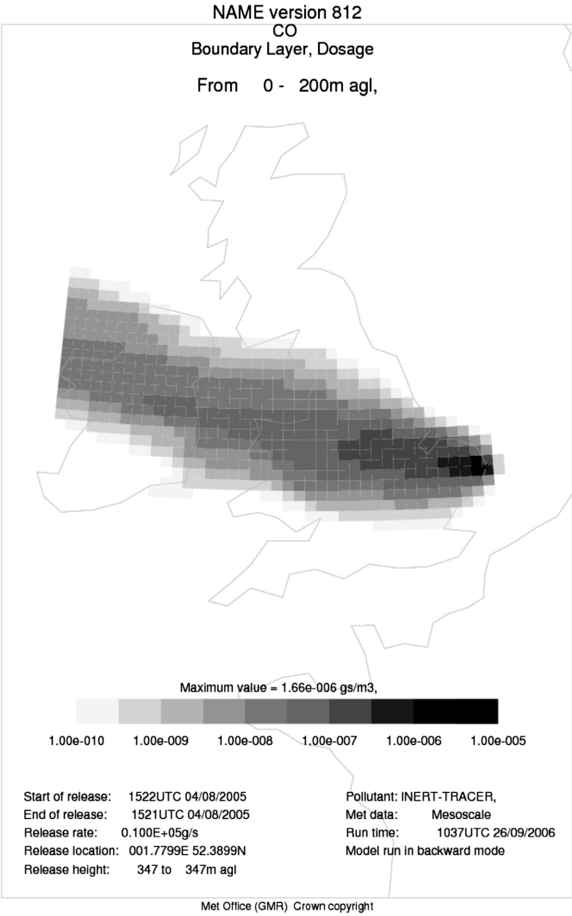


Figure 10: Inverse modelling of trace gas emissions over the UK using NAME

CONCLUSIONS

The high priority given to science underpinning climate change and the political processes in train leading to the delivery of the large scale reductions in emissions of greenhouse gases make this an important area of science. It is surprising given the time that we have been observing the growth in greenhouse gas emissions and concentrations that the methodology to validate national emissions has not been fully developed. The reality is that there are some very powerful methods in development to provide the necessary validation, but these are not fully operational. For atmospheric trace gases which have been subjected to international controls for a few decades, the monitoring and assessment is reasonably mature (EMEP, 2007). There are, therefore, lessons from other control strategies. However, in the case of greenhouse gases, the issues are fully global, they involve all nations and the monitoring and assessment has been focused to date on identifying the problem and attributing the cause to the different contributors to the radiative forcing of climate. As the focus increasingly moves towards the control process, much more attention will be paid to validation, if only because the costs of the controls are so large.

ACKNOWLEDGEMENTS

The authors acknowledge funding from the NERC Polluted Troposphere Thematic Research Programme within the project AMPEP.

REFERENCES

Baggott SL, Brown L, Milne R, Murrells TP, Passant N, Thistlethwaite G, Watterson JD (2005). UK Greenhouse Gas Inventory, 1990 to 2003. Annual report for submission under the Framework Convention on Climate Change. ISBN 0-9547136-5-6. AEAT/ENV/R/2037. National Environmental Technology Centre, AEA Technology plc, B551, Harwell, Didcot, Oxfordshire, OX11 OQJ, UK.

Baggott SL, Brown L, Cardenas L, Downes MK, Garnett E, Hobson M, Jackson J, Milne R, Mobbs, DC, Passant N, Thistlethwaite G, Thomson A, Watterson JD. (2006) “UK Greenhouse Gas Inventory, 1990 to 2004 Annual Report for submission under Framework Convention on Climate Change.” National Environmental Technology Centre, AEA Technology, April 2006.

Bellamy PH, Loveland PJ, Bradley RI *et al*. (2005) Carbon losses from soils across England and Wales 1978–2003. Nature 437, 245–248.

Bergamaschi P, Krol M., Dentener F, Vermeulen A, Meinhardt F, Graul R, Ramonet M, Peters W, and Dlugokencky EJ (2005). Inverse modelling of national and European CH4 emissions using the 5 atmospheric zoom model TM5. Atmos. Chem. Phys. 5, 2431–2460.

Bousquet P, Ciais P, Miller JB, Dlugokencky EJ, Hauglustaine DA, Prigent C, Van der Werf GR, Peylin P, Brunke EG, Carouge C, Langenfelds RL, Lathiere J, Papa F, Ramonet M, Schmidt M, Steele LP, Tyler SC, White J (2006). Contribution of anthropogenic and natural sources to atmospheric methane variability. Nature 443, 439–443.

Bremner JM (1997) Sources of nitrous oxide in soils. Nutrient Cycling in Agroecosystems, 49 (7) 16.

Buchwitz M, de Beek R, Burrows JP, Bovensmann H, Warneke T, Notholt J, Meirink JF, Goede APH, Bergamaschi P, Körner S, Heimann M, and Schulz A (2005a). Atmospheric methane and carbon dioxide from SCIAMACHY satellite data: Initial comparison with chemistry and transport models. Atmos. Chem. Phys., 5, 941–962.

Buchwitz M, de Beek R, No¨el S, Burrows JP, Bovensmann H, Bremer H, Bergamaschi P, K¨orner S, and Heimann M (2005b). Carbon monoxide, methane, and carbon dioxide retrieved from SCIAMACHY by WFM-DOAS: year 2003 initial data set. Atmos. Chem. Phys. 5, 3313–3329.

Buchwitz M, de Beek R, No¨el S, Burrows JP, Bovensmann H, Schneising O, Khlystova I, Bruns M, Bremer H, Berga maschi P, K¨orner S, and Heimann M (2006). Atmospheric carbon gases retrieved from SCIAMACHY by WFM-DOAS: version 0.5

CO and CH_4 and impact of calibration improvements on CO_2 retrieval. Atmos. Chem. Phys. 6, 2727–2751.

Christensen TR, Ekberg A, Ström L, Mastepanov M, Panikov N, Öquist M, *et al*., (2003). Factors controlling large scale variations in methane emissions from wetlands. Geophys Res Lett 30, 67-1 – 67-4 . 10.129/2002GLO16848

Conway TJ, Tans PP, Waterman LS, Thoning KW, Kitzis DR, Masarie K and Zhang N (1994). vidence for interannual variability of the carbon cycle from NOAA/CMDL global sampling network. J. Geophys. Res. 99, 22 831–22 855.

Dils B, De Mazi`ere M, M¨uller JF, Blumenstock T, Buchwitz M, de Beek R, Demoulin P, Duchatelet P, Fast H, Frankenberg C, Gloudemans A, Griffith D, Jones N, Kerzenmacher T, Kramer I, Mahieu E, Mellqvist J, Mittermeier RL, Notholt J, Rinsland CP, Schrijver H, Smale D, Strandberg A, Straume AG, Stremme W, Strong K, Sussmann R, Taylor J, van den Broek M, Velazco V, Wagner T, Warneke T, Wiacek A and Wood S (2006). Comparisons between SCIAMACHY and ground-based FTIR data for total columns of CO, CH_4, CO_2 and N_2O. Atmos. Chem. Phys. 6, 1953–1976.

Dlugokencky EJ, Walter BP, Masarie KA, Lang PM, and Kasischke ES (2001). Measurements of an anomalous global methane increase during 1998. Geophys. Res. Lett. 28, 499-502.

Frankenberg C, Meirink JF, van Weele M, Platt U and Wagner T (2005). Assessing methane emissions from global space-borne observations. Science 308, 1010–1014.

Gauci V, Dise N and Fowler D (2002). Controls on suppression of methane flux from a peat bog subjected to simulated acid rain sulfate deposition. Global Biogeochemical Cycles 16, 1–12.

Gauci V, Matthews E, Dise N *et al*. (2004). Sulfur pollution suppression of the wetland methane source in the 20th and 21st centuries. Proceedings of the National Academy of Sciences USA, 101, 12583– 12587; doi 10.1073/pnas.0404412101

Gedney N, Cox PM and Huntingford C (2004). Climate feedback from wetland methane emissions. Geophys. Res. Lett. 31, L20503. (doi:10.101029/2004GL020919).

Hargreaves KJ and Fowler D (1998). Quantifying the effects of water table and soil temperature on the emission of methane from peat wetland at the field scale. Atmospheric Environment, 32, 3275-3282.

Hein R, Crutzen PJ and Heimann M (1997). An inverse modelling approach to investigate the global atmospheric methane cycle. Global Biogeochem. Cycles 11, 43–76, 1997

Houghton JT *et al*. (eds) (1995). Climate Change 1994, Radiative Forcing of Climate Change, and an Evaluation of the IPCC IS92 Emission Scenarios. Cambridge Univ. Press.

Houghton J (2004). Global Warming - The Complete Briefing. Third Edition. Cambridge University Press, Cambridge, UK.

IPCC (1996). Climate Change 1995: The Science of Climate Change. Cambridge University Press, Cambridge, UK.

Kaminski T, Giering R, Heimann M (1996). Sensitivity of the seasonal cycle of CO_2 at remote monitoring stations with respect to seasonal surface exchange fluxes determined with the adjoint of an atmospheric transport model. Physics and Chemistry of the Earth 21, 457-462

MacDonald JA, Fowler D, Hargreaves KJ, Skiba UM, Leith ID, Murray MB (1998). Methane emission from peat wetlands: response to temperature, water table and transport. Atmospheric Environment 32, 3219-3227

Manning AJ, Ryall DB, Derwent RG, Simmonds PG and O'Doherty S (2003). Estimating European Emissions of ozone-depleting and greenhouse gases using Observations and a Modelling Back-Attribution technique. Journal of Geophysical Research 108, 4405 doi:10,1029/2002JD002312

Pathak H, Li C and Wassmann, R (2005). Greenhouse gas emissions from Indian rice fields: calibration and upscaling using the DNDC model. Biogeosciences 2, 77–102.

Paustian K, Six J, Elliott ET and Hunt HW (2000). Management options for reducing CO2 emissions from agricultural soils. Biogeochemistry 48 (1), 147–163.

Polson D (2008). Aircraft Measurement of Chemical Processing and Export fluxes of Pollutants over the UK (AMPEP) Doctoral Thesis, University of Edinburgh.

Prather M, Ehhalt D *et al*., (2001). Atmospheric chemistry and greenhouse gases, in Climate Change 2001: The Scientific Basis, IPCC Third Assessment Report. Cambridge University Press.

Prather M and Ehhalt D (2001). Atmospheric chemistry and greenhouse gases Climate Change 2001: The Scientific Basis Contribution of Working group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change ed Y D J T Houghton, D J Griggs, M Noguer, P J van der Linden, X Dai, K Maskell and C A Johnson (Cambridge: Cambridge University Press), pp. 239–387.

Prinn R, Cunnold D, Rasmussen R, Simmonds P, Aylea F, Crawford A, Fraser P and Rosen R (1990). Atmospheric emissions and trends of nitrous oxide deduced from 10 years of ALE-GAGE data. Journal of Geophysical Research 95, 18369–18385.

Simmonds PG, Derwent RG, McCulloch A, O'Doherty S and Gaudry A (1996). Long-term trends in concentrations of halocarbons and radiatively active trace gases in Atlantic and European air masses monitored at Mace Head, Ireland from 1987–1994. Atmospheric Environment 30, 4041–4063

Smith KA, Clayton H, Arah JR, Christensen S, Ambus P, Fowler D, Hargreaves KJ, Skiba U, Harris GW, Wienhold FG, Kledmedtsson L and Galle B (1994). Micrometeorological and chamber methods for measurements of nitrous oxide fluxes between soils and atmosphere: Overview and conclusions. Journal of Geophysical Research 99, 16,541-16,548.

Sozanska M (1999). Distribution and Amounts of Nitrous and Nitric Oxide Emissions from British Soils. Doctoral Thesis, University of Edinburgh.

Sozanska M, Skiba U and Metcalfe S (2002). Developing an inventory of N_2O emissions from British soils. Atmospheric Environment Elsevier 36 (6), 987-998.

Vermeulen AT, Pieterse G, Hensen A, van den Bulk WCM and Erisman JW (2006). COMET: a Lagrangian transport model for greenhouse gas emission estimation – forward model technique and performance for methane. Atmos. Chem. Phys. Discuss. 6, 8727-8779.

Walter B, Heimann M and Matthews E (2001). Modelling modern methane emissions from natural wetlands, 2. Interannual variations 1982-1993. J. Geophys. Res.

Wassmann R, Lantin RS, Neue HU, Buendia LV, Corton TM and Lu Y (2000). Characterization of methane emissions from rice fields in Asia. III. Mitigation options and future research needs. Nutr. Cycl. Agroecosys. 58, 23-36.

LAND USE, CLIMATE CHANGE AND GREENHOUSE GAS EMISSIONS

P Smith

Institute of Biological and Environmental Science, School of Biological Sciences, Cruickshank Building, St Machar Drive, University of Aberdeen, Aberdeen AB24 3UU, UK, E-mail: pete.smith@abdn.ac.uk

SUMMARY

Climate change has impacts on soils. Increasing temperatures will tend to increase decomposition but this will be limited where the soil water balance becomes very low. Where increasing temperatures increase net primary production (NPP), carbon inputs to the soil may increase which will work to decrease the direct impact of climate change on soils and may increase soil carbon. Organic soils store large quantities of carbon and some management practices, such as drainage or tillage, can release vast quantities of carbon. Land use affects emissions of greenhouse gases from soils. Further, some land-use changes can increase soil carbon sinks. There is significant potential for climate mitigation from agricultural land management and forestry, with climate mitigation in the land use sector being significant and cost competitive with other sectors. Improved land management may help to mitigate climate change and to reduce the vulnerability of ecosystems to climate change.

INTRODUCTION

Globally, soils contain about three times the amount of C in vegetation and twice the amount in the atmosphere (IPCC, 2000a), i.e. about 1500 Pg (1 Pg = 1 Gt = 10^{15} g) of organic C (Batjes, 1996). The annual fluxes of CO_2 from atmosphere to land (global Net Primary Productivity [NPP]) and land to atmosphere (respiration and fire) are each of the order of 60 Pg C y^{-1} (IPCC, 2000a). During the 1990s, fossil fuel combustion and cement production emitted 6.3 ± 1.3 Pg C y^{-1} to the atmosphere, whilst land-use change emitted 1.6 ± 0.8 Pg C y^{-1} (Schimel *et al*., 2001; IPCC, 2001). Atmospheric C increased at a rate of 3.2 ± 0.1 Pg C y^{-1}, the oceans absorbed 2.3 ± 0.8 Pg C y^{-1} with an estimated residual terrestrial sink of 2.3 ± 1.3 Pg C y^{-1} (Schimel *et al*., 2001; IPCC, 2001). The size of the pool of soil organic carbon (SOC) is therefore large compared to gross and net annual fluxes of C to and from the terrestrial biosphere (Smith, 2004). Small changes in the SOC pool could have dramatic impacts on the concentration of CO_2 in the atmosphere. The response of SOC to global warming is, therefore, of critical importance. One of the first examples of the potential impact of increased release of terrestrial C on further climate change was given by Cox *et al.* (2000). Using a climate model with a coupled C cycle, Cox *et al.* (2000) showed that release of terrestrial C under warming would lead to a positive feedback whereby C release would result in increased global warming. Since then, a number of coupled climate carbon cycle (so called C4) models have been developed. However, there remains considerable uncertainty concerning the extent of the terrestrial feedback, with the difference between the models amounting to 200 ppm CO_2-C by 2100 (Friedlingstein *et al.*, 2006). This difference is of the same order as the difference between fossil fuel C emissions under the IPCC SRES emission scenarios (IPCC, 2000b). It is clear that better quantifying the response of terrestrial C, a large proportion of which derives from the soil, is essential for understanding the nature and extent of the earth's response to global warming. Understanding interactions between climate and land-use change will also be critically important.

LAND USE IMPACTS ON SOIL CARBON AND GREENHOUSE GASES

Historically, soils have lost between 40 and 90 Pg C globally through cultivation and disturbance (Houghton, 1999; Houghton *et al*., 1999; Schimel, 1995; Lal, 1999). It is estimated that land use change emitted 1.6 ± 0.8 Pg C y^{-1} to the atmosphere during the 1990s (Schimel *et al*., 2001; IPCC, 2001). Land use change significantly affects soil C stock (Guo and Gifford, 2002). Most long term experiments on land use change show significant changes in SOC (e.g. Smith *et al*., 1997; 2000; 2001a; 2002). This is likely to continue into the future; in a recent modeling study examining the potential impacts of climate and land use change on SOC stocks in Europe, land use change was found to have a larger net effect on SOC storage than projected climate change (Smith *et al.*, 2005).

In a meta-analysis of long term experiments, Guo and Gifford (2002) showed that converting forest land or grassland to croplands caused significant loss of SOC, whereas conversion of forestry to grassland did not result in SOC loss in all cases. Total ecosystem C (including above ground biomass) does, however, decrease due to loss of the tree biomass C. Similar results have been reported in Brazil, where total ecosystem C losses are large, but where soil C does not decrease (Veldkamp, 1994; Moraes *et al.*, 1995; Neill *et al.*, 1997; Smith *et al.*, 1999), though other studies have shown a loss of SOC upon conversion of forest to grassland (e.g. Allen, 1985; Mann, 1986; Detwiller and Hall, 1988). In the most favorable case, only about 10% of the total ecosystem C lost after deforestation (due to tree removal, burning etc.) can be recovered (Fearnside, 1997; Neill *et al.*, 1997; Smith *et al.*, 1999).

The largest per-area losses of SOC occur where the C stocks are largest, e.g. in highly organic soils such as peatlands, either through drainage, cultivation or liming. Organic soils hold enormous quantities of SOC, accounting for 329-525 Pg C, or 15-35% of the total terrestrial C (Maltby and Immirizi, 1993) with about one fifth (70 Pg) located in the tropics. Studies of cultivated peats in Europe show that they can lose significant amounts of SOC through oxidation and subsidence; between 0.8 and 8.3 t C ha^{-1} y^{-1} (Nykänen *et al*., 1995; Leila *et al*., 2004; Maljanen *et al*., 2001; Maljanen *et al*., 2004). The potential for SOC loss from land use change on highly organic soils is therefore very large.

In short, SOC tends to be lost when converting grasslands, forest or other native ecosystems to croplands, or by draining, cultivating or liming highly organic soils. SOC tends to increase when restoring grasslands, forests or native vegetation on former croplands, or by restoring organic soils to their native condition. Where the land is managed, best management practices that increase C inputs to the soil (e.g. improved residue and manure management) or reduce losses (e.g. reduced impact tillage, reduced residue removal) help to maintain or increase SOC levels. Management practices to increase SOC storage are discussed in the next section.

The most effective mechanism for reducing SOC loss globally would be to halt land conversion to agriculture, but with the population growing and diets changing in developing countries (Smith *et al*., 2007a; Smith and Trines, 2007), more land is likely to be required for agriculture. To meet growing and changing food demands without encouraging land conversion to agriculture will require productivity on current agricultural land to be increased (Vlek *et al.,* 2004). In addition to increasing agricultural productivity, there are a number of other management practices that can be used to prevent SOC loss. These are described in more detail in the next section.

LAND USE CHANGE AND LAND MANAGEMENT TO RESTORE OR SEQUESTER SOC

Soil C sequestration can be achieved by increasing the net flux of C from the atmosphere to the terrestrial biosphere by increasing global C inputs to the soil (via increasing NPP), by storing a larger proportion of the C from NPP in the longer-term C pools in the soil, or by reducing C losses from the soils by slowing decomposition (Smith, 2007). For soil C sinks, the best options are to increase C stocks in soils that have been depleted in C, i.e. agricultural soils and degraded soils, or to halt the loss of C from cultivated peatlands (Smith *et al.,* 2008a).

The most recent estimate (Smith *et al.*, 2008a) is that the technical potential for SOC sequestration globally is around 1.3 Pg C y^{-1}, but this is very unlikely to be realized. Economic potentials for SOC sequestration estimated by Smith *et al.* (2008a) were 0.4, 0.6 and 0.7 Pg C y^{-1} at carbon prices of 0-20, 0-50 and 0-100 USD t CO_2-equivalent^{-1}, respectively. At reasonable C prices, then, global soil C sequestration seems to be limited to around 0.4-0.7 Pg C y^{-1}. Even then, there are barriers (e.g. economic, institutional, educational, social) that mean the economic potential may not be realized (Trines *et al*., 2006; Smith and Trines, 2007). The estimates for C sequestration potential in soils are of the same order as for forest trees, which have a technical potential to sequester about 1 to 2 Pg C y^{-1} (IPCC, 2000a), and economic potential for C sequestration in forestry (in trees and soil) is similar to that for soil C sequestration in agriculture (IPCC WGIII 2007).

Many reviews have been published recently discussing options available for soil C sequestration and mitigation potentials (e.g. IPCC, 2000a; Cannell, 2003; Metting *et al*., 1999; Smith *et al*., 2000; Lal, 2004; Lal *et al*., 1998; Nabuurs *et al*., 1999; Follett *et al*., 2000; Freibauer *et al*., 2004; Smith *et al.*, 2008a). Most of the estimates for the sequestration potential of activities range from about 0.3 to 0.8 t C ha^{-1} y^{-1}, but some estimates are outside this range (IPCC, 2000a; Lal, 2004a; Smith *et al*. 2000; Follett *et al*., 2000; Nabuurs *et al*., 1999; Smith *et al.,* 2008a). When considering soil C sequestration options, it is important also to consider other side effects, including the emission of other greenhouse gases. Smith *et al.* (2001b) suggested that as much as one half of the climate mitigation potential of some C sequestration options could be lost when increased emissions of other greenhouse gases (nitrous oxide; N_2O and methane; CH_4) were included, and Robertson *et al.* (2000) have shown that some practices that are beneficial for SOC sequestration, may not be beneficial when all greenhouse gases are considered. Smith *et al*. (2008a) showed that soil C sequestration accounts for about 90% of the total global mitigation potential available in agriculture by 2030.

CLIMATE CHANGE IMPACTS ON SOIL CARBON AND GREENHOUSE GAS EMISSIONS

The balance between the input of C via photosynthesis and losses by respiration is a key aspect of soil functioning that determines whether a soil is a C source or sink (Gorham, 1991; Freeman *et al*., 2004a). The rate of decomposition of above- and below-ground vegetative matter and litter to soil organic matter (OM), and further decomposition/mineralisation to dissolved organic carbon (DOC) and CO_2, respectively is dependent on a number of environmental factors including temperature, moisture, plant residue composition, and the capacity of the soil to stabilise soil OM (Coleman and Jenkinson, 1996; Moore *et al*., 1998; Martens, 2000; Blanco-Canqui and Lal, 2004). These parameters in turn affect the activity of the soil decomposer community comprising micro-organisms, fungi, and soil invertebrates.

Temperature, in particular, is a major factor regulating decomposition rate; it influences the observed seasonal patterns of higher soil CO_2 concentrations in the summer compared to the colder winter months (Castelle and Galloway, 1990; Jones and Mulholland, 1998; Hope *et al*., 2004). The temperature sensitivity of soil OM decomposition, however, remains a topic for debate (Davidson and Janssens, 2006). The overall response of soil C to global warming will depend upon the balance of increased C inputs to the soil due to increased plant productivity (which will tend to increase SOC stocks) and the increasing rate of decomposition at warmer temperatures (which will tend to decrease SOC stocks; Dawson and Smith (2007)). If warming-induced C emissions from soils exceed vegetation growth, soils could become sources of atmospheric CO_2 (Davidson and Janssens, 2006). It is not yet clear how these opposite effects will balance, either globally, or for specific regions. Though there is some evidence that terrestrial ecosystems in middle and high latitudes of the Northern Hemisphere have functioned as C sinks over the past 20 years (Schimel *et al.*, 2001), different studies from around the world report different trends in SOC stocks over the past decades, with a few authors suggesting that observed SOC losses may be due to climate change. Bellamy *et al.* (2005), for example, suggest a link to climate change to explain their observed mean loss of topsoil SOC of 0.6% yr^{-1}, between 1978 and 2003 in England and Wales. However, Smith *et al.* (2007b), reported modelling results which suggested that, at most, only about 10-20% of recently observed soil C losses in England and Wales could possibly be attributable to climate warming. Elsewhere in the world, increases in SOC have been reported over recent decades. Liao *et al*. (2008) report increases in cropland SOC stocks over the last two decades in China's Jiangsu province, but here the most likely explanation is a large contribution from a change in paddy field management over this period. Small-scale laboratory and field experiments and modelling studies suggest that climate change is likely to induce soil C loss from northern ecosystems (Goulden *et al*., 1998; Melillo *et al*., 2002), but little evidence comes from large-scale observations.

CONCLUSIONS

Soils contain a stock of C that is about twice as large as that in the atmosphere and about three times that in vegetation. Small percentage losses from this large pool could have significant impacts upon future atmospheric CO_2 concentrations, so the response of soils to global warming is of critical importance when assessing climate C cycle feedbacks. Models that have coupled climate and C cycles show a large divergence in the size of the predicted biospheric feedback to the atmosphere. Central questions which still remain when attempting to reduce this uncertainty in the response of soils to global warming are a) the temperature sensitivity of soil OM, especially the more recalcitrant pools, b) the balance between increased C inputs to the soil from increased production and increased losses due to increased rates of decomposition and c) interactions between global warming and other aspects of global change including other climatic effects (e.g. changes in water balance), changes in atmospheric composition (e.g. increasing atmospheric CO_2 concentration) and land-use change. Our tools for studying the response of soil C pools to climate change have improved greatly over recent years. Models have improved and have become more realistic and laboratory and field techniques have given us new tools with which to study the response of soil C to global warming. Projections of climate change have improved the full coupling of the biospheric C cycle into climate models will allow the feedbacks between global warming and soil C carbon responses to warming to be studied. Among the most vulnerable ecosystems are the northern peatlands and other regions either containing large C stocks, or within the permafrost zone, and those in arid regions currently on the verge of desertification. A number of possible technologies to begin to mitigate the worst impacts are available, mainly in managed systems. These technologies, which promote soil C sequestration, will also help to mitigate climate change itself (by reducing atmospheric CO_2 concentration) and are cost competitive with mitigation options available in other sectors. Some warming will occur and it is important that humans adapt management practices to cope with this change, but soils also provide a great opportunity, along with a raft of other measures, to slow that rate of warming. Identifying the “win-win” options that deliver both adaptation to, and mitigation of, climate change, and finding ways to implement these measures, remains one of our greatest challenges for this century (Smith *et al.*, 2008b).

REFERENCES

Allen JC (1985). Soil response to forest clearing in the United States and tropics: geological and biological factors. Biotropica 17, 15-27.

Batjes NH (1996). Total carbon and nitrogen in the soils of the world. European Journal of Soil Science 47, 151-163.

Bellamy PH, Loveland PJ, Bradley RI, Lark RM and Kirk GJD (2005). Carbon losses from all soils across England and Wales 1978–2003. Nature 437, 245-248.

Blanco-Canqui H and Lal R (2004). Mechanisms of carbon sequestration in soil aggregates. Critical Reviews in Plant Science 23, 481–504.

Cannell MGR (2003). Carbon sequestration and biomass energy offset: theoretical, potential and achievable capacities globally, in Europe and the UK. Biomass and Bioenergy 24, 97-116.

Castelle AJ and Galloway JN (1990). Carbon dioxide dynamics in acid forest soils in Shenandoah National Park, Virginia. Soil Science Society of America Journal 54, 252-257.

Coleman K and Jenkinson DS (1996). RothC-26.3- A Model for the turnover of carbon in soil. In Evaluation of Soil Organic Matter Models Using Existing, Long-Term Datasets. DS Powlson, P Smith and JU Smith (Eds.). NATO ASI Series I, Vol. 38, pp. 237-246. Springer-Verlag, Heidelberg, Germany.

Cox PM, Betts RA, Jones CD, Spall SA, Totterdell IJ (2000). Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. Nature 408, 184-187.

Davidson EA and Janssens IA (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature 440, 165-173.

Dawson JJC and Smith P (2007). Carbon losses from soil and its consequences for land management. Science of the Total Environment doi: 10.1016/j. scitotenv.2007.03.023.

Detwiller RP and Hall AS (1988). Tropical forests and the global carbon cycle. Science 239, 42-47.

Fearnside PM (1997). Greenhouse gases from deforestation in Brazilian Amazonia: Net committed emissions. Climatic Change 35, 321-360.

Freibauer A, Rounsevell M, Smith P and Verhagen A (2004). Carbon sequestration in the agricultural soils of Europe. Geoderma 122, 1-23.

Friedlingstein P, Cox P, Betts R, Bopp L, Von Bloh W, Brovkin V, Cadule P, Doney S, Eby M, Fung I, Bala G, John J, Jones C, Joos F, Kato T, Kawamiya M , Knorr W, Lindsay K, Matthews HD, Raddatz T, Rayner P, Reick C, Roeckner E, Schnitzler KG, Schnur R, Strassmann K, Weaver AJ, Yoshikawa C and Zeng N (2006). Climate-carbon cycle feedback analysis: Results from the (CMIP)-M-4 model intercomparison. Journal of Climate 19, 3337-3353.

Follett RF, Kimble JM and Lal R (2000). The potential of U.S. grazing lands to sequester soil carbon. In: Follett, RF, Kimble JM, Lal R (eds). The potential of U.S. Grazing lands to sequester carbon and mitigate the greenhouse effect, Lewis Publishers, Boca Raton, FL, pp. 401-430.

Gorham E (1991). Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. Ecological Applications 1, 182-195.

Goulden ML, Wofsy SC, Harden JW, Harden JW, Trumbore SE, Crill PM, Gower ST, Fries T, Daube BC, Fan SM, Sutton DJ, Bazzaz A and Munger JW (1998). Sensitivity of boreal forest carbon balance to soil thaw. Science 279, 214-217.

Guo L B and Gifford RM (2002). Soil carbon stocks and land use change: a meta analysis. Global Change Biology 8, 345-360.

Hope D, Billett MF and Cresser M S (1994). A review of the export of carbon in river water – fluxes and processes. Environmental Pollution 84**,** 301-324.

Houghton RA (1999). The annual net flux of carbon to the atmosphere from changes in land use 1850 to 1990. Tellus 50B, 298-313.

Houghton RA, Hackler JL and Lawrence KT (1999). The US carbon budget: contributions form land-use change. Science 285, 574-578.

IPCC (2000a). Special Report on Land Use, Land Use Change, and Forestry. Cambridge University Press, Cambridge, UK.

IPCC (2000b). Special Report on Emissions Scenarios. Cambridge University Press, Cambridge, UK.

IPCC (2001). Climate Change: The Scientific Basis. Cambridge University Press, Cambridge, UK.

IPCC WGIII (2007). Summary for Policy Makers. Working Group III contribution to the Intergovernmental Panel on Climate Change Fourth Assessment Report. Climate Change 2007: Mitigation of Climate Change. Cambridge University Press, Cambridge.

Jones JB and Mulholland PJ (1998). Carbon dioxide variation in a hardwood forest stream: an integrative measure of whole catchment soil respiration. Ecosystems 1, 183-196.

Lal R (1999). Soil management and restoration for C sequestration to mitigate the accelerated greenhouse effect. Progress in Environmental Science 1, 307-326.

Lal R (2004). Soil carbon sequestration impacts on global climate change and food security. Science 304, 1623-1627.

Lal R, Kimble JM, Follet RF and Cole C V (1998). The potential of U.S. cropland to sequester carbon and mitigate the greenhouse effect. Ann Arbor Press, Chelsea, MI.

Liao QL, Zhang XH, Li ZP, Pan GX, Smith P and Wu XM (2008). Increases in cropland soil organic carbon stocks over the last two decades in China's Jiangsu Province. Global Change Biology (in press).

Lohila A, Aurela M, Tuovinen JP and Laurila T (2004). Annual CO_2 exchange of a peat field growing spring barley or perennial forage grass. Journal of Geophysical Research 109, D18116, doi:10.1029/2004JD004715.

Maljanen M, Martikainen PJ, Walden J and Silvola J (2001). CO_2 exchange in an organic field growing barley or grass in eastern Finland. Global Change Biology 7, 679-692.

Maljanen M, Komulainen VM, Hytonen J, Martikainen P and Laine J (2004). Carbon dioxide, nitrous oxide and methane dynamics in boreal organic agricultural soils with different soil characteristics. Soil Biology and Biochemistry 36, 1801-1808.

Maltby E and Immirzi CP (1993). Carbon dynamics in peatlands and other wetlands soils: regional and global perspective. Chemosphere 27, 999-1023.

Mann LK (1986). Changes in soil carbon storage after cultivation. Soil Science 142, 279-288.

Melillo J M, Steudler P A, Aber J D, Newkirk K, Lux H, Bowles F P, Catricala C, Magill A, Ahrens T and Morrisseau S (2002). Soil warming and carbon-cycle feedbacks to the climate system. Science 298, 2173-2176.

Metting FB, Smith JL and Amthor JS (1999). Science needs and new technology for soil carbon sequestration. In: Rosenberg N J, Izaurralde R C, Malone E L (eds) Carbon sequestration in soils: Science, monitoring and beyond, Battelle Press, Columbus, Ohio, pp. 1-34.

Moore TR, Roulet NT and Waddington JM (1998). Uncertainty in predicting the effect of climatic change on the carbon cycling of Canadian peatlands. Climatic Change 40, 229-245.

Moraes JFL de, Volkoff B, Cerri CC and Bernoux M (1995). Soil properties under Amazon forest and changes due to pasture installation in Rondônia, Brazil. Geoderma 70, 63-86.

Nabuurs GJ, Daamen WP, Dolman AJ, Oenema O, Verkaik E, Kabat P, Whitmore AP and Mohren GMJ (1999). Resolving issues on terrestrial biospheric sinks in the Kyoto Protocol. Dutch National Programme on Global Air Pollution and Climate Change, Report 410 200 030 (1999).

Neill C, Melillo JM, Steudler PA, Cerri CC, Moraes JFL de, Piccolo MC and Brito M (1997) Soil carbon and nitrogen stocks following forest clearing for pasture in the Southwestern Brazilian Amazon. Ecological Applications 7, 1216-1225.

Nykänen H, Alm J, Lang K, Silvola J and Martikainen PJ (1995). Emissions of CH_4, N_2O and CO_2 from a virgin fen and a fen drained for grassland in Finland. Journal of Biogeography 22, 351-357.

Robertson GP, Paul EA and Harwood RR (2002). Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. Science 289, 1922-1925.

Schimel DS (1995). Terrestrial ecosystems and the carbon-cycle. Global Change Biology 1, 77-91.

Schimel DS, House JI, Hibbard KA, Bousquet P, Ciais P, Peylin P, Braswell BH, Apps MJ, Baker D, Bondeau A, Canadell J, Churkina G, Cramer W, Denning AS, Field CB, Friedlingstein P, Goodale C, Heimann M, Houghton RA, Melillo JM, Moore B, Murdiyarso D, Noble I, Pacala SW, Prentice IC, Raupach MR, Rayner PJ, Scholes RJ, Steffen WL and Wirth C (2001). Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. Nature 414, 169-172.

Smith JU, Smith P, Wattenbach M, Zaehle S, Hiederer R, Jones RJA, Montanarella L, Rounsevell M, Reginster I and Ewert F (2005). Projected changes in mineral soil carbon of European croplands and grasslands, 1990-2080. Global Change Biology 11, 2141–2152.

Smith P (2004). Soils as carbon sinks - the global context. Soil Use and Management 20, 212-218.

Smith P (2007). Land use change and soil organic carbon dynamics. Nutrient Cycling in Agroecosystems doi: 10.1007/s10705-007-9138-y.

Smith P and Trines E (2007). Agricultural measures for mitigating climate change: will the barriers prevent any benefits to developing countries? International Journal of Agricultural Sustainability 4, 173-175.

Smith P, Powlson DS, Glendining MJ and Smith JU (1997). Potential for carbon sequestration in European soils: preliminary estimates for five scenarios using results from long-term experiments. Global Change Biology 3, 67-79.

Smith P, Falloon P, Coleman K, Smith JU, Piccolo M, Cerri CC, Bernoux M, Jenkinson DS, Ingram JSI, Szabó J and Pásztor L (1999). Modelling soil carbon dynamics in tropical ecosystems. In: Lal R, Kimble J M, Follett R F and Stewart B A (eds) Global Climate Change and Tropical Soils. Advances in Soil Science, 341-364.

Smith P, Powlson DS, Smith JU, Falloon PD and Coleman K (2000). Meeting Europe's climate change commitments: quantitative estimates of the potential for carbon mitigation by agriculture. Global Change Biology 6, 525-539.

Smith P, Falloon P, Smith JU and Powlson DS (eds) (2001a). Soil Organic Matter Network (SOMNET): 2001 Model and Experimental Metadata, GCTE Report 7 (2nd Edition), GCTE Focus 3 Office, Wallingford, Oxon, 224pp.

Smith P, Goulding KW, Smith KA, Powlson DS, Smith JU, Falloon P and Coleman K (2001b). Enhancing the carbon sink in European agricultural soils: including trace gas fluxes in estimates of carbon mitigation potential. Nutrient Cycling in Agroecosystems 60, 237-252.

Smith P, Falloon PD, Körschens M, Shevtsova LK, Franko U, Romanenkov V, Coleman K, Rodionova V, Smith JU and Schramm G (2002). EuroSOMNET - a European database of long-term experiments on soil organic matter: the WWW metadatabase. Journal of Agricultural Science, Cambridge 138, 123-134.

Smith P, Martino D, Cai Z *et al.* (2007a). Policy and technological constraints to implementation of greenhouse gas mitigation options in agriculture. Agriculture, Ecosystems and Environment 118, 6-28.

Smith P, Chapman SJ, Scott WA, Black HIJ, Wattenbach M, Milne R, Campbell CD, Lilly A, Ostle N, Levy P, Lumsdon DG, Millard P, Towers W, Zaehle S and Smith JU (2007b). Climate change cannot be entirely responsible for soil carbon loss observed in England and Wales, 1978-2003. Global Change Biology 13, 2605-2609.

Smith P, Martino D, Cai Z *et al.* (2008a). Greenhouse gas mitigation in agriculture. Philosophical Transactions of the Royal Society, B. 363, 789-813.

Smith P, Fang C, Dawson J J C and Moncreiff J B (2008b). Impact of global warming on soil organic carbon. Advances in Agronomy 97: (in press).

Trines E, Höhne N, Jung M, Skutsch M, Petsonk A, Silva-Chavez G, Smith P, Nabuurs GJ, Verweij P and Schlamadinger B (2006). Integrating agriculture, forestry, and other land use in future climate regimes: Methodological issues and policy options.

Netherlands Environmental Assessment Agency, Climate Change - Scientific Assessment And Policy Analysis. Report 500102 002, 154 pp.

Veldkamp E (1994). Organic carbon turnover in three tropical soils under pasture after deforestation. Soil Science Society of America Journal 58, 175-180.

Vlek PLG, Rodríguez-Kuhl G and Sommer R (2004). Energy use and CO_2 production in tropical agriculture and means and strategies for reduction or mitigation. Environment, Development and Sustainability 6, 213-233.

MITIGATION OF NITROUS OXIDE AND METHANE EMISSIONS FROM AGRICULTURAL SOILS – A SUMMARY OF SAC EXPERIENCE

BC Ball[1], RM Rees[1] and AH Sinclair[2]

[1]SAC, West Mains Road, Edinburgh, EH9 3JG, UK, E-mail: Bruce.Ball@sac.ac.uk; [2]SAC, Craibstone Estate, Aberdeen, AB21 9YA, UK

SUMMARY

Nitrous oxide (N_2O) and methane (CH_4) are important greenhouse gases as they have a much greater global warming potential than carbon dioxide. We concentrate on N_2O as much more N_2O is emitted from agricultural soils than CH_4. Nitrous oxide emissions are highly dependent on crop type and locality, with soils under dairy grazing emitting most. Although some radical proposals like moving high N input cropping to drier areas are proposed, many other mitigation alternatives are possible, centred on the more efficient use of fertiliser and organic manure inputs. Other mitigation is possible by appropriate management of tillage, compaction and crop residues. Awareness of the need to align mitigation of gas emissions with leaching losses and other aspects of the 'carbon-nitrogen footprint' are important.

INTRODUCTION

Emissions of N_2O from agricultural soils contribute 77% of total N_2O emissions in Scotland, and 53% of the greenhouse gas emissions from the agriculture sector (Jackson *et al*., 2007). Most methane emissions are from cattle and sheep production. However, soils represent a small but significant sink for atmospheric CH_4 (Smith *et al*., 2000). The global warming potential of methane is 25 times greater than carbon dioxide and that of nitrous oxide is 298 times greater than carbon dioxide. Nitrous oxide also enhances depletion of the ozone layer in the upper atmosphere. Both gases are produced and/or consumed in the soil by microorganisms so that manipulation of soil conditions can influence their production. Thus assessment of the likely fluxes of these gases from soil and their mitigation are clear priorities in the abatement of climate change.

Land use and agricultural management have a strong influence on exchange of these greenhouse gases. SAC has extensive research experience which is ongoing in this area. The purpose of this paper is to summarise some of this experience and to identify and review the options for mitigation and reduction of N_2O and CH_4 emissions from soils in Scotland. Specific suggestions for mitigation in terms of modified management are given.

MATERIALS AND METHODS

Measurements of N_2O emissions from fields and field experiments in agricultural systems in Scotland have been made using static field chambers with the method described by Clayton *et al.* (1994). The chambers were sampled every few days through the growing season and more frequently following fertiliser application. Studies included the influence of crop type, grazing management, tillage, compaction, horticulture, organic farming, outdoor pig production and cattle on woodchip corrals.

Mineral and organic fertilisers, including slurry and sewage sludge, were included. Some of these data were upscaled using GIS spatial model at the national scale (Lilly *et al*., 2008). Measured data were used in calculating annual emission rates for the main agricultural land-use categories by summing the relevant daily emission values and using the soil wetness data and management operation dates to determine the numbers of days with 'wet' or 'dry' soil conditions. Methane emission and uptake were measured using the same static field chambers and studies on emissions after application of manures and uptake in different land uses were made.

RESULTS

Nitrous oxide emissions vary widely both spatially and temporally. Table 1 shows the range of calculated annual nitrous oxide emissions for each cultivated land-use type. Within each category the range of annual emissions is determined by climate, with the lower values in cooler, drier areas and the higher values in warmer, wetter areas.

Table 1: Range of annual nitrous oxide emissions from major cultivated land use crop types under different climate zones and soil wetness classifications.

Cultivated land use category		Annual emissions (kg N ha^{-1} yr^{-1})
Dairy Grass	Grazed	3.4 – 13.6
	Ungrazed	1.6 – 6.5
Beef Grass	Grazed	1.7 – 5.3
	Ungrazed	1.0 – 3.8
Winter Cereals		0.6 – 0.9
Spring Cereals		0.8 – 1.3
Root Crops		2.3 – 6.6
Extensive Grazing		0.5 – 0.9

Nitrous oxide emissions vary widely both with crop type and climate (Table 1). These data have been used to produce a map of nitrous oxide emissions for Scotland. This shows 'hot spot' areas of emission where dairy grazing and warm, moist soils coincide (Lilly *et al*., 2008). A consequence of this soil type and climatic interaction is that nitrous oxide emissions from grazed systems at Crichton farm, near Dumfries, are nearly ten times greater than from grazed mixed arable/grassland systems near Bush (Midlothian), for similar N inputs (ca. 200-250 kg N/ha). Thus, the first and most radical mitigation or adaptation is to re-locate high N input cropping to drier, cooler areas and to reduce the need for grazing by reducing intensity of animal production (Table 2). Relocating intensive dairies to Nitrate Vulnerable Zones (NVZs) in the east of Scotland will exacerbate the shortage of slurry storage that will result from the proposed amendment to the Action Programme for NVZ of a minimum of 22 weeks capacity (Scottish Executive Environment Group, 2006).

For a given crop and location, nitrous oxide emissions generally increase with N input. Thus a relatively simple way to reduce emissions is to reduce the amount of fertiliser applied. This may be achieved at no cost to productivity by following fertiliser recommendations. Since urea fertiliser is mobilised less quickly than nitrate in cooler conditions, urea fertiliser can be used early in the season (Dobbie and Smith, 2003).

Table 2: Options to mitigate emissions of nitrous oxide from soils by changing land use or by improving N use efficiency and changing tillage. These options include conclusions drawn by other groups of researchers working in Scotland

Type of change	Options
Land use	• Re-locate high N input cropping (e.g. dairy grazing) to drier, cooler areas • Reduce intensity of animal production, by having fewer animals and lower N inputs • Adopt organic farming - or its principles, using biological fixation to provide N inputs
Use less N fertiliser	• Reduce fertiliser N input, e.g. by following fertiliser recommendations, reduce losses by applying in dry conditions • Use urea fertiliser early in the season • Use nitrification or urease inhibitors or slow release fertiliser • Substitute for mineral fertiliser with organic manures or legumes
Use organic fertiliser	• Inject slurry. Separate slurry applications from fertiliser applications by several days • Use composts, straw-based manures in preference to slurry • Mix nitrogen rich crop residues with other residues of higher C:N ratio • Apply organic manures to arable land rather than to grassland
Use appropriate tillage	• Avoid no-tillage, consider occasional deep ploughing • Plough in early spring, spread crop residues evenly and control compaction • Maintain crop cover over winter
Soil condition	• Maintain the drainage system • Keep pH at an optimum for plant growth; pH 5.8–6.0 for grassland and pH 6.1–6.3 for arable in mineral soils

Nitrous oxide is mainly produced from nitrate, so the rate of production can also be decreased by use of nitrification inhibitors or the rate of release slowed down by use of slow release fertiliser. A risk with the use of slow release fertilisers is the potential for nitrate to be released after the crop demand for nitrogen has declined and soils are becoming wetter, leading to an increase in leaching loss of nitrate to watercourses.

Substitution of mineral fertiliser with organic manures is efficient in reducing the fossil fuel input associated with fertiliser manufacture. However, manures require careful management as N_2O losses from soils treated with liquid manures can be substantial (Ball *et al*., 2004). Jones *et al.* (2007) showed that the benefits of increased carbon sequestration associated with slurry applications were offset by high N_2O emissions, which resulted in large net negative global warming potentials. Composting is also efficient at mitigating nitrous oxide emission. In the above experiment the emission factor (the ratio of overall nitrous oxide emission to available N applied) was less under composted sludge (1.8) than under either dried sewage sludge pellets (2.1) or liquid sewage sludge (2.6).

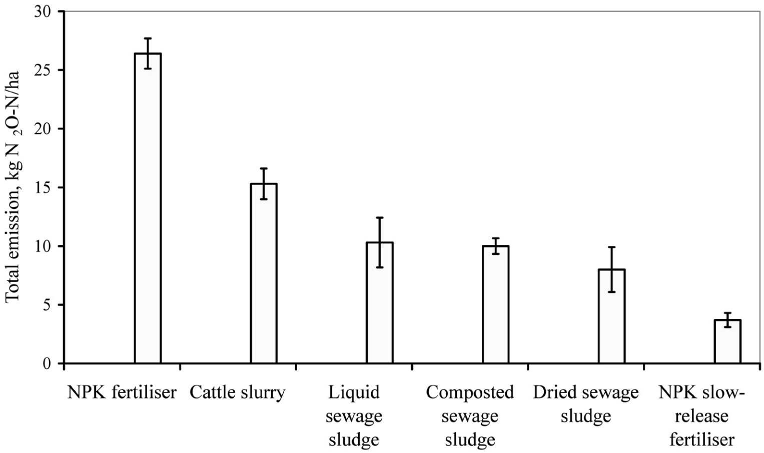


Figure 1: Cumulated N_2O emissions from repeatedly applied treatments (5 applications over 3 years) of alternatives to conventional mineral fertiliser on silage grass at Bush, near Edinburgh

Substitution of fertilisers with legumes that fix N in the soil is the main source of fertility in organic farming. Average nitrous oxide emissions from grass-clover leys under sheep-grazing in the organic experiment at Tulloch, near Aberdeen are low, typically less than 0.5 kg N_2O-N/ha/year. However, in organic systems, the N stored during the grass-clover ley is vulnerable to loss after spring ploughing or tillage, particularly when the soil is moist and warm when losses of up to 2 kg N_2O-N/ha can occur in the month after tillage. No-tillage or minimum tillage are possible alternatives, for conventional cereal production. However, use of no-tillage may increase N_2O fluxes and require greater N inputs to maintain yield than normal ploughing. This occurred at a site near Bush on imperfectly drained soil under spring barley, but not with winter barley, and was attributed to restricted aeration. The reduction in CO_2 under no-tillage was insufficient to make up for the overall increase in global warming potential resulting from the increased N_2O fluxes. Annual nitrate leaching losses at this site were small (6.4-19.6 kg/ha) with no significant treatment effects. Deep ploughing appeared to be the most environmentally friendly treatment, promoting the fixation of organic matter in the subsoil (Nieder *et al*., 2005). However, deep ploughing to incorporate organic residues may result in enhanced N_2O emissions (Baggs *et al*., 2002).

Compaction can influence N_2O fluxes, though compaction needs to be severe enough to create adverse conditions for plant growth before fluxes are increased.

Table 3: Nitrous oxide and carbon dioxide emissions under different tillage systems under spring and winter barley between March and June 1998. Fertiliser N was applied at 80 kg/ha under spring barley and 120 kg/ha under winter barley (from Vinten *et al*., 2002)

Tillage	Barley crop	N_2O emissions (kg N/ha)	CO_2 emissions (kg C/ha)
No tillage	Spring	7.76	1350
	Winter	1.89	2520
Normal ploughing	Spring	2.61	678
	Winter	1.96	4010
Deep ploughing	Spring	1.36	910
	Winter	2.06	3280

Methane emissions from soils are mainly from land spreading of wastes and are short lived (1-3 days). Perhaps of more importance is to maintain the uptake of atmospheric CH_4 by reducing fertiliser and tillage use whilst maintaining a well-aerated soil.

DISCUSSION

These mitigation options are mostly 'win-win'; improving the usage of nitrogen inputs and reducing their losses saves money. Choosing tillage timing and depth to conserve nitrogen stored in soils and crop residues is important. This is an area requiring further exploration, particularly in relation to organic systems and systems where mineral fertiliser use is being reduced by using inputs from legumes and residues. Mitigation options for N_2O are best concentrated in areas of potential 'hot spot' emission i.e. heavier soils in wet areas. Overall management choices should be made bearing in mind the overall 'carbon-nitrogen footprint' of the system. This might be helped by including consideration of both gaseous and leaching losses within fertiliser recommendations in the future.

ACKNOWLEDGEMENTS

This work was sponsored by the Scottish Government RERAD (Rural and Environment Research and Analysis Directorate) and the EU 'NitroEurope' project.

REFERENCES

Ball BC, McTaggart IP and Scott A (2004). Mitigation of greenhouse gas emissions from soil under silage production by use of organic manures or slow release fertiliser. Soil Use and Management 20, 287-295.

Baggs EM, Rees RM, Castle K, Scott A, Smith KA and Vinten AJA (2002). Nitrous oxide release from soils receiving N-rich crop residues and paper mill sludge in eastern Scotland. Agriculture, Ecosystems and Environment 90, 109-123.

Clayton H, Arah JRM and Smith K A (1994) Measurement of nitrous oxide emission from fertilised grassland using closed chambers. Journal of Geophysical Research-Atmospheres, 99, 16599-16607.

Dobbie, KE and Smith, KA (2003) Impact of different forms of N fertiliser on N2O emissions from intensive grassland. Nutrient Cycling in Agroecosystems 67, 37-46.

Jackson J, Li Y, Passant N, Thistlethwaite G, Thomson A and Cardenas L (2007). Greenhouse gas inventories for England, Scotland, Wales and Northern Ireland: 1990-2005. AEA Technology PLC, Oxon, pp. 40.

Jones SK, Rees RM, Skiba UM and Ball BC (2007) Influence of organic and mineral N fertiliser on N_2O fluxes from a temperate grassland. Agriculture Ecosystems and Environment 121, 74-83.

Lilly A, Ball BC, McTaggart IP and DeGroote J (2008). Spatial modelling of nitrous oxide emissions at the national scale using NitOx: the role of soils and land use. Submitted to Global Change Biology.

Nieder R, Kersebaum KC and Richter J (1995). Significance of nitrate leaching and long-term N immobilization after deepening the plow layers for the N regime of arable soils in NW Germany.

Scottish Executive Environment Group (2006). Action Programme for Nitrate Vulnerable Zones: Proposed amendments, Scottish Executive, Edinburgh.

Smith KA, Dobbie KE, Ball BC, Bakken LR, Sitaula BK, Hansen S, Brumme R, Borken W, Christensen S, Prieme A, Fowler D, MacDonald JA, Skiba U, Klemedtsson L, Kasimir-Klemedtsson A, Degórska A and Orlanski P (2000). Oxidation of atmospheric methane in Northern European soils, comparison with other ecosystems, and uncertainties in the global terrestrial sink. Global Change Biology 6, 791-803.

Vinten AJA, Ball BC, O'Sullivan MF and Henshall JK (2002). The effects of cultivation method, fertilizer input and previous sward type on organic C and N storage and gaseous losses under spring and winter barley following long-term leys. Journal of Agricultural Science 139, 231-243.

IMPLICATIONS OF FARM-SCALE METHANE MITIGATION MEASURES FOR UK METHANE EMISSIONS

U Dragosits[1], DR Chadwick[2], A del Prado[2], D Scholefield[2], JAN Mills[3], LA Crompton[3] and CJ Newbold[4]

[1]CEH Edinburgh, Bush Estate, Penicuik, Midlothian, EH26 0QB, UK, E-mail: ud@ceh.ac.uk; [2]IGER North Wyke, Okehampton, Devon, EX20 2SB, UK; [3]School of Agriculture, Policy and Development, University of Reading, Whiteknights, PO Box 237, Reading, RG6 6AR, UK; [4]Institute of Rural Sciences, The University of Wales, Aberystwyth, Ceredigion, SY23 3AL, UK

SUMMARY

Agriculture contributes ~40% of the total UK's emissions of methane (CH_4), mostly from enteric fermentation by ruminant livestock, with a smaller contribution associated with manure management. A number of CH_4 mitigation measures have been identified, but their effectiveness over broad spatial scales had not previously been investigated. Another question was whether widespread implementation would have consequences on production levels and emissions of other pollutants, such as ammonia (NH_3), nitrous oxide (N_2O), or leaching of nitrate (NO_3^-).

This project brought together models from rumen processes to the individual animal level, as well as at the herd, farm and national scale, for the first time (Chadwick *et al*., 2007). Emissions of CH_4, NH_3, N_2O, NO_x and NO_3^- leaching were quantitatively assessed for dairy cattle, beef cattle and sheep. Increasing milk yield in dairy cows (with associated reduction in numbers) results in the largest decrease in CH_4, with comparable decreases in N pollutants >20%. For beef cattle and sheep, the most effective CH_4 mitigation method is vaccination to reduce rumen methanogens by approximately 10%.

INTRODUCTION

Methane is the second most important greenhouse gas (GHG) after carbon dioxide (CO_2), contributing 20% to global warming. Agriculture accounts for ~40% of the UK's emissions of CH_4. In the UK GHG inventory, 85% of the agricultural CH_4 emissions are estimated to originate from enteric fermentation (39% dairy, 48% beef, 22% sheep), with the remaining 15% associated with manure management. Under the Kyoto Protocol the target is to reduce GHG emissions by 12.5% of the 1990 levels by 2008-2012, although this is now under re-negotiation.

A number of CH_4 mitigation measures has been identified for livestock sources, but there is a need to investigate their effectiveness over broad spatial scales, and whether widespread implementation would have other consequences, e.g. for production levels and emissions of other pollutants, such as ammonia (NH_3), nitrous oxide (N_2O), or leaching of nitrate (NO_3^-).

Potentially effective measures for reducing CH_4 emissions from ruminant livestock farming in the UK include:

- Increased productivity per dairy cow, i.e. increased milk production per kg CH_4 produced
- Increased fertility, i.e. reducing the number of followers required
- Improved forage composition and balanced energy/protein feeds
- Feed additives – to reduce rumen hydrogen production
- Vaccination – to reduce the rumen methanogens

In the modelling study presented here, the effectiveness of each of these methods is quantified at different scales, through spatial scenario exploration with a new modelling framework which links four existing models, at the rumen, herd/farm and national level.

METHODS

An overview of the modelling framework used in the study is represented in Figure 1, showing the links between the different models and scales. The rumen model of Reading University (e.g. Mills *et al*., 2001; 2003) generated CH_4 emissions from enteric fermentation for ruminant livestock under a range of intensities. Separate model estimates were obtained for three typical dairy farming typologies (extended grazing, conventional intensive and fully-housed intensive management), and upland and lowland farming systems for beef cattle and sheep, respectively (derived from IGER/ADAS, 2004a; 2004b; 2005; Defra, 2000; Smith *et al*., 2001). The effects of various mitigation strategies on CH_4 emissions were then predicted using a herd level model (Mottram and Mills, 2003), which allows herd management decisions and fertility factors to be incorporated.

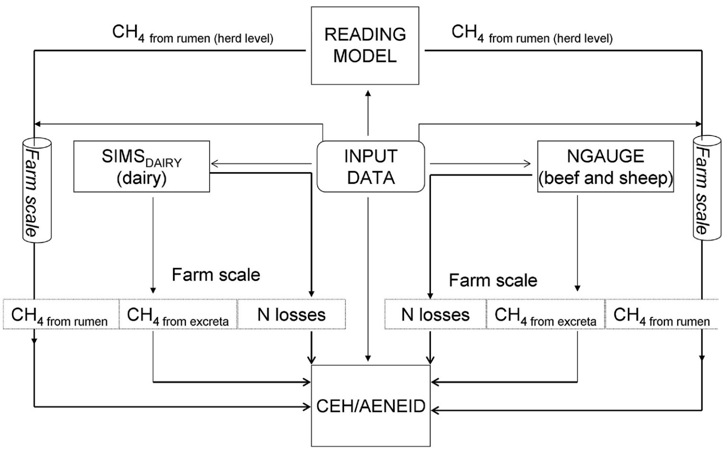


Figure 1: Schematic representation of the modelling system

In a second step, a single map was generated comprising a simplified overlay of climate, soil texture and altitude data, resulting in 121 zones (Figure 2a). Expert judgement was then used to apportion the dairy, beef and sheep typologies to these different zones. For example, it was assumed that 80% of dairy cows in the soil-climate zones in SW England and Wales were in the intensive conventional typology, 20% were in the extended grazing typology and 0% was in the fully housed intensive typology. The $SIMS_{DAIRY}$ (del Prado and Scholefield, 2006) and NGAUGE (Brown *et al*., 2005) models were then used to simulate emissions of CH_4 from manure management as well as emissions of NH_3 and N_2O and NO_3^- leaching, according to soil, agro-climatic factors and farm management for each the typologies (as well as the mitigation options) for all zones. Although CH_4 emissions were not assumed to be influenced by soil or climate, it was necessary to take the spatial variability of soil/climate into account when modelling changes in N_2O emissions and NO_3^- leaching due to CH_4 mitigation measures.

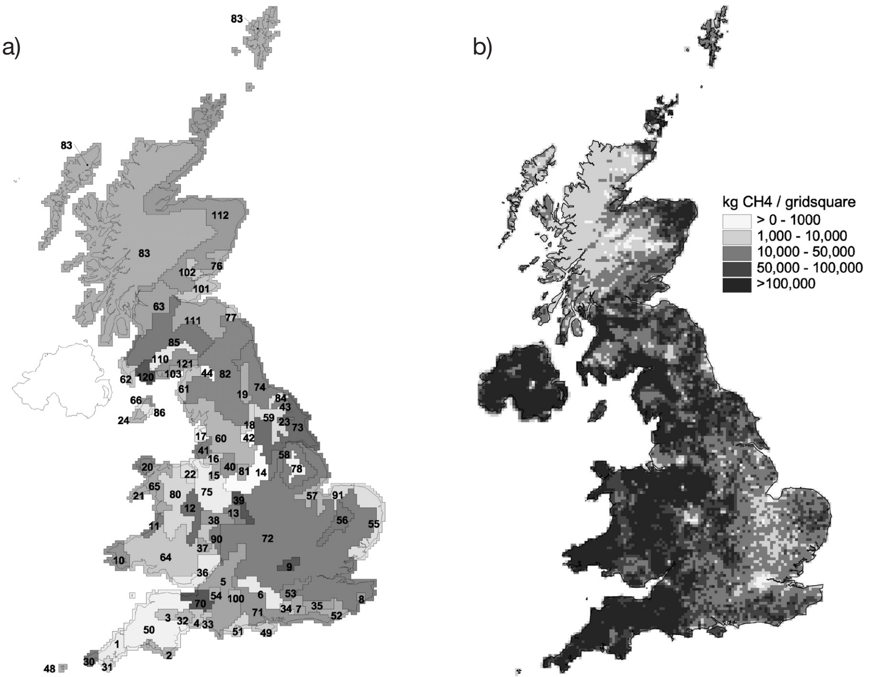


Figure 2: a) Soil/climate zones for farm scale and national scale modelling; b) baseline annual CH4 emissions (2005) from UK agricultural sources, modelled using AENEID (5 km grid)

This information was fed through to the UK scale modelling, by applying the spatially varying emission estimates on a per-animal basis in the AENEID model (Dragosits *et al*., 1998; Sutton *et al*., 2004; 2006) for each mitigation scenario and zone. AENEID (Atmospheric Emissions for National Environmental Impacts Determination) was originally developed for the spatial distribution of NH_3 for the UK and is currently used for annual modelling and mapping of the distribution of NH_3, CH_4 and N_2O emissions in the National Atmospheric Emissions Inventory (NAEI, www.naei.org). In this study, AENEID was used both to estimate baseline emissions and to assess the impacts of CH_4 mitigation methods against these baseline emissions. Figure 2b illustrates the baseline spatial distribution of CH_4 emissions from agriculture in the UK.

RESULTS AND DISCUSSION

At a per-breeding animal level (Table 1), losses from dairy cows (including followers) varied between management, mitigation methods (not shown; see Table 2 for UK data) and forms of N lost. Nitrate leaching from intensive management was larger than from extended grazing management, partly due to the greater proportion of concentrates ingested per animal. Nitrous oxide emissions were largest from the conventional intensive pastoral system, while NH_3 emissions from fully housed intensive management were almost double those from the other two typologies. For beef cattle, emissions per breeding animal were much larger in upland than in lowland situations. This may appear counter-intuitive, but this is due to the longer reproductive cycle and the resulting larger number of followers. Differences in losses due to soil and climate conditions were largest for NO_3^-, N_2O and NO_x. However, animal type and management were estimated by the models to have a lesser effect on NH_3 emissions.

Table 1: Estimated baseline emissions of CH_4, NH_3, N_2O, NOx and NO_3- leaching (in kg per breeding animal*) from the herd/farm models

Animal type and management	NO_3-N	CH_4	N_2O-N	NH_3-N	NO_x-N
	emission kg (including followers)$^{-1}$ yr^{-1*}				
Dairy cows					
Extended Grazing	12-36	103.9-104.6	0.2-5	31-35	0.002-0.003
Conventional Intensive	16-70	113.9-115.2	0.4-12.2	31-35	0.001-0.001
Fully-housed Intensive	23-67	107.3-107.4	0.2-6.4	68-68	0.001-0.002
Beef cows					
Lowland	6-35	169.9-171.2	0.2-12	21-37	0.022-0.356
Upland	1-14	214-214.4	0.1-6	8-16	0.012-0.237
Sheep					
Lowland	0.2-2.1	25.1-25.1	0.02-0.7	0.8-1.4	0.001-0.028
Upland	0.2-1.9	20.2-20.2	0.02-0.4	0.7-0.9	0.001-0.028

*Emissions relate to one adult dairy cow, beef cow or breeding ewe + the associated number of youngstock as calculated by applying typology-specific annual replacement rates. Ranges reflect different soil-climatic zones.

At a UK level (Table 2), an increase in milk yield per dairy cow (by 30% in the modelled scenario), coupled with a reduction in dairy cow numbers to maintain current national milk production, resulted in the largest reduction in CH_4 emissions (-24%). The next most effective mitigation strategy was a high fat diet, which provides a 14% saving, followed by increased heat detection rate (HDR) of cows in oestrus at 7% and a high starch diet at 5%. Changes in diet by feeding high quality forage did not appear to result in large differences in the national emission of CH_4 (-3%), whereas scenarios modelling an increase in low quality forage or decreased HDR resulted in marginal increases. A reduction in the milk yield per dairy cow by 30%, coupled with an increase in the number of dairy cows (to maintain national milk production), resulted in an increase in CH_4 emissions by almost 15%. The most effective CH_4 mitigation measure for beef cattle and sheep was vaccination (-10%), while a diet high in starch also appeared effective at reducing emissions from beef cattle at the national level (-5%). Diets high in water soluble carbohydrates (WSC) appeared to be counter-productive and actually increased modelled national CH_4 emission estimates slightly.

Table 2: Relative impact of methane mitigation methods at the UK scale on CH_4, NH_3, N_2O and NO_3^- leaching (for year 2003)

Mitigation scenario		UK 2003	Comparison with base scenario (%)			
		kt CH_4	CH_4	NH_3	N_2O	NO_3^-
Dairy Herd	base	277.3	100	100	100	100
	milk yield decrease: 30%*	318.6	115	118	113	121
	milk yield increase: 30%*	211.0	76	73	79	78
	high fat	238.5	86	99	100	104
	HDR decreased	298.9	108	106	104	105
	HDR increased	257.1	93	89	93	91
	high quality forage	269.5	97	100	99	99
	low quality forage	282.0	102	100	100	100
	high starch	264.0	95	99	100	100
Beef herd	base	391.6	100	-	-	-
	high starch	372.8	95	-	-	-
	high WSC	401.0	102	-	-	-
	high fat	391.6	100	-	-	-
	vaccine	352.8	90	-	-	-
Sheep flock	base	176.3	100	-	-	-
	high starch	174.0	99	-	-	-
	high WSC	176.8	100	-	-	-
	vaccine	158.3	90	-	-	-

*Numbers of dairy cows and associated followers increased/reduced to keep national milk yield constant

The effectiveness of increasing milk yield per cow to decrease CH_4 emissions was matched by similar decreases in emissions of NH_3 and N_2O and NO_3^- leaching. While high fat diets for dairy cows appeared to decrease CH_4 emissions by 14%, emissions of NH_3 and N_2O were only slightly decreased, but N_2O emissions and NO_3^- leaching showed a slight increase compared with the base scenario. Small decreases in CH_4 emissions through the introduction of high starch diets or high quality forage were not matched by similar decreases for the N compounds, which showed very marginal decreases.

In summary, the modelling framework provided a quantitative assessment of the effectiveness of selected CH_4 mitigation strategies, and the impacts of these on other forms of atmospheric (NH_3 and N_2O) and water pollution (NO_3^- leaching) at the farm scale, as well as nationally.

ACKNOWLEDGEMENTS

The authors are grateful to the Department for Environment, Food and Rural Affairs (Defra) for funding this work under project CC0270.

REFERENCES

Brown L, Scholefield D, Jewkes EC, Lockyer DR and del Prado A (2005). NGAUGE: a decision support system to optimise N fertilisation of British grassland for economic and/or environmental goals. Agriculture, Ecosystems and Environment 109, 20-39.

Chadwick DR, del Prado A, Mills JAN, Crompton LA, Dragosits U, Scholefield D and Newbold CJ (2007) The implications of farm-scale methane mitigation measures for long-term national methane emissions. Final report to Defra on Project CC0270.

DEFRA (2000). Fertiliser recommendations for agricultural and horticultural crops (RB209). The Stationary Office, UK.

Del Prado A and Scholefield D (2006). Use of SIMSDAIRY modelling framework system to specify sustainable UK dairy farms. Delivering Sustainability within profitable faming systems- is it possible? Aspects of Applied Biology 80, 73-80.

Dragosits U, Sutton MA, Place C J and Bayley AA (1998). Modelling the spatial distribution of agricultural ammonia emissions in the UK. Environmental Pollution 102, 195-203.

IGER/ADAS (2004a). Blueprints for sustainable nutrient use in grassland livestock production. Final report to Defra on Project IS0101.

IGER/ADAS (2004b). Cost curve of nitrate mitigation options. Final report to Defra on Project NT2511.

IGER/ADAS (2005). Nitrogen flows on farms with extended grazing. Final report to Defra on Project NT2509.

Mills JAN, Dijkstra J, Bannink A, Cammell SB, Kebreab E and France J (2001). A mechanistic model of whole-tract digestion and methanogenesis in the lactating dairy cow: Model development, evaluation, and application. Journal of Animal Science 79 (6), 1584-1597.

Mills JAN, Kebreab E, Yates CM, Crompton L A, Cammell SB, Dhanoa MS, Agnew RE and France J (2003). Alternative approaches to predicting methane emissions from dairy cows. Journal of Animal Science 81 (12), 3141-3150.

Mottram T and Mills JAN (2003). Defra project IS0102 final report. Environmental and Welfare Benefits of IMS for Dairy Cows.

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

Sutton MA, Dragosits U, Dore AJ, McDonald AG, Tang YS, van Dijk N, Bantock T, Hargreaves KJ, Skiba U, Fowler D, Misselbrook T, Brown L and Hobbs P (2004). The potential to use trace-gas changes following the 2001 outbreak of Foot and Mouth Disease in Great Britain to reduce the uncertainties in agricultural emissions abatement. Journal of Environmental Science and Policy 7, 177-194.

Sutton MA, Dragosits U, Simmons I, Tang YS, Hellsten S, Love L, Vieno M, Skiba U, di Marco C, Storeton-West RL, Fowler D, Williams J, North P, Hobbs P and Misselbrook T (2006). Monitoring and modelling trace-gas changes following the 2001 outbreak of Foot and Mouth Disease to reduce the uncertainties in agricultural emissions abatement. Journal of Environmental Science and Policy 9, 407-422.

NITROUS OXIDE EMISSIONS FOLLOWING LIVESTOCK MANURE APPLICATION TO LAND

RE Thorman[1], S Yamulki[2#], DR Chadwick[2], E Sagoo[1], JR Williams[1], TH Misselbrook[2], J Webb[3~] and BJ Chambers[4]

[1]ADAS Boxworth, Battlegate Road, Boxworth, Cambridge, CB23 4NN, UK, E-mail: rachel.thorman@adas.co.uk; [2]IGER North Wyke, Okehampton, Devon, EX20 2SB, UK ; [3]ADAS Wolverhampton, Wergs Road, Wolverhampton, WV6 8TQ, UK; [4]ADAS Gleadthorpe, Meden Vale, Mansfield, Nottinghamshire, NG20 9PF, UK; [#]currently at Forest Research, Alice Holt Lodge, Farnham, Surrey, GU10 4LH, UK; now at, AEA Energy and Environment, Didcot, OX11 0QR, UK

SUMMARY

In the UK, livestock manures are recognised as a major cause of controllable diffuse pollution. Changing the timing of manure applications from autumn/winter to spring and rapid soil incorporation following the spreading of manures are established techniques to reduce nitrate leaching and ammonia volatilisation losses, respectively. However, such management practice changes may result in greater losses of the greenhouse gas, nitrous oxide (N_2O). Experimental measurements showed that *direct* N_2O losses were greater (P<0.05) following slurry applications in late autumn/winter (1.10% total-N applied) than from those in spring (0.51% total-N applied) and that estimated *indirect* N_2O emissions, as a result of nitrate leaching losses, were also greatest (P<0.01) following late autumn/winter applications. The effect of manure incorporation by ploughing on *direct* N_2O loss was inconsistent. However, *indirect* N_2O losses were reduced (P<0.001) due to lower ammonia volatilisation losses. These results highlight the importance of taking into account the effects of manure management practice changes on both *direct* and *indirect* N_2O emissions.

INTRODUCTION

Around 90 million tonnes of livestock manure supplying 450,000 tonnes of nitrogen (N) are applied annually to agricultural land in the UK (Williams *et al*., 2000). These applications are a valuable source of plant available nutrients, however, they also pose a significant risk of diffuse pollution of the air and water environments. Nitrous oxide (N_2O) is a powerful greenhouse gas (GHG) due to its global warming potential (GWP). In the current UK GHG inventory (2005), the GWP of N_2O is 310 times that of carbon dioxide (Baggott *et al*., 2007). The inventory estimates that 67% of N_2O is produced from agriculture, of which *c*.62% is *directly* emitted from agricultural soils (e.g. following the application of livestock manure, mineral nitrogen fertiliser etc.) and *c*.32% is *indirectly* emitted from agricultural soils via two routes, *viz*: following N losses via ammonia (NH_3) volatilisation/NOx emission (*c*.20%) and nitrate (NO_3) leaching (*c*.80%) (Baggott *et al*., 2007).

As a result of the Kyoto protocol, the UK has agreed to a legally binding reduction in GHG emissions of 12.5% from 1990 levels by the period 2008-12. In order to comply with existing and forthcoming Directives (e.g. EC Nitrates Directive, National Emissions Ceilings Directive, etc.), the UK government is also committed to reducing NO_3 and NH_3 losses to the environment. Around 60% of NO_3 entering water systems is estimated to originate from agricultural land, with annual NO_3 leaching losses following autumn-winter livestock manure applications estimated at 58,000 tonnes N (Chambers and Smith, 1995). The deposition of emitted NH_3 can lead to soil acidification and N enrichment of sensitive habitats. In the UK, annual NH_3 emissions from agriculture have been estimated at 265,000 tonnes (equivalent to *c*.80% of total emissions), with losses following the land spreading of livestock manure responsible for *c*.36% of agricultural emissions (Defra, 2000). Measures implemented to mitigate nitrogen losses via NO_3 leaching (e.g. a change in slurry application timing from autumn/winter to spring) and NH_3 volatilisation (e.g. the rapid soil incorporation of manure) may, however, have implications for both direct and indirect N_2O emissions.

The current Nitrate Vulnerable Zone Action Programme (NVZ-AP) (which covers 55%, 14% and 3% of agricultural land in England, Scotland and Wales, respectively) restricts the application of high readily available N manures (e.g. slurries and poultry manures) on sandy and shallow soils in the autumn-early winter period in order to reduce NO_3 leaching losses (Defra, 2002). Proposed revisions to the NVZ-AP in Britain are likely to result in a 'closed-period' for slurry application on all soil types in late autumn/winter to minimise NO_3 leaching and other nutrient losses in surface runoff and drain flow. This will substantially increase the amount of slurry applied in the spring and summer periods. The different soil and climatic conditions in spring/summer may, however, lead to increased NH_3 and N_2O emissions.

Following the application of solid manures to arable land, rapid incorporation of manure into the soil has been identified as a practical technique to reduce NH_3 emissions (Webb *et al*., 2005). The reduced NH_3 loss, however, conserves N and thereby increases the pool of soil mineral N. This N may subsequently be available for microbial nitrification and denitrification, and the production of N_2O. Livestock manures are recognised as a major cause of controllable nutrient pollution. It is, therefore, essential that practical and integrated methods for mitigating diffuse pollution are developed as part of the programme of measures needed to minimise diffuse pollution from agriculture (under the Water Framework Directive). Such mitigation methods will also need to be developed within potential future revisions of the NVZ-AP and legislation to reduce NH_3 losses. This paper assesses the impact of manure management practice changes to reduce NO_3 leaching and NH_3 emissions on N_2O losses.

MATERIALS AND METHODS

N Losses from Cattle Slurry Applications

Cattle slurry was applied to free draining grassland soils on two commercial farms located in Cheshire (north-west England) and Somerset (south-west England). Slurry was applied using commercial shallow injection or trailing hose machinery to replicated (x3) plots (24 x 24 m). Slurry application rates were in the range 50-60 m^3 ha^{-1} for the trailing hose equipment and *c*.30 m^3 ha^{-1} for shallow injection equipment. The amount of total slurry N applied ranged from 85 to 160 kg ha^{-1}, with 40-65% present as readily available ammonium-N. Nitrous oxide emissions were measured following slurry applications in late autumn/winter and spring, and from an untreated control. Direct N_2O measurements were made using 5 static chambers (0.8 m^2 total surface area) per plot and analysed using gas chromatography. Measurements continued for up to 3 months after slurry application.

Ammonia losses were measured for *c*.7 days following slurry application, using the micrometeorological mass balance technique with passive samplers (Leuning *et al*., 1985) on masts (1 per plot and background). Nitrate leaching losses were measured using porous ceramic cups (10 per plot) installed at between 60 and 90 cm depth, during the period of over-winter drainage (Webster *et al*., 1993). Drainage volumes were estimated using IRRIGUIDE (Bailey and Spackman, 1996) and were combined with NO_3 concentrations to quantify the amounts of NO_3-N leached.

Indirect N_2O emissions were estimated from measured NO_3 leaching and NH_3 losses, and the Intergovernmental Panel on Climate Change (IPCC) default emission factors (EF) for the fraction of leached and volatilised N lost as N_2O-N, i.e. 2.5% and 1%, respectively (IPCC, 1996).

N Losses from Solid Manure Applications

At two sites in England: site 1 at ADAS Gleadthorpe, central England and site 2 at IGER North Wyke, south west England, N_2O and NH_3 emissions were measured from replicated (x4) plots (6 x 10 m) following spring (February/March) applications of solid manure. The plots were established on cereal stubble on a loamy sand soil (site 1) and on bare arable ground on a coarse sandy loam soil (site 2). Cattle farmyard manure (FYM), pig FYM, layer manure and broiler litter were spread at a target application rate of 250 kg N ha^{-1} and either left on the soil surface or immediately incorporated (to 20-25 cm depth) into the soil by ploughing. Control treatments were included where no manure was added.

Direct N_2O measurements were made using 2 static chambers (0.32 m^2 total surface area) per plot and analysed using gas chromatography. Measurements were carried out immediately following manure application and at regular intervals over a *c*.60-day period. Ammonia emissions were measured for up to 2 weeks after manure application, using wind tunnels (one per plot) based on the design of Lockyer (1984) and described in detail in Thorman *et al*. (2006a). Indirect N_2O emissions were estimated from measured NH_3 losses and the IPCC default EFs for the fraction of volatilised N lost as N_2O-N, i.e. 1% (IPCC, 1996).

RESULTS AND DISCUSSION

N Losses from Cattle Slurry Applications

Five pairs of measurements on free draining grassland soils showed that direct N_2O losses were greater ($P<0.05$) following late autumn/winter slurry applications (1.10% total-N applied) than from spring (0.51% total-N applied) timings (Figure 1). This difference was probably due to higher levels of crop N uptake in the spring, which would have depleted the soil mineral N pool available for N_2O production, compared with that in the late autumn/winter. Also, the difference may have reflected soil moisture and temperature differences between late autumn/winter and spring, although a simple relationship could not be established. Other workers have also reported differences between autumn/winter and spring N_2O emissions, e.g. much higher N_2O emission rates (*c*.17-fold) were reported after solid manure additions to a well drained grassland soil in autumn/winter than in spring (Allen *et al*., 1996), and following shallow injection of dairy slurry to grassland denitrification losses were 21% and 7% of total-N applied after winter and spring applications, respectively (Thompson *et al*., 1987).

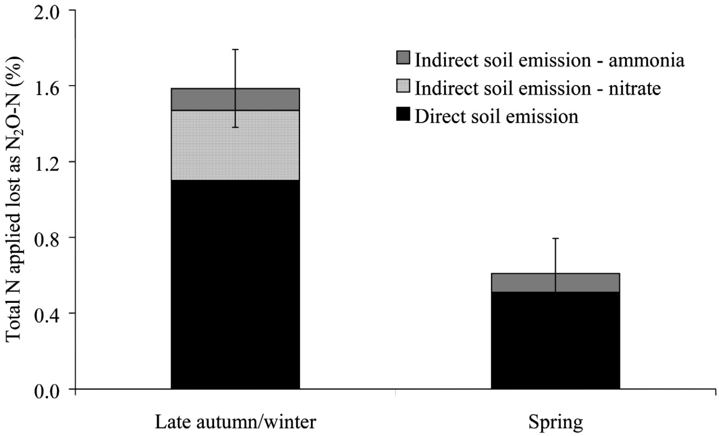


Figure 1: Total direct and indirect N_2O-N losses calculated over c.3 months following cattle slurry applications to free draining grassland soils. Error bars represent ± one standard error of the mean

Calculations based on the IPCC default values, *viz*: 2.5% of leached N and 1% of volatilised N lost as N_2O-N (IPCC, 1996), showed that *c*.30% (late autumn/winter application) and *c*.15% (spring application) of *total* N_2O emissions were accounted for by indirect losses. Similar to direct N_2O losses, indirect N_2O losses were also greater ($P<0.01$) following late autumn/winter slurry applications (0.49% total-N applied) than after spring applications (0.10% total-N applied). This largely resulted from the substantial contribution (*c*.75%) made by over-winter NO_3 leaching to the total indirect N_2O losses following autumn/winter slurry applications.

N Losses from Solid Manure Applications

At site 1, there was a significant ($P<0.001$) effect of incorporation on direct N_2O losses (Figure 2). Over all manure types, mean direct N_2O losses from the ploughed treatments were *c*.4-fold larger at 1.39% of total-N applied compared with losses of 0.36% of total-N applied from the plots where manure was left on the soil surface. In contrast at site 2, there was no effect of manure incorporation into the soil by ploughing on direct N_2O emissions ($P>0.05$), although there was a suggestion that ploughing reduced direct N_2O losses compared with surface application. Losses were 0.54% of total-N applied and 1.44% of total-N applied from the ploughed and surface applied treatments, respectively. The contrasting effects of ploughing on N_2O emissions between sites were probably a reflection of differences in soil moisture and temperature conditions.

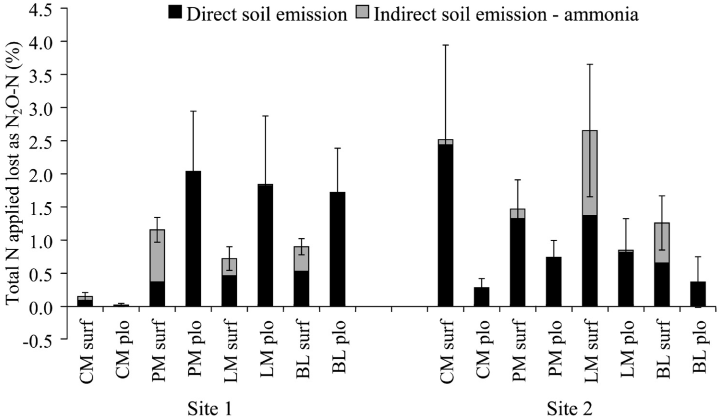


Figure 2: Total direct and indirect N_2O-N losses calculated over c.60 days following ploughing (plo) or surface (surf) application of solid manures. CM=cattle manure, PM=pig manure, LM=layer manure, BL=broiler litter. Error bars represent ± one standard error of the mean

At site 1, the mean soil temperature (5 cm depth) over the monitoring period was *c*.9ºC, with 41% of mean daily temperatures ≤ 8ºC. The mean soil temperature at site 2 was *c*.11ºC, with 19% of the mean daily temperatures 8ºC or less. Site 2 was wetter than site 1, with twice as much rain (*c*.175 mm) falling over the N_2O monitoring period compared with site 1 (*c*.85 mm). Numerous studies in the literature have shown that N_2O production increases with temperature and is stimulated with a rise in soil moisture status (Dobbie *et al*., 1999; Scott *et al*., 2000). Ploughing is likely to increase the length of the diffusion pathway from the site of N_2O production (i.e. the solid manure) to the soil surface and additionally soil structural conditions influence this diffusion rate. So at site 2, although the warm and moist conditions were more favourable for N_2O production than at site 1, the rate of N_2O diffusion through the soil was probably slower due to the heavier textured and wetter soil. This would provide a greater opportunity for N_2O reduction to N_2 and hence lower N_2O emissions.

Other work (Thorman *et al*., 2006b; Thorman *et al*., 2007) at the same sites had similarly shown an inconsistent effect of soil incorporation by ploughing following solid manure application on direct N_2O emissions. For example, following the application of broiler litter at site 1, ploughing resulted in greater (P<0.001) N_2O emissions (0.42% total-N applied) compared with surface application (0.11% total-N applied). In contrast at site 2, ploughing resulted in significantly (P<0.05) lower N_2O emissions (1.08% of total N applied) compared with surface application (3.76% total N applied). This difference was also attributed to soil temperature and moisture conditions, as well as site history.

Ploughing reduced mean NH_3 emissions (P<0.001) by *c*.97%, with losses <0.5% of total-N applied from the ploughed treatments. Calculations based on the IPCC default values, *viz*: 1% of volatilised N lost as N_2O-N (IPCC, 1996), showed that from the surface manure applications *c*.50% (site 1) and *c*.30% (site 2) of mean *total* N_2O emissions was accounted for by indirect losses via NH_3 volatilisation. At both sites, indirect N_2O losses were lower (P<0.001) from the ploughed than surface spread treatments reflecting the NH_3 abatement of ploughing (Figure 2). As the solid manures were spring applied there was no indirect N_2O loss via NO_3 leaching. At site 1, although the inclusion of indirect N_2O losses substantially increased the mean *total* N_2O emission from the surface applied treatments (0.67% total-N applied), *total* losses from the ploughed treatments (1.39% total-N applied) were still greater (P<0.05). In contrast at site 2, when indirect N_2O losses were taken into consideration *total* N_2O emissions following surface manure application (1.97% total-N applied) were greater (P<0.05) than following ploughing (0.55% total-N applied).

CONCLUSIONS

Experimental results indicated that spring slurry application timings on free draining grassland soils are likely to result in lower direct N_2O losses compared with late autumn/winter timings. This probably reflected the greater rate and extent of crop N uptake in spring compared with late autumn/winter timings, and differences in soil moisture/temperature conditions. Spring slurry applications are also likely to result in lower indirect N_2O emissions as a result of reduced NO_3 leaching losses, which is important in influencing the overall magnitude of total N_2O losses. Proposed revisions to the NVZ-AP, which will increase the amount of slurry applied in the spring, are likely to result in a 'win-win' situation in terms of reducing both NO_3 and N_2O losses.

The effects of ploughing solid manures into the soil compared with surface application on direct N_2O losses were inconsistent, with losses dependent on site specific conditions. However, because ploughing is an effective management practice to abate NH_3 emissions, the associated reduction in indirect N_2O losses can significantly influence *total* N_2O emissions.

Our measurement data highlight the importance of taking into account the effects of management practice changes and mitigation methods on both direct and indirect N_2O emissions (after NH_3 and NO_3 leaching losses). Also, there is a need to develop integrated manure N management practices, together with those for inorganic fertiliser N and biologically fixed N, that consider all N loss processes, to ensure that strategies to limit the loss of one N form do not exacerbate losses via another (so called 'pollution swapping').

ACKNOWLEDGEMENTS

Funding of this work by the UK Department for Environment, Food and Rural Affairs (Defra) is gratefully acknowledged.

REFERENCES

Allen AG, Jarvis SC and Headon DM (1996). Nitrous oxide emissions from soils due to inputs of nitrogen from excreta return by livestock on grazed grassland in the UK. Soil Biology and Biochemistry 28, 597-607.

Baggott SL, Cardenas L, Garnett E, Hobson M, Jackson J, Mobbs DC, Murrells T, Passant N, Thomson A and Watterson JD (2007). UK Greenhouse Gas Inventory, 1990-2005: Annual Report for submission under the Framework Convention on Climate Change. AEA Technology plc, Didcot, UK, April 2007.

Bailey RJ and Spackman E (1996). A model for estimating soil moisture changes as an aid to irrigation scheduling and crop water-use studies: I. Operational details and description. Soil Use and Management 12, 122-129.

Chambers BJ and Smith KA (1995). Management of farm manures: Economics and environmental considerations. Soil Use and Management 16, 157-161.

Defra (2002). Guidelines for Farmers in NVZs – England. Defra Publications, London (PB5505).

Defra (2000). Ammonia in the UK. Defra Publications, London (PB6865).

Dobbie KE, McTaggart IP and Smith KA. (1999). Nitrous oxide emissions from intensive agricultural systems: variations between crops and seasons, key driving variables, and mean emission factors. Journal of Geophysical Research 104, 26891-26899.

IPCC (1996). Climate Change 1995. The Science of Climate Change. Contribution of Working Group 1 to the Second Assessment Report of the Intergovernmental Panel on Climate Change. JT Houghton (Ed.), Cambridge University Press.

Leuning R, Freney J R, Denmead O T and Simpson J R (1985). A sampler for measuring atmospheric ammonia flux. Atmospheric Environment 19, 1117-1124.

Lockyer DR (1984). A system for the measurement in the field of losses of ammonia through volatilisation. Journal of the Science of Food and Agriculture 35, 837-848.

Scott A, Ball BC, Crichton IJ and Aitken MN (2000). Nitrous oxide and carbon dioxide emissions from grassland amended with sewage sludge. Soil Use and Management 16, 36-41.

Thompson R B, Ryden J C and Lockyer D R (1987). Fate of nitrogen in cattle slurry following surface application or injection to grassland. Journal of Soil Science 38, 689-700.

Thorman RE, Webb J and Yamulki S (2006a). Rapid incorporation of solid manure as a best management practice? In Gairns L, Crighton K and Jeffrey B (Eds.), Proceedings of the SAC/SEPA Conference Agriculture and the Environment IV: Managing Rural Diffuse Pollution, Edinburgh, Scotland, SAC, pp. 249-252.

Thorman RE, Chadwick DR, Boyles LO, Matthews R, Sagoo E and Harrison R (2006b). Nitrous oxide emissions during storage of broiler litter and following application to arable land. In: Proceedings of the 2nd International Conference on Greenhouse Gases and Animal Agriculture held in Zurich, Switzerland between 20 and 24 September 2005, International Congress Series 1293, Elsevier, Amsterdam, The Netherlands, pp. 355-358.

Thorman RE, Chadwick DR, Harrison R, Boyles LO and Matthews R (2007). The effect on N_2O emissions of storage conditions and rapid incorporation of pig and cattle FYM into arable land. Biosystems Engineering 97, 501-511.

Webb J, Menzi H, Pain BF, Misselbrook TH, Dämmgen U, Hendriks H and Döhler H (2005). Managing ammonia emissions from livestock production in Europe. Environmental Pollution 135, 399-406.

Webster CP, Shepherd MA, Goulding KWT and Lord EI (1993). Comparisons of methods for measuring the leaching of mineral nitrogen from arable land. Journal of Soil Science 44, 49-62

Williams JR, Chambers BJ, Smith KA and Ellis S (2000). Farm manure land application strategies to conserve nitrogen within farming systems. Proceedings of the SAC/ SEPA Conference: Agriculture and Waste Management for a Sustainable Future, Edinburgh, UK, pp. 167-179.

NOVEL QUANTIFICATION OF METHANE EMISSIONS FROM A CONSTRUCTED WETLAND IN THE SCOTTISH BORDERS

DS Reay and G Paul

School of Geosciences, University of Edinburgh, Crew Building, West Mains Road, Edinburgh EH9 3JN, UK, E-mail: David.Reay@ed.ac.uk

SUMMARY

Constructed wetlands are increasingly used to intercept inorganic N, P and sediment in agricultural drainage waters and surface run-off. From late winter to early summer 2007 we used a novel Tunable Diode Laser (TDL)-megachamber method to assess the extent of any pollution swapping via enhanced methane emissions in a constructed wetland in the Scottish Borders. We found that methane emissions at the site were consistently >3 mg m^{-2} h^{-1}, even in February and March, with large increases in methane emissions (>20 mg m^{-2} h^{-1}) during spring and summer related to elevated water temperatures. As such, elevated methane emissions appear to be a significant source of 'pollution swapping' in this wetland.

INTRODUCTION

Wetlands are capable of intercepting macronutrients such as nitrate, ammonium and phosphorous through the processes of sedimentation, nitrification and denitrification, and assimilation by microbes and plants. The use of constructed wetlands can be particularly useful in the treatment of agricultural and farmyard wastewater (e.g. Tanner *et al*., 2005). Of particular relevance in so-called 'Nitrate Vulnerable Zones' (where ground and surface waters exceed or are likely to exceed 50 mg NO_3^- l^{-1}) is the efficiency with which constructed wetlands are able to intercept nitrate in runoff and leachate. Nitrate interception by wetlands relies to a large extent on the process of denitrification, whereby denitrifying bacteria in the wetlands soils utilise the nitrate as a terminal electron acceptor, reducing the nitrate to N_2. Such removal of dissolved nitrate can be extremely effective, wetland soils having been shown to remove up to 3.5 g NO_3^--N m^{-2} day^{-1} (Mitsch *et al*., 2001).

Given a combination of recent reforms to the Common Agricultural Policy - including a greater environmental basis to land-use policy - with the relatively low cost of constructed wetlands as a nitrate interception strategy, their use is likely to grow rapidly. However, their true effectiveness in intercepting nitrate remains in question for many situations. In addition to this uncertainty, there is a risk that the enhancement of denitrification and methanogenesis in these wetland soils may also lead to significant emissions of the powerful greenhouse gases N_2O and CH_4 (Dosskey, 2001; Reay, 2004).

Previously we have reported the extent of such 'pollution swapping' due to N_2O emissions from a constructed wetland in the Scottish Borders (Reay *et al*., in press). Here we present a quantification of CH_4 emissions from the same wetland from late winter through to summer in 2007 using a novel flux chamber method.

MATERIALS AND METHODS

The study site (Lat. 55°48'-N, Long. 2°13'-W) consisted of a constructed wetland located within an area of the Scottish Borders designated as a Nitrate Vulnerable Zone (NVZ). The wetland was constructed in May 2004 and covers an area of 1,000 m^2. The surrounding farm (536 ha) consists of a mixture of grazed pasture (112 ha) and arable land (364 ha). The wetland receives wastewater from a hard steading, arable field drainage, and from overflow water from the farm's septic tank. The wetland itself is ~150 m in length and ranges in width from between 3 and 10 m.

Water and Gas Sampling

Every 1-2 weeks from late winter (early February) to the height of the growing season (end of June) water samples were taken at 5 m intervals along a 145 m transect running through the wetland. Water temperature, air temperature and windspeed were measured along the sample transect, with additional air temperature and windspeed data being obtained from a local meteorological station. Water samples were collected in headspace-free bottles and maintained at 4°C until analysis (normally within 2 days of sampling) for dissolved inorganic N using automated flow colorimetry (Bran and Luebbe, Norderstedt, Germany).

Megachambers were constructed at 4 sites at approximately 20 m intervals down the wetland. They each consisted of a curved framework of 3 m long fibreglass canes forming a tunnel 10 m in length and covered with highly reflective gas-impermeable sheeting to prevent super-ambient heating. Chambers were sealed using lengths of chain, with seals and total chamber volume checked by injection of SF_6 and measurement of its subsequent dilution. The Tunable Diode Laser (TDL) system (Gas Finder 2.0, Boreal Laser Inc.) provided methane detection along the entire chamber length and was linked to an external laptop computer for real-time observation (5 second average) of methane accumulation (Figure 1).

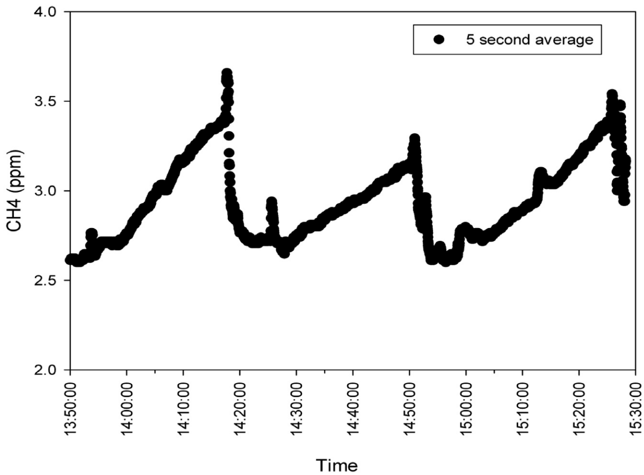


Figure 1: Methane accumulation curves for three replicate flux measurements using the TDL-megachamber method

Methane flux rates were determined by linear regression of accumulation curves, with triplicate measurements being made at each site. Chamber closure periods of 15 minutes were used, with manual flushing of chambers in between each closure period to bring CH_4 concentrations back to ambient (Figure 1). A mean flux for each site was derived from these triplicate measurements. Wetland-wide methane flux measurements hereafter represent the mean of all four sites unless stated otherwise. Wetland-wide CH_4 fluxes for the entire sampling period were calculated by extrapolating hourly fluxes to daily rates and interpolation between sampling days. This simple interpolation assumed CH_4 flux on a non-sampling day was equivalent to the average of the rates measured on the preceding and subsequent sampling days.

RESULTS

Average water temperatures in the wetland increased from 5 °C in February to 12 °C at the end of June 2007, with a peak water temperature of 14 °C at the end of May. Flow rates over the study period were relatively constant at ~1 l s^{-1} due to the constant inflow of water from the farm steading and septic tank.

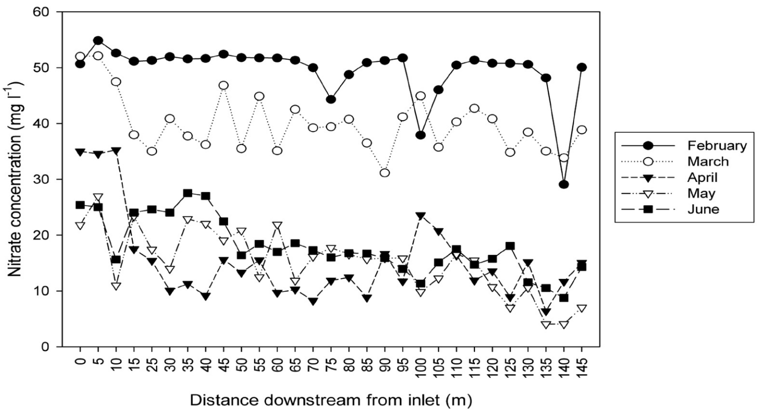


Figure 2: Monthly average NO_3^- concentration with distance from inlet

Monthly-averaged nitrate (NO_3^-) concentrations in water entering the wetland fell from more than 50 mg l^{-1} in February and March to less than 30 mg l^{-1} in May and June 2007. Dissolved NO_3^- concentrations in water leaving the wetland were little changed from those entering it during February, but became markedly lower in subsequent months (Figure 2). During May, average NO_3^- concentrations fell from ~20 mg l^{-1} at the inlet to <10 mg l^{-1} at the outlet, indicating significant interception. Dissolved ammonium (NH_4^+) concentrations followed a similar pattern, but with very low (<0.5 mg l^{-1}) NH_4^+ inputs throughout the study period (data not shown).

In terms of NO_3^- interception, the wetland appeared to be relatively ineffective in the late winter, but was able to intercept ~50 % of incoming dissolved NO_3^- later in the growing season. The increase in the interception efficiency in April, May and June coincided with a marked increase in water temperature at the site (Figure 3). For the entire study period, NO_3^-interception by the wetland was estimated at 140 kg NO_3^--N.

It should be noted that the interception efficiency of the wetland was calculated by dividing the difference in NO_3^- concentrations between the inlet and the outlet by the inlet concentration on each sampling day, with the assumption that water sampled at the outlet was representative of water entering the system, and its subsequent processing in the wetland.

Mean methane flux rates ranged from 3.8 to 24 mg m^{-2} h^{-1} during the study period with the mean flux across the 4 sites (~60 m^2 total area) showing a strong and positive relationship with water temperature (Figure 3). Flux rates remained below 5 mg m^{-2} h^{-1} in February, with an average water temperature of < 6°C. In early March the average water temperature rose to 9°C, but site-wide mean CH_4 flux rate remained at just 4.5 mg m^{-2} h^{-1} until late March where a rise in water temperature to 9.8°C coincided with a mean CH_4 flux rate of 11.3 mg m^{-2} h^{-1}. Mean CH_4 flux rates across the site further increased to 21.7 mg m^{-2} h^{-1} in late May, apparently in response to higher average water temperatures (13.8°C). On the subsequent sampling days in June, average water temperatures were slightly lower (11-12°C) but CH_4 fluxes remained at >20 mg m^{-2} h^{-1}. Extrapolation of these mean flux rates to a daily rate over the entire wetland area (~0.1 ha) resulted in a flux of approximately 0.5 kg CH_4 per day from the end of May through to the end of the study period in June. Interpolation of measured fluxes indicated that total wetland CH_4 emissions for the 4-month sampling period were of the order of 45 kg.

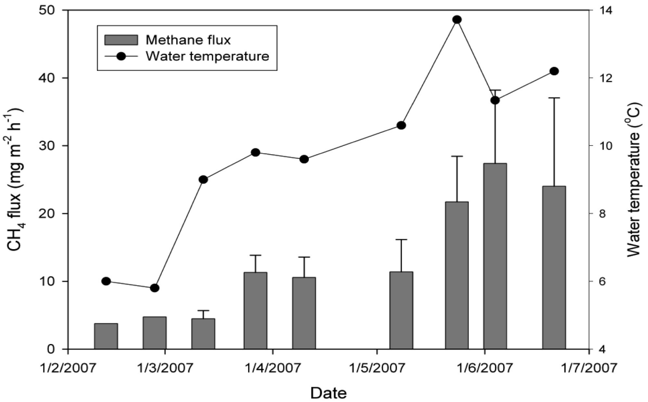


Figure 3: Mean CH_4 flux and average water temperature on all measurement days Error bars represent the standard error (n=4)

DISCUSSION

The efficiency of NO_3^- interception by the wetland increased over the study period, exhibiting strong seasonal dependence with highest efficiencies occurring in the summer months. Such seasonal dependency of nitrate interception efficiency has been previously shown for this wetland (Reay *et al*., in press) and for others (e.g. Tanner *et al.*, 2005). A key question that remains to be answered at this site is that of the retention time for intercepted reactive N. If the N intercepted during the growing season is then lost to downstream surface waters during the winter it can be argued that the net benefit of the wetland on an annual time-scale is minimal. Initial results indicate that winter-time N losses are small, but more detailed measurement of these losses is required.

The apparently strong positive relationship between average water temperature and site-wide CH_4 flux reported here supports the findings of previous studies of northern wetlands, where temperature is often found to be the dominant factor (e.g. Whalen, 2005; Bartlett, 1992; Christensen, 1996; Nakano *et al*., 2000). For example, over a number of northern wetland sites, soil temperature variations accounted for 84 % of the observed variance in methane emissions, with emissions showing a strong positive response to increased temperature (Christensen *et al.,* 2003). The CH_4 emission rates were within the range previously identified for such systems, with the high summer-time fluxes indicating this site to be a significant source of CH_4 (Søvik *et al*., 2006).

Quantifying methane fluxes from any wetland ecosystem can be a challenge given the great spatial and temporal variations in fluxes common to such environments (Wachinger *et al*., 2000; Wang *et al*., 2006). Traditional methane flux measurement involves the use of numerous small (<0.5 m^2) static chambers, with sampling of the chamber headspace after a set closure time (Dalva *et al*., 2001; Turetsky *et al*., 2002; Bubier *et al*., 2005). As methane production and consumption rates in wetland ecosystems can vary by several orders of magnitude within linear scales of a few metres (Hargreaves and Fowler, 1998), this technique can result in large inter-chamber variations in fluxes and introduce significant uncertainties when upscaling (e.g. Bubier *et al*., 2005; Van den Pol-Van Dasselaar *et al*., 1999). The novel megachamber technique, incorporating the TDL, used here provides integrated methane fluxes over a much greater area (~15 m^2 per site) than that possible with traditional static chambers, simultaneously capturing methane flux via diffusion, ebullition and plant aerenchyma. As such, we believe the CH_4 flux data from this study represent a more robust data set for use in estimating site-wide CH_4 fluxes, than that which would be provided by traditional techniques.

There are, however, some potentially significant caveats to our extrapolation of fluxes. First, all measurements in this study were made during the daytime and so any significant nocturnal fluctuations would not have been accounted for. In general, methanotroph activity increases with increasing temperature, though reported Q_{10} values span a very wide range (Walter and Heimann, 2000). In addition, the productivity of the autotrophic organisms in the wetland may respond to diurnal variation in light levels, with subsequent variations in oxygen concentrations and available carbon sources, such as root exudates, in the system. As these factors may all influence CH_4 production, our daytime-only measurements may have introduced significant bias to daily flux estimates and should therefore be viewed as indicative of the relative change in daytime CH_4 flux over the study period. As with nitrate interception, a more detailed examination of the CH_4 fluxes over an entire year is required before the net annual CH_4 flux can be determined.

The extent of 'pollution swapping' via enhanced CH_4 emission from this constructed wetland is an important issue, at least for the limited timeframe examined here. Using a simple comparison based on the external costs of reactive nitrogen (N_r) pollution of ground and surface waters (£0.034-0.048 kgN_r^{-1}) (Pretty, 2006), versus methane emissions (£1.30 per kg CH_4) (Stern, 2006), the estimated interception of 140 kg NO_3^--N over the study period would have effectively saved £6.74 in terms of avoided water treatment costs and other externalities, while the emission of 45 kg CH_4 would cost £58.50. Even with the caveats as to interpolation and extrapolation of CH_4 emissions noted above, this issue of pollution-swapping via enhanced CH_4 emissions is likely to remain significant.

ACKNOWLEDGMENTS

This work was supported by a Natural Environment Research Council fellowship awarded to David Reay.

REFERENCES

Bartlett KB, Crill PM, Sass RL, Harriss RC and Dise NB (1992). Methane emissions from tundra environments in the Yukon-kuskokwim delta, Alaska. Journal of Geophysical Research 97, D15, 16645–16660.

Bubier J, Moore T, Savage K and Crill P (2005) A comparison of methane flux in a boreal landscape between a dry and a wet year. Global Biogeochemical Cycles 19, GB1023, doi:10.1029/2004GB002351,200.

Christensen TR, Prentice IC, Kaplan J, Haxeltine A and Sitch S (1996). Methane flux from northern wetlands and tundra: An ecosystem source modeling approach. Tellus 48B, 652-661.

Christensen TR, Joabsson A, Ström L, Panikov N, Mastepanov M, Öquist M, Svensson BH, Nykänen H, Martikainen P and Oskarsson H (2003). Factors controlling large scale variations in methane emissions from wetland. Geophysical Research Letters 30, GL016848.

Dalva M, Moore T, Arp P and Clair T (2001). Methane and soil and plant community respiration from wetlands, Kejimkujik National Park, Nova Scotia: Measurements, predictions, and climatic change. Journal of Geophysical Research 106 (D3), doi: 10.1029/2000JD900500.

Dosskey MG (2002). Setting Priorities for Research on Pollution Reduction Functions of Agricultural Buffers. Environmental Management 30, 641-650.

Hargreaves KJ and Fowler D (1998). Quantifying the effects of water table and soil temperature on the emission of methane from peat wetland at the field scale. Atmospheric Environment 32, 3275-3282.

Johansson AE, Gustavsson A-M, Öquist MG and Svensson BH (2004). Methane emissions from a constructed wetland treating wastewater—seasonal and spatial distribution and dependence on edaphic factors. Water Research 38, 3960-3970.

Merritt R, Turetsky R, Wieder K and Vitt DH (2002). Boreal peatland C fluxes under varying permafrost regimes. Soil Biology and Biochemistry 34, 907-912.

Mitsch WJ, Day Jr JW, Gilliam JW, Groffman PM, Hey DL, Randall GW and Wang N (2001). Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin: Strategies to counter a persistent ecological problem. BioScience 51, 373-388.

Nakano T, Kuniyoshi S and Fukuda M (2000). Temporal variation in methane emission from tundra wetlands in a permafrost area, northeastern Siberia. Atmospheric Environment 34, 1205-1213.

Pretty J (2006). The Costs and Benefits of Nitrogen Enrichment. GANE symposium. London.

Reay DS (2004). Fertilizer 'solution' could turn local problem global. Nature 427, 485.

Søvik AK, Augustin J, Heikkinen K, Huttunen JT, Necki JM, Karjalainen SM, Kløve B, Liikanen A, Mander Ü, Puustinen M, Teiter S and Wachniew, P (2006). Emission of the Greenhouse Gases Nitrous Oxide and Methane from Constructed Wetlands in Europe. Journal of Environmental Quality 35, 2360-2373.

Stern N (2006). The Economics of Climate Change. Cambridge University Press, 712pp.

Tanner CC, Nguyen ML and Sukias JPS (2003). Constructed wetland attenuation of nitrogen exported in subsurface drainage from irrigated and rain-fed dairy pastures. Water Science and Technology 51, 55-61.

Van de Weg M, Reay DS, Gouriveau F and Grace J (2008). Pollution-swapping: nitrate interception and nitrous oxide production in a constructed wetland in the Scottish Borders. In: Bosch A, Teira MR, Villar JM (eds.) Towards a better efficiency in N use. Editorial Milenio, Lleida (Spain), 531pp.

Van Den Pol-Van Dasselaar A, Van Beusichem ML and Oenema O (1999). Determinants of spatial variability of methane emissions from wet grasslands on peat soils. Biogeochemistry 44, 221-237.

Wachinger G, Fiedler S, Zepp K, Gattinger A, Sommer M and Roth K (2000). Variability of soil methane production on the micro-scale: spatial association with hot spots of organic material and Archaeal populations. Soil Biology and Biochemistry 32, 1121-1130.

Wang Z-P, Han X-G and Li L-H. (2006). Methane emission patches in riparian marshes of the inner Mongolia. Atmospheric Environment 40, 5528-5532.

Walter BP and Heimann M (2000). A process-based, climate-sensitive model to derive methane emissions from natural wetlands: Application to five wetland sites, sensitivity to model parameters, and climate. Global Biogeochemical Cycles 14, 745-765.

Whalen SC (2005). Biogeochemistry of methane exchange between natural wetlands and the atmosphere. Environmental Engineering and Science 22, 73-94

THE POTENTIAL TO INCREASE CARBON STORAGE IN AGRICULTURAL SOILS

BJ Chambers[1], A Bhogal[1], AP Whitmore[2] and DS Powlson[2]

[1]ADAS Gleadthorpe, Meden Vale, Mansfield, Notts, NG20 9PF, UK, E-mail: Brian.Chambers@adas.co.uk; [2]Rothamsted Research, Harpenden, Herts, AL5 2JQ, UK

SUMMARY

The extent to which reduced tillage practices and organic material returns could increase the organic carbon (C) content of arable soils in the UK is reviewed in this paper. Our best estimate of the C storage potential of zero tillage under UK conditions was 310 kgC/ha/yr and reduced tillage 160 kgC/ha/yr, as an initial rate of increase (up to *c.*20 years). However, there are questions over the implications of reduced/zero tillage practices on nitrous oxide emissions and the overall balance of greenhouse gas emissions. At typical field application rates of organic materials (i.e. 250 kg/ha total N), estimated increases in topsoil organic C were in the range 630-1800 kgC/ha/yr, with the greatest retention estimated from biosolids (sewage sludge) cake, green compost and paper crumble additions, and least from farm manure additions. However, it is debatable whether these increases can be considered genuine additional C storage against a present/recent day baseline.

INTRODUCTION

Both reduced tillage and the recycling of organic materials to land have been promoted as a means of increasing the storage of carbon in agricultural soils. For example, the recently published Stern report on the economics of climate change promoted reduced tillage as a means of enhancing the storage of carbon in agricultural soils. Stern (2006) cited the example of the Chicago Climate Exchange, where a minimum 4 year commitment to continuous zero tillage on enrolled areas was valued at $1-2 per acre per year (equivalent to approximately £1.25-2.50/ha/yr). Also the 2006 UK Climate Change Programme included a policy commitment to "examine the scope and feasibility of a market based mechanism to facilitate trading of greenhouse gas reductions from agriculture, forestry and other land management sectors", which included looking at carbon storage in soils and forests as a potential abatement option. There are approximately 5 million ha of tillage land in the UK, so even small increases in soil organic carbon (SOC) storage per hectare of agricultural land could overall lead to important increases in carbon storage at a national level. This paper reviews the extent to which reduced tillage practices and organic material returns could increase the organic carbon content of arable soils in the UK.

EFFECT OF REDUCED TILLAGE PRACTICES ON SOIL ORGANIC CARBON

'Reduced tillage' is a term that is used to describe all non-plough based cultivation practices and has been suggested to increase SOC due to a reduction in soil disturbance and consequently the decomposition of organic matter (carbon). There have only been 6 studies of contrasting tillage systems in the UK (Cannell and Finney, 1973; Powlson and Jenkinson, 1981; Chaney, 1985; Ball, 1994). Taking an average of the SOC changes measured in these studies, our best estimate of the C storage potential of zero tillage under UK conditions was 310 kgC/ha/yr. This equates to *c.*0.35% of the typical carbon content of an arable soil in England and Wales (@ 91 t/ha, assuming 28 g/kg SOC in the topsoil; Webb *et al.,* 2001). Reduced tillage was estimated to have half the C storage potential of zero tillage at 160 kgC/ha/yr.

Changes in SOC are generally slow to occur and difficult to measure against the large background carbon content in arable soils (*c*.91 t/ha). After a change in management practice (e.g. the introduction of zero tillage or regular organic material additions) SOC will increase (or decrease) towards an equilibrium (after 100 years or more) that is characteristic of the soil type, land use and climate. Annual rates of SOC accumulation (or depletion) therefore change over time and gradually decline as the new equilibrium is approached, when they become zero. Typically *c.*50% of the SOC accumulation achieved after 100 years of introducing a management change, occurs within the first 20 years. Maintaining SOC at the new equilibrium level is then dependent on continuing the new management practice indefinitely. The estimates of potential C storage from zero and reduced tillage can therefore only be regarded as the *initial rate* of increase (up to *c*.20 years). They should also not be considered to be annually cumulative, as arable land in the UK is typically ploughed every 3 to 4 years to reduce the build-up in weeds, disease and soil compaction levels. It is arguable that much (if not most) of the stored C from reduced/zero tillage practices will subsequently be released as a result of the increased soil disturbance caused by periodic ploughing.

Reduced tillage has many benefits, besides protecting existing SOC levels and potentially increasing SOC; it can increase soil water infiltration rates and reduce water erosion, enhance soil water retention, and decrease production costs and fossil fuel (energy) consumption. However, zero tillage has also been shown to increase direct emissions of nitrous oxide (N_2O) by up to an equivalent of c.190 kg/ha/yr CO_2-C (compared with conventional tillage), due to an increase in topsoil wetness and/or reduced aeration as a result of less soil disturbance (MacKenzie *et al*., 1998). Nitrous oxide is a powerful greenhouse gas (GHG) with 310 times the global warming potential of CO_2, such that overall, increased N_2O emissions may completely offset the balance of greenhouse gas emissions compared with the amount of C potentially stored through changing from conventional to reduced/zero tillage practices. However, the evidence is not clear and further work is required to determine the effect of contrasting tillage systems on N_2O emissions, SOC storage and the overall balance of GHG emissions.

EFFECT OF ORGANIC MATERIAL ADDITIONS ON SOIL ORGANIC CARBON

The recycling of organic materials to land provides a valuable source of nutrients and organic matter, and could potentially increase SOC levels. Currently around 90 million tonnes of farm manures, 3-4 million tonnes of biosolids (sewage sludge) and 4 million tonnes of industrial ‘wastes’ are applied (on a fresh weight basis) annually to agricultural land in the UK (Hickman *et al.,* 2000).

Results from experimental studies (>4 years of duration) in the UK were used to estimate potential increases in SOC following the addition of a range of organic materials (Table 1). The results were based on average changes in SOC measured at 8 farm manure study sites (Johnston, 1975; Mattingly, 1975; Jenkinson and Johnston, 1977; Jenkinson, 1990; Bhogal *et al.,* 2007), 10 biosolids study sites (Johnston, 1975;

Gibbs *et al.,* 2006) and 4 green compost study sites (Wallace, 2005). We were not able to identify any medium-term UK studies measuring SOC following the application of paper crumble. However, measurements of the proportion of recalcitrant (lignin) and readily decomposable carbon within paper crumble (Bhogal *et al.,* 2007) suggested that it would behave similarly to farm manures, so the same C accumulation rate was assumed (Table 1). As with reduced tillage, the potential increases in SOC estimated in Table 1 should only be regarded as the initial (up to *c.*20 years) rate of SOC increase.

Table 1: Potential increases in SOC following the application of a range of organic materials at 250 kg/ha total N

Organic material	Application rate (t/ha dry solids)	SOC increase kg/ha/yr/t ds applied[b]	SOC increase kg/ha/yr	% of SOC stocks[d]
Farm manures	10.5	60 (20-100)	630	0.69
Digested biosolids	8.3	180 (130-230)	1500	1.64
Green compost	23	60 (40-80)	1400	1.54
Paper crumble	30[a]	60 (20-100)[c]	1800	1.98

[a] Typical application rate of primary or secondary chemical/physically treated paper crumble = 75 t/ha fresh weight, supplying 150 kg/ha total N (Gibbs *et al*., 2005); [b]mean with 95% confidence interval in parenthesis; [c]Average SOC increase per tonne dry solids applied assumed to be the same as for farm manures; [d]Assuming 28 g/kg SOC in the top 25 cm & a bulk density of 1.3 g/cm^3 (91tC/ha)

From the data summarised in Table 1, it is clear that SOC levels can be increased by the application of organic materials to arable soils in the UK. At typical application rates (i.e. 250 kg/ha total N, the maximum field application rate permitted in Nitrate Vulnerable Zones), 630-1800 kg/ha of additional carbon was estimated to be retained in the topsoil (depending on the type of material applied). This equates to 0.7-2.0% of the typical carbon content of an arable topsoil.

It is debatable whether increases in SOC following the application of farm manures can be considered genuine additional C storage (against a present/recent day baseline), as nearly all of these materials are already applied to land. The exception being *c*.580,000 tonnes of poultry litter that are used for electricity generation. Similarly, only 1% of biosolids are presently landfilled (Water UK, 2006), although historically greater amounts were landfilled. Compost and paper crumble applications may (probably) be regarded as genuine additional carbon storage (against a recent/present day baseline), as historically most of these materials were landfilled.

Other Implications from the Application of Organic Materials to Land

Increasing SOC levels through the application of organic materials can help to maintain soil structure, quality and aggregate stability, which in turn can increase soil water retention and water infiltration rates (thereby reducing the risks of soil erosion) and improve plant nutrient uptake. Most organic materials also provide a valuable source of plant available nutrients, particularly nitrogen (N), phosphorus (P), potassium (K), sulphur (S) and magnesium (Mg), thereby reducing the need for inorganic fertiliser inputs. This not only gives a reduction in manufactured fertiliser costs, but also a reduction in the amount of carbon dioxide (CO_2) emitted, as the production of inorganic fertiliser (particularly N) consumes energy (fossil fuel). However, paper crumble materials (particularly physically and chemically treated products) tend to have high C:N ratios and therefore typically immobilise soil N following land application, such that compensatory inorganic fertiliser N applications need to be made to ensure crop yields are not compromised (Gibbs *et al.,* 2005).

The application of organic materials also presents a risk of environmental pollution, if not handled and managed carefully. Applications therefore need to be managed to limit nitrate and phosphorus losses to water, and N losses by ammonia (NH_3) volatilisation and N_2O emission to air. Nitrous oxide emissions are important as they have the potential to affect the overall balance of GHG emissions, from increased SOC storage, through applying organic materials to land. However, reductions in inorganic fertiliser N usage (and hence direct N_2O emissions from this source) following organic material additions offset most of these losses. Also, the continued application of organic materials can result in nutrient accumulation (particularly phosphorus - P) and increased risks of diffuse pollution (e.g. increased P losses in surface runoff and drainflow waters). Additionally, there is a need to consider the long-term accumulation of heavy metals following repeated organic material additions.

NET CARBON STORAGE POTENTIAL OF REDUCED TILLAGE AND ORGANIC MATERIAL RETURNS

In order to determine the *net* carbon storage potential of reduced/zero tillage and organic material applications, potential decreases/increases in GHG emissions due to changes in fertiliser N manufacture (energy consumption), net N_2O emission decreases/increases from the applied organic materials and changes in fertiliser N application rates, all need to be taken into account. Figure 1 summarises the maximum *net* CO_2-C 'saving' potential of reduced/zero tillage and selected organic material applications at typical rates. The application of biosolids, green compost and paper crumble was estimated to offer the best opportunities for CO_2-C 'savings' at 1430-1640 kg/ha CO_2-C, with almost all of these due to increased SOC storage (rather than changes in energy use or N_2O emissions). However, as discussed above, it is questionable whether increases in SOC or CO_2-C savings following such organic material additions can be counted as genuine additional C storage (against a recent/present day baseline). This is probably the case for compost and paper crumble applications, with *c*.480,000 tonnes of green compost and *c*.700,000 tonnes of paper crumble currently recycled to agricultural land (Composting Association, 2005; Gibbs *et al.,* 2005). However, at current production and application rates they are only applied to relatively small areas of land (<50,000 ha), although compost use on agricultural land is expected to increase at least 3 to 5-fold over the next decade.

These management options are just some of those that have been proposed as potential methods for climate change mitigation (Smith *et al.,* 2007). It is probable that land-use change (e.g. from arable cropping to *permanent* willow/poplar biomass cropping, *permanent* grassland or woodland) offers the greatest potential for increased soil C storage and overall mitigation of GHG emissions from agricultural land. Also, of equal importance is the preservation of existing SOC stocks, particularly by avoiding the ploughing out of existing permanent grasslands and management of peatlands, which can result in significant SOC (and hence CO_2-C) losses.

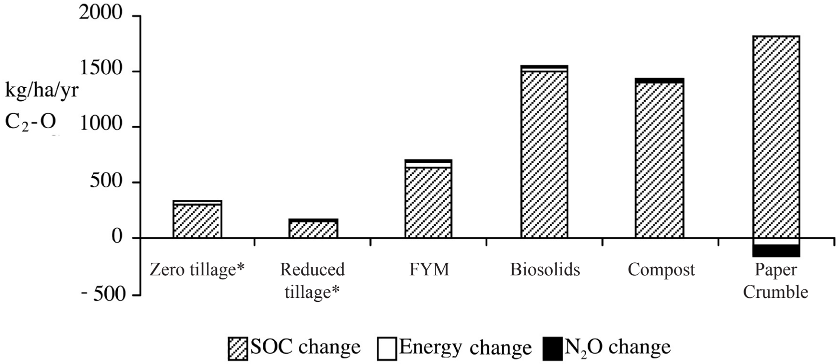


Figure 1: Maximum CO_2-C 'savings' from reduced/zero tillage and organic material additions to arable land
* Tillage figures exclude any potential change in N_2O emissions

CONCLUSIONS

This review has shown that there is limited scope for additional carbon storage from reduced/zero tillage. Indeed, there are questions over the implications of such practices on nitrous oxide emissions (as a result of increased topsoil wetness/reduced aeration compared with ploughing) and the overall balance of GHG emissions. Should reduced tillage be encouraged, it should be for its protection of existing SOC levels and benefits to soil water retention and prevention of erosion, as well as reduced production costs and energy use, rather than for additional carbon storage *per se.* Similarly, although the application of organic materials to land can make a valuable contribution to increasing soil C storage, there is limited scope for additional soil C storage (with the probable exceptions of compost and paper crumble applications), over and above a present/recent day baseline. The predominant justification for returning organic materials to soil should therefore be for maintaining and enhancing existing SOC levels, and completing natural nutrient and carbon cycles, not additional carbon storage for climate change mitigation *per se.*

ACKNOWLEDGEMENTS

Funding for this work from the Department for Environment, Food and Rural Affairs is gratefully acknowledged.

REFERENCES

Ball BC, Franklin MF, Holmes JC and Soane BD (1994). Lessons from a 26-year tillage experiment on cereals. In: Proceedings of the 13th International Conference of the International Soil Tillage Research Organisation, Aalborg, Denmark, Vol. 2, pp. 757-762.

Bhogal A, Nicholson, FA and Chambers, BJ (2007). Organic carbon additions – effects on soil bio-physical and physico-chemical properties. In: Organic Matter Dynamics in Agro-Ecosystems. Proceedings. 16-19 July 2007, Poitiers, pp. 334-335.

Cannell, RQ and Finney JR (1974). Effects of direct drilling and reduced cultivation on soil conditions for root growth. Outlook on Agriculture 7, 184-189.

Chaney K, Hodgson DR and Braim MA (1985). The effects of direct drilling, shallow cultivation and ploughing on some soil physical properties in a long-term experiment on spring barley. Journal of Agricultural Science, Cambridge 104, 125-133.

Composting Association (2005). The State of Composting in the UK 2003/2004. The Composting Association. ISBN 0-9547797-1-1.

Gibbs P, Muir I, Richardson S, Hickman G and Chambers B J (2005). Landspreading on agricultural land: nature and impact of paper wastes applied in England and Wales. Science Report SC030181/SR, Environment Agency, Bristol.

Gibbs PA, Chambers BJ, Chaudri AM, McGrath S P, Carlton-Smith CH, Bacon JR, Campbell CD and Aitken MN (2006). Initial results from a long-term, multi-site field study of the effects on soil fertility and microbial activity of sludge cakes containing heavy metals. Soil Use & Management 22, 11-21.

Hickman GAW, Chambers BJ and Pierrepont PJ (2000). Addressing customer concerns – Development of the 'Safe Sludge Matrix' for the sustainable application of sewage sludge to agricultural land. In: Agriculture and Waste Management for a Sustainable Future. SAC, pp. 4-10.

Jenkinson DS (1990). The turnover of organic carbon and nitrogen in soil. Philosophical Transactions of the Royal Society London 329, 361-368.

Jenkinson DS and Johnston AE (1977). Soil Organic Matter in the Hoosfield Continuous Barley Experiment. Rothamsted Experimental Station Report, Part 2, 87-101.

Johnston AE (1975). The Woburn Market Garden Experiment, 1942-1969 II. The Effects of the Treatments on Soil pH, Soil Carbon, Nitrogen, Phosphorus and Potassium. Rothamsted Experimental Station Report for 1974, Part 2, 102-131.

Mattingly GEG, Chater M and Johnston AE (1975). Experiments made on Stackyard Field, Woburn, 1876-1974 III. Effects of NPK Fertilisers and Farmyard Manure on Soil Carbon, Nitrogen and Organic Phosphorus. Rothamsted Experimental Report 1974, Part 2, 61-78.

Mackenzie AF, Fan MX and Cadrin F (1998). Nitrous oxide emission in three years as affected by tillage, corn-soybean-alfalfa rotations, and nitrogen fertilisation. Journal of Environmental Quality 27, 698-703.

Powlson DS and Jenkinson DS (1981). A comparison of the organic matter, biomass, adenosine triphosphate and mineralisable nitrogen contents of ploughed and direct-drilled soils. Journal of Agricultural Science, Cambridge 97, 713-721.

Smith P, Martino D, Cai Z, Gwary D, Janzen H, Pushpam K, McCarl B, Ogle S, O'Mara S, Rice C, Scholes B, Sirotenko O, Howden M, McAllister T, Pan G, Romanenkov V, Schneider U, Towprayoon S, Wattenbach M and Smith J (2007). Greenhouse gas mitigation in agriculture. Philosophical Transactions of the Royal Society B. 363. doi, 10.1098.

Stern N (2006). Stern Review Report on the Economics of Climate Change. http://www.hm-reasury.gov.uk/independent_reviews/stern_review_economics_climate_change

Wallace P (2005). Compost Use in Agriculture.
www.compost.me.uk

Water UK (2006). Recycling of Biosolids to Land. Water UK, 1 Queen Anne's Gate, London.

Webb J, Loveland PJ, Chambers BJ, Mitchell R and Garwood T (2001). The impact of modern farming practices on soil fertility and quality in England and Wales. Journal of Agricultural Science, Cambridge 137, 127-138.

SIMPLE VISUAL SCORING OF SOIL STRUCTURE

BC Ball[1], AM Liesch[2], T Batey[3] and LJ Munkholm[4]

[1]SAC, West Mains Road, Edinburgh EH9 3JG, UK, E-mail: Bruce.Ball@sac.ac.uk; [2]University of Wisconsin River Falls, WI-54022-5001, USA; [3]Plant and Soil Science Department, University of Aberdeen, AB24 3UU, UK; [4]Agroecology and Environment, Århus University, DK-8830, Tjele, Denmark

SUMMARY

Soil structure is critical to supporting sustainable land use. We describe a simple method to assess soil structure. This involves breaking-up a block of soil and examining the resulting natural soil aggregates and broken-up material. Soils are allocated to one of five categories of soil structure. The characteristics and types of soil found in each category are described. Changes required to restore and maintain soil structure at a satisfactory quality are indicated.

INTRODUCTION

Soil structure is the basic arrangement of soil particles into aggregates, groups of aggregates, clods and slabs. Soil structure affects root penetration, uptake of nutrients by roots, water storage and flow in soils and gas exchange. The preservation of good soil structure is essential to maintain these soil functions. We present a quick and simple method of determining the quality of soil structure which requires little equipment or expertise. The technique is designed for use by anyone with an interest in soil including consultants, farmers and gardeners. We describe the types of soil management producing each structural quality and relate this to soil and plant measures. The structural quality will determine if there are any needs to change crop or soil management.

MATERIALS AND METHODS

The method is split into three sections. First a block of topsoil 25 cm deep by 10 cm thick is dug out with a spade. The block is then examined and broken up and, if necessary, some of the resultant aggregates are broken apart. The block is then scored by comparison with a visual key and allocated to one of five classes of structural quality (Sq) (Table 1). This key is available in laminated plastic from the authors. The best quality category (Sq1) has many fine aggregates whereas the worst quality category (Sq5) has large slabs with low porosity and often contains soil of grey-blue colour typical of anaerobism. The Sq score is also confirmed by consideration of the ease of block extraction, shape and size of aggregates and distribution of roots. The method allows for assessment of horizontal layers of different structure within the spadeful, if present. Where this occurs, an average score is given for the block which is weighted for depth. The test takes 5-15 minutes per location and enough replicates can be obtained for statistical comparison of datasets. Typically, ten spots per field are assessed. An overall assessment for the field is given by averaging over spots. One of the sites where the method was developed was in a tillage experiment in Denmark where parallel measurements of penetration resistance and relative vegetation index (a measurement related to leaf area index) were made using the methods of Olesen and Munkholm (2007).

Table 1: Description of categories of scores within the key

Structure quality	Ease of break-up (moist soil)	Size and appearance of aggregates (after break-up)	Visible porosity	Roots	Distinguishing Feature
Sq1 Friable	Aggregates crumble easily	< 6 mm	Highly porous	Throughout soil	Small aggregates
Sq2 Intact	Aggregates easily broken with one hand	Porous, rounded 2 mm-7 cm	Porous aggregates	Throughout soil	High aggregate porosity
Sq3 Firm	Most aggregates break with one hand	2 mm-10 cm 30% are <1 cm. Clods may be present	Macropores and cracks may be visible	Most roots are around aggregates	Low aggregate porosity
Sq4 Compact	Considerable effort to break aggregates with one hand	Most >10 cm and subangular or platy <30% are <7 cm	Few macropores and cracks	All roots clustered in macropores and around aggregates	Distinct macropores
Sq5 Very compact	Difficult	Most >10 cm and angular	Very low, anaerobic zones are often present	Few, if any, restricted to cracks	Grey-blue colour associated with anaerobic conditions

The method is described in further detail by Ball *et al.* (2007).

RESULTS

About half an hour is needed to learn the test. Problems are occasionally encountered in interpretation of resistance to break-up when the soil is either very wet or very dry. We occasionally found that we could identify structures which had properties of two adjacent categories so we gave intermediate scores, e.g. Sq2.5 is between Sq2 and Sq3. We also found soils contained two or occasionally three layers with different Sq values. For example, ploughed soil could have a finer structure (Sq2) at 0-10 cm overlying a coarser structure (Sq3) at 10-25 cm. These differences will influence the storage and release of nutrients in relation to root exploration. Measurements from the tillage experiments in Denmark (Figure 1) show that as Sq values increase and structure becomes poorer, penetration resistance also increases and relative vegetation index decreases, indicating poorer conditions for crop growth.

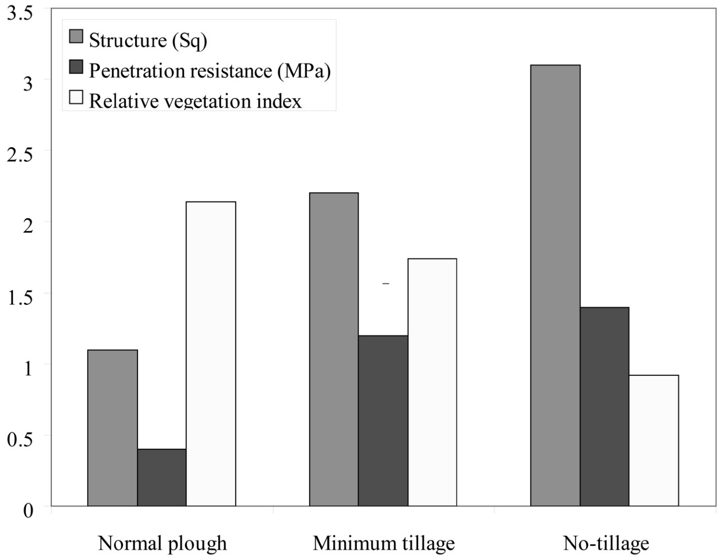


Figure 1: Soil structure quality (Sq) in relation to penetration resistance and relative vegetation index (divided by 10, see Olesen and Munkholm (2007)) in a tillage experiment at Foulum, Denmark

DISCUSSION

There are a wide variety of soil and management conditions under which the different structural qualities are found. Some of these are summarised in Table 2.

Table 2: Typical land uses in each Sq category

Sq	Land use
1	Long term pasture, seedbeds on coarse textured arable soils
2	Typical pasture; typical arable seed beds
3	Lightly poached pasture, post harvest of root crops in dry seasons
4	Moderately poached pasture, post harvest of root crops in wet seasons
5	Field roadways, tramlines and headlands, areas of repeated compression; severe poaching of wet soils

The soil structural qualities in Sq1 and Sq2 are satisfactory. The quality in Sq3 is adequate, though soil management may need adjustment to sustain satisfactory quality and crop productivity. The qualities of Sq4 and Sq5 require change in management such as loosening by tillage, drainage, incorporation of organic material or establishment of deep rooting plants to improve quality. We are currently working on guidelines for changes in soil management required in Sq3-5 to improve or sustain structural quality. A summary which includes the visual key is available from the senior author.

ACKNOWLEDGEMENTS

This work was sponsored by the Scottish Government RERAD (Rural and Environment Research and Analysis Directorate) and by the Danish Ministry of Food, Agriculture and Fisheries.

REFERENCES

Ball BC, Batey T and Munkholm LJ (2007). Field assessment of soil structural quality: a development of the Peerlkamp test. Soil Use and Management 23, 329-337.

Olesen JE and Munkholm LJ (2007). Subsoil loosening in a crop rotation for organic farming eliminated plough pan with mixed effects on crop yield. Soil and Tillage Research 94, 376-385.

NUTRIENT AND SEDIMENT EXPORT FROM A SMALL AGRICULTURAL CATCHMENT IN ANGUS, SCOTLAND

DJ Ballantine and JP Bowes

Scottish Environment Protection Agency, Carseview House, Castle Business Park, Stirling, FK9 4SW, UK, E-mail: jonathan.bowes@sepa.org.uk

SUMMARY

Intensively cultivated agricultural land is known to be a signficant contributor to nutrient and sediment loadings in any catchment. Export of nutrients and sediment from such catchments has the potential to cause eutrophication and reduced water quality in receiving waters, particularly in the case of a loch being a receptor. Nutrient and sediment export is generally event driven with highest loads being recorded in high flow events. Rescobie Loch is a hypertrophic loch in the Lunan Water catchment (Angus) in which phosphorus loads far exceed those outlined for good status as defined by the EU Water Framework Directive. To gain realistic estimates and to better understand loads of phosphorus and sediment to Rescobie Loch, a programme of sampling through high flow events was initiated on a 7 km^2 subcatchment of the loch. This subcatchment, the Burnside Burn, is thought to be a signficant contributor of phosphorus to Rescobie Loch. Land use within this subcatchment is mainly intensive agriculture, with cereals, potatoes and root crop cultivation dominating. Three events were recorded in the winter of 2006/2007, during which flow ranged from 0.050 to 0.259 cu mecs. Results from chemical analysis showed that phosphorus concentrations increased during the high flow events and were far in excess of the standards outlined for good status as defined by the EU Water Framework Directive. Similarly, concentrations of suspended solids increased through the high flows. Concentrations of SRP ranged from 0.041 mg/l to 0.841 mg/l over the three events while suspended solids concentrations reached a maximum of 820 mg/l. Loads of SRP and suspended solids exported through the three events totalled 2.68 kg and 2770 kg respectively. These results from initial sampling suggest that this subcatchment is a signficant source of phosphorus and sediment to the loch, contributing to its poor trophic status and any remediation efforts within the Lunan Water catchment should be directed at better land management in small subcatchments such as the Burnside Burn.

TOWARDS A NATIONAL RESEARCH FACILITY AT NORTH WYKE

D Beaumont, D Hatch, R Bol, P Murray and L Firbank

North Wyke Research, Okehampton, Devon, EX20 2SB, UK, E-mail: david.hatch@bbsrc.ac.uk

SUMMARY

Agriculture is increasingly required to deliver multiple environmental and production outputs as demand on the land intensifies. Urgent research is required to provide the evidence base for the future management of complex agricultural systems that will improve our understanding of the interactions between soil, microbes, plants, animals, water and the atmosphere within grassland-dominated landscapes

North Wyke Research intends to provide a unique national research facility to develop and test hypotheses to compare a conventional, lowland beef production system with alternative multifunctional land use options in grassland landscapes.

We propose to develop an instrumented experimental platform at North Wyke to monitor a range of indicators at various spatial and temporal scales. The indicators will account for a wide range of ecosystem functions including yield, emissions to air and water, biodiversity, and landscape quality. Appropriate indicators for long-term monitoring will be selected to enable successful environmental evaluation. Where possible, sensing will be automated using a wireless network to allow real-time data collection that will contribute to the evidence base for the sustainable land management systems that are: i) adaptable to different production and/or environmental goals; ii) help protect our natural resources; iii) help mitigate against, and adapt to climate change and iv) take account of changing social, economic, policy and environmental conditions.

The platform will provide quality assured data and a facility for innovative experiments for scientists and collaborators. Researchers will be able to explore sustainability, system resilience and develop a range of ecosystem heath indices for different land use scenarios. Data from the platform will be used to test and develop predictive models for future management systems

Research outcomes will enable farmers, agri-food industry, regional and national policy makers, agencies, NGOs and local people to make informed management decisions to deliver environmental, social and economic benefits in lowland grasslands. We will seek to develop resilient funding streams from multiple sources and exploit the outputs of our research to ensure economic and societal impact by supporting training and innovation in the agri-food industry of south-west England. The Research facility will also provide a valuable national resource for training of postgraduate students.

INFLUENCE OF SOIL ORGANIC MATTER AND NITROGEN FERTILISER ON N_2O EMISSION FROM A LOAMY SAND SPODOSOL OF NORTH-WESTERN REGION OF RUSSIA

NP Buchkina, EY Rizhiya, EV Balashov and SV Pavlik

Agrophysical Research Institute, St. Petersburg, 14 Grazhdansky prospect, 195220, Russia, E-mail: natalya_buchkina@yahoo.com

SUMMARY

The purpose of the study was to find out how properties of light-textured soils changed by high rates of FYM affect subsequent N_2O emissions from the soils. Direct N_2O fluxes were measured from a loamy sand spodosol of north-western Russia during the growing season of 2006. Two plots containing different amounts of soil organic carbon (SOC) due to earlier amendments in one of them with farmyard manure (FYM) were compared in the experiment. The soil in the two plots had different chemical, biological and physical properties. Barley and cabbage were grown on the two soils with and without extra N application as ammonium nitrate. In 2006 generally positive effects of N-input and SOC content on N_2O flux were masked by dry weather conditions in the growing season. Cumulative N_2O fluxes from all the plots were low and the only significantly higher cumulative N_2O fluxes were found from the "FYM" and "no FYM" soils under barley receiving extra mineral N fertiliser.

INTRODUCTION

It is very common in many countries, including Russia, to use FYM on light-textured soils to improve such soil properties such as increasing soil organic matter and nutrient content, water-holding capacity, and reducing soil bulk density. If rates of FYM are very high, some properties of light-textured soils can be affected quite substantially for several years. The main purpose of this study was to find out how high rates of FYM change properties of light-textured soils and whether these changes affect direct nitrous oxide emissions from the soils, especially if extra nitrogen is added to the soil as mineral fertilisers.

In many previous studies it was shown that application of manures into soils encouraged N_2O production but there were studies indicating that there was as yet no conclusive evidence on the influence of application of organic manure on direct N_2O emissions from soils (Chadwick *et al*., 1999).

MATERIALS AND METHODS

The field experiment was conducted at Menkovo Experimental Station of the Agrophysical Research Institute in St. Petersburg region of Russia (59°34'N, 30°08'E) in 2006. The soil of the region was well-drained(?) loamy sand spodosol. Two plots were included in the experiment. One was amended with FYM twice – in spring 2003 and 2004 - receiving 160 t ha^{1} of FYM (656 kg N ha^{-1}) each time ("FYM" soil). The other plot was left with no FYM ("no FYM" soil). In 2006 the "FYM" and "no FYM" plots were divided into two: one half of each plot received extra nitrogen (on 10 May) as ammonium nitrate - 110 kg N ha^{-1} for "FYM" soil and 60 kg N ha^{-1} for "no FYM" soil (higher mineral N application into the "FYM" soil was chosen as higher crops were expected from the soil with higher SOC) - while the other half did not receive any extra nitrogen. Two crops were grown on each fertilised and not-fertilised plots – barley *(Hordeum vulgare* L.*)* and cabbage *(Brassica oleracea* L.*)*. The closed chamber technique was used to measure direct N_2O fluxes from the soil (Buchkina *et al*., 2006) with chamber area being 0.03 m^2. Gas samples were collected between noon and 2 pm two-three times a week throughout the growing seasons (May-September). Four replicate chambers were used on all barley plots and eight (four in furrows and four on ridges) on all cabbage plots. Average daily fluxes and cumulative N_2O fluxes for the growing seasons were calculated for each plot with errors and standard deviations.

SOC content, content of soil available nitrogen (N), pH, soil bulk density (BD), and gravimetric soil water content (W) were measured in top 10-cm layer with conventional methods (Rastvorova, 1983; Rastvorova *et al*., 1995). Microbial biomass content (MBC) was measured with the method of substrate-induced respiration (Anderson and Domsch, 1978). Basal respiration (BR) was measured as CO_2 production after a 1-day incubation of the soil samples at field capacity and at 30°C.

Soil of the cabbage plots was mechanically cultivated to 10-12 cm depth at least once in every two weeks (to kill the weeds) for about 1.5 months after cabbage seedlings were planted while the soil of the barley plots was not disturbed from the planting date until harvest.

RESULTS

Soil Properties

Application of high rates of FYM to the spodosol in 2003 and 2004 resulted in changes of some soil properties and these changes were still substantial in April 2007 (Table 1). The "FYM" soil had higher SOC content, MBC, BR, pH and contained more available mineral nitrogen (N_{min}). At the same time it had lower soil bulk density (BD) and higher water-holding capacity than "no FYM" soil.

Table 1: Soil properties of light-textured loamy sand spodosol of north-western region of Russia in April 2007 (without and with FYM amendment in 2003 and 2004)

	SOC, g C kg^{-1} soil	pH	BD, g cm^{-3} (top 10 cm layer)	W, %	MBC, mg C kg^{-1} soil	BR, mg C-CO_2 kg^{-1} soil h^{-1}	N_{min}, mg N kg^{-1} soil
Without FYM	17.0	6.4	1.5	19.8	329±3	4.6±0.3	20.2
With FYM	25.4	6.9	1.2	32.1	439±15	7.8±0.5	49.5

SOC – soil organic carbon, BD – soil bulk density, W – soil water content, MBC – microbial biomass content, BR – basal respiration, N_{min} – soil available nitrogen

During the growing season of 2006, which was mostly drier than average (Figure 1e), water-filled pore space (WFPS) of the top 10-cm layer of the soils on cabbage ridges changed from 9 to 20% for both “FYM” and “no FYM” soils. In cabbage furrows, as well as on barley plots, the WFPS was somewhat higher but still low, at 15-30% *per se*.

Weather Conditions

The growing season of 2006 was drier than the 20-year average - there were 88, 63 and 91 mm of rain in June, July and August, respectively, when the 20-year average values for the same months were 106, 120 and 124 mm respectively. There were 113 and 101 mm of rain in May and September 2006, respectively, and these values were 20-30 mm higher than the 20-year average values for these months, but despite the greater rainfall in May the cabbage crop was suffering from drought at the end of July.

N_2O Flux

Cumulative N_2O fluxes for all the plots during the growing season of 2006 are shown in Table 2. Where no extra mineral nitrogen was applied, the plots produced from 300 to 500 g N_2O-N ha^{-1} for the growing season under both crops. For barley the difference between the cumulative N_2O flux of “FYM” and “no-FYM” soils was significant whereas the difference for the cabbage crop was not significant ($p<0.05$).

Application of extra mineral nitrogen into the soils in 2006 led to significant increase in cumulative N_2O flux only from the “no-FYM” soil under barley. The cumulative N_2O flux from the “FYM” soil under barley also increased after the mineral fertiliser application, but the difference with the “FYM” soil receiving no extra nitrogen was not significant.

Table 2: Cumulative N_2O flux from the light-textured spodosol with different treatments for the growing seasons of 2006

Crop	Barley	Barley	Barley	Barley	Cabbage	Cabbage	Cabbage	Cabbage
Manure/ Fertiliser application	no FYM, no N	no FYM, 60 kg ha^{-1} N	FYM, no N	FYM, 110 kg ha^{-1} N	no FYM, no N	no FYM, 60 kg ha^{-1} N	FYM, no N	FYM, 110 kg ha^{-1} N
Cumulative flux*, g N_2O-N ha^{-1}	313 ± 70	662 ± 163	488 ± 88	728 ± 181	404 ± 80	387 ± 46	363 ± 89	466 ± 108
Crop yield, 10^3 kg ha^{-1}**	2.1	2.32	3.87	4.53	41.4	59.9	81.1	103.32

* - Errors shown are standard errors; ** - Data of Dr. E A Oleychenko (not published)

Application of mineral nitrogen to “FYM” and “no-FYM” soils with cabbage did not result in significant increase of cumulative N_2O flux in either soil. Cumulative N_2O fluxes from all the four cabbage plots were about the same irrespective of whether they received FYM or extra N with the mineral fertiliser.

There were statistically significant differences between daily N_2O fluxes from the barley plots receiving extra N and the ones receiving no extra N for both "no-FYM" and "FYM" soils during most of the growing season (Figure 1). For the cabbage plots, N-application increased daily N_2O fluxes only from "FYM" soil and had no effect on the fluxes from "no FYM" soil.

Yields

Amendment of the light-textured soil with FYM in 2003 and 2004, as well as fertilisation of "FYM" and "no FYM" soils with mineral fertiliser in 2006 significantly affected both barley and cabbage yields in 2006 (Table 2). Yields of barley and cabbage produced by the "FYM" soil were higher than the yields of the "no FYM" soil. Extra mineral N input into both soils increased the yields even more and not fertilised "FYM" soil produced higher yields than the fertilised "no FYM" for both barley and cabbage.

DISCUSSION

Cumulative N_2O fluxes were very low from all the plots for the growing season of 2006 compared to previous measurements at the same site in wetter growing seasons (Buchkina *et al.*, 2006). The differences between "FYM" and "no FYM" soils were mostly not significant for this growing season even when extra mineral nitrogen was applied. The only soils where extra mineral N resulted in the production of more N_2O were "FYM" and "no FYM" soils under barley but even these soil emitted quite low quantities of N_2O for the growing season of 2006 compared to the same soils of the earlier, wetter seasons (data not shown).

It was reported by other scientists (Dobbie *et al*., 1999) and also shown in our work on the same soil in 2004 (Buchkina *et al*., 2006) that cumulative N_2O fluxes from soil where crops were grown on ridges were higher than fluxes from soils under cereals. This results from the higher WFPS in the furrows, causing the soil to emit substantially more N_2O than from ridges or cereal cropping. Conversely, in this study, cumulative N_2O fluxes from all the cabbage plots were significantly lower than from the fertilised plots under barley for both "FYM" and "no FYM" soils. These differences were attributable to a combined effect of several factors: very dry weather of this particular growing season, the fact that the cabbage crop was mechanically cultivated once in every two weeks in June and July (to stop weeds growing and drying the soil), and the fact that soils of the cabbage plots were more exposed to the sun as large areas of the plots were not covered by the crop for most of the growing season. We observed in our earlier studies on the same soil that even when there was plenty of rain the correlation between N_2O flux and rainfall for the crops grown on ridges was not as strong as for cereals in June and July, when the soils with ridges were mechanically cultivated (not published). We concluded that these mechanical cultivations increased soil aeration and pore space reducing WFPS even in the soils in the furrows. Probably in dry seasons the effect of these mechanical cultivations on soil properties was even higher and soil conditions of the cabbage plots were less suitable for N_2O production than those of the barley plots which were not disturbed from seeding until harvest. High yearly variations of cumulative N_2O emissions and the effect of weather conditions, especially rainfall, on the N_2O flux were also reported by other scientists (Smith *et al*., 1998).

The FYM plots for both crops received almost twice as much of NH_4NO_3 in 2006 than the "no FYM" plots. As expected the yields of both crops were significantly affected by extra N application but cumulative N_2O fluxes from these plots did not differ significantly from each other for either crop. N_2O emissions from soils are generally strongly affected by amount of available nitrogen as well as by soil organic carbon content (Smith *et al*., 1998; Dobbie and Smith, 2003) in wet growing seasons. However, in dry seasons the effect of available N and SOM was probably masked by the strong influence of weather conditions on soil WFPS.

a) Barley, no FYM

b) Barley, FYM

c) Cabbage, no FYM

d) Cabbage, FYM

e) Precipitation

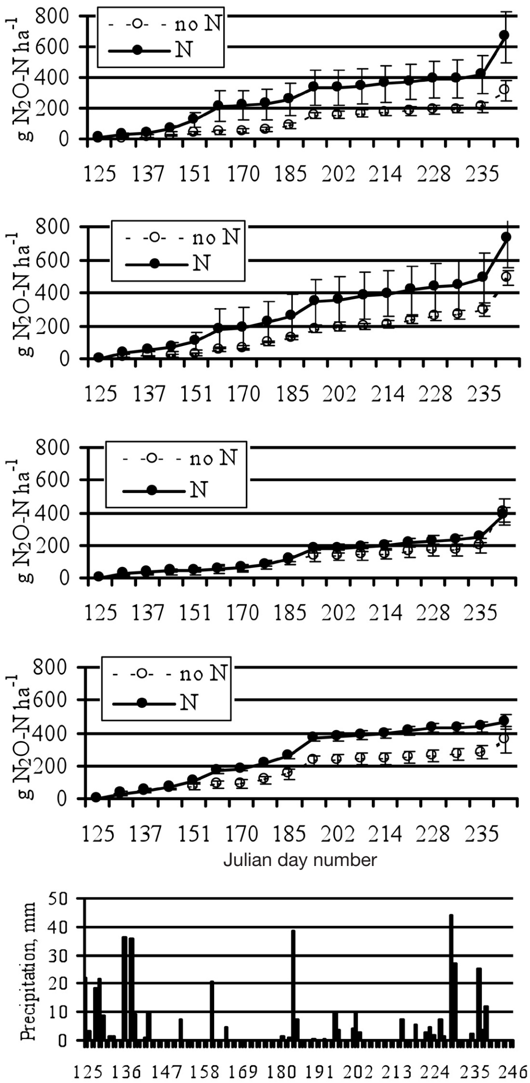


Figure 1: Daily precipitation and cumulative N_2O flux for barley and cabbage crops in the growing season of 2006

As was concluded by Sehy *et al*. (2003) from their study of site-specific fertiliser application on N_2O emissions and maize yields, annual variations in climate, especially rainfall pattern, need to be taken into account for site-specific N-fertiliser application if N_2O emissions from soils are to be reduced and crop yields are to be increased. The effect of high rates of N-application in soils appears to be more significant in wet growing seasons that in dry ones.

ACKNOWLEDGEMENTS

We thank Prof. V A Semenov and Dr E A Oleychenko for the data on the crop yields in 2006. The N_2O measurements were partly financed by the NATO Collaborative Linkage Grant. We also thank Dr B D Soane for his help with preparation of the paper for publication.

REFERENCES

Anderson JPE and Domsch KH (1978). A physiological method for the quantitative measurement of microbial biomass in soils. Soil Biology and Biochemistry 10, 215-221.

Buchkina NP, Balashov EV, Rizhiya EY and Smith KA (2006). Nitrous oxide emissions from a light agricultural soil of North-Western Russia in relation to applied N fertiliser and manures. Proceedings of the 17th International conference ISTRO Kiel Germany, 19-20.

Dobbie KE, McTaggart IP and Smith KA (1999). Nitrous oxide emission from intensive agricultural systems: variations between crops and seasons, key driving variables, and mean emission factors. Journal of Geophysical Research 104 (D21), 26891-26899.

Chadwick DR, Sneath RW, Phillips VR and Pain BF (1999). A UK inventory of nitrous oxide emissions from farmed livestock. Atmospheric Environment 33, 3345-3354.

Dobbie KE and Smith KA (2003). Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water-filled pore space and other controlling variables. Global Change Biology 9, 1-15.

Rastvorova OG (1983). Soil physics (in Russian). Leningrad University press, Leningrad, Russia, 196 p.

Rastvorova OG, Andreev DP, Gagarina EI, Kasatkina GA and Fedorova NN (1995). Chemical analyses of soils (in Russian). St Petersburg University press, St Petersburg, Russia, 254 p.

Sehy U, Ruser R and Munch JC (2003). Nitrous oxide fluxes from maize fields: relation to yield, site-specific fertilisation, and soil conditions. Agriculture, Ecosystems and Environment 99, 97-111.

Smith KA, Thomson PE, Clayton H and McTaggart IP (1998). Effects of temperature, water content and nitrogen fertilisation on emissions of nitrous oxide by soils. Atmosphere Environment 32 (19), 3301-3309.

THE EFFECT OF DIFFERENT DAIRY SYSTEMS ON ENTERIC METHANE PRODUCTION AND NON-MILK NITROGEN DURING THE WINTER FEEDING PERIOD

MGG Chagunda, DAM Römer, JF Bell and DJ Roberts

SAC, Dairy Research Centre, Midpark House, Bankend Road, Dumfries, DG1 4SZ, UK, E-mail: Mizeck.Chagunda@sac.ac.uk

SUMMARY

The objectives of the current study were to quantify the impact on the environment of different dairy production systems. Two sets of traits were studied in four dairy production systems. In the first set, traits representing the polluting effect of dairy cows to either the air or the underground water resources were calculated. Enteric methane production were calculated to quantify the impact of dairy systems on the atmospheric environment while non-milk nitrogen (N) represented potential pollution into the underground water resources. In the second set, energy corrected milk yield and body energy content were determined to represent traits indicating production efficiency from a cow productivity perspective. After correcting for systematic factors, highly significant differences in environmental impact were found between the production systems on their impact on the environment. Cows with increased productivity in terms of milk yield and body energy reserves were associated with a decrease in methane production per kg of milk but an increase in nitrogen loss. The importance of the pollution cost of dairy production on the environment suggests the need to incorporate environmental issues in dairy systems analysis.

INTRODUCTION

Methane and nitrates are some of the important pollutants from dairy production. Quantification of methane (CH_4) in dairy cattle production is a subject of great interest because large emissions occur from enteric fermentation in cows and also from anaerobic storage of manures (Hensen *et al*., 2006). Further, the Intergovernmental Panel for Climate Change (IPCC) has recently reported sharp increases in concentrations of greenhouse gases of which methane is one (IPCC, 2007). Together, enteric fermentation and manure represent some 80% of agricultural methane emissions and about 30 to 40 percent of the total anthropogenic methane emissions (FAO, 2006). Nitrates coming from dairy cattle represent the feed nitrogen component that is not utilised by cows and potentially can pollute through runoff, volatilisation and leaching (Jonker *et al*., 2002). Understanding the effect of different genotypes and feeding regimes and their interactions on the production of polluting agents is important not only for dairy productivity but also for developing mitigation strategies for the contribution of agricultural activities to anthropogenic greenhouse gas (GHG) emissions (Weiske *et al*., 2006) and underground water pollutants. Hence the objective of the current study was to determine the effect of genotype and feeding regime on enteric methane and non-milk nitrogen production during the winter feeding period.

MATERIALS AND METHODS

Animals and Production Systems

Data were obtained for the winter feeding period from a herd of Holstein Friesian cows, which are on a long-term 2 x 2 factorial, genotype x management system project based at the SAC Dairy Research Centre, Dumfries, Scotland. Two contrasting approaches to dairy herd management systems were practiced. The two systems were a high forage system (HF) and low forage system (LF). In HF system, the cows grazing when sufficient herbage was available and fed a complete diet containing between 70% and 75% forage in the dry matter when grass heights fell below set values and in the winter months. In the LF system, the cows were housed throughout the year and had access to a roofed loafing area for approximately eight hours per day during the summer months. The cows in the LF system were fed a complete diet containing between 45% and 50% forage dry matter (DM). A summary of the feed composition for the feed offered to the two groups is presented in Table 1. Each management system consisted of cows that belonged to one of the two genetic lines (Select and Control) based on merit for kilograms milk fat plus protein. The Select group cows were sired by bulls with high predicted transmitting abilities (PTA) for fat plus protein yield, whereas the Control cows were sired by bulls of UK average merit for fat plus protein (Pryce *et al*., 1999). This breeding programme has been running since 1970. Approximately equal numbers of cows formed the Select line-low forage (LFS), Select line-high forage (HFS), Control line-low forage (LFC), and Control line-high forage (HFC) groups. The same herdsman was responsible for cows in both management systems which were milked through the same 24:24 herringbone parlour. The cows were milked three times per day at approximately 05.00; 13.00 and 20.00 hours. The cattle, when housed, were in the same building with cubicles and concrete passageways. Passageways were scraped clean automatically every four hours.

Table 1: A summary of the feed composition for the feed offered to the two groups during the study period on dry matter (DM) basis

Feed Constituent	Group					
	Low Forage			High Forage		
	Average	Min	Max	Average	Min	Max
ME (MJ/kg DM)	12.3	12.1	12.5	11.5	11.3	11.7
Crude Protein (g/kg DM)	185	180	190	180	175	185
Oil (g/kg DM)	60	55	665	50	45	55
Starch (g/kg DM)	180	160	200	130	110	150
Sugar (g/kg DM)	70	60	80	50	40	60
NDF (g/kg DM)	345	330	360	390	360	420
NDF from forage (g/kg DM)		240			270	
Dry matter (%)		45			30	

Milk yields of individual cows were recorded at each milking and individual cow milk samples taken weekly for analysis of fat, protein and somatic cell contents. Live weights were measured after each milking and conditions cores carried out weekly using the tail head systems (Mulvanny, 1977). The feed intake was recorded on 3 days out of six using Hoko gates (Insentec BV, Marknesse, The Netherlands). The chemical compositions of the silages and concentrate were determined at the SAC Analytical Lab, Edinburgh, Scotland. Grass heights were recorded twice weekly, when the cattle were grazing, using a rising plate meter. The cattle in the HF herd were grazed for three periods each day on perennial ryegrass swards when compressed grass heights exceeded 9 cm. The grazing periods were reduced to two and one period when grass heights fell below 9 and 7 cm, respectively. The cattle were grazed as one group, except for the period after turnout when they were only grazing for one period/day. At this time they were grazed as two groups corresponding to their winter feeding groups. Health records were kept throughout the experiment and locomotion scores were recorded weekly, using a 1 to 5 scale (Manson and Leaver, 1988).

Environmental Impact and Cow Productivity Traits

For the purposes of the current study, two sets of traits were studied. In the first set, traits representing the polluting effect of dairy cows to either the air or the underground water resources were calculated. Enteric methane production was calculated to quantify the impact of dairy systems on the atmospheric environment while non-milk N (representing faecal N, urine N, and retained N) was calculated as an indicator for potential pollution into the underground water resources. In the second set, energy corrected milk yield and body energy content were determined to represent traits indicating production efficiency from a cow productivity perspective. To estimate these two environmental-impact traits, equations obtained from literature were used. Different equations of varying levels of complexity for estimating methane and non-milk N per cow per day exist in the literature. However, the selection of the equations used in the current study was based on both the biological relevance and the practical availability on the farm of the parameters in the equation. Enteric methane production per cow per day was calculated using two different equations, one that uses total DM intake, DM intake from concentrate component of the feed (CDMI), and the neutral detergent fibre (NDF) in the feed (Yates *et al*., 2000), i.e. CH_4 (MJ/day) = 1.36 + 1.21 * DMI - 0.825 * CDMI + 12.8 * NDF. The other equation uses DM intake, feed characteristics and milk characteristics i.e. Methane energy (% gross energy) = 2.927-0.0405*milk (kg/d) + 0.335 * milk fat(%) – 1.225 * milk protein (%) + 0.248 * CP (%DMI) – 0.448 * ADF (%DMI) + 0.502 * forage ADF (%DMI) + 0.0* ADF digestibility(%) (Holter and Young, 1992). In both cases, the methane energy output was initially converted to grams of methane per cow per day and further to grams of methane per kg of milk. Non-milk N was calculated as the difference between N intake and milk N.

Statistical Analysis

Data were analysed using a univariate mixed model which included the following fixed factors: production system, parity, days in milk, year of production, and days in milk. The individual cow within parity and random residual were included as random effects. Cows in any parity higher than three were combined with those in parity three.

Analysis was undertaken using restricted maximum likelihood (REML) methodology implemented by the MIXED procedure of SAS version 8.2 (SAS Inst. Inc., 2001). Least square means for DIM were used to generate profiles for methane and manure N over the first part of the lactation. To determine the relationships among the dairy production efficiency traits and environmental-impact traits, correlation analysis was used.

RESULTS

Highly significant differences were found among the four production systems on their impact on the environment. Effects of systematic factors on the studied traits are presented in Table 2.

Table 2: Estimates (least square means) and SEM for enteric Methane (CH_4) using two different methods, Non-milk Nitrogen (Nnm), Energy corrected milk (ECM) and Body Energy Content (EC) for cows in different production systems

Factor	Effect	CH_4 g/kg milk ([2]method 1)		CH_4 g/kg milk (method 2)		Nnm g/cow/d		ECM (kg/d)		EC (MJ/d)	
		lsmean	SEM	Lsmean	SEM	lsmean	SEM	lsmean	SEM	lsmean	SEM
Parity	1	15.9	0.14	23.0	0.37	433.7	4.67	23.9	0.31	2172.1	34.9
	2	15.0	0.21	22.6	0.54	523.1	6.77	28.8	0.44	2116.7	48.8
	3	14.2	0.34	21.8	0.86	547.2	10.85	30.6	0.71	2006.0	78.8
System[1]	LFS	12.2	0.25	20.8	0.60	583.2	7.71	33.3	0.51	2234.8	56.5
	HFS	16.4	0.24	22.1	0.58	459.7	7.61	26.3	0.51	2011.7	57.8
	LFC	13.5	0.23	22.8	0.57	523.6	7.35	27.9	0.48	2224.7	53.9
	HFC	18.5	0.26	24.3	0.63	438.7	8.04	23.4	0.53	1921.8	58.8
Year	2003	16.6	0.30	28.1	1.03	536.3	9.16	27.7	0.46	1908.8	38.3
	2004	13.7	0.20	19.2	0.55	487.6	6.22	28.5	0.36	2135.9	34.8
	2005	14.5	0.15	20.8	0.39	488.1	4.67	27.9	0.30	2125.9	32.9
	2006	15.4	0.14	21.9	0.35	493.2	4.39	26.8	0.29	2222.4	32.8

[1]LFS = low forage, high genetic merit group; HFS = High forage, high genetic merit group; LFC = Low forage, low genetic merit group; HFC = High forage, low genetic merit group. [2]Method 1 = Yates *et al*. (2000); Method 2 = Holter and Young (1992)

After correction for different animal and environmental traits it was found that both the genotype and the feeding regime had significant effect on enteric methane and non-milk nitrogen production during the winter feeding period. Generally, the cows in the low forage systems had high milk production and also high body energy reserves. From a production point of view, the low forage systems could be considered as high productivity ones. Methane production per kg milk estimated by the two methods gave results which were in general agreement. The correlation coefficient between the two methods was 0.4 ($p<0.001$). In both methods CH_4/kg milk was lowest in the LFS group and highest in the HFC group. For non-milk N, LFS had the highest values while HFC had the lowest values. On the cow productivity traits, cows in LFS had the highest milk yield and cows in HFC had the lowest milk yield. Similar to milk yield, cows in LFS had the highest body energy reserves while cows in HFC had the lowest body energy reserves.

Between-system differences for CH_4 /kg milk indicate clusters. Using method 1, the values of methane per kg milk for the cows in LFS and LFC groups clustered together while HFS and HFC formed another cluster. With method 2, HFS and LFC clustered together. In both methods, LFS had the lowest while HFC had the highest enteric methane production per kg of milk. For non-milk N, LFS and LFC had numerically higher values than HFS and HFC groups.

DISCUSSION

This study demonstrated that the various dairy systems had different environmental impact during the winter feeding period. Generally, high forage systems had higher methane production per kg of milk than low forage groups. However, low forage groups had higher nitrogen loss than high forage groups. Hence, cows with increased productivity in terms of milk yield and body energy reserves were associated with a decrease in methane production per kg of milk but an increase in nitrogen loss. The results highlight the importance of the need to incorporate the dynamics of emission burden and pollution potential of different dairy systems in characterising different dairy systems. By including the environmental issues in calculations of productivity for dairy systems, production should be improved without compromising the environment.

ACKNOWLEDGEMENTS

We are grateful to the staff at the SAC Dairy Research Centre, from where data used in this study were obtained, and to Ross McGinn, the database manager. SAC receives financial support from The Scottish Government.

REFERENCES

FAO (2006). Livestock's long shadow: environmental issues and options. Food and Agriculture organization, Rome, Italy.

Hensen A, Groot TT, van der Bulk WCM, Vermeulen AT, Olesen JE and Schelde K (2006). Dairy farm CH_4 and N_2O emissions, from one square metre to the full farm scale. Agriculture Ecosystems and Environment 112, 146-152.

Holter JB and Young AJ (1992). Methane prediction in dry and lactating cows. Journal of Dairy Science 75, 2165-2175.

Intergovernmental Panel on Climate Change (IPCC) (2007). Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the

Fourth Assessment Report of the Intergovernmental Panel on Climate Change. http://ipcc-wg1.ucar.edu/wg1/docs/WG1AR4_SPM_Approved_05Feb.pdf

Jonker JS, Kohn RA and High J (2002). Dairy herd management practices that impact nitrogen utilization efficiency. Journal of Dairy Science 85, 1218-1226.

Manson FJ and Leaver JD (1988). The influence of concentrate amount on locomotion and clinical lamaness in dairy cattle. Animal Production 47, 185-190.

Mulvanny P (1977). A body condition scoring technique for use with British Friesian cows. Animal Production 24, 157-158.

Pryce JE, Nielsen BL, Veerkamp RF and Simm G (1999). Genotype and feeding system effects and interactions for health and fertility traits in dairy cattle. Livestock Production Science, 57(3), 193-201.

SAS Institute Inc (2001). SAS Institute Inc., Cary, NC, USA.

Weiske A, Vabitsch A, Olesen JE, Schelde K, Michel J, Friedrich R and Kaltschmitt M (2006). Mitigation of greenhouse gas emissions in European conventional and organic dairy farms. Agriculture Ecosystems Environment 112, 221-232.

Yates CM, Cammell SB, France J, Beever DE, (2000). Prediction of methane emissions from dairy cows using multiple regression analysis. Proceeding of the British Society of Animal Science 2000, 94.

THE SAC ENVIRONMENTAL FOCUS FARMS PROJECT - YEAR ONE

C Christian[1], W Crooks[2], R Crerar[2], E Hart[3], W Jeffrey[1], A Jones[2], AH Sinclair[4] and J Skinner[1]

[1]SAC, Bush Estate, Penicuik EH26 0PH, UK, E-mail: Carole.Christian@sac.co.uk; [2]SAC, Auchincruive, Ayr, KA6 5HW, UK; [3]SAC Forfar, 77 North Street, Forfar, DD8 3BL, UK; [4]SAC, Craibstone Estate, Aberdeen, AB21 9YA, UK

SUMMARY

SAC's Environmental Focus Farm (EFF) Project has been established in two catchments, one in Angus and the second in Ayrshire. The catchments have been chosen by the Scottish Environment Protection Agency (SEPA) because they represent land use patterns typical of east-coast arable agriculture and west-coast dairying respectively. Both catchments are challenged by diffuse pollution and water quality issues and may not meet the environmental objectives of the Water Framework Directive (WFD).

The EFF project, working with farmer Community Groups to implement Best Management Practices (BMPs) to reduce agricultural diffuse pollution, aims to encourage farmer Group members to initiate these workable, affordable and effective BMPs on their own farms.

In Year One of the programme, the Groups have been established, baseline data collected and Diffuse Pollution Audits completed. An ongoing programme of events and activities is in hand with input from specialists directed in part by the farmers' own objectives.

INTRODUCTION

The Environmental Focus Farm project has been established by the Scottish Agricultural College (SAC) as a response to the selection of two catchments in Scotland as Monitored Priority Catchments (MPCs) by the Scottish Environment Protection Agency (SEPA). The selections have been made to enable SEPA to carry out detailed studies of the potential sources of pollution in the catchments and to identify the risk that they pose to water quality. The two catchments are the Lunan Water in Angus and the Cessnock in Ayrshire; both are shown in Figure 1 overleaf.

They were chosen by SEPA because they represent land use patterns typical of east-coast arable agriculture and west-coast dairying respectively and because they are at risk of not meeting the environmental objectives of the Water Framework Directive (WFD). Initial characterisation work undertaken by SEPA indicates that agriculture comprises a main contributor to diffuse pollution in these two catchments.

The project comprises several strands of work. As well as SAC's contribution, The Macaulay Institute are carrying out monitoring in the Lunan Water catchment and SEPA are monitoring various surface and ground water parameters in both catchments.

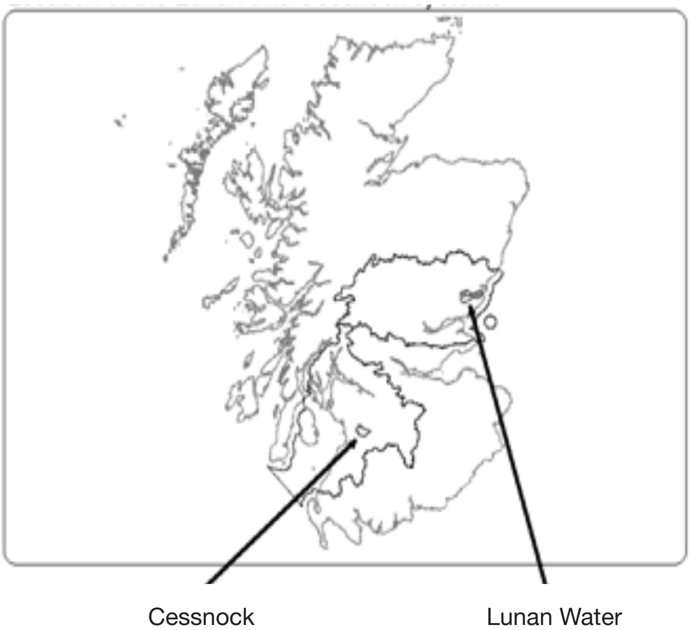


Figure 1: Map showing locations of Lunan Water in Angus and Cessnock in Ayrshire

The Lunan Water catchment includes two lochs that are Sites of Special Scientific Interest (SSSIs); Rescobie Loch and Loch of Balgavies. Rescobie Loch in particular has experienced significant nutrient over-enrichment, and algal blooms are a problem. The whole catchment is at risk of not meeting the environmental objectives of the WFD. As well as nutrients, sediment and, to a lesser extent, pesticides are under study in the catchment.

The Cessnock Water is a tributary to the River Irvine which discharges at Irvine Beach. This is a designated Bathing Beach and the condition of bathing waters here have a historically poor quality record for Faecal Indicator Organisms (FIOs). As well as agriculturally-derived faecal matter, the River Irvine catchment also contains sewer overflows, emergency outfalls and septic tanks. Nutrient over-enrichment of the Cessnock is a problem with high Biochemical Oxygen Demand (BOD) materials being of particular concern because of their potential for water de-oxygenation and consequent impacts on freshwater ecology.

Agriculture in the Catchments

Agriculture in the Lunan Water catchment is predominantly arable including potatoes with some livestock enterprises and vegetable and soft fruit growing. The average farm size is 300ha and there are approximately 37 farm businesses in the catchment.

The Cessnock catchment is mainly dairy-based with suckler beef playing an increasingly important role. The average farm size in the catchment is 81ha and there are approximately 50 farm businesses, 24 of which are dairy operations.

THE FOCUS FARMS

Mains of Balgavies Farm, Angus

Mains of Balgavies is owned and farmed by Mr Tom Sampson. It comprises around 290 ha and is mainly arable with grass, permanent set-aside and woodland. It extends in altitude from 60 m at Loch of Balgavies to 110 m. Cropping is mainly autumn-sown wheat, oil-seed rape and oats, spring barley and typically about 30 ha ground rented out for potatoes. Summer grazing is provided for up to 40 suckler cows and calves and occasionally for up to 200 autumn ewes. Shed accommodation is rented to a neighbour for fattening cattle. Minimum cultivation techniques are practiced on the arable ground and all fields have a 2 m buffer strip along all headlands as part of a Rural Stewardship Scheme Agreement. Soils are mostly Forfar and Balrownie Association and are freely or imperfectly drained iron podzols of predominantly sandy loam texture (Muir *et al.,* 1964). The entire farm is within a Nitrate Vulnerable Zone (NVZ) (SEERAD, 2003).

Low Holehouse Farm, Ayrshire

Low Holehouse is owned and farmed by Mr Willie Campbell and his family. It is a slurry-based dairy and beef operation and extends to approximately 56 ha at Low Holehouse, with a further 156 ha of owned or rented land within the business. Cattle and sheep are grazed and both grass and whole-crop silage are made. The dairy herd comprises about 110 cows. A limited number of ewes and lambs are grazed at Low Holehouse but up to 250 ewes that run elsewhere are housed for lambing. The farm is approximately 120 m above sea level and experiences about 1100 mm of rainfall per annum. The soils are predominantly sandy clay loams and peat (Mitchell and Jarvis, 1956). The Killoch Burn and tributaries run through the property and close to the steading.

DIFFUSE POLLUTION AUDITS

Mains of Balgavies Farm, Angus

The Diffuse Pollution Audit for Mains of Balgavies identified risks of diffuse losses of nutrients, sediments and pesticides. The Audit ranked the significance of these risks in terms of their Environmental Sensitivity and the Magnitude of Impact. Together, these factors gave a Significance value. These rankings are presented in Table 1 below.

Table 1: Significance of Diffuse Pollution risks at Mains of Balgavies

Pollutant	Environmental sensitivity	Magnitude of impact	Significance
Phosphorus	high	medium	substantial
Nitrate	medium/high	medium	moderate
FIOs	low	medium/low	slight
Suspended Solids	medium	medium	moderate
Pesticides	high	medium/high	substantial

Nutrient loss was a strong possibility in many fields with slopes sufficiently steep to be erodable by overland flow, and from other fields where soil loss in drains was observed. In both cases phosphorus was of most concern because of its capacity to bind to soil particles. Field scale nitrate losses are likely through leaching (SEERAD, 2003) so are also of concern.

As well as soil loss through erosion, sediment losses were evaluated at points on the farm where watercourses were accessed for drinking water by grazing livestock. In one instance access was to the margins of Loch of Balgavies, the other was to a canalised stream.

BMPs have been proposed to mitigate some of the identified risks. A detailed nutrient budget has been completed to better inform fertiliser application decisions. This has included data derived from a yield meter installed on the farm's combine by the project.

The focus of interest for pesticide loss was in the sprayer filling and washdown area although it was also observed that minimum cultivation techniques may lead to a more intensive than locally-average use of pesticides at the field level. A biobed for the sprayer filling area will be considered by the Focus Farm Group. The possibility of employing Integrated Pest Control to reduce pesticide utilisation will be investigated.

Low Holehouse, Ayrshire

The Diffuse Pollution Audit separated identified issues into those with "steading" and "field" origins. Amongst the "Steading" issues, the main problems are related to slurry, wash water and silage effluent collection and storage, and the capacity and integrity of existing stores and surfaces. The overall storage capacity for slurry and other collected non-solids is equivalent to 80 days. There are a number of problem areas on the steading that result from non-optimal siting of cattle accommodations, feed storage areas and difficulties of maintaining some structures in a sound condition. Additionally, some surfaces drain non-optimally making the collection of run-off from these areas problematic.

"Field" issues are related to livestock movements, slurry and manure application, and nutrient management. In all these cases, there is a high risk of loss of nutrients to the water environment. Additionally, deposition of faecal material by grazing livestock and the spreading of fresh slurries may be contributing to the entry of faecal pathogens into the watercourses. For nutrient management in the dairy and beef, rations could be better optimised and recorded.

MONITORING

Low Holehouse, Ayrshire

During the bathing season (June-September) the monitoring at Low Holehouse has focused on sampling water in the associated burn and farm ditches at the time of significant rainfall events since such events have been shown to be an important causal mechanism of delivery of bacterial inputs to bathing waters (Kay *et al*., 1998). In addition, samples are taken each fortnight throughout the year. All samples are analysed for the determinants shown in Table 2. Samples taken during major events are also analysed for faecal indicator organisms (FIOs). The samples are taken at five locations on the farm to provide a means of apportioning any diffuse pollution loading to specific aspects of the farming operation as well as providing a measure of total loading.

Table 2: List of determinants for water analysis at Low Holehouse

Total Nitrogen	Nitrate	Total Organic Nitrogen	Nitrite
Ammonia	Total Phosphorus	Orthophosphate	TOC
Chloride	Solids	Alkalinity	Turbidity
Conductivity	pH	FIOs	

Lunan Water, Angus

Long-term surface water monitoring programmes are undertaken by SEPA and measurements include water chemistry, hydrology, turbidity and ecology. This project is piloting the use of continuous turbidity measurement to evaluate the impact of suspended sediment on mitigations, and to provide information for an evaluation of the effect of turbidity on ecology. Additionally, borehole analysis is being undertaken to assess the response time of groundwater quality to mitigations. The Macaulay Institute is monitoring nitrate concentration at 10 locations within the catchment and sediment concentration upstream of Mains of Balgavies.

FARMER FOCUS GROUPS

The Farmer Focus Groups in each catchment were launched alongside the Focus Farm Project itself at events in Spring of 2007. The launch events were attended by catchment farmers, landowners, local authorities, farmers representatives, regulators and Scottish Government officials. The events were co-ordinated and facilitated by SAC staff and attracted considerable local and national press interest in both general and trade publications.

At the events, the outline results of the Diffuse Pollution Audits were presented. The farmers and facilitation teams led farm and steading walks and specialists presented the identified issues to the attendees. Farmers were recruited at these events and subsequently to the Farmer Focus Groups.

The Groups have now been fully established with inaugural and technical meetings having taken place. Responsibility for setting the agenda rests between the facilitator and the farmers themselves. Both Groups expressed a clear and strong interest in learning more about SEPA's assessments of their catchments' water quality issues and about the science behind diffuse pollution. The opportunity has been taken to involve specialists in their fields in presenting base information to the Focus Groups and the response and feedback from farmers has been very positive.

Where appropriate, the results from early monitoring have been presented and Focus Group farmers are being encouraged to think about how the BMPs proposed for the Focus Farm might apply on their own farms.

FUTURE PROGRAMMES

The Project has a five year duration and, after the initial phase of baseline data gathering, the assessment of existing practices and the provision of background information and support to farmers Group members, the BMP evaluations and implementations will begin. Individual candidate BMPs will be investigated, using existing examples and experience where feasible, and their inclusion in the project will be considered. Where appropriate, specialist advice will be sought from outwith the project's immediate personnel. The BMPs' effectiveness, using pre- and post-implementation monitoring, as well as the costing implications, and "practicality" of the BMPs selected, will be evaluated. Their fit within existing and proposed agri-environment support schemes will be considered as part of the mechanism for rolling out the favoured BMPs to the wider farming community and, eventually, across Scotland.

ACKNOWLEDGEMENTS

This project is being undertaken co-operatively between SEPA, The Macaulay Institute and SAC. We acknowledge the support of The Scottish Government for project funding, SEPA for the provision of a water sample analysis facility and The Macaulay Institute for considerable monitoring effort. The Focus Farmers themselves - Mr Tom Sampson, Mains of Balgavies, Angus and Mr Willie Campbell, Low Holehouse, Ayrshire give generously of their time, expertise and enthusiasm to the project and this is gratefully acknowledged.

REFERENCES

Kay D, Crowther J and Wyer MD (1998). Enteric bacterial concentrations in Ayrshire bathing waters. A report to West of Scotland Water. CREH, University of Wales, pp. 43.

Mitchell BD and Jarvis RA (1956). The Soils of the Country round Kilmarnock [Sheet 22 and part of Sheet 21]. Department of Agriculture for Scotland.

Muir JW, Romans JCC, Laing D, Smith J and Glentworth R (1964). Soil map of Forfar [Sheets 57 and 57A]. Southampton: Ordnance Survey.

SEERAD (2003). Guidelines for farmers in nitrate vulnerable zones.

THE BIODIVERSITY VALUE OF RIPARIAN FIELD MARGINS IN INTENSIVELY MANAGED GRASSLANDS

LJ Cole, D Robertson, W Harrison and DI McCracken

SAC, Auchincruive, Ayr, KA6 5HW, UK, E-mail: Lorna.Cole@sac.ac.uk

SUMMARY

Riparian field margins (i.e. fenced areas adjacent to watercourses) are becoming more widespread in intensively managed grassland in the UK as a means of controlling diffuse pollution. This study is examining a range of riparian zones (both fenced and unfenced) with the aim of determining their influence on farmland biodiversity. The activity abundance of key invertebrate groups (i.e. leatherjackets and sawfly larvae) was greater in fenced riparian margins when compared to unfenced margins. Furthermore, the ecological structure of ground beetle assemblages in fenced margins differed from that of intensively managed grassland fields with the fenced margins supporting a higher proportion of nocturnal plant feeders and small, nocturnal wingless predators. Fenced riparian field margins may therefore enhance biodiversity both at the field level by increasing the abundance of invertebrates important in the diet of farmland birds, and at the landscape level by adding additional habitats supporting different invertebrate assemblages. However, some form of limited management may be needed in the margins to enhance the availability of these invertebrates to foraging birds. The nature of such management needs careful consideration to ensure that it would not adversely affect either the invertebrates within the sward or the diffuse pollution mitigation rationale behind the establishment of the margins in the first place.

INTRODUCTION

In Scotland, the contamination of watercourses by intensively managed dairy farms is a cause for concern, from both a water quality and a public health perspective (SEPA, 2002). The establishment of fenced riparian field margins, to physically exclude livestock from the watercourse and create a barrier between the watercourse and grazing livestock and associated management practices (e.g. slurry spreading) in the adjacent field, is one way of reducing diffuse pollution from intensively managed grassland. The installation of riparian field margins is encouraged by agri-environment schemes and consequently is becoming more widespread in the UK. While such margins are primarily being established to mitigate diffuse pollution, they also provide an opportunity to offset the declines in farmland biodiversity associated with intensive livestock production. The establishment of similar margins along the edge of arable fields (i.e. conservation headlands) is widespread and has been shown to benefit wildlife such as butterflies (Rands and Sotherton, 1986), spiders (Pfiffner and Luka, 2000) and many declining farmland birds (Fuller, 2000). Initial investigations of non riparian margins in intensively managed grasslands indicate that may also benefit a range of invertebrates (Haysom *et al*., 2004; Cole *et al*., 2007; Woodcock *et al*., 2007). This ongoing study is investigating a range of riparian zones in intensively managed grassland farms in the Cessnock catchment, Ayrshire, Scotland. The aim of the analyses reported here was to determine the key factors influencing invertebrate diversity and the ecological structure of ground beetle assemblages within the riparian zone.

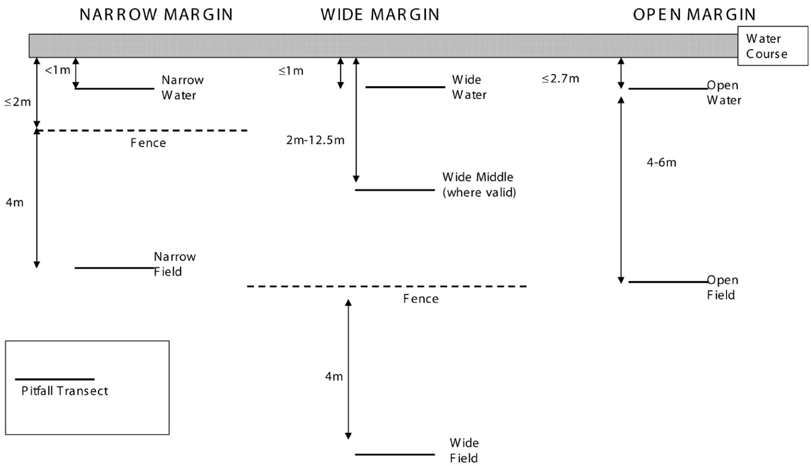


Figure 1: Illustration of the three types of riparian zones (Narrow, Wide and Open) being considered in the study, together with an indication of the location of the two or three pitfall trap transects placed within each. Each study site contains only one of these type of riparian zone

MATERIALS AND METHODS

To-date, twenty-two sampling locations across seven intensive dairy farms in the Cessnock catchment area, Ayrshire, Scotland (UK National Grid Reference: NS 53) have been studied over a three year period (2004-2006). Each sampling location was allocated into one of three categories: Open Margins (i.e. sites with no fence between the field and watercourse), Narrow Margins (i.e. sites with narrow, <2 m, fenced riparian margins) and Wide Margins (i.e. sites with wide, >4 m, fenced riparian margins). At each site, two sampling transects were established, one adjacent to the water course (Water) and the other 4 m into the field (Field). For Wide Margins a third transect was established in the middle of the riparian margin (Middle). A total of seven treatment transects were therefore investigated: Open Water, Open Field, Narrow Water, Narrow Field, Wide Water, Wide Middle and Wide Field (Figure 1).

At each transect, nine pitfall traps (75 mm diameter and 100 mm deep) were installed at 2 m intervals to monitor the ground active invertebrates. Each year, invertebrates were collected during two trapping periods, each of four weeks duration (June/July and July/August). On collection, the nine pitfall samples in each line were pooled. Both the activity and abundance of invertebrates can influence pitfall catches and consequently the abundance of invertebrates caught by pitfall trapping is referred to as the activity abundance (Thiele, 1977). Along each transect, information on vegetation height, density and composition was collected annually in August/September. Vegetation data was collected following pitfall removal to minimise interference during the trapping period and to facilitate identification of the full range of plant species present. Vegetation composition was determined by randomly placing four 1 m x 1 m quadrats along each pitfall line while the Robel pole visual obscurity method (Robel *et al.,* 1970) was used to measure sward density and height. Linear mixed models using the method of residual maximum likelihood (REML) were fitted to test for effects of treatment, vegetation density and year (after adjusting for effects of farm and sampling location) on the abundance of key invertebrate groups and the ground beetle species richness (log transformed where required). The following model was applied in GENSTAT:

Fixed Effects=VegetationDensity+Year+Treatment; Random Effects=Farm/Location/Year).

Table 1: Results of REML analyses on the activity abundance of key invertebrate groups and the richness of carabid beetles showing the Wald statistic (W) and probability value

Invertebrate Group*	Vegetation density (df=1)	Influence of density	Treatment (df=6)	Location of treatment differences (F=Field, W=Water, M=Middle)
Leatherjacket N	W=14.63 P<0.001	+ve	W=12.57 P<0.05	Narrow W>All F&Open W Wide W&M>Wide F Wide W>Open W
Sawfly larvae N	W=23.12 P<0.001	+ve	W=19.60 P<0.001	All W > Wide/NarrowF NarrowW > OpenF WideM > WideF
Ground beetle N	W=30.96 P<0.001	-ve	W=19.36 P<0.001	All F&Open W>Narrow/Wide W & Wide M
Ground beetle S	W=2.67 P=NS		W=5.37 P=NS	

* N = Number of individuals, S = Number of Species

RESULTS

REML analyses indicated that the activity abundance of sawfly larvae and leatherjackets (both key prey items for farmland birds) was positively influenced by vegetation density indicating that these species were more abundant in longer denser vegetation (Table 1). Such vegetation is typical of fenced riparian margins. The influence of treatment on leatherjacket activity abundance reflected this with higher densities occurring in fenced margins (both narrow and wide) when compared to the adjacent fields and open margins. While the activity abundance of sawfly larvae was highest in the fenced riparian margins, the abundance of this group was also greater in the open water sites when compared to the adjacent fields supporting the view that even grazed riparian zones can be naturally rich in biodiversity (Carbacho *et al*., 2003).

The influence of vegetation density on ground beetle activity density was quite different from that found for leatherjackets and sawfly larvae, and ground beetles were actually less abundant in dense vegetation (Table 1). Furthermore, treatment effects indicated that the activity abundance of ground beetles was greater in the intensively managed fields and open water margins than the fenced-off riparian margins. While the abundance of ground beetles was greater in the fields and open water margins, the number of species recorded did not differ between treatments.

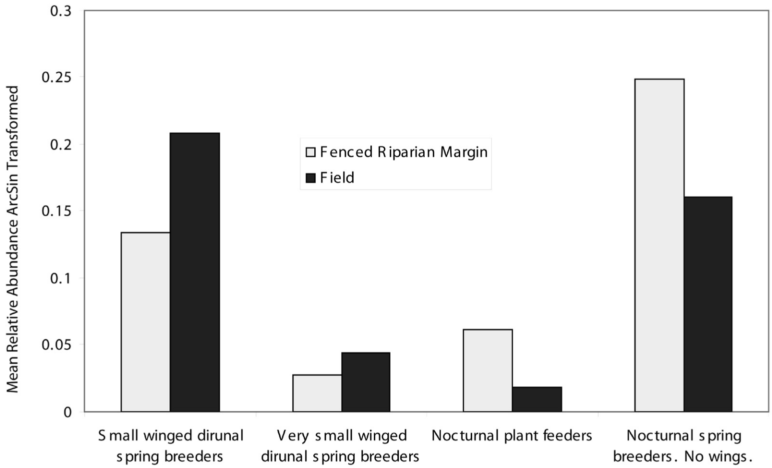


Figure 2: Influence of habitat on the relative abundance of ground beetle ecological groups

When looking at the ecological composition of the ground beetle assemblages, however, clear differences were found between fenced riparian margins (narrow and wide combined) and intensively managed grassland fields (Figure 2). While nocturnal plant feeders were rarely recorded in intensively managed grassland fields, they occurred relatively frequently in the fenced riparian margins. In addition, the relative abundance of nocturnal spring breeders without wings was greater in the fenced margins, while the field has a higher proportion of very small and small diurnal spring breeders.

DISCUSSION

Erecting fences adjacent to watercourses in intensively managed grassland fields not only has the potential to reduce diffuse pollution by excluding livestock from the watercourse, but may also help to address some of the biodiversity concerns associated with such intensively managed fields. The current study found that fenced riparian margins enhanced the activity density of sawfly larvae and leatherjackets which are important prey items for farmland birds. It is, however, important to bear in mind that higher densities of prey does not necessarily equate to richer foraging grounds for birds and other factors such as prey accessibility must also be considered (Vickery *et al*., 2001; McCracken *et al*., 2004). Vegetation in the fenced margins was typically long and dense and, consequently may decrease the detectability of prey and impede the movement of foraging birds (McCracken and Tallowin, 2004). For riparian margins to achieve their full biodiversity potential it is therefore likely that they may need to be subjected to some form of limited management (such as grazing or cutting) to help open up the vegetation more and, thus, enhance the accessibility of prey to foraging birds (Olson and Wäckers, 2007). Such management practices would, however, need careful consideration to ensure that they did not adversely affect either the invertebrates within the sward or the diffuse pollution mitigation rationale behind the establishment of the margins in the first place.

As found by Haysom *et al*. (2004), the activity density of ground beetles was greater in intensively managed fields and riparian zones open to grazing than in fenced riparian margins. While the erection of fences along riparian zones did not promote beetle numbers, it did influence the ecological composition of the beetle assemblages, with fenced margins supporting quite distinct assemblages from intensively managed grassland fields. A higher proportion of nocturnal plant feeders was recorded in the fenced margins, when compared to the intensive grassland fields which is a likely consequence of the margins supporting richer plant communities, providing better resources for phytophagous ground beetles. The proportion of flightless nocturnal spring breeders was greater in the closed riparian margins, while the field has a higher proportion of diurnal spring breeders with wings. Diurnal predators rely on visual cues to capture their prey and, consequently, the shorter, more open vegetation of the grazed field facilitates hunting. Likewise, nocturnal predators that rely on tactile stimuli to hunt are not impeded by the long, dense vegetation in the fenced riparian margins. The higher proportion of wingless predators in the fenced margins may be the result of such margins providing a more stable habitat that is able to retain a residential population of beetles. Intensively managed grassland fields, on the other hand, are exposed to frequent and regular disturbances and consequently, they favour invasive species that are adapted to recolonising following disturbance (Blake *et al*., 1994).

It would appear, therefore, that the installation of fences along riparian zones in intensively managed grassland has the potential to promote farmland biodiversity at the field level by increasing the abundance of invertebrates and, at the landscape level, by providing an additional habitat that adds heterogeneity to the otherwise homogenous intensively managed grassland and supports different invertebrate assemblages. However, further work is required in order to consider the most effective way of maximising the benefits to foraging birds to be gained from such margins.

ACKNOWLEDGEMENTS

We are indebted to the farmers who provided access to their land for this study. SAC receives financial support from the Scottish Government Rural and Environment Research and Analysis Directorate (RERAD).

REFERENCES

Blake S, Foster GN, Eyre M and Luff ML (1994). Effects of habitat type and grassland management practices on the body size distribution of carabid beetles. Pedobiologia 38, 502-512.

Cole LJ, McCracken DI, Baker L and Parish D (2007). Grassland conservation headlands: Their impact on invertebrate assemblages in intensively managed grasslands. Agriculture, Ecosystems and Environment 122, 252-258.

Corbacho C, Sanchez JM and Costillo E (2003). Patterns of structural complexity and human disturbance of riparian vegetation in agricultural landscapes of a Mediterranean area. Agriculture, Ecosystems and Environment 95, 495-507.

Fuller R (2000) Relationships between recent changes in lowland British agriculture and farmland bird populations: an overview. In: Ecology and Conservation of Lowland Farmland Birds, (eds.) Aebischer N J, Evans A D, Grice P V and Vickery J A. British Ornithologist's Union, Tring, UK.

Haysom KA, McCracken DI, Foster G N and Sotherton NW (2004). Developing grassland conservation headlands: response of carabid assemblage to different cutting regimes in a silage field edge. Agriculture, Ecosystems and Environment 102, 263-277.

McCracken DI and Tallowin JR (2004). Sward and structure, the interactions between farming practices and bird food resources in lowland grassland. Ibis 146, 108–114.

McCracken DI, Bignal EM, Blake S and Foster GN (2004) Productivity and profitability: the effects of farming practices on the prey of insectivorous birds. In: Insects and Bird Interactions, (eds.) van Emden H F and Rothschild M. pp. 75-87. Intercept, Andover.

Olson DM and Wäckers FL (2007). Management of field margins to maximise multiple ecological services. Journal of Applied Ecology 44, 13-21.

Pfiffner L and Luka H (2000). Overwintering of arthropods in soils of arable fields and adjacent semi-natural habitats. Agriculture, Ecosystems and Environment 78, 215-222.

Rands MRW and Sotherton NW (1986). Pesticide use on cereal crops and changes in the abundance of butterflies on arable farmland. Biological Conservation 36, 71-83.

Robel RJ, Briggs JN, Dayton AD and Hulbert LC (1970). Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23, 295-297.

SEPA (2002). Protecting Scotland's bathing waters. In SEPA View: Farming special (ed. Budge C). Scottish Environmental Protection Agency, Stirling.

Thiele HU (1977). Carabid Beetles in Their Environment. Springer Verlag, Berlin.

Vickery JA, Tallowin JR , Feber RE, Asteraki EJ, Atkinson PW, Fuller RJ and Bown VK (2001). The management of lowland neutral grasslands in Britain, effects of agricultural practices on birds and their food resources. Journal of Applied Ecology 38, 647–664.

Woodcock BA, Potts SG, Pilgrim E, Ramsay AJ, Tscheulin T, Parkinson A, Smith REN, Gundrey AL, Brown VK and Tallowin JR (2007). The potential of grass field margin management for enhancing beetle diversity in intensive livestock farms. Journal of Applied Ecology 44, 60–69.

WOODY BIOMASS ENERGY CROPS: POSSIBILITIES FOR PHYSICAL AND BIOLOGICAL CARBON SEQUESTRATION

AJC Coles[1], REH Sims[2] and KA Lewis[1]

[1]Agriculture and Environment Research Unit, Science and Technology Research Institute, University of Hertfordshire, College Lane, Hatfield, Hertfordshire, AL10 9AB, UK, E-mail: A.J.C.Coles@herts.ac.uk; [2]Energy Research Centre, Institute of Technology and Engineering, Massey University, Palmerston North, Private Bag 11 222, New Zealand

SUMMARY

Bioenergy through short rotation forestry could hold potential for fossil fuel displacement thereby mitigating climate change by using various combustion, carbon capture technologies, and sequestration technologies. Two routes, gasification and pyrolysis, show potential for enabling bioenergy to become carbon negative rather than carbon neutral. One further relatively unexplored route is biochar, a naturally occurring material that may offer a unique link between bioenergy and sequestration that is both simple and energy bearing.

INTRODUCTION

Human induced global warming is now unequivocal (IPCC, 2007). Attributed to anthropogenic greenhouse gas (GHG) emissions, climatic change associated with global warming is considered to be the greatest threat facing mankind today (IPCC, 2001). A primary GHG, carbon dioxide (CO_2) is responsible for more than half of the total warming potential of all GHGs (IPCC, 2001) placing it in firm focus of GHG reduction strategies (Jaber, 2002). The IPCC (2007) Fourth Assessment Working Group I Report identifies the primary source the post-industrial revolution increase of atmospheric CO_2 to be fossil fuel use, although land use change has also been identified as another significant, but lesser, source (IPCC, 2007).

Bioenergy systems could play an important role in the displacement of fossil fuels, a vital step in fighting climate change (Cook and Bayea, 2000). Generally assumed to be a CO_2-neutral energy carrier (not including processing and transport), bioenergy allows carbon (C) to be stored in plants, emitted through decomposition or combustion, and taken up once again during new re-growth (Schlamadinger *et al.*, 1995; Schlamadinger and Marland, 1996). Grown in short rotation forestry (SRF) plantations, an intensive form of silviculture, trees, harvested over short periods of time, provide a constant supply of bioenergy feedstock (Heller *et al.*, 2003).

Contemporary coal-fired electricity plants can co-fire 3-5% biomass of its feedstock as biomass with minimal modifications and efficiency penalty (Cook and Beyea, 2000; Hallam *et al.*, 2001). Technically, wood chips for co-fired electricity production in existing coal fired power plants could be realised relatively easily, requiring few engineering modifications (Hartmann and Kaltschmitt, 1999). However, widespread incorporation of bioenergy will inevitably lead to land use change and soil disturbance due to crop establishment (Paul *et al.*, 2002).

Soil disturbance plays an important role in soil organic carbon (SOC) dynamics (Lal, 2002). SOC is the largest actively cycling C pool within the terrestrial system, estimated to contain around 2000Gt C predominantly within the top metre of the soil profile (Follett, 2001; Janzen, 2004). However, soil disturbance often leads to SOC depletion and organic matter loss (Lal, 2002). With 75% of total terrestrial C stored within the soil, should woody biomass production for energy be applied globally, significant impacts on the global C budget would be seen (Paul *et al.*, 2002). Although providing a means of renewable C neutral energy, to maintain C neutral the resultant emissions from SRF bioenergy logistics operations would need to be offset. But can woody biomass energy crops be used as a 'CO_2 pump' to link biological and physical sequestration technologies for enhanced climate change mitigation?

BIOENERGY AND 'TRADITIONAL' CARBON SEQUESTRATION

Gasification or the thermal decomposition of organic matter in a low oxygen environment generates CO_2 and other waste gases (Sadaka *et al.*, 2002). Carbon Dioxide Recovery (CDR), originally developed by the fossil fuel industry, is an end-of-pipe emissions scrubbing technology (Kraxner *et al.*, 2003) that in theory can be combined with gasification systems. Combining biomass gasification, CDR, and physical C sequestration, bioenergy has the potential to reduce CO_2 emissions (Möllersten *et al.*, 2003). Although commercially available, CDR is as yet to be applied to conventional combustion systems (IPCC, 2005).

Although, physical C sequestration doesn't represent a 'permanent' solution to C emissions, it does provide storage for an extended period of time (Hutchinson *et al.*, 2007). However, C captured using CDR can be a costly process requiring financial incentives such as enhanced oil recovery, hydrogen production, tax breaks, or C credits to produce a more commercially viable process (IEA, 2002). These economic implications mean few commercial C-emitting ventures employ CDR for large-scale sequestration purposes (IEA, 2002). Should CDR and physical C sequestration be employed on a viable commercial basis and connected to a bioenergy system, a traditionally C neutral energy source would become C negative (Kraxner *et al.*, 2003).

BIOENERGY AND BIOCHAR SEQUESTRATION

Pyrolysis, a form of gasification, can be optimised to convert biomass into bio-oil (or bio-crude), biochar, and non-condensable gases (Dembiraş, 2000). Heating the feedstock in the absence of air, pyrolysis produces high-energy fuels at lower temperatures than in gasification. Pyrolysis is an efficient biomass conversion process capable of 95.5% fuel-to-feed efficiency, successfully competing with non-renewable fossil fuels (Dembiraş, 2000). Pyrolysis biochar yield is dependent on carbonisation temperature, and can increase from 25.6% at 800°C to 66.5% at 300°C, increasing the fixed carbon from 55.79% to 93.15% (Ogawa *et al.*, 2006).

Formed as a result of incomplete combustion, charred materials such as biochar (also known as black carbon, charcoal, and char) are ubiquitous in many terrestrial environments, and have unique physical and chemical properties (Forbes *et al.*, 2006). Generally porous, biochar retains water, breathes well, and holds great potential as a soil conditioner for improving permeability whilst preventing the leaching of fertilisers and improving plant growth and yield (Kadota and Niimi, 2004). A naturally occurring derivative of forest fires, the aromatic structure of biochar suggests it to be relatively inert for extended periods, although this has yet to be quantified (Forbes *et al.*, 2006; Ogawa *et al*., 2006). Recognised in Japan by the Ministry of Agriculture, Forestry and Fisheries as a soil improver (Ogawa, 1998), biochar may play a significant role in the soil C pool and is becoming of increasing interest for C sequestration purposes (Hamer *et al.*, 2004; Forbes *et al.*, 2006).

INVESTIGATING BIOCHAR STABILITY IN SOIL

To investigate the stability of biochar in soil, three contrasting soil types were harvested from the North Island, New Zealand. The soil types were Manawatu fine sandy loam (40° 23' 30" S 175° 37' 54" E), Tokomaru silt loam (40° 23' 54" S 175° 36' 49" E) and, Egmont black silt loam (39° 49' 01" S 174° 56' 35" E). Harvested from the top 20cm of the soil profile, the field capacity, field moisture, C percentage of each soil type was determined.

A homogenised soil sample was passed through a 1mm-test sieve to remove large organic debris and stones. Extracting from the sample four sub-samples of 135g equivalent oven dry weight, each were placed in a beaker and fitted into an air-tight, septum fitted, preserving jar of a known headspace with a test tube holding 10ml of distilled water to regulate atmospheric moisture. The sealed jars, or incubation chambers, were then covered with thick polythene creating a darkened microcosm maintained at a temperature of $20^oC \pm 1^oC$ within a temperature-controlled laboratory. Respiration levels were monitored for any resultant increase in due to soil disturbance during preparation. Upon respiration reaching a plateau, the C within the soil sub-sample was raised by 4.8% of the C equivalent of the soil dry mass (restricted by soil water content and field capacity) via the addition of biochar. The incubations were then adjusted to -10kPa using distilled water (monitored and maintained throughout the study). Four sub-samples and replicate controls were monitored for each soil type.

Incubation chamber atmosphere was measured periodically for CO_2 by extracting 1ml samples using a calibrated gas-tight glass-syringe. The gas sample was then injected into a gas-calibrated CO_2 Analyser and the %C determined. Post analysis, the atmosphere of each incubation chamber was flushed with ambient air and re-sealed. By monitoring soil respiration over time within incubation chambers and comparing the results of amended and unamended soils, the effects of biochar application may be isolated and determined.

Biochar treatment results were observed through accumulated mean C flux (mg) over a 250-day period (Figure 1). All amended soil samples showed C sequestration trends (sequestration = (soil C + conditioner C input) – C emitted) (Figure 2). However, statistically significant sequestration results were only found in the Manawatu ($B=0.202$, $p=0.046$) and Egmont ($B=-0.552$, $p<0.001$) incubations. Where: B = beta coefficient of X, i.e. the "effect" of applying X compared to the control; and, p = the probability of the sample been drawn randomly from the population tested). No statistically significant difference was found between the Tokomaru control and amended soil.

Discussion and Concluding Remarks

The increase of soil C by 4.8% of the C equivalent of the soil dry mass, was achieved by the Manawatu and Egmont soils receiving a single dose of total of 0.21 g of C per 100 g of soil, while the Tokomaru soil received 0.15.5 g of C per 100 g of soil. However, over the 250-day period, the Egmont soil sequestered 42% more C than the Manawatu soil. Conversely the Tokomaru soil sequestered 78% of the total C stored within the Manawatu soil at the end of the trial.

The study indicates that under laboratory conditions the C sequestration potential of biochar is dependent on soil type and that its real world effects should be determined by long term field trial examination under different soil and climate conditions. The variation in soil respiration response to biochar additions between soil types may possibly be due to differences in soil chemistry and/or microbiology and also requires further investigation. As relatively small changes in soil respiration can alter C flux, should woody bioenergy crops be established over large areas, dramatic changes in the C balance in turn, atmospheric concentration may be seen (Bowden *et al.*, 2004). The question whether can woody biomass energy crops be used as a 'carbon dioxide pump' to link biological and physical sequestration technologies for enhanced climate change mitigation remains? The answer… definitely maybe.

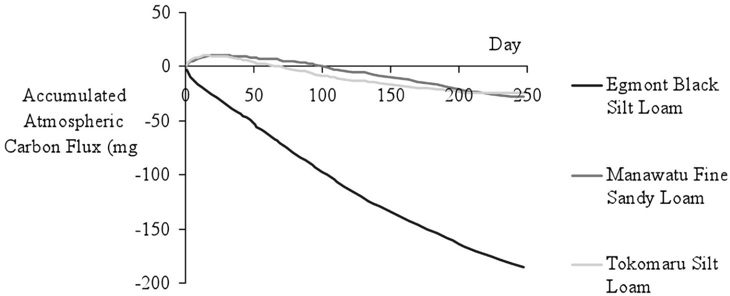


Figure 1: Accumulated (cumulative) mean carbon flux (mg) per soil type when amended with biochar

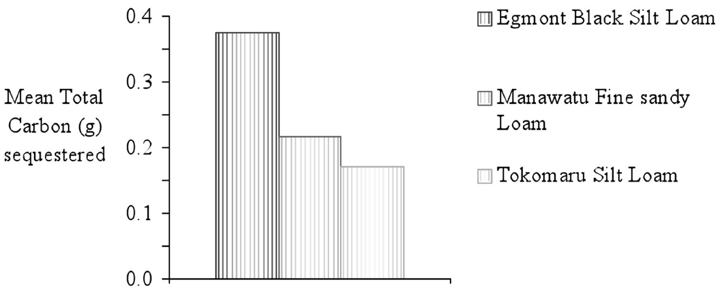


Figure 2: Mean net carbon sequestration (g) over a 250-day period per soil type post basal soil respiration, background atmospheric carbon (CO_2) levels, and carbon additions through biochar conditioner application have been accounted for via carbon mass balance

ACKNOWLEDGEMENTS

Thank you to the C. Alma Baker Fund and Ravensdown Agricultural Services, Foundation for Research Science and Technology, Commodities NZ Limited, Fonterra Research Centre (Palmerston North), Kinleith Pulp and Paper Mill and Professor Surinder Saggar, Dr Val Snow and Dr Peter Read.

REFERENCES

Bowden RD, Davidson E, Savage K, Arabia C and Steudler P (2004). Chronic nitrogen additions reduce total soil respiration and microbial respiration in temperate forest soils at the Harvard Forest. Forest Ecology and Management 196, 43-56.

Cook J and Beyea J (2000). Bioenergy in the United States: progress and possibilities. Biomass and Bioenergy 18, 441-455.

Dembiraş A (2000). Mechanisms of liquefaction and pyrolysis reactions of biomass. Energy Conversion and Management 41, 633-646.

Follett RF (2001). Soil management concepts and carbon sequestration in cropland soils. Soil and Tillage Research 61, 77-92.

Forbes MS, Raison RJ and Skjemstad JO (2006). Formation, transformation and transport of black carbon (charcoal) in terrestrial and aquatic ecosystems. Science of the Total Environment 370, 190-206.

Hallam A, Anderson IC and Buxton DR (2001). Comparative economic analysis of perennial, annual, and intercrops for biomass production. Biomass and Bioenergy 21, 407-424.

Hamer U, Marschner B, Brodowski S and Amelung W (2004). Interactive priming of black carbon and glucose mineralization. Organic Geochemistry 35, 823-830.

Hartmann D and Kaltschmitt M (1999). Electricity generation from solid biomass via co-combustion with coal. Energy and emission balances from a German case study. Biomass and Bioenergy 16, 397-406.

Heller MC, Keoleian GA and Volk TA (2003). Life cycle assessment of a willow bioenergy cropping system. Biomass and Bioenergy 25, 147-165.

Hutchinson JJ, Campbell CA and Desjardins RL (2007). Some perspectives on carbon sequestration in agriculture. Agricultural and Forest Meteorology 142, 288-302.

IEA (2002). Solutions for the 21st Century - 2 - Zero Emissions Technologies for Fossil Fuels. Technology Status Report. International Energy Agency, Committee on Energy Research and Technology, Working Party on Fossil Fuels (Chairman: Barbara McKee). IEA, Paris, France.

IPCC (2001). Climate Change 2001: Mitgation. Contribution of Working Group III to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, Cambridge University Press.

IPCC (2005). IPCC Special Report on Carbon Dioxide Capture and Storage. Prepared by Working Group III of the Intergovernmental Panel on Climate Change. Cambridge, Cambridge University Press.

IPCC (2007). Discussion meeting on the science of climate change: A Royal Society showcase of the IPCC 4th Assessment Working Group 1 Report. London. Intergovernmental Panel On Climate Change. IPCC Special Report. Emissions Scenarios. Summary for Policymakers. (Draft Report). Working Group I. London, IPCC.

Jaber JO (2002). Greenhouse gas emissions and barriers to implementation in the Jordanian energy sector. Energy Policy 30, 385-395.

Janzen HH (2004). Carbon cycling in earth systems – a soil science perspective. Agriculture, Ecosystems and Environment 104, 399-417.

Kadota M and Niimi Y (2004). Effects of charcoal with pyroligneous acid and barnyard manure on bedding plants. Scientia Horticulturae 101, 327-332.

Kraxner F, Nilsson S and Obersteiner M (2003). Negative emissions from Bioenergy use, carbon capture and sequestration (BECS) – the case of biomass production by sustainable forest management from semi-natural temperate forests. Biomass and Bioenergy 24, 285-296.

Lal R (2002). Soil carbon dynamics in cropland and rangeland. Environmental Pollution 116, 353-362.

Möllersten K, Yan J and Moreira JR (2003). Potential market niches for biomass energy with CO2 capture and storage – Opportunities for energy supply with negative CO2 emissions. Biomass and Bionenergy 25, 273-285.

Ogawa M, Okimori Y and Takahashi F (2006). Carbon sequestration by carbonization of biomass and forestation: three case studies. Mitigation and adaptation Strategies for Global Change 11, 429-444.

Ogawa M (1998). Utilization of Symbiotic Microorganisms and Charcoal for Desert Greening. Green Age-News/Technical Papers March 1998, 5-11.

Paul KI, Polglase PJ, Nyakuengama JG, Khanna PK (2002). Change in soil carbon following afforestation. Forest Ecology and Management 168, 241-257.

Sadaka SS, Ghaly AE and Sabbah MA (2002). Two phase biomass air-stream gasification model for fluidized bed reactors: Part I – model development. Biomass and Bioenergy 22, 439-462.

Schlamadinger B and Marland G (1996). The role of forest and bioenergy strategies in the global carbon cycle. Biomass and Bioenergy 10, 275-300.

Schlamadinger B, Spitzer J, Kohlmaier GH and Ludeke M (1995). Carbon balance of bioenergy from logging residues. Biomass and Bioenergy 8, 221-234.

THE SEPA "REGULATORY" SOIL MONITORING STRATEGY: INVESTIGATING THE IMPACT OF WASTE TREATMENTS ON AGRICULTURAL SOIL QUALITY

AP Cundill[1] and KE Dobbie[2]

[1]Scottish Environment Protection Agency, Greyhope House, Greyhope Road, Aberdeen, AB11 9RD, UK, E-mail Alan.Cundill@sepa.org.uk; [2]Scottish Environment Protection Agency, Heriot Watt Research Park, Edinburgh, EH14 4AP, UK

SUMMARY

Scottish Environment Protection Agency (SEPA) have devised a regulatory soil monitoring strategy for investigating the impact of applying organic waste material to agricultural land. This is driven by the Sludge (Use in Agriculture) Regulations 1989 (as amended) and the Waste Management Licensing Regulations 1994. Soil quality indicators are used to assess the impact of waste spreading. A risk assessment procedure identified appropriate fields from which to take samples. Wastes applied included distillery waste, wood processing waste, dairy waste and off-specification compost as well as sewage sludge. Where possible at each farm, a "control" field, to which no waste had been applied, was also sampled. Results obtained to date indicate that potentially toxic elements in all soils sampled are below the limits specified in the Sludge (Use in Agriculture) Regulations 1989 (as amended). There are early indications that Zn and Cu concentrations may be slowly building-up in soil at one farm in Inverness-shire, however, more data is required before any conclusions can be drawn. Results from fields receiving wood processing waste show that the relationship between soil nutrient concentrations and waste application is complex.

INTRODUCTION

The SEPA Corporate Plan includes six environmental outcomes, one of which is the Land Outcome. One of the aims of the land outcome is to achieve 'Good land quality with healthier soils'. Specific targets set out under this aim include using soil quality indicators to assess the impact of land treatments using waste materials, with the objective of ensuring that 95% of all soil samples taken by SEPA are compliant with the potentially toxic element (PTE) limits stated in the Sludge (Use in Agriculture) Regulations 1989 (as amended), and reporting results on SEPA's website. In order to achieve these targets, SEPA has developed a regulatory soil monitoring strategy driven by the Sludge (Use in Agriculture) Regulations 1989 (as amended) and the Waste Management Licensing Regulations 1994.

METHODS

Each year, SEPA receives a register showing which fields in Scotland have had sewage sludge applied. From this register, a number of fields which could potentially be at risk from further sludge addition were selected for sampling. In addition, fields where the application of exempt waste spreading to land was of concern to local SEPA environment protection teams were selected for sampling. Samples were taken from 2-4 fields which had received applications of sludge/exempt waste at each farm. If possible, samples were also taken from a nearby reference field which had never received waste. Soil samples had been taken at two of these farms under an earlier SEPA organic waste to land action plan, which provided limited time series data. Soil sampling followed the method outlined in the Sludge (Use in Agriculture) Regulations 1989 (as amended). Soil samples were analysed for pH, extractable P, K and Mg, total organic carbon, total nitrogen, total metals (Cu, Zn, Ni, Pb, Cd, Hg). Latterly soils were also analysed for microbial biomass carbon, potentially mineralisable nitrogen and earthworms. In future soil bulk density will also be measured.

Farms across Scotland were sampled in an attempt to determine the impacts of sewage sludge and waste application on soil quality at the national scale.

The location of farms sampled and the type of exempt waste or sewage sludge applied is shown in Figure 1 while Figure 2 shows the type of farm sampled.

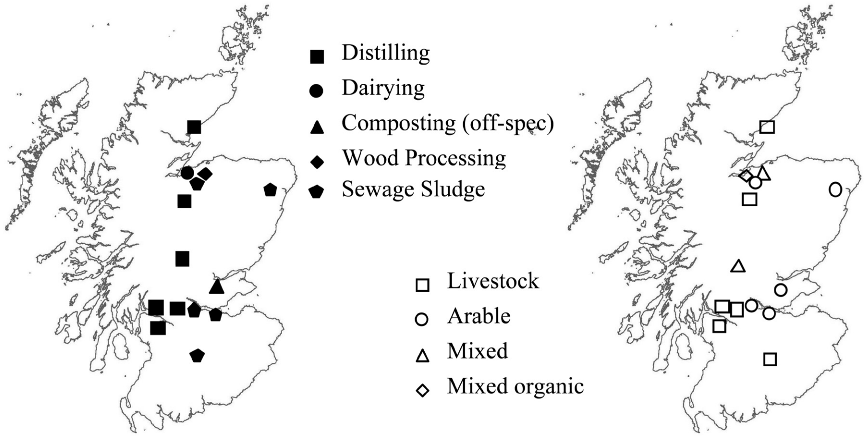


Figure 1: Farm locations and waste treatments used

Figure 2: Farm types

RESULTS

Initial results showing trends in zinc and copper concentrations in soil from a field in Inverness-shire which has received distillery waste over at least 4 years are shown in Figure 3. Although these appear to increase slightly over 4 years, they remain well within regulatory limits.

Figure 4 shows extractable nutrient concentrations (analysed using the SAC method) in soils from two fields receiving wood processing waste, along with a third field to which waste had not been applied.

Figure 5 compares soil organic carbon in two fields at two different locations which have received sewage sludge and adjacent fields to each of these to which sludge has never been applied. The results illustrate that organic carbon concentrations were higher in soils in the fields which have received sewage sludge applications.

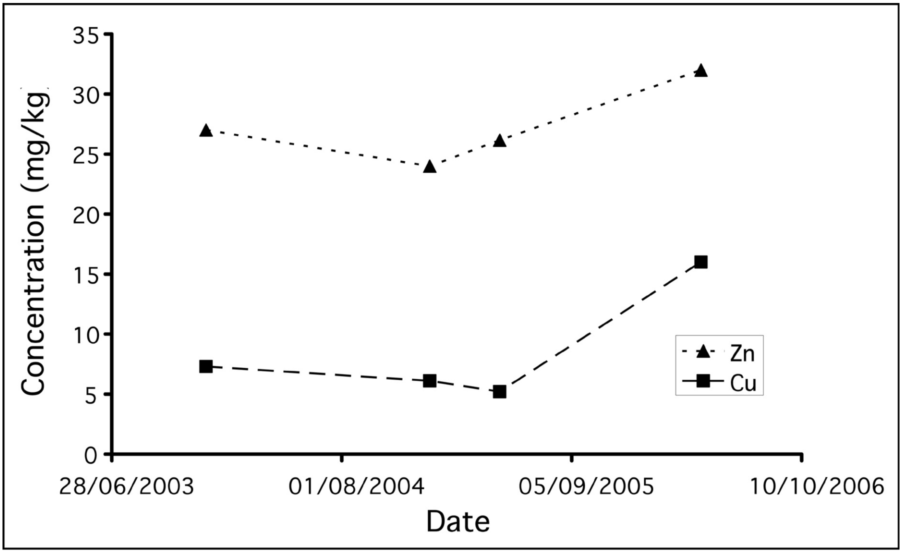


Figure 3: Zinc and Copper concentrations for soil from a field which has received repeated applications of distillery waste

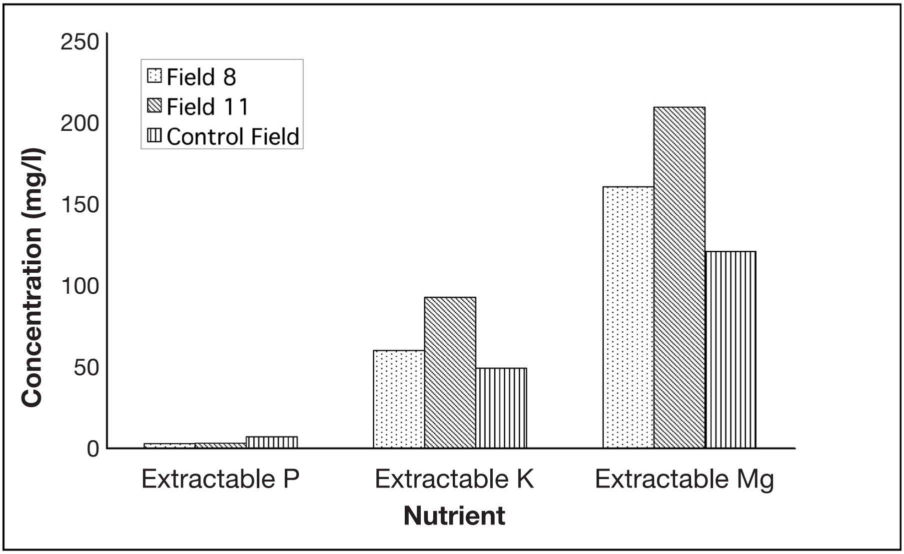


Figure 4: Nutrient concentrations in soil for three fields under silage grass cover. Fields 8 and 11 had received applications of wood processing waste, while the control field had no waste applied

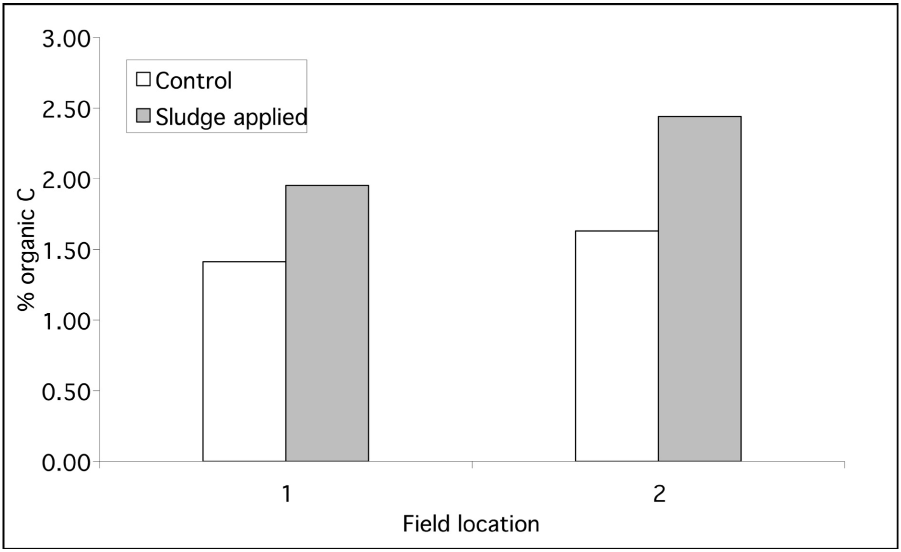


Figure 5: Comparison of organic carbon concentrations in two fields which have received sewage sludge and two fields which have not received applications of this material

DISCUSSION

Currently, the regulatory soil monitoring being undertaken by SEPA is at an early stage of development. A number of soil quality indicator results are still pending, whilst in other cases there is too little data to allow a meaningful interpretation of results.

Results from a field in Inverness-shire where distillery waste has been routinely applied suggest that zinc and copper concentrations have increased slightly through time (Figure 3). However, concentrations remain well below the limits set out in the Sludge (Use in Agriculture) Regulations, which are also used as guidance when determining whether exempt waste applications are acceptable. Indeed, metal concentrations on all fields receiving distillery waste for which results have been obtained to date show that limits set out in the Sludge (Use in Agriculture) Regulations have not been exceeded (Table 1).

Table 1: Maximum concentrations for potentially toxic elements (PTEs) in soils receiving applications of sewage sludge under the Sludge (Use in Agriculture) Regulations. Figures are in mg kg-1 extractable by aqua regia digest. Maximum allowable concentrations for lead, cadmium and mercury are independent of soil pH. Application of sewage sludge to soils with pH <5.0 is not permitted

Element	Limit according to soil pH			
	5.0-5.5	5.5-6.0	6.0-7.0	>7.0
Zinc	200	250	300	450
Copper	80	100	135	200
Nickel	50	60	75	110
Lead	300			
Cadmium	3			
Mercury	1			

Soil samples taken from another farm in Inverness-shire where wood processing waste had been applied revealed that soils in two fields which had received waste contained much higher concentrations of extractable Mg and K than a third field which had not received waste (Figure 4). Furthermore, although all three fields were, at time of sampling, under a silage grass crop which had yet to receive its first seasonal cut, the grass was visibly taller in the two fields which had received the wood processing waste than in the control field. On the other hand, Figure 4 shows that extractable P concentrations were higher in the control field. The higher extractable P concentration in the control field may represent the inability of the crop to use available P in the soil, due to another nutrient limiting the crop growth, whereas in the fields that have received the waste, this nutrient limitation has been removed, enabling the growing crop to utilise available P more effectively. Hence, the results shown in Figure 4 could represent the effects of changes in plant-soil nutrient interactions brought about by applications of the wood processing waste. The waste application does not necessarily increase all soil nutrient concentrations, as this is also affected by crop response. This illustrates the complex interaction between initial soil nutrient status, crop nutrient requirement and waste nutrient content. Although it is not possible to tell from the soil chemistry whether “agricultural improvement”, as required by the Waste Management Licensing Regulations, has been achieved, it is also important to show there is no ”disbenefit”, for example build up of P in soil.

The addition of sewage sludge to soil is shown to increase the organic C content of the soil compared to adjacent fields to which no sludge has been applied (Figure 5). In light of the current Defra target to “To halt the decline of soil organic matter caused by agricultural practices in vulnerable soils by 2025, whilst maintaining, as a minimum, the soil organic matter of other agricultural soils, taking into account the impacts of climate change.” (Defra, 2006), increasing soil organic carbon concentrations through application of sewage sludge may be regarded as desirable. However, although at present there is no evidence of PTE concentrations in these soils exceeding the limits set out in Table 1, it is important to ensure there is no PTE pollution in the long term, and also that there is no excessive build up of soil P.

CONCLUSIONS

Implementation of the SEPA soil monitoring strategy is at an early stage. The work to date has been focused on soils receiving waste under either Waste Management Licensing Exemptions or under the Sludge (Use in Agriculture) Regulations. Comparison of results for soil quality indicators before and after applications of organic waste, and examination of trends over time can be used to assess the impact of the waste application on soil quality. To date, no exceedence of the levels specified in the Sludge (Use in Agriculture) Regulations have been found, however it is important to continue to monitor soils to ensure there is no risk to the environment through dangerous build up of PTEs or nutrients as a result of these organic waste applications.

ACKNOWLEDGEMENTS

The authors would like to thank farmers and landowners for permission to collect soil samples and the waste operating companies for supplying waste spreading information. Soil analysis was carried out at SEPA, SAC, AlControl and the Macaulay Institute laboratories. The contribution from colleagues in the SEPA Soil Science Working Group is also acknowledged Assistance with sampling from colleagues in SEPA's Field Chemistry unit was also greatly appreciated.

REFERENCES

Defra (2006). Web reference http://www.defra.gov.uk/environment/land/soil/research/indicators/organic-matter.htm accessed 4/12/2007, last update 18/9/2006.

Sludge (Use in Agriculture) Regulations (1989). Statutory Instrument 1989 No. 1263.

Waste Management Licensing Regulations (1994). Statutory Instrument 1994 No. 1056.

PLANET – THE NATIONAL STANDARD DECISION SUPPORT AND RECORD KEEPING SYSTEM FOR NUTRIENT MANAGEMENT ON FARMS IN ENGLAND

P Dampney and E Sagoo

ADAS Boxworth, Battlegate Road, Boxworth, Cambridge, CB23 4NN, UK, E-mail: Peter.Dampney@adas.co.uk

SUMMARY

PLANET (Planning Land Application of Nutrients for Efficiency and the environmenT) is a decision support system for routine use by farmers and their advisers to plan and record fertiliser and manure use on fields in an easy and user-friendly way that is technically correct and meets compliance requirements. Fertiliser recommendations are given for each crop, and users may then devise their own nutrient application plan and keep field-level records. The recommendations mimic those in Defra's 'Fertiliser Recommendations (RB209)' publication (7th edition). Released in January 2005, there are now over 6500 registered users of the standalone version of PLANET. In addition, 4 companies have integrated PLANET into their commercial software used by farmers. PLANET version 2 will contain new calculation and recording modules that will help farmers comply with the revised NVZ Action Programme rules, including Whole Farm Manure N Loading, Organic Manures Inventory and Storage Requirements and Farmgate Nutrient Balance.

INTRODUCTION

Reducing nutrient pollution of water and air are key policy objectives of Defra (Department for Environment, Food and Rural Affairs) and the Environment Agency (EA) whilst also to develop and foster a competitive farming industry. It is estimated that about 60% of nitrate and 25% of phosphorus in English waters originate from agricultural land. A mix of advisory, incentivised and regulatory measures are in place or being developed to help farmers adapt where necessary so that production methods are economically and environmentally sustainable. The expansion of Nitrate Vulnerable Zones (NVZs) to 55% of the land area of England is a measure specifically designed to reduce diffuse nitrate pollution. As part of the Cross Compliance requirement to qualify for the Single Farm Payment, farmers with land inside a designated NVZ must comply with the NVZ Action Programme (AP) rules (Anon, 2002). The NVZ AP rules are currently being revised. Future implementation of the requirements of the Water Framework Directive is likely to place further emphasis on nutrient management planning.

In order to achieve these objectives, Defra has sought to provide information, advice and help to farmers that is clear and effectively adopted in practice. It must be accessible, understandable and usable, and with integrated environmental and economic messages. National standard recommendations for the use of fertilisers are contained in Defra's 'Fertiliser Recommendations for Agricultural and Horticultural Crops (RB209)' publication (Anon, 2000). RB209 was first published in 1973 and is currently in its 7th edition. These recommendations are widely used in practice, not only in England but also in Wales and Northern Ireland. A wide range of other

Defra and EA publications provide supporting information and advice on nutrient management.

DEVELOPMENT OF PLANET VERSION 1

PLANET (Planning Land Application of Nutrients for Efficiency and the environmenT) has been developed as a software tool designed for routine use on farms so that farmers or their advisers can plan and manage nutrient use on individual fields in a quick, easy and user-friendly way that is also technically correct and meets compliance requirements. It is designed as a 'tool' for practitioners, to help and encourage careful planning and recording of nutrient applications to crops and to help farmers comply with the legal and scheme requirements that are in place, notably the NVZ AP rules. PLANET is publicly owned and the CD is provided free of charge to farmers and their advisers. Technically, the national standard (RB209) recommendations generated by PLANET mimic those that would be generated by the source publication (Anon, 2000). This publication is currently being revised with publication expected in late 2008. It is expected that PLANET will also be revised so that there continues to be only one national standard recommendation system in use but that can be accessed in either hard or soft formats.

The structure and functionality of PLANET version 1 has been described by Gibbons *et al.* (2005). In summary, new users need to set up a few relevant details of the farm and each field (e.g. farm name, average annual rainfall, field name, soil type). To obtain the RB209 recommendations for each crop, information is needed for past cropping (at least one previous year), soil analysis, use of organic manures and use of fertilisers and lime. Some minimum information must be available in order to obtain a recommendation (e.g. crop type, soil type, organic manure type), but default information is used for other data types if specific information is not available. However, users are encouraged to obtain specific information so that the recommendations can be as accurate as possible. Once the recommendations have been generated, users can devise their own nutrient application plan for each field by rate and timing of application of each nutrient. At the end of the season, the details of the nutrient plan may be edited and then confirmed as an accurate record of what actually happened. The field record for one harvest year then becomes the basis for generating RB209 recommendations for the next harvest year. There are a range of reports that can be generated, printed off or exported, providing output of past field records, the forward nutrient application plan for each field, the RB209 recommendations for each field or a statement of compliance with the NVZ AP rules. The cycle of how PLANET is intended to be used is illustrated in Figure 1.

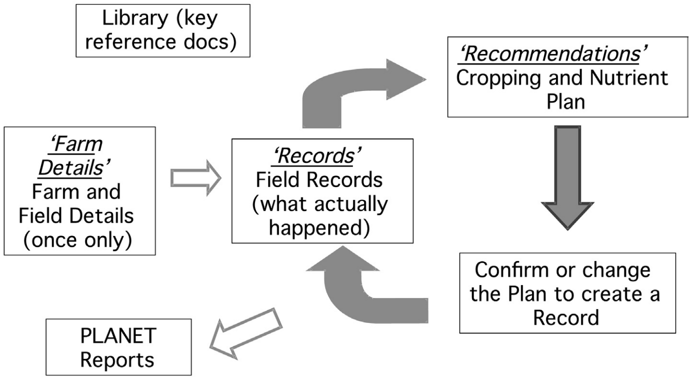


Figure 1: PLANET version 1 – content and the annual cycle of use

From the outset of the PLANET project, it was recognised that there was a need for a close working relationship with the agricultural software industry. It was clear that farmers with existing commercial farm recording software do not want to enter field-level information into a separate software system. Also, since field records for the majority of the national arable area are held on commercial software systems, effective integration of PLANET provides rapid market penetration. To date, the PLANET RB209 recommendations DLL (Dynamic Link Library) has been integrated into 4 commercial software systems. Each company is required by their licence to ensure that their database can hold and present all of the data-types required by the DLL. Each company designs its own Graphical User Interface (GUI) for data entry and output though technical aspects of the integration are subject to scrutiny by Defra/ADAS.

USER SUPPORT

Effective user support to PLANET users is regarded as essential to ensure that PLANET is used correctly on farms and is widely regarded as a worthwhile, reliable and credible tool. Several components of user support have been developed as follows.

Website

The PLANET website (www.planet4farmers.co.uk) provides an easily accessible focus for potential new users to find out about PLANET and to download documents, the PLANET software and updates. Users can select from 'About PLANET', 'News and Events' and 'Software Download' pages. The website also provides a mechanism for registering to receive a copy of the standalone version. Simple contact details are logged to the database including the user's email address.

Helpline

All users of the PLANET standalone version can contact the PLANET Helpline free of charge by telephone, email or from the PLANET website. All aspects of the Helpline are operated by ADAS staff. Queries are routed via PLANET Admin reception where trained staff deal with as many of them as possible. Complex queries are routed by email to a duty software specialist or duty nutrient management specialist depending on the nature of the query. Queries and the responses given are logged to the PLANET Helpline database, allowing interrogation of the Helpline activities.

The Helpline also provides help to those commercial software companies who have signed the Defra DLL Licence Agreement. ADAS technical specialists are available to help these companies integrate the PLANET DLL in an effective way. Help to users of these commercial software packages is provided by the software company, not ADAS.

Seminars, Training Workshops and Demonstrations

Farmers and advisers have the opportunity to be briefed and/or trained in the use of PLANET in various ways. Most events are free, being funded by Defra either as part of the PLANET project or as part of other focused advice activities to farmers.

- PLANET is commonly described and demonstrated as part of seminar type meetings that focus on or include nutrient management within the programme.
- Many whole day workshops have been held across England to train users in the PLANET standalone version. Each workshop has a maximum of 25 delegates, each with their own laptop, either their own or provided by ADAS. To start with, an overview of PLANET is given followed by a demonstration, with delegates following the trainer using their own laptops. In the second half of the workshop, delegates work on their own using a provided exercise to enter field information and obtain nutrient recommendations. The format of these workshops has proved to be very successful. Over 60 workshops have been held since the launch of PLANET with an average of about 20 delegates at each workshop.
- Each year, the PLANET standalone software has been demonstrated at selected national farming events (e.g. Cereals event, Dairy event, Grassland/Muck event, Smithfield show).

Email Updates

When needed, emails are sent by ADAS to all registered PLANET users covering topics including:

- Notification of new versions of PLANET or software updates.
- Notification of relevant events and training workshops.
- Technical information, for example advice on spring nitrogen use.

USER FEEDBACK

In February 2006 (13 months following release of version 1), a questionnaire was sent by email to all registered users of the PLANET standalone version and to farmers/advisers who had access to the PLANET module in the Muddyboots CropWalker package (Dampney, 2006). A total of 463 responses were obtained from PLANET standalone users (9% response rate) and 108 responses from users of CropWalker PLANET. Key points were as follows:

- Approximately 1,260 farms and 184,000 ha of land owned by or advised on by respondents, have received RB209 recommendations via PLANET. The majority of use of the standalone version was by advisers rather than farmers (just over 70% of both farms and land area was from use by advisers). Most land and farms influenced were in arable cropping systems, but less than 10% of land or farms were in a dairy system.
- 87% of all respondents expected to use PLANET either regularly or occasionally in future; 35% of respondents who had not yet made any serious use of PLANET standalone said that this was due to lack of time. 19% said this was because they were waiting for PLANET to be delivered via a commercial software package; however, all of these were concerned with arable not grass cropping. This reflects the importance of rolling out PLANET via commercial software but that the impact of this route is much stronger in the arable rather than the grassland sector. 47% of all 374 arable farm respondents said that they are either already using, or expect to use PLANET provided as part of commercial software. Only 23% of grass farm respondents expected to use PLANET in this way.
- 66% of farmers, but only 37% of advisers strongly agreed or agreed that PLANET standalone was ‘quick and easy to use’. Around 50% of respondents said that the PLANET operation and facilities were very good or good. This emphasises the importance of keeping the ease, simplicity and user-friendliness of the software as a key, high priority design criterion. Comments on the ease of use of PLANET standalone were variable with some criticisms of certain aspects, notably the data entry process. Adverse comments were more common from advisers than farmers – this may reflect that advisers use PLANET more often and are therefore more demanding.
- The PLANET website and Helpline had been used by 57% and 37% respectively of all respondents, and 30-40% of these had found them either very good or good. Nearly half of respondents had attended a PLANET workshop, and 60% had found it to be very good or good. There were several requests for more training.
- The main motivating factors for using PLANET were to help with complying with the requirements of the NVZ AP rules and the ELS Nutrient Management Plan option. Improved farm profitability (through better decisions on nutrient use), or ease of nutrient planning, were less important. Meeting compliance requirements was less important to grass farmers who were generally less motivated by any of the reasons.

The survey responses showed that PLANET has had significant use and impact within the first year since its release, but more in areas of arable cropping than grassland cropping. Bearing in mind that many registered users have probably not yet fully realised their intentions to use PLANET, and that the number of registered users is continually increasing (currently 6500), the future prospects for the use of PLANET are encouraging. However, there is a need to stimulate and support the use of PLANET in the grassland sector.

FUTURE DEVELOPMENTS

Version 2 of the PLANET standalone software is currently under development and will contain the following additional calculation modules and improvements. It is expected to be ready for release in spring 2008. The new modules will provide methods that have been approved by Defra and the Environment Agency for the calculation, recording and reporting of the measures contained in the revised NVZ AP. The modules are based on the proposed measures outlined in Defra's NVZ Consultation document (August 2007). These measures are subject to change following the comments received from the consultation, and any changes will be reflected in the new modules developed for PLANET. Each module is being developed as a DLL that will be made available under licence to commercial software companies that wish to incorporate the DLL into their software.

- An ***Organic Manures Inventory and Storage Requirements*** module that will calculate monthly quantities and nutrient content of slurries, solids and dirty water, and the minimum slurry storage requirement as required for compliance with the proposed NVZ AP measures.
- An ***Organic Manure Storage Capacity*** module that will calculate the storage capacity of existing slurry and solid manure stores based on store dimensions.
- A ***Whole Farm Livestock Manure N Loading*** module, that will calculate the current whole-farm loading as required for compliance with the proposed NVZ AP measures.
- ***'ENCASH'*** module that will calculate the annual N and P production of different livestock types based on diet. The output may be used when calculating compliance with the proposed NVZ whole farm livestock manure N loading limit.
- A ***Compliance with Nmax*** function that will calculate the farm average maximum N rate (Nmax) for individual crop types on the farm as required in the proposed NVZ AP measures.
- A ***Farmgate Nutrient Balance*** that will calculate the balance of nitrogen, phosphate and potash coming onto a farm (e.g. in feeds, fertilisers, organic manures) against these nutrients exported off the farm (e.g. in farm produce, organic manures).
- A ***Data Export function*** to allow users to export data from the PLANET database into an external spreadsheet for additional calculations (e.g. converting a nutrient plan into a buying requirement for commercial fertiliser products).
- Various improvements to functionality to make the PLANET standalone version more user-friendly.

It is expected that a further version of PLANET will be developed to take account of the revisions to Defra's 'Fertiliser Recommendations (RB209)' publication. These revisions are currently under discussion and due for completion by summer 2008.

ACKNOWLEDGEMENTS

The PLANET software has been developed by ADAS with funding from Defra, the Environment Agency (EA) and the Department for Agriculture and Rural Development in Northern Ireland (DARDNI). Guidance is given by a Steering Group with membership drawn from a wide range of stakeholder organisations.

REFERENCES

Anon (2000). Fertiliser recommendations for agricultural and horticultural crops (RB209), 7th edition. The Stationery Office, Norwich.

Anon (2002). Guidelines for Farmers in NVZs. Defra Publications PB5505.

Anon (2005). Entry level Stewardship Handbook. Defra Publications PB10355.

Dampney PMR (2000). Development and revision of national fertiliser recommendations for the use of fertilisers and organic manures in England and Wales. Proceedings of the International Fertiliser Society No. 457, York.

Dampney PMR (2006). PLANET Satisfaction Survey. Report to Defra. http://www.planet4farmers.co.uk/downloads

Gibbons MM, Fawcett CP, Waring RJ, Dearne K, Dampney PMR and Richardson SJ (2005). PLANET nutrient management decision support system – a standard approach to fertiliser recommendations. EFITA/WCCA conference, Portugal, pp. 89-94.

BASELINE SCOTLAND: TOWARDS BETTER GROUNDWATER QUALITY MANAGEMENT

BÉÓ Dochartaigh[1], AM MacDonald[1], MT Graham[1] and V Fitzsimons[2]

[1]British Geological Survey, Murchison House, West Mains Road, Edinburgh, EH9 3LA, UK, E-mail: beod@bgs.ac.uk; [2]Scottish Environment Protection Agency, Clearwater House, Heriot Watt Research Park, Avenue North, Riccarton, Edinburgh, EH14 4AP, UK

SUMMARY

The natural chemistry of groundwater is influenced by geology, rainfall and soil characteristics, but can be modified by direct and indirect anthropogenic activities. A thorough understanding of the natural background groundwater chemistry and the processes impacting on it is needed to identify and characterise the impacts of pollution, and to help to effectively protect valuable groundwater resources.

A new project, Baseline Scotland, running from 2005 to 2010, is being jointly funded by the British Geological Survey (BGS) and SEPA, and seeks to improve data availability and general understanding of the chemistry of Scotland's groundwater. Its primary aim is to provide core hydrogeochemical data and interpretation to help in the implementation of the Water Framework Directive and in particular the goal of achieving good groundwater quality.

By December 2007 four areas have been surveyed: Strathmore, southern Scotland, Aberdeenshire, and the Moray and Invernesshire coast.

INTRODUCTION: WHY BASELINE GROUNDWATER CHEMISTRY?

Groundwater chemistry is an important control on many surface water and terrestrial ecosystems, including wetlands, lochs and streams. In its natural state groundwater is generally of excellent quality. However, the natural quality of groundwater is continually being modified by human influence. A thorough knowledge of groundwater quality, including a good understanding of the controlling physical and chemical processes, is essential for effective management of this valuable resource.

The natural chemical properties of groundwater are largely determined by geochemical processes that take place as rain or surface water infiltrate the ground and react with rock-forming minerals. This natural baseline groundwater chemistry varies from one rock type to another because of the different minerals present. Evolution of groundwater chemistry occurs over time and space, even within the same rock type, as groundwater flows through aquifers. Variations in groundwater quality are seen at their simplest in the different areas of hard and soft water across the country.

A number of geochemical processes can shape the unique natural characteristics of groundwater, including:

- oxidation and reduction (which control natural concentrations of elements such as iron, manganese, arsenic and chromium);

- mineral solubility (which controls many element concentrations, including fluoride and barium); and
- sorption and exchange with mineral surfaces (which affect the concentrations of many trace elements).

Each area is therefore underlain by an almost unique natural groundwater quality resulting from the local geology. Until recently, however, these natural variations in groundwater chemistry across Scotland were generally unknown.

Baseline Scotland is a targeted programme of data collection and interpretation to improve our understanding, support effective environmental management and meet the requirements of the Water Framework Directive. Without information from Baseline Scotland it will be difficult to:

- identify polluted groundwater;
- assess the impacts of human activity, including the mitigating effects of changing land management policies; or
- plan for remediation of any contamination.

PRE-BASELINE: PREVIOUS STUDIES OF GROUNDWATER CHEMISTRY IN SCOTLAND

Before the onset of the Baseline Scotland project in 2005 there was little information on the natural chemistry of groundwater in Scotland. Existing data tended to be old, of variable quality, limited (often to a few major ions only), and skewed to areas of groundwater contamination, particularly related to mining.

At the start of the project we reviewed previous studies of groundwater chemistry and available groundwater chemistry data in Scotland (MacDonald and Ó Dochartaigh, 2005). A total of 428 good quality major ion analyses for groundwater were identified – those with an error in ionic balance of less than 10%. They are not distributed evenly but are biased towards the more productive Scottish aquifers, in particular the Devonian aquifers of Fife, Strathmore and Morayshire, and the Dumfries Permian aquifer. Most of the samples were collected from boreholes, but many came from springs, particularly on lower productivity aquifers. Most of the available analyses include only those trace elements that are likely to be present at high concentration – often only Mn, Fe and Zn. Even where trace elements were analysed, the laboratory's analytical detection limits are often too high to reveal any detail in element chemistry. These major and trace element data provide a limited picture of groundwater quality across Scotland based on information available at the start of the Baseline Scotland project, and allow gaps in information and understanding to be identified. However, the lack of analytical detail means they are not being used further during the project.

The Baseline Scotland Project

This project, carried out by the British Geological Survey (BGS) in collaboration with SEPA, will improve data availability and general understanding of the chemistry of Scotland's groundwater, and thereby support the implementation of the Water Framework Directive (WFD). The twin aims of the project are:

- To characterise the ranges in natural background groundwater quality in the main aquifer types in Scotland, by carrying out groundwater sampling surveys that as far as possible incorporate representative areas of each aquifer, allowing extrapolation of the interpreted results to the remaining parts; and
- To provide a scientific foundation to underpin Scottish, UK and European water quality guideline policy, notably the Water Framework Directive, with an emphasis on the protection and sustainable development of high quality groundwater.

The project runs from 2005 to 2010. Each year, high quality new groundwater chemistry data are generated by collecting and analysing new groundwater samples from one or more hydrogeological area, analysed for a detailed range of determinands by BGS laboratories (e.g. see Ó Dochartaigh *et al*., 2006), and the resulting data interpreted in the light of known hydrogeological conditions.

BASELINE SCOTLAND STUDY AREAS

To help interpret the groundwater chemistry data, seven hydrogeological units have been defined within Scotland on the basis of geological age (the six bedrock units are shown in Figure 1).

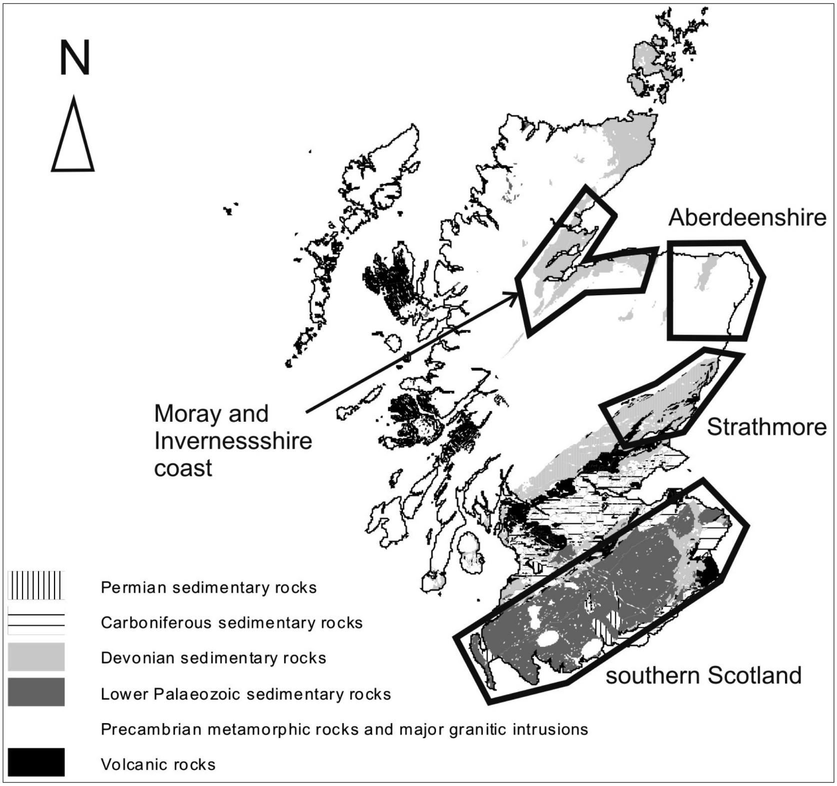


Figure 1: Bedrock hydrogeological units and study areas already surveyed (superficial deposits aquifers are on too small a scale to be shown)

Many of these hydrogeological units are widespread across Scotland: for example, major outcrops of Devonian sedimentary rocks occur in Orkney and Caithness; the Moray and Invernessshire coast; Strathmore; and the Borders. To enable effective data collection and interpretation, study areas have been defined based on the hydrogeological unit, geographical location, and general groundwater usage (Table 1).

Table 1: Study areas and aquifers in the Baseline Scotland project

Geographical area	Aquifer
Strathmore	Devonian sedimentary and volcanic rocks
Southern Scotland	Lower Palaeozoic, Devonian and Carboniferous sedimentary, volcanic and intrusive igneous rocks
Aberdeenshire	Weathered Precambrian metamorphic and granitic rocks
Moray and Invernesshire coast	Devonian sedimentary rocks
Midland Valley	Carboniferous sedimentary and volcanic rocks
Caithness and Orkney	Devonian sedimentary rocks
Fife	Upper Devonian sedimentary rocks
Highlands	Precambrian metamorphic and granitic rocks
Throughout Scotland	Permian sedimentary rocks
Throughout Scotland	Superficial valley-fill deposits

PRELIMINARY RESULTS

Between 2005 and 2007, targeted sampling was carried out in the first four of the Baseline Scotland study areas (Figure 1). Analysis and interpretation of the new groundwater chemistry data from the first of these, the Devonian aquifer in Strathmore, are presented in Ó Dochartaigh *et al.* (2006). This, and preliminary interpretation of the new data from the next three areas, is already providing a more detailed and more accurate understanding of the baseline groundwater chemistry across Scotland.

In Strathmore groundwaters are typically oxygenated, slightly alkaline (pH 7.3-7.6), moderately hard (HCO_3 130-210 mg/l), and of Ca-HCO_3 type (Figure 2). Where dolomitic cement is present in the sandstone aquifer the groundwaters show elevated Mg concentrations and a distinct Ca-Mg ratio. Coastal groundwaters show evidence of the impacts of the sea and saline intrusion: they are often reducing and of Na-Cl type. Fe and Mn concentrations are low almost everywhere, except in reducing coastal groundwaters. The median nitrate concentration is 6.7 mg/l as N, with a 75th percentile of 11.7 mg/l and a 90th percentile of 15.7 mg/l, both as N. The median phosphate concentration is 30 µg/l as P, ranging up to a 90th percentile of 100 µg/l as P. Stable isotope and CFC gas analysis indicates that most of the groundwater is of relatively young age – recharged within the last 50 years ago – and is well mixed within the top 100 m of the aquifer. Active recharge also means there is a route for contaminants to enter the aquifer, and groundwater is therefore vulnerable to contamination. The widespread presence of elevated nitrate concentrations throughout the top 100 m of the aquifer is another indicator of the vulnerability of groundwater: the main source of nitrate in groundwater in the UK is from agricultural contamination (e.g. Dunn *et al.*, 2004; Johnson *et al*., 2007).

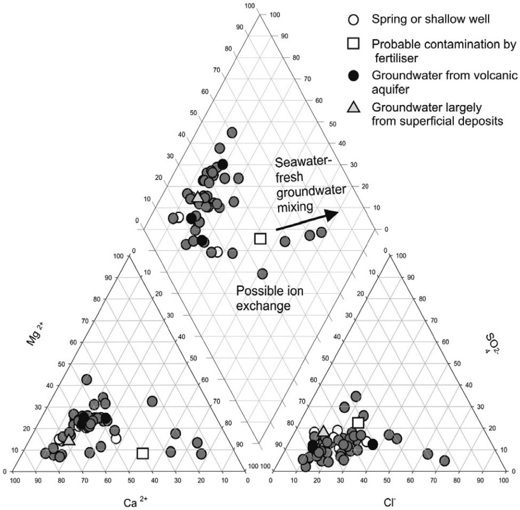


Figure 2: Piper diagram illustrating the major ion groundwater chemistry of the Strathmore Devonian aquifer, including seawater-fresh groundwater mixing

By contrast, initial interpretation of new data from the Moray and Invernessshire Devonian aquifer indicates that the groundwaters here are often reducing, with elevated Fe concentrations.

In the Lower Palaeozoic aquifers of southern Scotland, groundwaters are typically oxygenated, with near-neutral pH (6.8-7.7), and variable hardness (HCO_3 85-250 mg/l). Median nitrate concentrations are less than 5 mg/l as N, ranging up to a 90th percentile of 10.6 mg/l as N. Elevated nitrate concentrations show a correlation with land use, particularly with pasture land used for dairy poultry or pigs, and secondarily with arable land (Figure 3). Land use has been assessed by field observations and by means of the Land Cover of Scotland 1988 dataset (MLURI, 1993). CFC analysis indicates that all the sampled groundwaters had a component of modern water (recharged within the past 50 years), and nitrate concentrations show a strong correlation with the proportion of modern water (Figure 3).

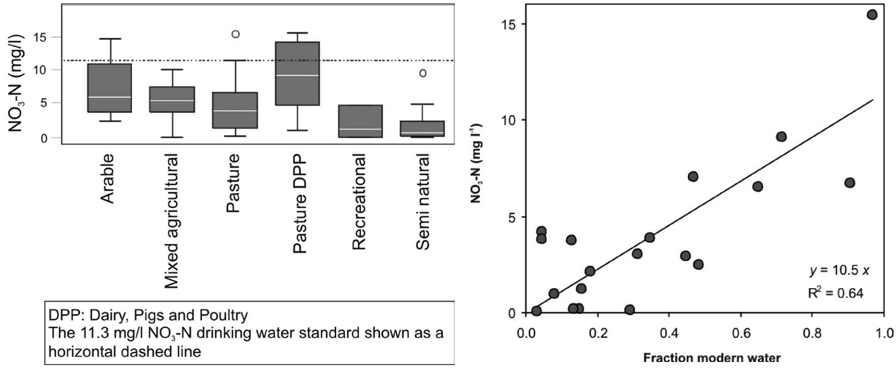


Figure 3: Box plot (left) variation in groundwater nitrate concentrations with land use type; and (right) correlation between nitrate concentration and the proportion of modern water. Both for southern Scotland

The median phosphate concentration in groundwater in southern Scotland was 29 µg/l as P, and in all samples was less than 100 µg/l as P (Figure 4). A number of well-known mineralised springs occur that have quite different chemical compositions. These often issue from shale bands and show elevated concentrations of a variety of minerals, including SO_4, Fe, Na and Cl, and in one spring a pH of 3.7.

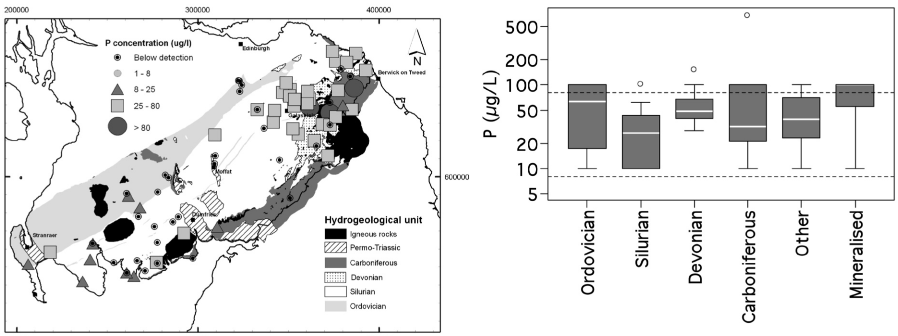


Figure 4: Spatial distribution of P (µg/l) concentrations in groundwater across southern Scotland (left), and (right) box plot of P variations in different hydrogeological units. The SEPA management limits for phosphate in oligotrophic and eutrophic surface water bodies are shown (8 and 80 µg/l as P, respectively)

CONCLUSIONS

Baseline Scotland offers a unique opportunity to characterise the chemistry of groundwater across Scotland and estimate the influence of anthropogenic activity, including land use, on groundwater quality. The dataset will be of most use when completed in 2010, but the following preliminary conclusions can be drawn.

In most groundwaters sampled to date there is a high proportion of modern water, and therefore recent land use activities which affect the chemistry of recharge can potentially exert considerable influence on groundwater quality.

The most significant groundwater quality problem identified is the widespread presence of elevated nitrate concentrations, sometimes higher than the drinking water standard. There is a strong correlation between nitrate concentration and the proportion of modern water, agreeing with the results of previous studies (e.g. MacDonald *et al*., 2003). The correlation between elevated nitrate concentrations and land use also agrees with other studies, which have shown that groundwater in areas of land used for arable farming and for dairy, pigs and poultry shows generally higher nitrate concentrations (e.g. MacDonald *et al*., 2005). There appears to be little correlation between nitrate concentration and depth, probably because groundwaters in the uppermost 100 m of most Scottish aquifers are well mixed.

The results also indicate that phosphate concentrations in groundwater may be highly significant in areas where there is significant groundwater baseflow to surface water bodies. Many samples to date show phosphate concentrations that exceed the SEPA limit for eutrophic surface waters (80 µg/l as P), and the majority exceed the SEPA limit for oligotrophic surface waters (8 µg/l as P). The relationship with land use is much less straightforward than nitrate, due to the more complex geochemical behaviour of phosphate, but groundwater in areas of arable agriculture generally has higher phosphate concentrations than in areas of semi natural vegetation or pasture agriculture (Figure 4).

REFERENCES

Dunn SM, Vinten AJA, Lilly A, DeGroote J, Sutton MA and McGechan M (2004). Nitrogen Risk Assessment Model for Scotland: I. Nitrogen leaching. Hydrology and Earth System Sciences 8, 191–204.

Johnson D, Fletcher SW and Bray K (2007). Integrated assessment of nitrate concentrations for the designation of groundwater Nitrate Vulnerable Zones in England and Wales. Quarterly Journal of Engineering Geology and Hydrogeology 40**,** 407-415.

MacDonald AM, Darling WG, Ball DF and Oster H (2003). Identifying trends in groundwater quality using residence time indicators: an example from the Permian aquifer of Dumfries, Scotland. Hydrogeology Journal 11 (4), 504-517.

MacDonald AM and Ó Dochartaigh BÉ (2005). Baseline Scotland: an overview of available groundwater chemistry data for Scotland. British Geological Survey Commissioned Report CR/05/239N.

MacDonald AM, Griffiths KJ, Ó Dochartaigh BÉ, Lilly A and Chilton PJ (2005). Scotland's groundwater monitoring network: its effectiveness for monitoring nitrate. British Geological Survey Commissioned Report CR/05/205N.

MLURI (1993). The Land Cover of Scotland 1988. Final Report. The Macaulay Land Use Research Institute, Aberdeen.

Ó Dochartaigh BÉ, Smedley PL, MacDonald AM and Darling WG (2006). Baseline Scotland: the Lower Devonian aquifer of Strathmore. British Geological Survey Commissioned Report CR/06/250N.

H_2O (HEURISTICS FOR HYDROLOGICAL OBSERVATORIES): A PROTOTYPE APPLICATION TO THE LUNAN WATER CATCHMENT

MN Futter[1], R Ferrier[1], D Ballantine[2], A Vinten[1], K Blackstock[1] and SM Dunn[1]

[1]Macaulay Land Use Research Institute, Craigiebuckler, Aberdeen, AB15 8QH, UK, E-mail: m.futter@macaulay.ac.uk; [2]Scottish Environmental Protection Agency, Carseview House, Stirling FK9 5AD, UK

SUMMARY

The term “heuristic” has several meanings. It can mean “a replicable method or approach for directing one’s attention in learning, discovery, or problem-solving”; it may also mean a “rule of thumb” or “’fag-packet’ calculation”. Hydrological Observatories will provide a platform contributing to understanding of water resources which encompasses both natural variability and anthropogenically induced changes on regional-scale hydrologic systems, land-atmosphere interactions, and the biogeochemical cycles that control contaminant transport. Here, we show how simple heuristic ‘fag-packet’ calculations can lead to an improved understanding of the source and potential costs of diffuse pollution in Scottish waters, and how this understanding can be applied to the Hydrological Observatory concept.

INTRODUCTION

Understanding of diffuse pollution pressures can come through monitoring or modelling. The Screening Tool (SNIFFER, 2006) provides a “broad brush” picture of environmental conditions in Scotland. However, most of the results are based on modelled as opposed to measured data. Using measured data and simple models may aid in understanding the health of Scottish waters. SEPA, the Scottish Environmental Protection Agency, has collected large amounts of monitoring data throughout Scotland. When used as inputs to simple calculations, these data can provide further understanding of the pressures and stresses faced by waters in Scotland. One of the simplest calculations that can be performed with flow and concentration data is the calculation of loads, which are obtained by multiplying flow by concentration. Loads are an estimate of the mass of a chemical passing a particular point in the catchment. These calculations can offer additional insight to those obtained through use of the Screening Tool and monitoring of concentrations. For example, high summer concentrations may be less of an issue for diffuse pollution of downstream waters due to low base-flows; most of the impact may be felt during periods of relatively low concentration but high flows.

Eutrophication of inland and coastal waters is a serious problem in Scotland. Many Scottish catchments are affected by diffuse non-point source and point source pollution. Excessive phosphorus (P) inputs cause eutrophication in lochs while excessive inorganic nitrogen (N) causes eutrophication of coastal waters. Inputs of both N and P are required to farm in Scotland. There are a number of pieces of legislation designed to manage diffuse pollution in the EU. These include the Nitrate Directive (91/676), which designates Nitrate Vulnerable Zones (NVZ) and the Water Framework Directive (WFD) (2000/60/EC) which requires environmental and economic characterization of water bodies and the development of programmes of measure to ensure good ecological status. Diffuse pollution arises from historical and current sources. Eutrophication is partly a legacy issue. For example, more P is coming out of some Scottish lochs than is going in. N may be a legacy issue in groundwater or a current issue in surface water run-off.

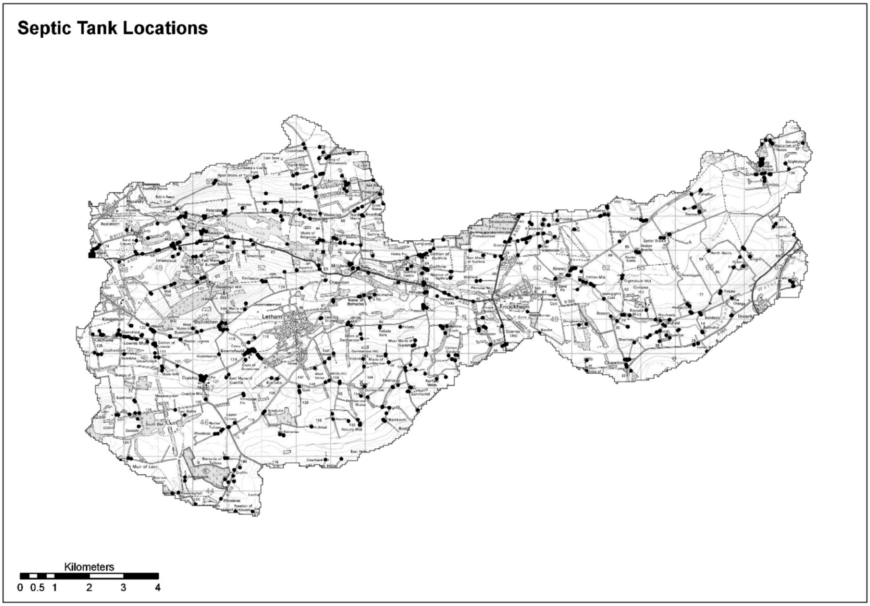


Figure 1: Map of the Lunan Water catchment showing the location of 830 septic systems

There are a number of reasons for prototyping the H_2O approach at the Lunan Water. First, smaller catchments are easier to understand than large ones. The Lunan Water is subject to many of the same pressures (diffuse pollution and hydromorphological alteration) as the Tay, Scotland's largest river. Increasing NO_3 concentrations in the Lunan Water are a concern. It is important to determine whether it originates in surface waters or ground water. Eutrophication of Rescobie Loch is currently an issue. Lunan Bay is an important recreational bathing beach but it is not currently affected by riverine eutrophication as there is not a significant estuary. Understanding diffuse pollution in the Lunan Water will aid in understanding diffuse pollution in the Tay. If the H_2O approach presented here aids in understanding the diffuse pollution pressures on the Lunan Water, it should be possible to incorporate it into the planned Hydrological Observatory on the Tay.

There are perceptual issues around diffuse pollution arising from agricultural practices, rural residency, pollutant sources and management options. Farmers are often unaware of the off-farm impact of their actions (Macgregor and Warren, 2006). Monitoring agencies may be unaware of the sources of pressures. Regulators may be unaware of the impact of their decisions on farm business and the non-farm rural and peri-urban population may be unaware of their environmental footprint.

RESULTS

Flow and chemical data were obtained from the SEPA database. Locations of septic systems in the catchment (Figure 1) were derived from a household survey. The Lunan Water catchment has an area of 134 km^2; Approximately 124 km^2 are drained through the SEPA hydrochemical monitoring site at Kirkton Mill. Loads to the marine environment were estimated by multiplying Kirkton Mill fluxes by 134/124.

Understanding diffuse pollution inputs requires the separation of surface and ground water derived inputs. The hydrograph at Kirkton Mill (Figure 2) shows a great deal of seasonality with high winter flows and low base flows in summer. It appears that most of the water flowing through Kirkton Mill is derived from surface waters. A different picture is obtained when flows are summarized on a monthly basis (Figure 3). Hydrograph separation was conducted in a simplistic manner. It was assumed that all summer flows (June-September) were driven by base flow and that flows in the other months were a combination of surface and groundwater flow. The amount of base flow in October-May was assumed to be equivalent to the average summer (June-September) flow. Thus, base flow contributed approximately 4% of the total annual flow each month, or in the neighbourhood of half the annual total flow at Kirkton Mill.

Potential septic system contributions to diffuse pollution were calculated in the following manner. Using a household size of 4 and per-capita septic system loads from SNIFFER (2006) and Defra (2002), it was estimated that each septic systems delivers between 0.3-1.2 kg P/yr and 6.5-16 kg N/yr to the Lunan Water catchment. With 830 septic systems in the catchment, there is a potential input of ~250–1000 kg P/yr and 5400-13,300 kg N/yr.

Table 1: Phosphorous source apportionment for Lunan Water

Source	Input (kg/yr)	Output (kg/yr)
Kirkton Mill		2500
Septics	250-1000	
Lochs	350	
Agriculture (by difference)	1150-1900	

Lochs in the Lunan Water may be a major source of P (Figures 4 and 5). Summer P concentrations in the Rescobie Loch outflow were higher than the inflow (Figure 4), suggesting that loch sediments are a net source of P in the catchment. Loads were calculated in the following manner: at Rescobie Loch, it was assumed that hydrological inflows were equal to outflows and temporal patterns of flow at loch inlet and outlet were the same as at Kirkton Mill. Loads from Balgavies Loch were calculated by using area-weighted Rescobie estimates. Net P export from Rescobie Loch sediments was calculated to be ~13.25 kg/month (Figure 5), or 160 kg/yr. If Balgavies Loch is behaving in a similar manner, it may export ~190 kg/yr.

It can be seen from Table 1 that the direct effects of agriculture contribute 45-75% of the P inputs to the Lunan Water. The effects of septic systems are worthy of further study as they may contribute up to 40% of the observed P load.

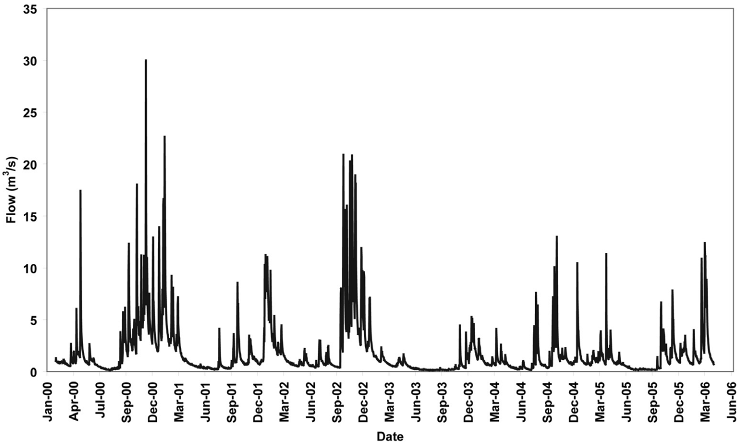


Figure 2: Hydrograph of daily flows at Kirkton Mill

A large amount of inorganic N as NO_3 (approximately 42 tonnes/yr) is lost from the catchment (Figure 6). Almost all of this is agricultural in origin. Using numbers from the high-end of possible septic loads, approximately 1 tonne/month (or between 1 and 2.5%) of the inorganic N leaving the catchment may come from septic inputs

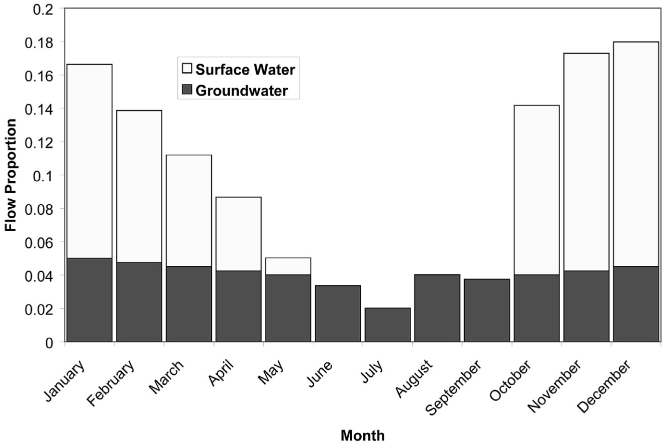


Figure 3: Monthly flow proportions at Kirkton Mill

Kirkton Mill exports approximately 2.5 tonnes P/yr and close to 500 tonnes N/yr. The majority of N is exported as NO_3. To put these figures into perspective, inorganic N fertilizer currently retails for approximately £270 tonne in the UK. This fertilizer is 34.5%N by weight. Thus, the cost of 1 tonne of inorganic N is approximately £780. Using these figures, it can be seen that approximately £390,000 worth of inorganic N is lost from the Lunan Water catchment every year.

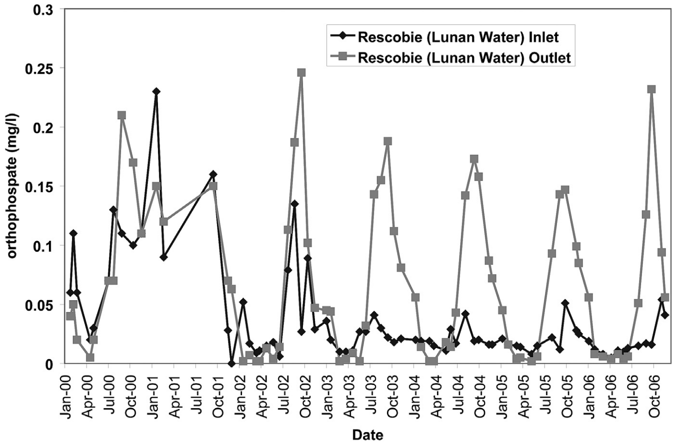


Figure 4: Orthophosphate concentrations in the Rescobie Loch inflow and outflow

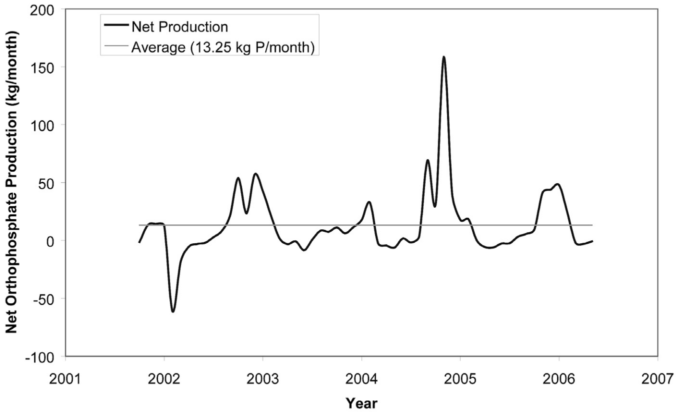


Figure 5: Net export of P from Rescobie Loch sediments (13.25 kg/month)

DISCUSSION

An analysis of the SEPA monitoring data reveals that while almost all the inorganic N in the catchment is delivered by agriculture, the same may not be true of P. The method presented here is probably most suited to catchments with diffuse pollution pressures. It should work for catchments subject to point source pollution pressures but may not work too well for those subject to hydromorphological alteration.

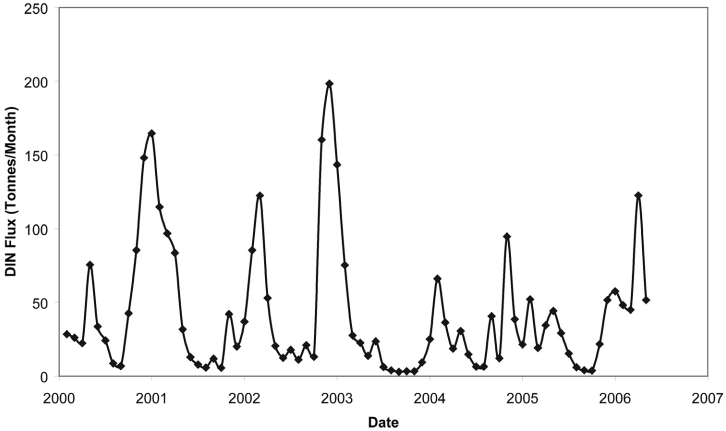


Figure 6: Monthly dissolved inorganic nitrogen fluxes at Kirkton Mill

There are four main points arising from the application of the H_2O concept to the Lunan Water. The first is the possibility that much of the phosphorus entering surface waters in the catchment originates from septic systems as opposed to agricultural activity. The use of phosphate-free detergents and better septic system maintenance may reduce the septic system contribution to surface water eutrophication. The second is that alternate loch management strategies need to be employed on Rescobie and Balgavies Loch to reduce summer time mobilisation of phosphorus from the sediment. These strategies may include more focussed management of the fish community in the loch or attempts to reduce the potential for anoxia at the sediment water interface. Third, as approximately half of the flows in the Lunan Water are derived from ground water, it is imperative that a groundwater sampling campaign be conducted. Finally, the high cost of nitrogen being lost from the land to the marine environment is worthy of further attention.

REFERENCES

Defra (2002). Development of a risk assessment and decision-making tool to control diffuse loads of phosphorus and particulates from agricultural land, PE0202.

Macgregor CJ and Warren CR (2006). Adopting sustainable farm management practices within a Nitrate Vulnerable Zone in Scotland: The view from the farm. Agricultural Ecosystems and Environment 113, 108-119.

SNIFFER (2006). Provision of a Screening Tool to Identify and Characterise Diffuse Pollution Pressures: Phase II WFD19 (230/8050).

CONSTRUCTED FARM WETLANDS (CFWs) FOR REMEDIATION OF FARMYARD RUN-OFF: WATER TREATMENT EFFICIENCY AND ECOLOGICAL VALUE

F Gouriveau[1], KV Heal[1], G Russell[1] and A Vinten[2]

[1]The University of Edinburgh, Crew Building, West Mains Road, Edinburgh, EH9 3JN, UK, E-mail: fabrice.gouriveau@ed.ac.uk; [2]The Macaulay Institute, Craigiebuckler, Aberdeen, AB15 8QH, UK

SUMMARY

This research evaluates the treatment efficiency and ecological value of two Scottish Constructed Farm Wetlands (CFW 1 and 2) receiving run-off from farmyards, fields, roofs, tracks and septic tank effluent. Water and sediment quality, vegetation and aquatic macroinvertebrates have been monitored since 2006. CFW 1 receives low pollutant loadings and releases water close to river standards except for nitrate and faecal indicators. It hosts a diverse flora and fauna due to its habitat heterogeneity and low contamination. CFW 2 achieves concentration reductions for all studied pollutants, but outflow concentrations during storm events are much higher than river water quality targets. Its ecological value is poor due to its high pollution and low habitat heterogeneity. To consistently achieve water quality targets, CFWs should comprise a sedimentation pond followed by several large enough and fully vegetated cells, to provide sufficient residence time and enhance treatment. Sustainable CFWs represent a significant investment for farmers, hence support schemes are recommended to promote their adoption.

INTRODUCTION

Farmyard run-off is a significant source of diffuse water pollution that results in public health concern, eutrophication, siltation and subsequent degradation of aquatic ecosystems. It causes many of the lochs and rivers in Scotland to fail the Water Framework and Bathing Water Directives and incurs considerable costs (D'Arcy *et al.*, 2000; Scottish Executive, 2005).

Farmers are required by law to collect, store and spread even lightly contaminated farmyard water, which may be impractical and costly and is sometimes not implemented in practice. Therefore, among other Best Management Practices, Constructed Farm Wetlands (CFWs), surface flow systems comprising a series of shallow vegetated ponds, are proposed for collection and treatment of farmyard run-off including run-off from yards, roofs, tracks, silage pits and vegetable washings (Kadlec and Knight, 1996; EPA, 2000; Dunne *et al.*, 2005). The legislative framework is being modified to clarify their status and ensure that they are properly implemented and managed. However, their long-term treatment efficiency, optimal design and cost-effectiveness are not well known and require further investigation.

The main aims of this study are to: 1) Evaluate the treatment performance of two CFWs and the link between performance and design; 2) Assess their ecological value; and 3) Propose guidelines for the design, construction and aftercare of sustainable CFWs.

MATERIALS AND METHODS

Research focuses on two Constructed Farm Wetlands (CFW 1 and 2) built in southeast Scotland and receiving run-off from farmyards, roofs, tracks, fields and small inputs from septic tanks.

CFW 1 was built in April 2004 in a naturally wet area on a mixed beef (120 cows) and arable farm, comprising five ponds (<1 m deep) in series separated by vegetated wetland areas, occupies ~0.9 ha and has a drainage catchment of ~35.5 ha including 1.8 ha of impermeable surfaces. It was designed following the Treatment Volume Approach to accommodate 2 x V_t of 1400 m^3, i.e. 2800 m^3. The vegetation comprises mainly *Phragmites australis*, *Juncus effusus*, *Typha latifolia*, *Nasturtium officinale* and grasses such as *Glyceria fluitans*, *Holcus lanatus* and *Agrostis stolonifera*.

CFW 2 was built in October 2004 in an improved pasture on a large dairy farm (400 cows). It comprises a 40 m swale draining into a single pond (2000 m^2, 1715 m^3) and collects run-off from 3.22 ha, of which 2.28 ha are impermeable, as well as septic tank effluent. It was designed to accommodate 5 x V_t of 340 m^3, i.e. 1700 m^3. The system was planted only sparsely and colonization is occurring slowly on the edges mainly by *Typha angustifolia*, *Phragmites australis*, *Phalaris arundinacea*, *Juncus effusus* and *Agrostis stolonifera*.

Rainfall, water levels and flow at the inlets and outlets of the CFWs are continuously monitored using raingauges, pressure transducers and ISCO flow meters respectively. Evaporation is estimated using evaporation pans and local meteorological data. Pollutant removal is assessed from water samples collected manually along the systems from inlet to outlet every month, or during storm events at the inlets and outlets using automatic water samplers. Water samples are analysed using standard methods for Biological Oxygen Demand (BOD_5), Total Suspended Solids (TSS), nitrate/nitrite (NO_3^-/NO_2^-), ammonium (NH_4^+), inorganic phosphorus (IP), faecal coliforms (FC) and *streptococci*, and total phosphorus (TP).

In addition, annual surveys of sediment depth and nutrient content are conducted by taking sediment cores at regular intervals along the CFWs between inlet and outlet. CFW habitat value is assessed three times a year using vegetation and macroinvertebrate surveys following the methodology recommended by Pond Action (Pond Action, 1998). Interviews with farmers and experts allow collection of technical and economic data on farm practices, pond construction and maintenance, and help to assess farmer acceptance of CFWs.

RESULTS AND DISCUSSION

Water Quality

CFW 1

Results so far (Table 1) show that CFW 1 receives only lightly contaminated run-off and discharges relatively low ammonium and phosphorus concentrations close to river water quality standards. This can be explained by the type of farm (mixed beef and arable), the low number of cattle, the measures already implemented to control pollution at source (e.g. roofing of feeding areas), but also by the fact that dirty water from the farm is not all intercepted.

Table 1: Concentrations of pollutants at inlet and outlet of CFW 1 and mean concentration reduction (combining monthly and storm samples)

	No. samples	Mean inlet	Max inlet	Mean outlet	Max outlet	Treatment Efficiency (concentration)
BOD^5 ($mg\ l^{-1}$)	(n≥14)	0.8	3	2.1	10	< 0 %
NH_4^+ ($mg\ l^{-1}$)	(n≥73)	0.6	3	0.3	1.5	50 %
NO_3- ($mg\ l^{-1}$)	(n≥73)	27.7	58	14.5	73	47 %
IP ($mg\ l^{-1}$)	(n≥73)	0.2	0.9	0.1	0.5	50 %
TSS ($mg\ l^{-1}$)	(n≥17)	65	703	8	55	88 %
FC (CFU/100 ml)	(n=2)	1800	3500	75000	> 150000	< 0 %

However, during heavy and prolonged rainfall, and especially in winter when large amounts of field run-off (from existing drains and surface run-off from the adjacent field) enter the CFW, NO_3^- is frequently present at high concentration (around 50 $mg\ l^{-1}$) throughout the system (Figure 1), while the outflow reaches 12 $l\ s^{-1}$. Indeed, residence time in winter is significantly shorter and lower temperatures may impede removal by bacterial denitrification. In summer, faecal indicators are also released in large quantities due to the presence of waterfowl (swans, ducks and moorhens).

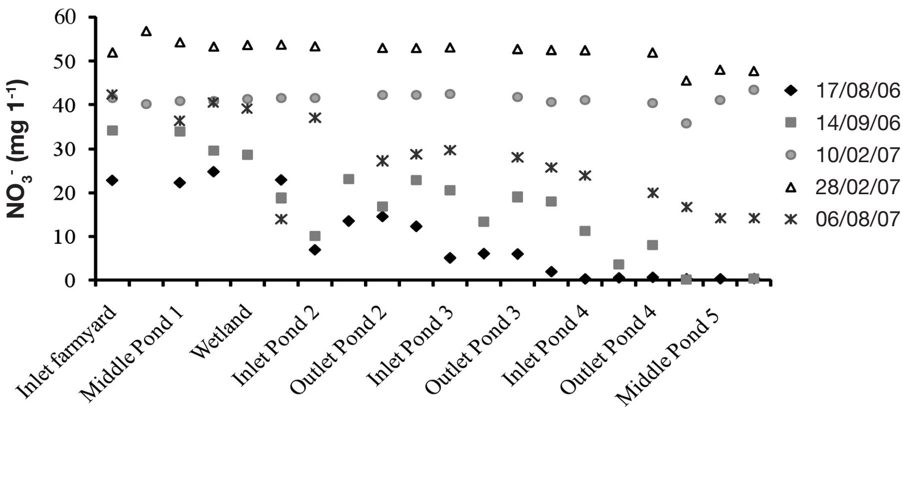


Figure 1: NO_3^- concentration in water samples taken along the length of CFW 1 from inlet to outlet in summer, autumn and winter

During storm events, elevated IP concentrations occur at the inlet, but concentrations are attenuated by the outlet of the system (Figure 2), and the treatment efficiency in terms of mass removal reaches 55%. Phosphorus removal may be explained by adsorption to sediment, uptake by plants and algae, and burial with organic and mineral matter.

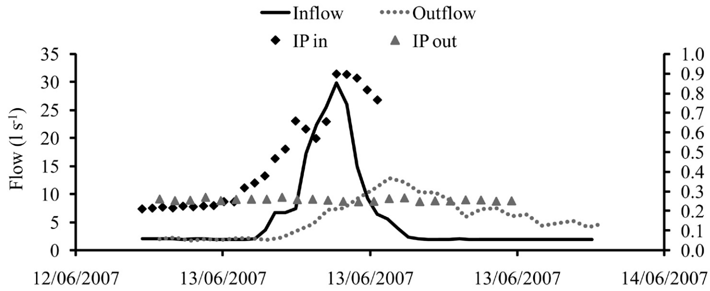


Figure 2: Flow and inorganic phosphorus concentration at the inlet and outlet of CFW 1 during a storm event in June 2007 (~21 mm rain in 15 h)

CFW 2

CFW 2, in contrast, intercepts large amounts of significantly contaminated farmyard run-off (Table 2), with an inflow reaching 40 l s^{-1} during heavy rainfall. This higher level of contamination is mainly explained by the size of the impermeable area draining into the CFW and by the high number of cattle at the steading. Despite its simple design and young age, it achieves substantial mean concentration reductions for all studied pollutants between the inlet and outlet. However, monthly and storm event sampling show that pollutants are often released at the CFW outlet at concentrations much higher than water quality targets while the outflow reaches 9 l s^{-1} (Table 2).

Table 2: Concentrations of pollutants at inlet and outlet of CFW 2 and mean concentration reduction (combining monthly and storm samples)

	No. samples	Mean inlet	Max inlet	Mean outlet	Max outlet	Treatment Efficiency (concentration)
BOD_5 (mg l^{-1})	(n≥18)	107	500	21	50	80 %
NH_4^+ (mg l^{-1})	(n≥90)	17	65.1	9.7	31.7	43 %
NO_3- (mg l^{-1})	(n≥90)	17.7	153	7.2	45	59 %
IP (mg l^{-1})	(n≥90)	1.7	9.2	1.4	2.5	18 %
TSS (mg l^{-1})	(n≥38)	239	1700	66.5	160	72 %
FC (CFU/100 ml)	(n=2)	104500	> 150000	6625	12250	93 %

BOD_5 levels at the outlet are slightly in excess of 20 mg l^{-1} on average, but never exceed 50 mg l^{-1}. NH_4^+ and NO_3^- concentrations at the outlet vary considerably, between 0 and 32 mg l^{-1} and 45 mg l^{-1} respectively, while IP varies between 0 and 2.5 mg l^{-1}.

No clear seasonal variation in water treatment performance is observable. Indeed, concentrations of the different pollutants within the system seem predominantly influenced by the extent and frequency of the inputs during storm events. However, in drier summer conditions there is more time for treatment between storm events, which results in higher treatment efficiencies and usually lower concentrations in the outflow.

Since macrophytes only cover a small part of the system (pond edges), the treatment observed may be mainly due to nitrification/denitrification, uptake by microorganisms and algae (blooms are observable in summer), sedimentation and adsorption, and organic matter degradation by macroinvertebrates such as cladocerans.

Ecological Value of CFWs

Macroinvertebrate and vegetation surveys conducted in 2006 and 2007 reveal strong differences in terms of ecological value between the two CFWs. Figure 3 shows the BMWP (Biological Monitoring Working Party) scores and interpretation in terms of water quality for the different ponds studied, including a 10-year old amenity pond which does not receive any farmyard run-off.

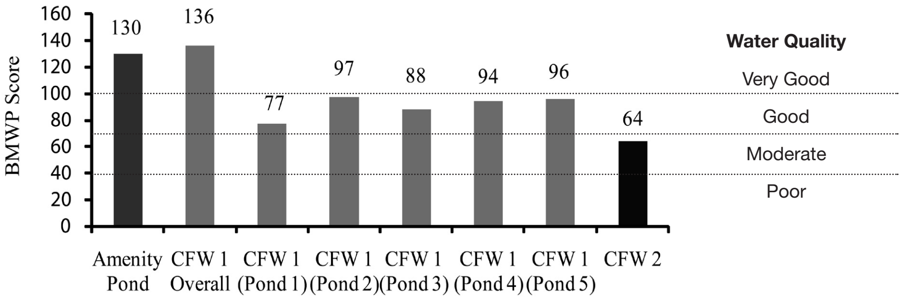


Figure 3: BMWP scores for CFW 1, CFW 2 and an Amenity Pond

CFW 1, as a whole, has a rather diverse invertebrate community, due to its habitat heterogeneity, open water areas and relatively low contamination. It hosts more than 30 families, among which are pollution sensitive taxa such as stoneflies (Nemouridae), dragonflies (Libellulidae) and damselflies (Coenagrionidae). The vegetation is relatively diverse, but diversity appears to be decreasing over time as *Phragmites australis* is progressively colonizing the system and outcompeting other species.

The ecological value of CFW 2 and the colonization rate by plants and macroinvertebrates are very low due to its high pollution level, considerable fluctuations in pH (from 7.2 to 9.5) and low habitat heterogeneity. The diversity is expected to increase over time but will probably never reach a status as good as in CFW 1.

CONCLUSIONS

CFW 1 receives a lightly polluted run-off, discharges a good quality effluent meeting river water standards, except periodically for nitrate and faecal coliforms, and has good ecological value. However, its effectiveness is limited because not all farmyard dirty water is conveyed properly to the CFW and inputs of field drainage and groundwater decrease residence time and treatment (e.g. denitrification), especially in winter. In addition, preferential flow occurs in the system due to improper levelling and biomass build-up, reducing the residence time and area of exchange between dirty water and biota. CFW 2 receives significantly polluted run-off, achieves some treatment but discharges a poor quality effluent which does not meet river water standards. It has only a moderate ecological value. The poor efficiency and ecological value are mainly due to the lack of vegetation, the small size of the system, the lack of subdivision into several cells, and consequently a very reduced residence time. However, shallow fully vegetated cells could be added to the system to enhance water treatment.

To consistently achieve water quality targets, CFWs should comprise a sedimentation pond followed by several wetland cells, shallow, large enough, fully vegetated and properly levelled to provide sufficient residence time and contact between water and biota. Although the cost of CFWs is lower than the cost incurred by conventional dirty water management options, they represent a significant investment for farmers. Appropriate external funds and support schemes are therefore recommended to promote more widely their adoption and proper management.

ACKNOWLEDGEMENTS

The authors are grateful to the following for funding and support: the David Kinloch Michie Studentship, the Torrance Bequest, the University of Edinburgh Development Trust, the Scottish Environment Protection Agency (SEPA), The Macaulay Institute, Scottish Agricultural College (SAC), Alan Frost, Marjan Van de Weg, Martha Lucia Gouriveau, Carole Christian, Alison Cole, Rob Briers, Andrew Gray, John Morman and all the farmers involved.

REFERENCES

D'Arcy BJ, Ellis JB, Ferrier RC, Jenkins A and Dils R (Eds.) (2000). Diffuse pollution impacts. The environmental and economic impacts of diffuse pollution in the UK. Terence Dalton Publishers, Suffolk, UK.

Dunne EJ, Culleton N, O'Donovan G, Harrington R and Olsen AE (2005). An integrated constructed wetland to treat contaminants and nutrients from dairy farmyard dirty water. Ecological Engineering 24, 221-234.

EPA (2000). Guiding principles for constructed treatment wetlands: providing for water quality and wildlife habitat. United States Environmental Protection Agency (EPA). Office of Wetlands, Oceans and Watersheds, Washington, DC, pp. 41.

Kadlec RH and Knight RL (1996). Treatment wetlands. CRC Press LLC, Lewis Publishers, Boca Raton, New York, USA, pp. 893.

Pond Action (1998). A guide to the methods of the National Pond Survey. Pond Action, Oxford, UK, pp. 16.

Scottish Executive (2005). Diffuse water pollution from rural land use. Paper 2005/35, December 2002. Scottish Executive Environment Group, Edinburgh, UK.

DIFFUSE NITRATE POLLUTION AND POLLUTION SWAPPING

W McAuley, D Reay, KV Heal and K Smith

I.A.E.S., The University of Edinburgh, EH9 3JN, UK, E-mail: lena.mcauley@ed.ac.uk

SUMMARY

Agricultural practices have and continue to increase the amount of leached N from farmland through intensification, the inefficient use of fertilisers and manure, the ploughing of established grasslands, and the removal of natural buffer systems - allowing for the undisturbed flow of nutrients from terrestrial to aquatic ecosystems.

Increased inputs of synthetic N to agroecosystems have four principle consequences: increased losses of NO_3^- from soils, concerns over human health, environmental degradation and increases in emissions of the greenhouse gas N_2O.

The environmental problem is complex as NO_3^- is rarely lost from soil in any great quantity directly from fertiliser application. NO_3^- losses occur post-harvest as soil microbes breakdown organic residues into forms of reactive N that are either washed out of soils and into the aquatic environment or released into the atmosphere.

Diffuse pollution from agriculture adversely affects 83% of polluted lochs and contributes to over 2000 km of polluted watercourses in Scotland. Defra estimates that 80% of rivers, 50% of lakes, 25% of estuaries and coasts and 75% of groundwaters are at risk of failing the Water Framework Directive (WFD) targets, principally due to diffuse pollution from agriculture. CAP reforms, Catchment Sensitive Farming (CST) and Diffuse Pollution Initiatives (DPI) aim to reduce diffuse pollution from farmland by encouraging farmers to follow best management practises and implement mitigation measures such as buffer strips and wetlands.

Field-edge buffer strips and natural or constructed wetlands are two remediation strategies that have great potential for mitigating diffuse nitrate pollution whilst improving water quality and enhancing biodiversity. The main processes of nitrate removal are denitrification and plant uptake. However microbial processes (denitrification and nitrifier-denitrification) produce the powerful greenhouse gas N_2O as either an end- or by-product suggesting the potential for high indirect N_2O emissions during NO_3^- mitigation.

EU policy generally addresses NO_3^- pollution of water however little attention has been paid to the potential problem of 'pollution swapping' and its potential impact on climate change. The increased land coverage of buffer strips and wetlands throughout the UK as a result of mitigation strategies could become an important indirect source of N_2O that needs further investigation.

The effectiveness of buffer strips and wetlands at intercepting diffuse N pollution and the extent of pollution swapping are being investigated at a field site in North East England near Newcastle.

Preliminary results reveal high N pollution, with NO_3^- concentrations exceeding the water quality standard of 50 mg NO_3 l^{-1} and reaching peak concentrations in excess

of 100 mg l^{-1}. Decreased NO_3^- concentrations and nitrous oxide emissions in both the wetland and buffer strips occurred during the growing season. However as yet there is no clear divergence in N_2O emissions between the control and saturated buffer strip. This is anticipated to occur at the end of the growing season with the saturated buffer strip expected to show greater N_2O emissions due to the supply of NO_3^- rich stream water.

ACCOUNTING FOR EXTERNALITIES FROM UK AGRICULTURE

D Moran[1], A McVittie[1], I Spencer[2] and V Caldwell[2]

[1]SAC, West Mains Road, Edinburgh, EH9 3JG, UK, E-mail: Dominic.Moran@sac.ac.uk; [2]Jacobs, 27 Abercromby Place, Edinburgh, EH3 6QE, UK

SUMMARY

Agricultural production generates positive and negative environmental impacts; so-called externalities. There is interest from government in developing a set of environmental accounts for agriculture that both quantify and place economic values on these externalities. This paper reports the results of a recent research project that sought to refine previous work in developing a set of environmental accounts for agriculture in the UK. The negative impacts from agriculture arise mainly from emissions to air and have a total value of £4.7 billion. Positive environmental impacts total £1.2 billion arising mostly from the provision of landscapes, habitats, and biodiversity. However, values for broad agricultural landscape remain a significant data gap resulting in an underestimate of the true scale of the positive impacts. These flows of positive and negative impacts result in a net environmental cost of £3.5 billion, which compares to gross value added from agriculture of £4.9 billion in 2006.

INTRODUCTION

While agriculture's share of UK Gross Domestic Product is small, the sector exerts a significant influence on UK land use. Agricultural activity also impacts on our environment in a variety of ways, such as through the active management of agricultural landscapes and their wildlife; generation of waste; emissions of pollutants and greenhouse gases; and abstraction and use of water. The environmental accounts for agriculture are a framework for measuring and valuing the positive and negative impacts of agriculture on the environment. When viewed alongside the conventional sector accounts, they help to provide a clearer picture of agriculture's overall impact on welfare, including allowance for its impacts on income in other sectors. This paper presents the findings from a review and update of the framework for the environmental accounts for agriculture as developed by Eftec/IEEP (2004) that was recently undertaken by Jacobs and SAC (2007). The review stage of the project sought to consider the outstanding methodological and conceptual issues that arise in development of an environmental accounts framework for agriculture. We then proceeded to update the relevant externality calculations.

REVIEW OF THE 2004 FRAMEWORK FOR THE ENVIRONMENTAL ACCOUNTS FOR AGRICULTURE

The review of the 2004 framework, and work to refine it, showed that the gap between theory and data for environmental accounts remains quite wide, although some key improvements were possible.

In the absence of a standardised set of physical data and damage cost curves for environmental impacts, calculations of environmental impacts from agriculture are frequently simplistic, requiring many potentially critical assumptions. As a result, some estimates have a high degree of uncertainty; and some significantly more than others. Further, economic and physical figures may change considerably as new data emerge. At present, adjustments are not based on consensus valuation figures as there is disagreement about measurements. Careful consideration should be given to publication of any sector environmental accounts beyond demonstrative and advocacy purposes. There are many useful ways in which the environmental accounts can be developed in the future. Whilst the results from sector level environmental accounts are useful and can be practically employed to analyse the contribution of agriculture to environmental impacts, care should be applied in the treatment and use of the figures.

A wide range of modifications have been made to the 2004 framework, representing substantial progress in terms of the coverage of the impacts included in the environmental accounts, the regional data coverage, and various improvements to the physical and economic data used within the calculations.

KEY CONCEPTUAL ISSUES UNDERPINNING THE ENVIRONMENTAL ACCOUNTS FOR AGRICULTURE

We clarified and moved forward some of the key theoretical and conceptual issues that arise in the compilation of the 2007 environmental accounts for agriculture. Crucially, we clarify the determination of the appropriate counterfactual against which adjustments in the environmental accounts are to be made, and assess the extent to which estimates of impacts derived from economic valuation studies are consistent with the accounting counterfactual of 'no activity' and in this case, 'no agriculture'.

Conventional economic accounts that are a part of the national income account typically assume a counterfactual of no economic activity. In other words, industry produces output, and the value of that output measures the extent of wealth that would be lost in a 'no industry' counterfactual. Under a counterfactual of 'no agriculture', for a given area of land there would strictly speaking be no environmental outcomes within the accounts. For example, no landscape, biodiversity or habitats from which any environmental benefit could flow with respect to the agricultural accounts. Whether these occur to some extent under an alternative land use is not of concern in the agricultural environmental accounts. Rather, these benefits would be part of the sectoral environment accounts for the alternative land uses.

The concept of a 'no agriculture' counterfactual nevertheless has generated concern about the type of land use that would result if agriculture did not exist. Importantly, we conclude that this concern is misplaced. This is because (as with conventional production accounts) the objective of the accounting activity is not equivalent to a comparison of scenarios. It is simply to measure the current level of the positive and negative stocks or flows attributable to an activity. For example, conventional sector accounts would measure the value or stocks and/or flows for an economic activity in monetary terms. If a factory ceased operations then, eventually, an alternative activity would take its place – the important thing for the accounts is the value of the factory's activity, rather than a speculative comparison between what activity could be there if it ceased to exist.

The crucial question therefore is: how well can the environmental accounts represent the value of the environmental stocks and flows which we apportion to agriculture? Theoretically, most of the methodologies for valuing impacts covered in the 2007 framework are conceptually consistent with the accounting counterfactual. For example, there are few conceptual challenges to valuation against the accounting counterfactual where:

a) total environmental impacts are quantifiable in physical terms;
b) apportionment of impacts to agriculture is quantifiable, and;
c) economic values use impact thresholds which can reflect the accounting counterfactual.

The most straightforward example of this is for an air emission with linear impacts (i.e. each unit of emissions causes a similar level of damage). The concept of valuing total agricultural air emissions for an accounting purpose does not cause conceptual issues. However, the focus of the above question inevitably leads to the most significant positive flows from agricultural, i.e. the positive flows from agricultural landscapes habitats and biodiversity. Specifically, the issue around how valuation studies can capture the value of a stock or flow against the accounting exercise's 'no activity' counterfactual, without introducing an alternative scenario as a baseline against which to measure relative value. This is a key observation of Swanwick *et al*. (2007), a scoping study on agricultural landscape valuation. The problem being that willingness to pay for current stocks and flows of landscapes habitats and biodiversity, need to be determined versus a believable alternative or baseline scenario. The idea that 'nothing' would exist in place of current stocks is hard to communicate within a credible valuation scenario. Therefore valuation studies for these features must assume a more tangible alternative land use scenario and are therefore not easily transferable to an accounting framework.

Consequently there is considerable imbalance within the accounts between the relative conceptual straightforwardness of capturing key negative impact valuations such as air emissions, in their entirety, versus the difficulty of capturing only part of the key positive flows. The conceptual difficulty around rectifying the imbalance leads to suggestions that the accounts are fundamentally undermined. Whilst we recognise this difficulty, we focus on what could be done to progress towards a more balanced set of accounts.

Working from the principle of a 'no agriculture' counterfactual, the important issue is how stocks or flows are apportioned to agriculture. In most cases, this is relatively straightforward, with environmental impacts that can be unequivocally tied to agriculture. In other cases, agriculture is only one contributor to a particular environmental impact, such as water pollution, and therefore apportionment data is required to disentangle agriculture's impact.

A further contentious issue relating to application of the accounting counterfactual is the extent to which landscapes, biodiversity and habitats are attributable to agriculture. Some landscape, habitats and/or biodiversity are highly valued and are of a direct consequence of agricultural management. This evidence and wider evidence that the agricultural accounts should include the values attributable to landscapes, habitats and biodiversity assets is presented and explored in Jacobs and SAC (2007). In doing so, that report sets out and clarifies the theoretical basis for attributing these assets and the flows from them to agriculture.

IDENTIFICATION OF ADDITIONAL AVAILABLE VALUATION STUDIES AND ECONOMIC DATA

A review of the data used in 2004 was conducted which identified a wide range of gaps and weaknesses, mainly due to data limitations at the time. New sources of economic and physical data were reviewed and considered for use in the 2007 framework. In most categories of environmental impact, data were updated or the source of the data was revised. Crucially, new categories of impact, such as transitional waters and soil carbon sequestration were introduced.

New economic data were found in many areas, including water quality and quantity, air quality and greenhouse gas emissions. The most significant additional economic data are those for air quality emissions, in particular ammonia (NH_3). The economic values (damage cost) for NH_3 (the most significant agricultural air quality pollutant) were revised upwards by a factor of around 115 (from £87 to £10,000 per tonne emitted) based on recommendations of recent work on NH_3 damage costs (Entec, 2007) and detailed discussion with Defra. This alone has a very significant impact on the accounts, amounting to a negative flow of nearly £3b. Whilst these revisions were discussed carefully with Defra, caution is advised in the use of these figures and further work is due to be undertaken by Defra to refine the NH_3 damage costs. This refinement would be in respect of both the physical damage arising from NH_3 and the basis for the valuation of that damage. Damage costs for other air emissions (nitrous oxide and sulphur dioxide) were also revised upwards significantly.

The most significant gaps remain in the landscapes, habitats and biodiversity category. This is the largest positive environmental contribution that agriculture makes. Careful consideration has been given to the potential to develop the approach to this category of impact versus the 2004 approach, but data limitations make this particularly difficult. Consideration was also given to making this section of the framework consistent with categories of landscapes indicated within other ongoing Defra work on landscapes valuation (Swanwick *et al.*, 2007), although this would further increase the data constraints and challenge the robustness of any calculations.

KEY RESULTS OF THE 2007 FRAMEWORK

The key differences versus the 2004 framework are:

- The negative flows from air emissions are far larger than those in the 2004 framework. This is due to the updated (and much higher) economic cost value for ammonia, sulphur dioxide and oxides of nitrogen.
- New environmental impact categories have been added which increase the negative flows i.e. transitional waters, eutrophic lakes and soil carbon sequestration (which is negative and therefore a net emissions).
- A new benefit flow has been introduced, i.e. agriculture as a waste sink.

The results in Tables 1 and 2 below are headline figures. More detailed tables and information on sources are available in Jacobs and SAC (2007). Table 3 presents the summarised positive and negative environmental flows from agriculture. It also presents a combined 'net' figure. Careful consideration was given to whether, given the uncertainties inherent in the valuation of key flows from agriculture, it is misleading to present a combined 'net' impact figure. The conclusion is that a net figure itself does not represent a complete picture of the net flows from agriculture due to data gaps and key uncertainties. These mainly relate to underestimation of positive flows from landscapes, habitats and biodiversity due to incomplete physical and economic data, versus complete physical data for the most significant negative impacts, i.e. air (and particularly ammonia) emissions.

Presentation of this net figure alongside key caveats was deemed the most appropriate means of avoiding misinterpretation of the net figure, which can readily be calculated (and could be expected to be calculated) should we have refrained from doing so.

The net figure for environmental flows from agriculture therefore should not be interpreted or represented as an accurate estimate of net welfare impacts. Positive impacts are expected to be significantly underestimated by the current valuations. This reflects the need to develop better physical and economic information on agricultural landscape types.

Table 1: Negative environmental adjustments to UK agriculture sector accounts (2007, £m)

Impact category	Negative accounting adjustment	E	W	S	NI	Total UK
Rivers	Rivers of less than 'good' quality due to agricultural diffuse pollution	44.58	0.81	11.54	4.62	61.56
Lakes	Eutrophication in lakes due to agricultural diffuse pollution	26.55		Not Available		26.55
Drinking water	Removal of contaminants	96.87		Not Available		96.87
Pollution Incidents	Point source pollution events due to agriculture	0.27		0.05	0.21	0.53
Bathing waters	Bathing waters failing to meet FIO standards	7.95	0.75	2.24	0.17	11.10
Abstraction	Value of water abstracted	36.73		16.41	8.52	61.66
Flooding	Apportionment of flood damage and prevention costs	Not Available				233.80

Table 1 continued

Climate change	Value of greenhouse gas emissions (net of sequestration)	772.53	149.84	278.86	113.34	1,314.56
Air quality	Value of air quality pollutant emissions	1,868.79	308.83	371.25	330.82	2,879.70
Soil	Apportionment of soil erosion damage costs			Not Available		9.41
Waste	Value of waste treated off-site			Not Available		8.10
					TOTAL	**£4,703.83**

Table 2: Positive environmental adjustments to UK agriculture sector accounts (2007, £m)

Impact category	Positive accounting adjustment	E	W	S	NI	Total UK
Landscape and habitats	Value of area of habitats	374.91		345.05	45.69	853.51
Linear features	Value of length linear features	1.67		0.14	0.63	2.44
Biodiversity	Value of farmland bird species			Not Available		307.44
Waste	Benefit of avoided sewage sludge incineration	33.83		1.33		35.16
					TOTAL	**£1,198.55**

Table 3: Summary of total environmental adjustments to agricultural accounts (2007, £m)

Summary results	UK
Total Negative Flows	£4,703.83
Total Positive Flows	£1,198.55
Net	**-£3,505.28**

ACKNOWLEDGEMENTS

This work was jointly funded by Defra, the Welsh Assembly Government, the Scottish Government and the Department of Agriculture and Rural Development (Northern Ireland).

REFERENCES

Eftec and IEEP (2004). Framework for Environmental Accounts for Agriculture. Final Report to Defra, Department of Agriculture and Rural Development (Northern Ireland), Scottish Government (RERAD) and Welsh Assembly Government.

Entec (2007). Ammonia Damage Costs: Final Report to Defra.

Jacobs and SAC (2007). Environmental Accounts for Agriculture. Final Report to Defra, Welsh Assembly Government, Scottish Government and Department of Agriculture and Rural Development (Northern Ireland).

Swanwick C, Hanley N and Termansen M (2007). Scoping Study on Agricultural Landscape Valuation. Final Report to Defra. Universities of Sheffield, Stirling and Leeds.

CONTROL OF NUTRIENTS WITH INTEGRATED CONSTRUCTED WETLANDS

A Mustafa[1], M Scholz[1], R Harrington[2] and P Carroll[3]

[1]Institute for Infrastructure and Environment, School of Engineering and Electronics, The King's Buildings, The University of Edinburgh, EH9 3JL, UK, E-mail: m.scholz@ed.ac.uk; [2]National Parks and Wildlife, Department of Environment, Heritage and Local Government, The Quay, Waterford, Ireland; [3]Waterford County Council, Kilmeaden, Waterford, Ireland

SUMMARY

Integrated constructed wetlands (ICW) treat farmyard runoff rich in nutrients. A performance assessment of ICW showed that these systems were successful in consistently removing nutrients even after entering the seventh year of operation. For most of the years, the annual mean effluent concentrations for ammonia-nitrogen, nitrate-nitrogen and molybdate reactive phosphorus were <1.5, <2.0 and 1 mg/l, respectively, indicating that the recorded nutrient concentrations after ICW treatment were in agreement with Irish urban wastewater standards for discharge to sensitive waters. Molecular microbiological techniques were employed to assess the presence of ammonia-oxidising and denitrifying organisms in sediment and litter samples collected from representative ICW sites. The number of denitrifying bacteria detected in different ICW systems was higher than the number of ammonia oxidising bacteria. The presence of anoxic conditions provided conditions suitable for denitrification. The overall benthic-microbial community contained sufficient denitrifying bacteria. Litter and sediment components of ICW systems supported denitrification.

INTRODUCTION

The Integrated Constructed Wetland (ICW) Initiative was developed with an approach that endeavoured to achieve 'water treatment', 'landscape fit' and 'biodiversity enhancement' targets by an innovative wetland design methodology. Most systems were commissioned in 2001 to treat farmyard runoff rich in nitrogen and phosphorus, which potentially posed a serious threat to the receiving water bodies. The conventional practice in Ireland is land spreading of farmyard dirty water, and this method has resulted in increased levels of nitrogen and phosphorus in surface and ground waters (Healy *et al.*, 2007). The ICW concept is founded on the holistic use of land to control water quality. These systems are areas of land-water interface that form an integral part of the environmental and ecological structure of the landscape (Dunne *et al*., 2005; Scholz *et al.*, 2007). They act as buffer areas that control the transfer and storage of farmyard dirty water rich in nutrients.

The main characteristics of ICW such as shallow water depth, emergent vegetation and the use of *in situ* soils mimic those found in natural wetland ecosystems. Scholz *et al.* (2007) reported on the detailed concept of these synergistic, robust and sustainable systems.

The contaminated effluent within ICW is treated through various physical, chemical and biological processes involving plants, micro-organisms, water, soil and sunlight. The extent of treatment by ICW depends upon the wetland design, microbial community and types of plants involved. Water quality improvements are predominantly caused by bacteria (Ibekwe *et al*., 2003). The main processes for nitrogen transformation in constructed wetlands are nitrification and denitrification. Both nitrification and denitrification are processes mediated by microorganisms (Scholz, 2006).

Microbes play an important role in the nutrient transformation and removal processes within ICW systems. The diversity of microorganisms in the wetland environment is likely to be critical for their proper functioning and maintenance (Ibekwe *et al.*, 2003). In ICW systems, the litter from decaying macrophytes provides surface area for attachment of biofilms, and is therefore important for microbial processes such as the transformation of nutrients within wetlands. For most aquatic systems, the bulk biological conversions are undertaken by microorganisms immobilized in sediments (Scholz, 2006). Therefore, sediment and litter components play a vital role in supporting these microbial mediated processes. Samples should be collected to gain an insight into the microbial transformations taking place in removing nutrients from ICW. The aim of this investigation was therefore to characterise bacterial communities present in sediment and litter components of ICW. The objectives were (a) to assess the long-term performance of these systems; (b) to identify the presence of ammonia-oxidizing bacteria; and (c) to identify the presence of denitrifying bacteria.

MATERIALS AND METHODS

Study Sites Within the Case Study Catchment

The study sites are located in the Waterford County (south-east of Ireland). The ICW 3, 9 and 11 were designed and constructed between 1999 and 2000 to intercept and treat farmyard dirty water from three representative working farms located in Annestown-Dunhill catchment (Table 1). The site suitability was assessed by identifying different indicator variables such as good agricultural practice, site access and historical data availability. Components of farmyard dirty water discharged to the wetlands were variable and the runoff typically consisted of yard and dairy washings, rainfall on open yards and farmyard roofed areas along with silage and manure effluents. All ICW were in operation for at least seven years.

Table 1: Site characteristics of farms and corresponding integrated constructed wetland (ICW) systems in Waterford, Ireland

ICW no.	Farm type	Farmyard area (m^2)	Dairy washings (cow number)	Effective ICW area (m^2)	Number of ICW cells
3	Dairy	5400	Yes (50)	10288	5
9	Mixed	4800	Yes (55)	7964	4
11	Dairy	5000	Yes (77)	7676	4

Sampling and Analytical Methods

Grab samples for each wetland cell inlet and outlet were taken on an approximately fortnightly basis. Water analysis was conducted at the Waterford County Council water laboratory using predominantly American Public Health Association standard methods (APHA, 1998) unless stated otherwise.

It important to better understand nutrient removal processes in different parts and components of selected ICW to improve their design. Therefore, sediment and litter samples were collected in April and May 2007 from three different representative wetlands (ICW 3, 9 and 11), frozen and sent off to Linköping University (Sweden) for subsequent molecular microbiological analysis (extraction, deoxyribonucleic acid (DNA), polymerase chain reaction (PCR), 16S ribosomal deoxyribonucleic acid (rDNA) fragments and gel electrophoresis). Samples from ICW 3 and 9 were taken on 24 April 2007, while samples for ICW 11 were taken on 8 May 2007.

Deoxyribonucleic Acid Extraction

Sediment and litter samples were subjected to DNA extraction using a FastDNA® SPIN kit for Soil (Bio 101, Inc., La Jolla, CA, USA). Samples (0.25 g) were suspended in a sodium phosphate buffer supplied with the FastDNA® SPIN kit as stipulated by the manufacturer, and homogenised for 180 s with a hand-held blender (DIAX 900 Homogeniser Tool G6, Heidolph, Kelheim, Germany).

Deoxyribonucleic acid was extracted from soil samples by bead beating, a procedure in which soil aggregates are disrupted and bacterial cells are lysed mechanically. Bead beating was extended to 3 × 30 s to achieve good homogenization of the samples. The subsequent centrifugation was prolonged to 2 × 5 min and the centrifugation after washing with SEWS-M, a salt and ethanol wash solution (Qbiogene, Inc., USA), was extended to 5 min. The extracted DNA was stored at -20ºC.

Polymerase Chain Reaction

The ammonia-oxidising bacterial community was investigated using group-specific PCR primers targeting the 16S ribosomal ribonucleic acid (rRNA) gene, while the denitrifying bacterial community was assessed using the functional gene nitrous oxide reductase (*nosZ*), which is the gene for the terminal enzyme in denitrification. A primer is a nucleic acid strand that serves as a starting point for DNA replication, and is required because most DNA polymerases (i.e. enzymes that catalyze the replication of DNA) cannot synthesise a new DNA strand from ‘scratch’. Ribosomal ribonucleic acid is one of the three major types of ribonucleic acid (RNA), part of ribosomes and composed of RNA of different sizes such as 5S, 16S and 23S in prokaryotes. Ribosomal ribonucleic acid and the genes that encode them are ideal biomarkers, which are molecules containing information concerning the evolutionary identity of organisms.

The extracted DNA from all samples was diluted 10-fold to avoid inhibition of the PCR by humic substances. This was determined by testing for different dilution ratios. Polymerase chain reaction amplification was undertaken using forward and reverse primers (CTO189fA/B -GC; CTO189fC-GC and CTO654r) for ammonia-oxidizing bacteria. The PCR was performed on a PTC-100TM thermal cycler (MJ Research Inc., San Francisco, CA, USA) in a 50µl mixture (Sundberg *et al.*, 2007).

The forward and reverse primers (nosZF and nosZ1622R-GC) targeting the *nosZ* gene were used in the next PCR. The PCR was performed on a PTC-100TM thermal cycler in a 50 µL mixture including 1.33 U of Taq polymerase, 5 µl of the supplied buffer (1.5 mM MgCl2; Roche Diagnostic GmbH, Manheim, Germany), 200 µM each nucleotide, 0.125 µM for each primer, 600 ng µl^{-1} of bovine serum albumin and 2 µl of the DNA template (Sundberg *et al.*, 2007).

Agarose Gel Electrophoresis

The PCR products of DNA extraction and PCR reactions were examined by agarose gel electrophoresis. The agarose was melted by heating the mixture (agarose plus buffer) and then poured into the agarose gel casting tray. The gel was covered with an electrophoresis buffer before running electrophoresis. The electrophoresis buffer was the same as the one used to prepare the agarose.

The PCR products and dye supplied with the DNA extraction kit (2μl of dye and 4μl of PCR products) were placed into the loading wells formed by the gel comb. The first well of each row was loaded with 2μl of Gene Ruler (1 kb DNA ladder; 1000 base pairs for ammonia-oxidizing bacteria and nitrous oxide reductase nosZ) and 4μl of distilled water. The electrophoresis was run for 40 min at 120 V (Owl Scientific, Inc., Woburn, MA, USA). The gel was then placed in ethidium bromide solution (immersed for 15 min) located in the fume cupboard and washed subsequently with tap water. The ethidium bromide stained gel was then visualized by UV illumination.

RESULTS AND DISCUSSION

Performance of Three Distinct ICW

To assess the performance of three representative ICW in nutrient removal, ammonia-nitrogen (NH_4-N), nitrate-nitrogen (NO_3-N) and molybdate reactive phosphate (MRP) concentrations at the inlet and outlet were used to calculate the overall removal efficiency (Table 2). Removal of nutrients is very good. The mean effluent NH_4-N and MRP concentrations are less than 1 mg/L for ICW 9 and 11. The efficiency of removal of NH_4-N for the three integrated constructed wetlands is excellent with ICW 3, ICW 9 and ICW 11 having 97.3%, 98% and 99% respectively. The NO_3-N removal efficiency is good with ICW 3, ICW 9 and ICW 11 having 69.4%, 72.4% and 74%. ICW 9 and 11 have excellent MRP removal efficiency of 94.5 and 91.8% respectively while ICW 3 has a good efficiency of 77.4%. In general, ICW 9 and 11 are more efficient systems as compared to ICW 3. The nutrient concentrations after ICW treatment are in agreement with Irish urban wastewater standards for discharge to sensitive waters.

The long-term (August 2001-August 2007) water quality monitoring suggests that integrated constructed wetlands are efficient systems for nutrient removal from agricultural wastewater. In contrast to pond systems these systems are more robust. ICW 9 and 11 showed very good removal efficiency as compared to ICW 3 because of more dense vegetation stands in the former systems as compared to the latter.

Table 2: Mean inlet and outlet nutrient concentrations and removal efficiency, and overall removal efficiency for three distinct ICW sites, August 2001-2007

ICW no.	NH_4-N (mg/l)	NO_3-N (mg/l)	MRP (mg/l)	Overall nutrient removal efficiency
3	51.0/1.35 (97.3%)	3.30/1.01 (69.4%)	15.2/3.43 (77.4%)	81.4%
9	30.5/0.59 (98%)	4.79/1.32 (72.4%)	8.57/0.47 (94.5%)	88.3%
11	39.6/0.37 (99%)	3.81/0.99 (74%)	11.5/0.94 (91.8%)	88.3%

Comparison of Ammonia-oxidising and Denitrifying Microorganisms

In comparison to ammonia-oxidizers, the denitrifiers are more abundant in most of the litter and sediment samples collected from the three ICW sites. Since the nitrate concentrations within the ICW systems were low, it was likely that oxygen and nitrate have served as electron acceptors in the supporting layer of the ICW bottom, and this might have promoted the growth of denitrifying bacteria. Each ICW system contains denitrifying bacteria, but they are present in varying quantities. For example, ICW 11 has lower denitrifying bacteria than ICW 3 and ICW 9. Samples analyzed from ICW 3 and ICW 9 do not indicate the presence of ammonia-oxidising bacteria (Table 3).

Table 3: Relative presence or absence (%) of bacteria related to ammonia-nitrogen and nitrate concentrations in ICW

ICW no.	Ammonia oxidising bacteria	Denitrifying bacteria	NH_4-N (mg/L)	NO_3-N (mg/L)
3	0	72.5	13.9	0.27
9	0	80.2	5.20	1.69
11	26.6	53.2	9.73	1.73

Ammonia-oxidizing bacteria were present at ammonia-nitrogen concentrations between approximately 5 and 20 mg/L. Denitrifying bacteria were present at nitrate-nitrogen concentrations between 0.1 and 4.5 mg/L. In comparison to ammonia-oxidising bacteria, more denitrifiers were present in different ICW systems. Ammonia-oxidising bacteria were found in samples collected from ICW 11 only.

Concerning denitrifying bacteria, ICW 3 has lower denitrifying bacteria numbers than ICW 9 but higher numbers than ICW 11. The high numbers of denitrifying bacteria in ICW 9 were linked to higher concentrations of nitrates present in this wetland system as compared to ICW 3. Also the decaying plants contributed organic matter that became a source of carbon and energy for denitrifying bacteria and ICW 9 had a higher plant cover density than ICW 3.

There was a reduced availability of organic matter at the bottom of ICW 11 leading to decreased numbers of heterotrophic bacteria and consequently created conditions in which ammonia-oxidizing bacteria proliferate. Also the decaying plants contributed organic matter that became a source of carbon and energy for denitrifying bacteria and ICW 9 had a higher plant density than ICW 11.

The organic material present in the ICW systems has an indirect impact on the bacterial community. The litter on top of the sediment limits the diffusion of oxygen to lower sediment layers, creating anoxic conditions and hence making conditions favourable for denitrification.

ACKNOWLEDGEMENTS

The authors acknowledge the financial support by the Federation of European Microbiological Societies for the molecular work.

REFERENCES

American Public Health Association (APHA) (1998). Standard Methods for the Examination of Water and Wastewater, 20th edition, APHA-AWWWA-WEF, Washington DC, USA.

Dunne EJ, Culleton N, O'Donovan G, Harrington R and Olsen A E (2005). An Integrated constructed wetland to treat contaminants and nutrients from dairy farmyard dirty water. Ecological Engineering 24 (3), 221-234.

Healy MG, Rodgers M and Mulqueen J (2007). Treatment of dairy wastewater using constructed wetlands and intermittent sand filters. Bioresource Technology 98, 2268-2281.

Ibekwe AM, Grieve CM and Lyon SR (2003). Characterisation of microbial communities and composition in constructed dairy wetland wastewater effluent. Applied and Environment Microbiology 69 (9), 5060-5069.

Knight RL, Payne Jr WE, Borer RE, Clarke Jr RA and Pries JH (2000). Constructed Wetlands for livestock wastewater management. Ecological Engineering 15 (1-2), 41-55.

Scholz M (2006). Wetland Systems to Control Urban Runoff. ISBN 0-444-52734-6, Elsevier, Amsterdam, The Netherlands.

Scholz M, Harrington R, Carroll O and Mustafa A (2007). The Integrated Constructed Wetlands (ICW) Concept. Wetlands 27 (2), 337-354.

Sundberg C, Tonderski K. and Lindgren P-E (2007). Potential nitrification and denitrification and the corresponding composition of the bacterial communities in a compact constructed wetland treating landfill leachates. Water Science and Technology 56 (3), 159-166.

THE TESTING OF 'PROMPT VALUES' FOR ASSESSING SITE SPECIFIC SOIL QUALITY

F Nicholson[1], D Barraclough[2], T Boucard[2], B Chambers[1], R Creed[2], K Dobbie[3], G Merrington[4], R Parkinson[5], I Schoeters[6], E Smolders[7], J Stuart[8] and W Piatkiewicz[9]

[1]ADAS, Gleadthorpe, Meden Vale, Mansfield, Nottinghamshire, NG20 9PF, UK; [2]Environment Agency, Howbery Park, Wallingford, Oxon, OX10 8BD, UK, E-mail: tatiana.boucard@environment-agency.gov.uk; [3]SEPA, Heriot Watt Research Park Avenue North, Edinburgh, EH14 4AP, UK; [4]Watts and Crane Associates, 23 London Street, Faringdon, Oxon, SN7 7AG, UK; [5]University of Plymouth School of Biological Sciences, Drake Circus, Plymouth, Devon, PL4 8AA, UK; [6]ECI - European Copper Institute, Av. de Tervueren 168 box 10, 1150 Brussels, Belgium; [7]Katholieke Universiteit Leuven, Kasteelpark Arenberg 20, 3001 Heverlee, Belgium; [8]Department for Environment Food and Rural Affairs, 17 Smith Square, London, SW1P 3JR, UK; [9]Zinc Information Centre, Wren's Court 56 Victoria Road, Sutton Coldfield, West Midlands, B72 1SY, UK

SUMMARY

Numerical limit values are used to protect soil quality in a number of regulatory and advisory regimes. However, there are few instances where these values have been validated to determine whether they afford adequate levels of protection. As part of a risk-based approach to regulation, it is also important to determine where there may be over or under protection. Soil metal 'prompt values' in existing regulations and advisory documents/regimes were tested against field experimental soil biology/crop quality data (e.g. soil microbial activity, microbial community size, grain Cd concentrations) to assess their performance in protecting soil functions. This paper outlines the process used to test the performance of existing soil metal 'prompt values' and outlines some initial results. Initial analysis of the results indicates that current regulatory regimes offer a range of protection for soil microbial activity and soil health.

INTRODUCTION

The EU Thematic Strategy for Soil Protection (EC, 2002), Defra's "First Soil Action Plan for England: 2004-2006" (Defra, 2004) and the Environment Agency's Soil Strategy (EA, 2007a) have all highlighted the need for information on the status of and recent changes in soil properties to ensure the long-term protection of soil quality and fertility. The UK Soil Indicators Consortia (UKSIC) is a group of public stakeholders who are developing a set of soil indicators and a soil monitoring scheme to meet this need for the UK (www.defra.gov.uk/environment/land/soil/research/indicators/consortium/index.htm).

Potential indicators for key soil functions (e.g. environmental interaction, food and fibre production) have been identified and 'change' or 'prompt values' derived (EA, 2006). The purpose of these values is to provide a 'prompt' when a level of change is considered to be important in terms of a soil's fitness for a specific use or function. If the 'prompt value' is exceeded, there should be a move to another level within a tiered risk assessment process, which may require the reinterpretation of data or the collection of more data. 'Prompt values' associated with national soil quality monitoring should provide a means of focusing future effort on those soils most at risk of degradation. To assess whether 'prompt values' are set at an appropriate level requires testing against soils datasets independent of those from which they were derived.

This paper describes the process used to test the performance of soil metal 'prompt values' used in existing and proposed regulatory regimes to protect selected soil functions using data collected from 'real-world' scenarios (e.g. microbial activity, microbial community size).

MATERIALS AND METHODS

Selection of Regimes for Testing

The regimes selected for testing were all either in existing use or proposed for use in the UK. All relied on a comparison of site specific measurements of soil metal concentrations with a numerical value, considering to a greater or lesser extent other soil physico-chemical conditions (e.g. soil pH).

The Sludge (Use in Agriculture) Regulations aim to protect the environment, in particular the soil, when sewage sludge is used in agriculture (SI, 1989). The Regulations state that sludge use should "prevent harmful effects on soil, vegetation, animals and man", that account is taken of the nutrient needs of the plants and that the quality of the soil and of the surface and groundwater is not impaired. The metal limit values for sludge amended soils are in Table 1.

Table 1: Maximum permissible total metal concentrations (mg/kg dry weight) in sludge amended soils (SI, 1989; DoE, 1996)

	Soil pH			
	5.0<5.5	5.5<6.0	6.0<7.0	>7.0
Zinc (Zn)[1]	200	250	300	450
Zinc (Zn)[2]	200	200	200	300
Copper (Cu)	80	100	135	200
Nickel (Ni)	50	60	75	110
Cadmium (Cd)	3	3	3	3
Lead (Pb)	300	300	300	300
Mercury (Hg)	1	1	1	1

[1]Statutory maximum (SI, 1989)
[2]Advisory limit (DoE, 1996)

The Code of Practice for Agricultural Use of Sewage Sludge (DoE, 1996) contains lower maximum soil Zn concentrations than stipulated in the Regulations (SI, 1989) at 200 mg/kg for soil pH in the range 5-7 (Table 1) as a 'precautionary measure' following the recommendations of an Independent Scientific Committee review of the soil fertility aspects of heavy metals (MAFF, 1993).

The EC "Working Document on Sludge" 3rd Draft' (EC, 2000) states that sewage sludge should be used in such a way as to minimise the risk of negative effects on human, animal and plant health; the quality of groundwater and/or surface water; the long-term quality of the soil, and the biodiversity of the microorganisms living in the soil. This document was produced as a technical document and has no regulatory or guidance status. The limit values proposed are shown in Table 2 and are considerably lower than those currently in use in the UK and many other EU countries.

Table 2: Proposed maximum permissible total metal concentrations (mg/kg dry weight) in sludge amended soils (EC, 2000)

	Soil pH		
	5.0<6.0	6.0<7.0	>7.0
Zn	60	150	200
Cu	20	50	100
Ni	15	50	70
Cd	0.5	1	1.5
Pb	70	70	100
Hg	0.1	0.5	1

The Compost Quality Protocol (EA, 2007b) specifies that quality compost can be used in agriculture and horticulture as a soil 'improver or mulch' provided that it does not pose a risk to human health or the environment and its use does not compromise the future sustainable use of the soil. Soil heavy metal analysis is required prior to the first compost application and again when predicted concentrations approach 75 % of the limit values, which are the same as those set out in the "Code of Practice for Agricultural Use of Sewage Sludge" (DoE 1996, Table 1).

The spreading of industrial wastes on agricultural land is controlled by the Waste Management Licensing Regulations (WMLR; SI, 2005) and must be shown to provide benefit to agriculture or ecological improvement under a WMLR exemption. Definitions of agricultural benefit or ecological improvement are not given in the legislation, but statutory guidance indicates that the waste going to land must serve a useful purpose by replacing substances that otherwise would have been used for that purpose (e.g. replacing the need for manufactured fertiliser or lime applications). Such exemptions are needed when recycling composts (outside the Compost Quality Protocol), canal dredgings, paper pulp, etc. The application of these materials must not cause the concentration of any of the metals in soils to exceed the specified limits in the "Code of Practice for Agricultural Use of Sewage Sludge" (DoE 1996, Table 1).

The British Standard for Topsoil (BSI, 2007) specification for multipurpose topsoil stipulates that it should be capable of supporting grass, trees, shrubs and other plantings. Two categories of contaminants are identified i.e. 'phytotoxic contaminants', including Zn Cu and Ni for which the limits are the same as those specified in DoE (1996), and 'chemical contaminants (of concern to human health and the environment) for which no numerical values are given, but reference is made to ensure suitability for purpose.

The values the Environment Agency is proposing to use as soil screening values (SSVs) in the Contaminated Land regime (under Part 2a of the Environment Protection Act 1990) and as 'trigger values' in assessing UK Soil Quality are based on the values derived under the auspices of the EU Existing Substances Regulations (Directive 98/8/EC) and are predicted no-effect concentrations (PNECs) based on soil ecotoxicological test data. The values from this assessment are bioavailability-based and represent a step change in the way in which ecological effects risks from metals in soils are assessed (Smolders *et al*., 2004; Rooney *et al*., 2006; Broos *et al*., 2007).

The EU Risk Assessment approach considers that the ecotoxicity of metals to soil organisms is dependent on soil physico-chemistry (pH, CEC, etc.) and contact time between the metal and the soil (reduction of bioavailability over time). The methodology uses metal specific regression relationships derived from lab-based ecotoxicity data and a lab to field correction to derive predicted no effect concentrations (PNEC) (as total metal concentration). These are then compared with the measured metal concentration in the field. If the field measured value divided by the PNEC is greater than unity, then there is a potential risk from that specific metal under those soil conditions.

Selection of Soils Data

Soils data were collated from a number of field experiments conducted in Britain where applications of organic manures (e.g. sludge cake, liquid sludge, cattle, pig and poultry manures, green compost, paper crumble) had been made and elevated soil heavy metal concentrations were present compared with background metal concentrations at the site (Bhogal *et al*., 2003; Gibbs *et al*., 2006). Soils were selected for inclusion in the database where there was experimental data on soil heavy metal contents and other soil properties required by the specific regimes, such as pH, organic matter and clay content. In addition, each soil selected had supplementary data available on one or more biological properties measured at the site (e.g. soil microbial activity, microbial community size, wheat grain cadmium concentration, earthworm numbers) which were not required by the regime *per se*, but could help to determine whether there had been detrimental effects to the soil at that site following the addition of organic materials. Data were only included where it was possible to measure a significant ($p<0.05$) change in the biological property relative to an untreated control soil, or where a biological property exceeded a specified limit (e.g. the wheat grain Cd concentration was greater than 0.2 mg/kg fresh weight).

Soils Database

The soils database contained details of site location, soil texture (e.g. % sand, silt, clay content), sampling technique (e.g. depth of soil sampled, number of samples taken), experimental design (e.g. number of replicates, plot area) and all soil physical, chemical and biological properties for which data were available. At most sites, there was more than one experimental treatment so the database was structured in such a way as to provide one record per treatment (i.e. each treatment was considered to be a soil scenario). Where there were data for more than one year, only the most recent were included. A summary of the ranges of metal concentrations and other soil properties for soils in the database (comprising data collated from five independent studies) is given in Table 3.

A soil was deemed to have failed based on the biological criteria if one or more of the following effects were observed:

- There was a statistically significant ($p<0.05$) reduction in crop yield compared to a relevant reference plot.
- The wheat grain Cd concentration exceeded the EU limit (0.2 mg Cd/kg fresh weight).
- There was a significant ($p<0.05$) reduction in biomass and/or rhizobia numbers compared to a reference plot.
- There was a significant ($p<0.05$) increase in respiration rates compared to a reference plot.
- There was a significant ($p<0.05$) reduction in earthworm, nematode or enchytraeid numbers compared to a reference plot.

Table 3: Summary of the soil metal concentrations and other soil properties for the soil database

	5.0<5.5	Maximum	Limit*	Count
Zn (mg/kg)	23	474	200	297
Cu (mg/kg)	<1	373	135	297
Cd (mg/kg)	<0.1	33	3	507
Pb (mg/kg)	7	783	300	240
Ni (mg/kg)	3	80	75	69
pH	5.4	8.3	-	507
OC (%)	<1	14	-	507
Clay (%)	6	30	-	297

*Limit value for soils of pH 6.0-7.0 (DoE, 1996)

Assessing Regime Performance

Testing of the soil biological/crop quality indicators against the performance of each regime was undertaken by an independent scientist using a 'generic' guidance note comprising information on the regimes to be tested and detailed work instructions. The assessor was asked to produce a summary table for each of *c.*100 soils selected from the database, indicating whether each soil had passed or failed each regime and on which metal(s) it had failed, to assess whether the protection goals for the respective regimes had been met.

The summary table from the assessor was matched against 'real-world' field soil biology/crop quality measurements and a regime performance table was constructed to highlight how successful each regime had been in protecting soil function. Each soil/regime combination was assigned one of the following 4 performance categories:

- Pass/Pass – the soil was below the 'prompt value' and no adverse soil biological/crop quality effects were observed.

- Fail/Fail – the soil was above the ‘ prompt value’ and an adverse soil biological/ crop quality effect was observed.

For soils in these two performance categories, the regimes were performing correctly, i.e. a correct prediction was made by the regime.

- Pass/Fail – the soil was below the ‘prompt value’ but an adverse soil biological/ crop quality effect was observed.

For soils in the category, the regime was not sufficiently protective of the soil.

- Fail/Pass – the soil was above the ‘prompt value’ but no adverse soil biological/ crop quality effects were observed.

For soils in this category, the regime was overly protective of the soil.

RESULTS AND DISCUSSION

Final results will be available in summer 2008.

The analysis carried out to date indicates that existing regulatory regimes afford a range of protection to soil microbial activity and soil health generally. Further work is required to assess the significance of these results in terms of policy development.

ACKNOWLEDGEMENTS

This work is a collaborative partnership between the Environment Agency, SNIFFER, SEPA, Defra, the European Copper Institute, the Lead Development Association, the International Cadmium Association, the International Zinc Association-Europe and the Nickel Producers Environmental Research Association.

REFERENCES

Bhogal A, Nicholson FA, Chambers BJ and Shepherd MA (2003). Effects of past sewage sludge additions on heavy metal availability in light textures soil : Implications for crop yield and metal uptakes. Environmental Pollution 121, 413-423.

BSI (2007). Specification for Topsoil BS 3882:2007. British Standards Institution, London.

Broos K, Warne M StJ, Heemsbergen DA, Stevens D, Barnes MB, Correll RL, McLaughlin MJ (2007). Soil factors controlling the toxicity of Cu and Zn to microbial processes in Australian soils. Environmental Toxicology and Chemistry 26, 583-590.

Chaudri A, McGrath S, Gibbs , Chambers B, Carlton-Smith C, Godley A, Bacon J, Campbell C and Aitken M (2007). Cadmium availability to wheat grain in soils treated with sewage sludge or metals salts. Chemosphere 66, 1415-1423.

Defra (2004). The First Soil Action Plan for England: 2004-2006. PB9411. Defra Publications, London.

DoE (1996). Code of Practice for Agriculture Use of Sewage Sludge. DoE Publications.

EA (2006). The Development and Use of Soil Quality Indicators for Assessing the Role of Soil in Environmental Interactions. Environment Agency, Rio House, Bristol. Science Report SC030265, pp. 249.

EA (2007a). Soil: A Precious Resource. Environment Agency, Rio House, Bristol.

EA (2007b). Quality Protocol: Compost. Environment Agency, Rio House, Bristol.

EC (2000). Working Document on Sludge Third Draft (ENV.E.3/LM April 2000).

EC (2002). Towards A Thematic Strategy for Soil Protection.
http://ec.europa.eu/environment /soil/index.htm

Gibbs PA, Chambers BJ, Chaudri AM, McGrath SP, Cartlon-Smith CH , Bacon JR, Campbell CD and Aitken MN (2006). Initial results from a long-term multi--site field study of the effects on soil fertility and microbial of sludge cakes containing heavy metals. Soil Use and Management 22, 11-21.

MAFF (1993). Review of the Rules for Sewage Sludge Application to Agricultural Land – Soil Fertility Aspects of Potentially Toxic Elements. PB 1561. MAFF Publications, London.

Rooney C, Zhao FJ and McGrath SP (2006). Soil factors controlling the expression of Cu toxicity to plants in a wide range of European soils. Environmental Toxicology and Chemistry 25, 726-732.

SI (1989). The Sludge (Use in Agriculture) Regulations 1989, SI 1263, HMSO, London.

SI (2005). The Waste Management Licensing (England and Wales) (Amendment and Related Provisions) Regulations 2005, SI 883. The Stationary Office, Norwich.

Smolders E, Buekers J, Oliver I, McLaughlin MJ (2004). Soil properties affecting toxicity of Zn to soil microbial properties in laboratory-spiked and field-contaminated soils. Environmental Toxicology and Chemistry 23, 2633-2640.

INTERCROPPING CAN REDUCE ENVIRONMENTAL IMPACTS

VA Pappa[1,2], RM Rees[2], RL Walker[2] and JA Baddeley[2]

[1]School of GeoSciences, The Crew Building, The University of Edinburgh, West Mains Road, Edinburgh, EH9 3JN, UK, E-mail: Valentini.Pappa@sac.ac.uk; [2]SAC, West Mains Road, Edinburgh, EH9 3JG, UK

SUMMARY

Intercropping can improve the utilisation of available resources and result in increased yields. The main objectives of the experiment were to determine N uptake of the intercrop treatments compared with their associated monocrops and to explore the effects of intercropping on post-harvest N dynamics. Two sites were compared in NE and SE Scotland. At the experiment in the SE, hydrologically-isolated plots enabled the analysis of drainage water losses of N. The treatments were a spring barley (*Hordeum vulgare* cv. *Westminster)* monoculture and intercrops of barley/ white clover (*Trifolium repens* cv. *Alice*) and barley/pea (*Pisum sativum* cv. *Zero4* or cv. *Nitouche)*. No fertilisers, herbicides or pesticides were used. The Land Equivalent Ratio (LER) for the barley/ clover (1.83) was significantly greater than the barley/pea (1.6 and 1.4 for cv. *Zero4* and for cv. *Nitouche*, respectively) intercrops. Two (pea cv. *Nitouche* and white clover) out of the three intercrops showed greater N_2O loss than the barley monocrop. Nitrate leaching from the intercrop containing pea cv. *Nitouche* was lower than from other intercrop treatments.

INTRODUCTION

Intercropping, defined here as any system of multiple synchronous cropping has the potential to offer a range of environmental benefits. Not only has the technique been shown to increase yields, but it is also a useful means of spreading risk: if one crop fails another may still provide sufficient food until the next harvest (Trenbath, 1993). In developed countries and conventional cropping systems, monoculture has proved the rule, with the exception of some grass–clover mixtures, probably because of the ease of combining or lifting a single crop with machinery. Despite this, theoretical and experimental work has pointed to the potential benefits of mixtures of species or varieties. It has been found that where two annual grasses do not compete for a resource, yield per m^2 may be significantly greater than under monocropping; Bulson *et al*. (1997) and Hauugaard-Nielsen *et al*. (2006) have demonstrated this more widely. These results point to clear benefits in productivity by planting intercrops that do not compete with each other, because resources are used efficiently. Hauugaard-Nielsen *et al*. (2003) found a small reduction in nitrate leaching from lysimeters cropped with a pea–barley mixture compared with sole crops, although much of this difference may be attributable to differences in the N-content and rate of decomposition of roots and residues. Where the intercrops have a sequential demand for that nitrogen, yields (and profit) might be maintained but the losses of N reduced.

The objectives of the present experiment were to: 1) determine whether there was any yield benefit of intercrops compared with their associated monocrops; 2) investigate the effects of intercrops of different legume species and varieties on N_2O emissions and NO_3^- leaching from cropping systems.

MATERIALS AND METHODS

A drained-plot experiment near Edinburgh (55.9°N, 3.2°W), consisting of twelve hydrologically-isolated plots was established in an area had been fallow for three years pririor to this experiment. The soil was a sandy loam (Eutric Cambisol, Macmerry Series) developed from partially sorted glacial till. In Aberdeen an experiment was established on a sandy loam (Leptic Podzol, Countesswells Series) in a field which had previously been under grass/clover. The treatments (Table 1) were arranged in three randomised blocks. In the intercrops the seed rates for the pea and barley followed a 50:50 replacement design. Thus, the target intercrop density was 50% of the monoculture density of each crop. Seed rates were 125 kg ha^{-1}of pea, 100 kg ha^{-1} of barley and 5 kg ha^{-1}of white clover. No manure, fertiliser, herbicide or other agrochemicals were applied to the plots. N_2O fluxes were measured at intervals of between one and four weeks by the static chamber technique and gas chromatography. Nitrate and ammonium concentrations in the water samples were determined by continuous flow analysis. Grain yields were calculated using values obtained from combine harvesting of plots. During the winter the plots remained fallow and in the spring, oats were grown in all plots.

Table 1: Combinations of cereals and intercrops used in the experiment together with seed rates (kg ha-1)

Cereal	Intercrop
Barley *Westminster* (200)	None
	Clover *Alice* (5)
Barley *Westminster* (100)	Pea *Nitouche* (125)*
	Pea *Zero4* (125)

* Edinburgh site only

RESULTS

The total barley yield of the barley/clover (≈3.3 ton ha^{-1}) treatment was significantly greater than the barley/pea and barley monocrop for the both sites (Figure 1). The Land Equivalent Ratio (LER) for the barley/ clover (1.83) was significantly greater than the barley/pea (1.6 and 1.4 for cv. *Zero4* and for cv. *Nitouche*, respectively) intercrops. Two out of the three intercrops showed greater N_2O loss than the barley monocrops, although this differed with variety. The two varieties of peas showed large differences in N_2O losses at the Edinburgh site (Table 2). Intercrops also contributed to varying reductions in the amount of N leached from the plots with great differences between the barley/ clover and barley/peas treatments (Figure 2).

Table 2: Cumulative N2O emissions (kg N ha-1) from plots planted with barley/ legume intercrops during summer and winter 2006. Periods are "summer" (June 2006 – September 2006) and "winter" (October 2006 – March 2007). Similar letters within a column indicate treatments not significantly different from each other (P>5%)

Treatment 2006	Aberdeen		Edinburgh	
	Summer	Winter	Summer	Winter
Barley	0.63a	0.61a	0.19a	0.21a
Barley/Clover	0.53a	0.85a	1.23b	0.51b
Barley/Pea *Zero4*	0.75b	0.92b	0.19a	0.19c
Barley/Pea *Nitouche*	-	-	0.92c	0.33d

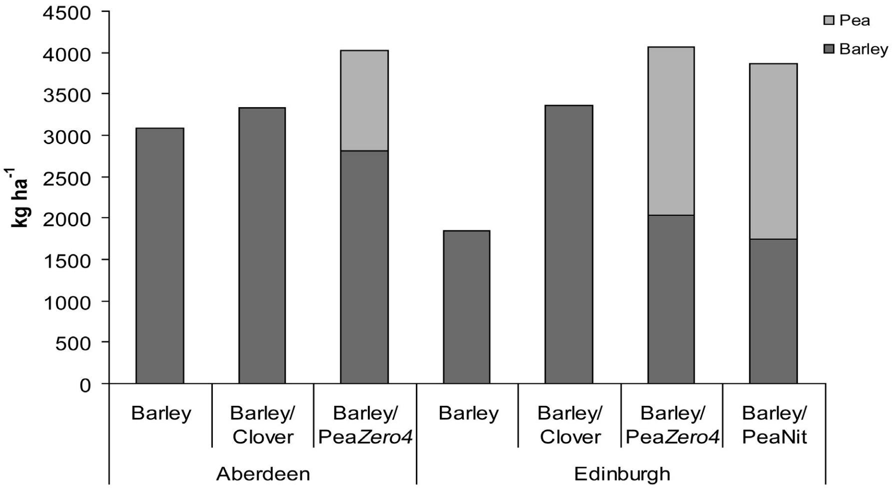


Figure 1: Grain yield for barley and pea in Aberdeen and Edinburgh site

DISCUSSION

Intercropping can result in significantly higher biomass production and nutrient accumulation in the crop. However, N_2O emissions from the legume intercrops were greater than those from the barley monocrop except the barley/ pea cv. *Zero4*. The two pea varieties showed significant differences in N loss with pea cv. *Nitouche* contributing to greater losses of N by N_2O emissions (p< 0.001) and NO_3^- leaching. The underlying mechanisms driving these losses are unclear, although they may be linked to differential rates of root growth and turnover in the monocropped and intercropped treatments. There is a need to take account of the overall nitrogen balance when assessing the environmental impact of farming systems. Finally, this experiment will provide immediate information to farmers on the potential benefits of intercropping systems and evaluate the real benefits in this particular environment.

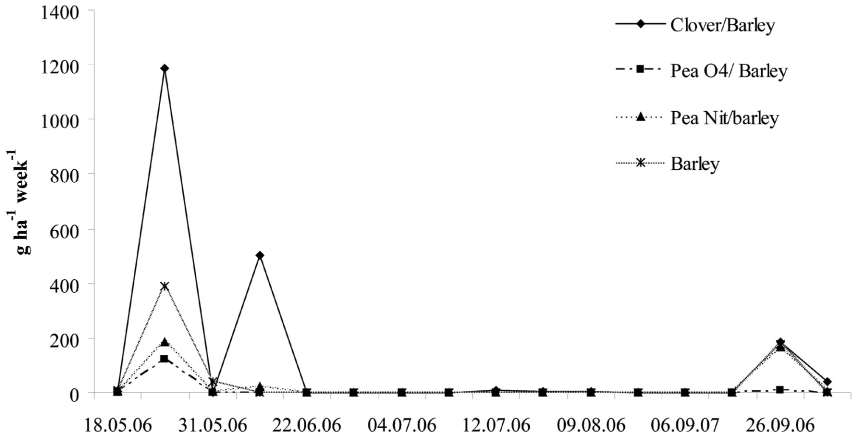


Figure 2: The nitrate leaching rates for the four treatments at the first growing season

ACKNOWLEDGEMENTS

The author wishes to thank the Greek State Scholarship Foundation and SAC for the funding of her studies; Colin Crawford, John Parker, Amy Milne and Derek Simpson for helping with the technical work. SAC receives funding from the Scottish Government Rural and Environment Research and Analysis Directorate.

REFERENCES

Bulson BAJ, Snaydon RW and Stopes CE (1997). Effects of plant density on intercropped wheat and field beans in an organic farming system. J. Agric. Sci. 128, 59–71.

Hauugaard-Nielsen H, Andersen MK, Jørnsgaard B and Jensen ES (2006). Density and relative frequency effects on competitive interactions and resource use in pea–barley intercrops. Field Crops Res. 95, 256–267.

Hauugaard-Nielsen H, Ambus P and Jensen ES (2003). The comparison of nitrogen use and leaching in sole cropped versus intercropped pea and barley. Nutr. Cycl. Agroecosyst. 65, 289–300.

Trenbath BR (1993). Intercropping for management of pests and diseases. Field Crops Res. 34, 381–405.

DOES ENHANCED NITROGEN DEPOSITION REPRESENT A THREAT TO SPHAGNUM AND THUS THE SUSTAINABILITY OF SCOTTISH PEATLANDS?

LJ Sheppard[1], ID Leith[1], SK Kivimaki[1], N van Dijk[1], J Hall[2], P Bruneau[3], J Grace[4] and RM Rees[5]

[1]Centre for Ecology and Hydrology, Bush Estate, Penicuik, EH26 0QB, UK, E-mail: ljs@ceh.ac.uk; [2]Centre for Ecology and Hydrology, Orton Building, Deiniol Road, Bangor, LL57 2UP, UK; [3]SNH, Anderson Place, Edinburgh, EH6 5NP, UK; [4]School of GeoSciences, Crew Building, West Mains Road, Edinburgh, EH9 3JN, UK; [5]SAC, West Mains Road, Edinburgh, EH9 3JG, UK

SUMMARY

Nutrient limited ombrotrophic bogs and peatlands support high conservation valued ecosystems, potentially susceptible to current elevated levels of reactive nitrogen (N) deposition. Here, we present the effects and consequences of different N forms, wet, dry, reduced and oxidised N on the functioning of a bog moss, *Sphagnum capillifolium*. *Sphagnum* mosses maintain the acid, low nutrient conditions, crucial for the sustainability of peat lands, where productivity must exceed decomposition. Dry deposited ammonia substantially elevated shoot N status, which led to tissue breakdown loss of function and death in *S. capillifolium.* Wet deposited nitrate and ammonium also negatively affected *S. capillifolium*, significantly reducing shoot extension and cover and significantly elevating N status. These effects occurred over 5 years and were significant even at the lowest reduced N dose, 8 kg N ha^{-1} y^{1} (background = 8-10 kg N ha^{-1} y^{-1}), highlighting the threat N poses for the effective functioning of bog ecosystems.

INTRODUCTION

Blanket and raised bogs are peat based ecosystems, with a restricted world distribution, covering around 1.5 million hectares in Great Britain. Peat based bogs can represent thousands of years of organic matter accumulation and are amongst Britain's most ancient natural/semi-natural ecosystems. Bogs are valued for their specialised plant and bird communities and their ability to act as a sink for carbon. These plant communities are adapted and restricted to nutrient limited conditions sustained by the wet, often anoxic, acidic conditions. These conditions, which restrict decomposition, are generated by the unique properties of the keystone species belonging to *Sphagnum* spp. or 'bog moss' (Clymo and Hayward, 1982). Today, apart from reclamation, drainage for other land uses, one of the major threats to the sustainability of bogs comes from the enhanced deposition of reactive nitrogen. *Sphagnum* mosses are efficient scavengers of mineral nitrogen, this N 'sequestration' while increasing their N sensitivity (Limpens *et al.,* 2004), helps to exclude faster growing plant species with higher transpiration rates and the potential to lower the water-table. Without *Sphagnum* many bogs would be transformed into drier grass/tree dominated habitats, at the expense of all the specialized species (Aldous, 2002).

Atmospheric reactive nitrogen comes in the oxidised form from fossil–fuel combustion and in the reduced form from food production, especially via intensive livestock units and fertilizer use. These pollutants can be deposited locally (ammonia) or transported for varying distances in the atmosphere to affect air and rainfall quality across the UK. In Scotland bog and peatland ecosystems are found predominantly in the wetter north and west, and thus receive most of their N load as wet deposition. Bogs can also be affected by nitrogen eutrophication from runoff or local sources of ammonia. In Europe, many bogs occurred in regions of intensive livestock farming, here the detrimental effects of reduced nitrogen, ammonia have transformed such bogs into grassy plains (Heil and Diemont, 1983). The majority of British bogs, with the exception of those in Northern Ireland, by comparison do not occur in areas of high nitrogen deposition, although most sites already receive the Critical Nitrogen Load of 5-10 kg N ha^{-1} y^{-1}.

Our study compares the *in situ* effects of three different reactive N forms: gaseous ammonia (NH_3), wet ammonium (NH_4^+) and wet nitrate (NO_3^-) on an ombrotrophic bog under real world conditions and at realistic concentrations. This paper reports the effects of these 3 N forms on the growth, cover and N status in a key hummock dwelling *Sphagnum*, *S. capillifolium* and discusses the implications of elevated N deposition for the sustainability of peat land ecosystems.

MATERIALS AND METHODS

In 2002 a unique N manipulation experiment comprising an ammonia fumigation (Leith *et al.,* 2004), simulating ~ 20,000 broiler hens (Sheppard *et al*., 2008), and a wet automated spray system (Sheppard *et al*., 2004) was established on Whim bog in the Scottish Borders. The ammonia release was programmed so that gaseous ammonia mixed with air was released over a 100 m long transect, from a 10 m perforated pipe, 1 m above the vegetation, into the prevailing wind, providing a concentration and N deposition gradient. Equivalent N doses to those applied to the wet treatments, (8, 24 and 56 kg N $ha^{-1}y^{-1}$) were estimated at 8, 16 and 32 m from the NH_3 source (Cape *pers comm*.). Data are reported for these distances. The wet treatments were provided from revolving sprayer heads, in the centre of each plot, as either sodium nitrate or ammonium chloride at 3 N doses: 8, 24 and 56 kg N $ha^{-1}y^{-1}$, over and above the 8-10 kg N $ha^{-1}y^{-1}$ background, at a maximum concentration of 4 mM N, to 4 replicate plots per treatment. A water only control was included to assess the effects of the additional 10 % precipitation. Treatments were fully replicated, one 12.5 m^{-2} treatment plot per each of 4 blocks and applications were coupled to rainfall, no rain no treatment. Treatments have been applied throughout the year since summer 2002, as and when meteorological conditions permitted.

Species cover was assessed initially in May 2002, in 2004 and in summer 2007 for 3 x 0.25 m^{-2} permanent quadrats, subdivided into 16 squares, per plot. The overall percent cover of *Sphagnum* was also estimated by 2 observers independently, for each plot in 2004 and 2007 and reported as % change. *Sphagnum* plot was highly variable and this variability was not replicated in the quadrats, likewise the 4 replicate plots had differing initial amounts of *Sphagnum* and were not replicates in the true sense. The data presented are weighted cover estimates of the degree to which cover in the quadrats has changed over 5 years. Extension growth was assessed between May and November 2007 using10 modified crank wires (Limpens *et al*., 2004) per plot inserted into clumps of healthy *S. capillifolium*. Plot means (4) were analysed using GenStat Release 10.1 (ANOVA, General treatment structure, no blocking: model based on N addition, N dose and N form). In 2005 samples of the apical 2 cm of stem were removed, cleaned and frozen, then extracted in ultra-pure water for 4 h, filtered and the soluble NH_4^+ measured using an ammonia flow injection analysis system. Results are presented in µg N g^{-1} dry wt after log transformation.

RESULTS

After 5 years the *S. capillifolium* within 16 m of the NH_3 source had died, and at 32 m only red pigmented shoots survive. No 'vital' shoots were available for growth or N status monitoring in the ammonia treatments. In the wet treatment plots, *S. capillifolium* has increased in the control by ~ 20% (Figure 1) but decreased in response to N, particularly reduced N, by ~ -40%, with the greatest effects at 56 kg N $ha^{-1}y^{-1}$. Effects of oxidised N were also detrimental, ~ -30%. In the quadrats also, N additions at the highest doses or as reduced N decreased cover, whereas with oxidised N, up to 24 kg N $ha^{-1}y^{-1}$ (Figure 2) there was still a small positive effect on cover, albeit substantially less than in the control quadrats.

After 5 years the *S. capillifolium* within 16 m of the NH_3 source had died, and at 32 m only red pigmented shoots survive. No 'vital' shoots were available for growth or N status monitoring in the ammonia treatments. In the wet treatment plots, *S. capillifolium* has increased in the control by ~ 20% (Figure 1) but decreased in response to N, particularly reduced N, by ~ -40%, with the greatest effects at 56 kg N $ha^{-1}y^{-1}$. Effects of oxidised N were also detrimental, ~ -30%.

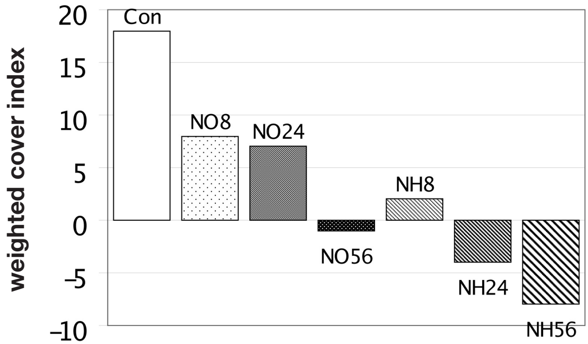


Figure 1: The weighted cover index represents the change in the cover of *Sphagnum capillifolium* between 2002 (pre-treatment) and 2007 in response to wet deposition, applied as a spray, of sodium nitrate (NO) or ammonium chloride (NH) at 8, 24 and 56 kg N ha^{-1} y^{-1} chloride (NH) at 8, 24 and 56 kg N ha^{-1} y^{-1}

When extension growth was measured in the wet plots, after 5 years of continuous treatment, the detrimental effects of N were highly significant (P=0.003). Reduced N additions reduced shoot extension, irrespective of dose (Figure 3), even at 8 kg N $ha^{-1}y^{-1}$ dose in contrast to oxidised N where the effects were moderated by the dose. The soluble N (NH_4^+) status of the apical stem section was significantly enhanced in response to wet deposited N (P=0.019) responding to both dose (P<0.001) and form (P=0.099), with reduced N causing the largest N increase (Figure 4).

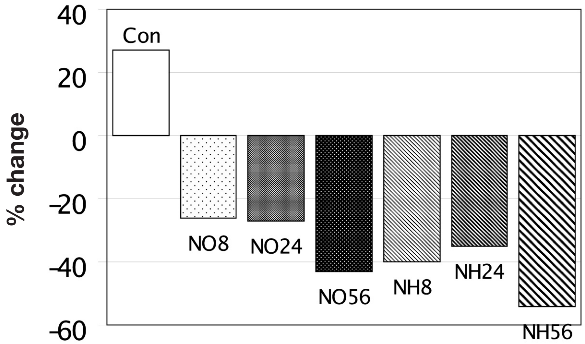


Figure 2: Mean percent change in the cover of *Sphagnum capillifolium* growing in the four 12.5 m^2 plots, between 2004 and 2007 in response to wet deposition, applied as a spray, of sodium nitrate (NO) or ammonium chloride (NH) at 8, 24 and 56 kg N ha^{-1} y^{-1}

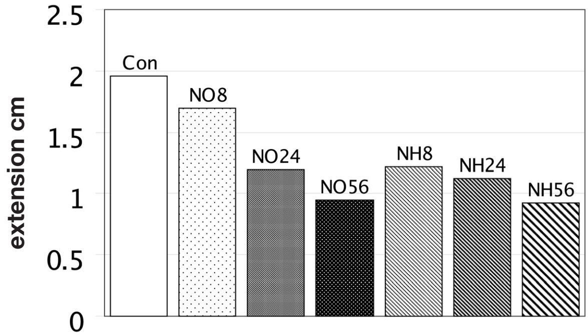


Figure 3: Extension growth in *Sphagnum capillifolium* measured from a fixed position, cranked wire, 10 per plot, between May and November 2007 in response to wet deposition, applied as a spray, of sodium nitrate (NO) or ammonium chloride (NH) at 8, 24 and 56 kg N ha^{-1} y^{-1}

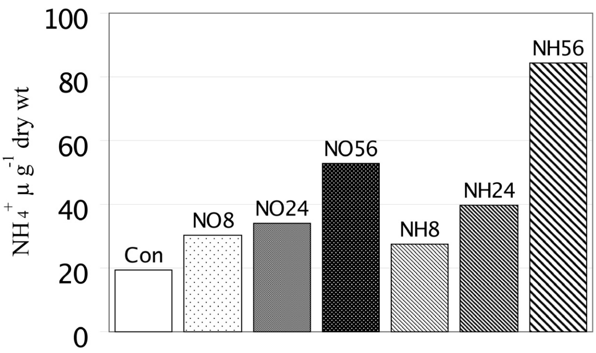


Figure 4: Soluble ammonium in *Sphagnum capillifolium*, apical 2 cm frozen, then extracted in water, in October 2005 in response to wet deposition, applied as a spray, of sodium nitrate (NO) or ammonium chloride (NH) at 8, 24 and 56 kg N ha^{-1} y^{-1}

DISCUSSION

This experiment, the first to study the effects of different N forms on an ombrotrophic bog ecosystem has shown conclusively the vulnerability of such ecosystems to elevated N deposition, fully corroborating the low Critical Load adopted for these ecosystems at the UNECE Expert workshop Berne 2002. Critical Loads are set in order to protect ecosystems, providing a temporal protection ~ 20 years to take account of the potential for cumulative effects. The responses reported for Whim bog are important because unlike many previous manipulation studies which were conducted in areas of high background N deposition, and where acclimation to N may have already occurred, the N deposition at Whim is relatively low and more similar to that found where the majority of Scottish bogs occur. At Whim bog where the wet treatment application is coupled to rainfall and N concentrations are relatively low, by experimental standards, although still twenty-fold higher than occult deposition and two orders of magnitude higher than concentrations in rainfall, the ambient deposition is within the Critical Load. In response to ambient N inputs *S. capillifolium* appears quite healthy, maintaining growth rates consistent with other hummock species (Limpens *et al*., 2004) and thus its competitive ability. However, at just double the Critical Load and after only 5 years, deleterious effects of N in precipitation can be identified, especially when the N deposition is dominated by reduced N. The accumulation of potentially toxic NH_4^+ ions may be contributing to the loss of vitality, however, competition from nitrophytes also appears to be important. At Whim *S. capillifolium* is being overgrown and out competed by more N tolerant pleurocarpous mosses such as *Hypnum jutlandicum* and *Pleurozium schreberi*. In addition, the additional N, especially at the lower doses, has increased the growth of ericoid shrubs, which in turn will restrict the light reaching these understorey mosses and potentially increase evapo-transpiration and lower the water-table. The detrimental effects of N on *S. capillifolium* are strongly exacerbated during droughts, even though the absence of precipitation restricts N deposition (Carfrae *et al.,* 2007). Likewise, at reduced light levels, the potential to detoxify N via assimilation is reduced.

These results corroborate the accumulating literature regarding N impacts on *Sphagnum,* which also highlight the acute N sensitivity of hummock dwelling *Sphagnum* species (Gerdol *et al.,* 2007). *S. capillifolium* is the only constant *Sphagnum* in the M19 (National Vegetation Classification) mire category (Rodwell *et al.,* 1991), competing with the hypnaceous mosses for the ground carpet. Given that M19 has already been almost eradicated from south of the Border through a combination of drainage, pollution and erosion, we need to be concerned for this type of mire in Scotland. Comparing the N deposition maps for moorland (closest surrogate for bogs) and the distribution of *S. capillifolium* (Rodwell *et al.,* 1991) reveals that current levels of N deposition are approaching levels, shown here to cause detrimental effects leading to reduced cover, in the south of Scotland, northern England and the Cairngorms. Nitrogen depositing as ammonia gas was the most damaging N form, however, this threat is more restricted to the co- location of bogs and localized NH_3 sources most commonly found in Northern Ireland.

This N manipulation study has demonstrated that enhanced nitrogen deposition, at doses in excess of the Critical Load will have a profound negative impact on the hummock forming species *S. capillifolium* the keystone *Sphagnum* moss in *Calluna vulgaris – Eriophorum vaginatum* blanket mire. Although, in Scotland the distribution of *S. capillifolium* means that many of its locations receive N deposition at doses below the Critical Load, we also know that changes in the water table level, especially lowering, considerably exacerbate the detrimental effects of elevated N deposition (Carfrae *et al.,* 2007). We currently need a better understanding of these interactions, together with the responses of pool inhabiting *Sphagnum* so that we can reduce the uncertainties in predicting the response of *Sphagnum* species and peatlands to the combination of climate change and N deposition.

ACKNOWLEDGEMENTS

The Whim experiment was initiated under the NERC GANE thematic programme and is now supported through CEH, the NERC-DEFRA Terrestrial Umbrella contract (CPEA 18) and NITROEUROPE. William Sinclair Horticulture Ltd is thanked for access. Sanna Kivimaki is funded by an SNH studentship.

REFERENCES

Aldous AR (2002). Nitrogen translocation in *Sphagnum* mosses: effects of atmospheric nitrogen deposition. New Phytologist 156, 241-253.

Carfrae JA, Sheppard LJ, Raven J A *et al.* (2007). Potassium and phosphorus additions modify the response of *Sphagnum capillifolium* growing on a Scottish ombrotrophic bog to enhanced nitrogen deposition. Applied Geochemistry 2007 (in press).

Clymo RS and PM Hayward (1982). The ecology of *Sphagnum*. In A J E Smith, editor. Bryophyte Ecology. Chapman and Hall Ltd, London.

Gerdol R, Petraglia A, Bragazza L, Iacumin P and Brancaleoni L (2007). Nitrogen deposition interacts with climate in affecting production and decomposition rates in *Sphagnum* mosses. Global Change Biology 2007 (in press).

Heil GW and Diemont WH (1983). Raised nutrient levels change heathland into grassland. Vegetatio 53: 113-120.

Leith ID, Sheppard LJ, Fowler D *et al.* (2004). Quantifying dry NH_3 deposition to an ombrotrophic bog from an automated NH_3 release system. Water, Air, and Soil Pollution: Focus 4 207-218.

Limpens J, Berendse F and Klees H (2004). How phosphorus availability affects the impact of nitrogen deposition on *Sphagnum* and vascular plants in bogs. Ecosystems 7:793-804.

Rodwell JS (ed.) (1991*)* British plant communities*.* Vol 2. Cambridge University Press.

Sheppard LJ, Crossley A, Leith I D *et al.* (2004). An automated wet deposition system to compare the effects of reduced and oxidised N on ombrotrophic bog species: practical considerations. Water, Air, and Soil Pollution: Focus 4 197-205.

Sheppard L J, Leith I.D, Crossley *et al*. (2008). Stress responses of *Calluna vulgaris* to reduced and oxidised N applied under 'real world conditions'. 2008. Environmental Pollution 2008 (in press).

THE IMPACT OF CLIMATE CHANGE ON GREENHOUSE GAS EMISSION FROM GRASSLAND SWARDS: A SCOTTISH PERSEPECTIVE

CFE Topp and RM Rees

SAC, West Mains Road, Edinburgh, EH9 3JG, UK, E-mail: Kairsty.Topp@sac.ac.uk

SUMMARY

Scottish agriculture contributes 12% of the Scottish greenhouse gas emissions, with grasslands being the biggest single source. The DNDC model has been used to explore the impact of an inorganic fertiliser and a slurry application strategy on greenhouse gas emissions from grassland cut for silage at two different sites in Scotland. In order to assess the effect of current and future climate conditions on global warming potential the weather generator Earwig was used to create baseline and 2020 scenarios based on the UKCIP02 low and medium high emissions scenarios. Although the grasslands continue to act as net sinks for greenhouse gases, the sink strength would decline by up to 40%. This study highlights that over a relatively short period of time the greenhouse gas sink strength of Scottish grasslands may show a significant decline, as a consequence of our changing climate.

INTRODUCTION

In 2003, Scottish agriculture contributed 12% of the Scottish greenhouse gas emissions (GWP) (Scottish Executive, 2006). More significantly, Scottish agriculture was responsible for 83% of the nitrous oxide (N_2O) emissions, much of which is associated with the use of organic and inorganic fertilisers on grasslands. Despite this, grasslands tend to operate as net greenhouse gas sinks largely as a consequence of their high CO_2 sink strength (methane exchange from cut grasslands is generally close to zero) (Soussana *et al*., 2007). It is essential therefore to take into account the opposing effects of C uptake and N_2O release when assessing future climate and management scenarios in order to calculate a net greenhouse gas balance. Improved grasslands occupy a significant part of the Scottish landscape covering about 22% of the agricultural land area in 2005. The manure and inorganic fertiliser management of these grasslands is known to have a significant impact on their N_2O emissions and hence greenhouse gas balance. Modifications to their management based upon an assessment of current and future climatic conditions could therefore play a valuable role in reducing greenhouse gas emissions. The DNDC (DenitrificationDecomposition) model (Li *et al*., 1992; Li *et al*., 2006; Saggar *et al*., 2004) simulates the daily fluxes and pool sizes of carbon and nitrogen in agroecosystems. It has been extensively applied around the world and is widely acknowledged as a state-of-the-art model for assessing greenhouse gas emissions and nutrient fluxes from agriculturally managed soils. This model has been used to explore the impact of different inorganic fertiliser and manuring strategies for a grassland sward utilized for silage production on greenhouses gas emissions at two sites in Scotland.

MATERIALS AND METHODS

Two sites in Scotland were used to assess the impact of inorganic fertiliser and manure applications on the GWP of grassland, one located at Cowpark on the Bush Estate, near Edinburgh and one at Crichton Royal Farm, Dumfries. The DNDC model was used to simulate management for 30 years, with the results presented for the final 20 years as this gives time for the soil pools to stabilise. During the 20 years for which the results are presented, it was assumed that either 300 kg N of inorganic fertiliser or cattle slurry were applied. The inorganic fertiliser was applied in 3 applications of 100 kg N in mid March, mid May and mid July. Slurry applications were made in mid April and mid June with 150 kg of available N being applied at each application. Three cuts of silage were taken per year with cuts in mid June, late August and late October. The weather generator Earwig (Kilsby *et al*., 2007) was used to create 30 years of baseline (1961-1990) and 2020 UKCIP scenarios (http://www.ukcip.org.uk/). For 2020, the low and medium high emissions scenarios were used in order to assess the sensitivity of the emissions to climate variability.

RESULTS

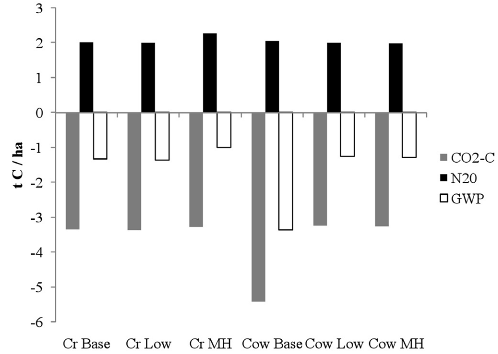
The results indicate that for both the inorganic fertiliser and slurry treatments, the emissions of methane were insignificant by comparison with N_2O emissions and carbon uptake. At both Crichton and Cowpark there was a net uptake of GHGs (negative GWP) driven by the large C sink strength and all management and climate combinations. However, at both sites there was a reduction in the GHG sink strength (less negative GWP) when baseline conditions were compared with either of the 2020 climate scenarios (Figure 1). At Crichton this was due to the combination of a small increase in N_2O emission and a small decline in the C sink, while at Cowpark the change was mostly due to a reduction in the C sink.

Both the inorganic fertiliser and slurry treatments were net GHG sinks under all climates, (Figure 1), but the slurry treatment at both sites resulted in a greater net carbon uptake than the inorganic fertiliser treatment. However, the overall environmental benefits of the slurry were counterbalanced by the increase in nitrate leaching (Figure 2). This was particularly noticeable at the wetter Crichton site. There was also a significant increase in the soil organic matter pool size for the slurry treatment relative to the inorganic fertiliser treatment.

DISCUSSION

The results suggest that the net sink strength for greenhouse gases in the grasslands that have been studied will decline by up to 40% over the next 20 years. This change is driven largely by predicted changes in the climate, and the effects highlight the potential for strong regional differences in ecosystem responses. In the west of Scotland, the warmer conditions are predicted to lead to increased losses of N_2O. In the east, drier summers and overall increases in temperature would reduce the carbon sink strength but have little impact on N_2O emissions. There are clearly uncertainties associated with these predictions. This is partly a consequence of model uncertainties, although predicted greenhouse gas fluxes are broadly consistent with those from measurements at Scottish sites (Jones *et al*., 2006; Jones *et al*., 2007). There are also clearly uncertainties about future climates. However, this study highlights that over a relatively short period of time the GHG sink strength of Scottish grasslands may show a significant decline as a consequence of our changing climate. These feedback effects will make targets for significant reductions in greenhouse gas emissions even more challenging.

a)



b)

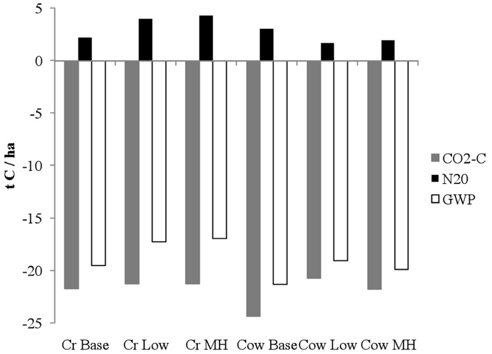


Figure 1: Carbon sequestration, N_2O and global warming potential for the (a) fertiliser treatment and (b) slurry treatment for Crichton (Cr) and Cowpark (Cow) weather conditions for current (Base), and the low (Low) and medium-high (MH) UKCIP02 emission scenarios

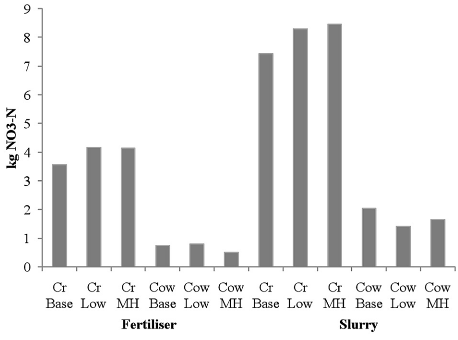


Figure 2: NO_3-N leaching for the fertiliser and slurry treatments for Crichton (Cr) and Cowpark (Cow) weather conditions for current (Base), and the low (Low) and medium-high (MH) UKCIP02 emission scenarios

ACKNOWLEDGEMENTS

We acknowledge the support of RERAD and EU project NitroEurope IP (Contract no: 017841).

REFERENCES

Jones SK, Rees RM, Kosmas D, Ball BC and Skiba UM (2006). Carbon sequestration in a temperate grassland; management and climatic controls. Soil Use and Management 22, 132-142.

Jones SK, Rees RM, Skiba UM and Ball BC (2007). Influence of organic and mineral N fertiliser on N_2O fluxes from a temperate grassland. Agriculture Ecosystems and Environment 121, 74-83.

Kilsby CG, Jones PD, Burton A, Ford AC, Fowler HJ, Harpham C, James P, Smith A and Wilby RL (2007). A daily weather generator for use in climate change studies. Environmental Modelling and Software 22, 1705-1719.

Li C, Frolking S and Frolking A (1992). A model of nitrous oxide evolution from soil driven by rainfall events: 1. Model structure and sensitivity. J. Geophys. Res 97, 9759-9776.

Li CS, Farahbakhshazad N, Jaynes DB, Dinnes DL, Salas W and McLaughlin D (2006). Modeling nitrate leaching with a biogeochemical model modified based on observations in a row-crop field in Iowa. Ecological Modelling 196, 116-130.

Saggar S, Andreew RM, Tate KR, Hedley CB, Rodda NJ and Townsend JA (2004). Modelling nitrous oxide emissions from dairy-grazed pastures. Nutrient Cycling in Agroecosystems 68, 243-255.

Scottish Executive (2006). Changing our ways; Scotland's climate change programme.

Soussana JF, Allard V, Pilegaard K, Ambus P *et al.* (2007). Full accounting of the greenhouse gas (CO_2, N_2O, CH_4) budget of nine European grassland sites. Agriculture Ecosystems and Environment 121, 121-134.

WAgriCo – WATER RESOURCE MANAGEMENT IN CO-OPERATION WITH AGRICULTURE

JL Whitehead and MA Shepherd

ADAS UK Ltd, Gleadthorpe, Meden Vale, Mansfield, Nottinghamshire, NG20 9PF, UK, E-mail: jodie.whitehead@adas.co.uk

SUMMARY

The EU LIFE and Defra funded project WAgriCo ('Water resource management in co-operation with agriculture') aims to develop and demonstrate integrated catchment management in pilot catchments in Germany and England with the intention of decreasing nitrate losses to groundwater. Central to the approach is the large-scale use of 'Programme of Measures'. It is a supportive project, working with the local farming community, to help in providing a model for others to follow.

INTRODUCTION

Effective land management within a catchment is vital in meeting the UK's target obligations under the Water Framework Directive by 2015. The EU LIFE and Defra funded project WAgriCo (Water resource management in co-operation with agriculture) is a three year project that aims to develop and demonstrate integrated catchment management in pilot catchments in Germany and England with the intention of decreasing nitrate and pesticide losses to groundwater. However, this paper will only focus on diffuse nitrate losses and its impact on drinking water standards.

In England, the project is centred within the South West catchments of the Frome, Piddle and Wey, where major groundwater abstraction boreholes have peak nitrate concentrations above or approaching the 50 mg l^{-1} NO_3 drinking water standard. The project is therefore working with farmers to raise awareness of diffuse pollution and to identify and implement measures to tackle it. The catchments cover 560 km^2. The area is predominately rural and is extensively farmed, with approximately 650 farm holdings. Agricultural land use is mixed with both arable (including cereal, horticulture and non rotational crops) and pasture grazing. Farming within the catchments is undertaken in very close proximity to the groundwater boreholes as no buffer zones exists to protect the aquifers serving the boreholes. Therefore, these boreholes are at risk from diffuse pollution as a result of nutrient and pesticide contamination.

APPROACH

Initial Farm Assessments

Catchment advisors undertook preliminary farm assessments in all five catchments. The aim was to discuss the project and its objectives with the farmers, to identify borehole/stream sampling points already on the farm, and to gain co-operation with the project. An initial risk assessment of the farm was also undertaken.

Detailed farm audits were undertaken to collate farming system and fertiliser and manure management data for the last five years. These data have allowed the assessment of the nitrogen pollution issues on each farm, culminating in the calculation of nutrient balances (see below). These data also serve as a baseline against which to compare improvements so as to assist in the evaluation of success later in the project.

Farm Gate Nutrient Balances

Nutrient budgets summarise nutrient inputs to, and outputs from, a defined system over a defined period of time. A farm-gate budget records the amount of nutrients in all kinds of products that enter and leave the farm via the farm-gate (Goodlass *et al*., 2006). The data collected from the farm audits were used to construct average farm gate nutrient budgets for each of the farms. These data can be used as an indicator of the efficiency of the system and the likelihood of environmental damage, though the correlation is not always straightforward (Oborn *et al.,* 2005). It is also useful for benchmarking farms against national results. As part of the project this approach will also be trialled with the farmers as a management tool.

Water and Soil Sampling

A range of measurements have been taken throughout the project (Table 1). These measurements are used for several purposes: supporting farmers and their co-operation in the project; assessment of effectiveness; identifying problems; demonstration of effects to farmers.

Table 1: Summary of catchment monitoring

Type of Monitoring	Description
Soil nutrient status	Analysis of topsoil for P, K, Mg status and Ph
Soil mineral N (SMN) to 90 cm depth	Analysis of soil for SMN status (nitrate and ammonium content) can be used to estimate leaching risk (when taken in autumn at the return of the soil to field capacity) or for fertiliser N recommendations (when sampled in the spring)
Water sampling	Borehole and well sampling will allow groundwater quality and quantity to de determined
Porous pots	To allow the monitoring of nitrate leaching losses from the soil

Programme of Measures (PoMs)

The farm audits and risk assessments indicated that there were few examples of severely poor practice that could be immediately identified as the main cause of diffuse pollution problems within the catchment. It was therefore decided that mitigation methods should focus on 2 levels:

- **Good Agricultural Practice (GAP)** – focusing on good fertiliser and manure management practices; and
- **Enhanced GAP** – including approaches that were currently outside usual practices or are potential areas of improvement, e.g. the use of cover crops, fertiliser spreader calibration.

It was considered that much could be tackled by focusing on GAP and by helping farmers understand the linkage between farm practices and N loss. A total of six measures were compiled which form the PoMs for the priority catchments within the study area; fertiliser recommendations (including soil testing), enhanced manure management plan, cover crops, fertiliser spreader calibration, moving from autumn to spring application of manures and encouraging the use of N efficiencies as a management tool.

Farmer Support

To bring about change at the local level, the influencers are best based locally; therefore, WAgriCo uses catchment advisers to offer on-farm assistance and advice on farm management issues. There are four catchment advisers covering the WAgriCo priority areas who act as points of contact for the farmers. The role of the catchment adviser is two-fold; to offer advice and support to the farmer and then to gather data and feedback on the PoMs implemented on the farms.

Farm advice has been delivered through one-to-one farm visits, newsletters and training workshops. Advice has mainly concentrated on improving nutrient and manure management on the farms. For example, this has involved discussing fertiliser recommendations and taking account of nutrients applied as manures. This was achieved through farm visits and a training workshop on the use of nutrient management software. As well as on-farm advice, the catchment adviser with other members of the project team have set up Local Farmer Groups, to discuss specific problems within catchments and how measures, supported through funding in some circumstances, can be put in place to improve the situation.

RESULTS AND DISCUSSION

Farm Gate Nutrient Balances

Farm gate nutrient balances were calculated for 26 farms within the Milborne St Andrew and Dewlish catchments based on farm data from 2001-2005 (Figure 1). When averaged across the catchment, Milborne St Andrew farms performed well compared with the average of 177 farms across England (Chambers, 2006) (Figure 1). Overall, these results reflected the initial (qualitative) farm assessments undertaken by the catchment advisers; 'higher risk' farms showed the greatest N surpluses. This demonstrates the value of an experienced catchment advisor. When compared to national data the results suggest that the WAgriCo catchments are typical of the rest of England and that farms within the project face similar N management challenges to those of the rest of the country under similar agriculture.

Soil Mineral N (SMN)

Figure 2 shows the autumn 2006 SMN samples from Milborne St Andrew and Dewlish plotted with the results from England-wide measurements taken from the Nitrate Vulnerable Zone (NVZ) monitoring scheme (Lord *et al*., 2007). Although from only one year, the data point to the fact that there is room for improvement in N management, with greater residual soil N levels than average.

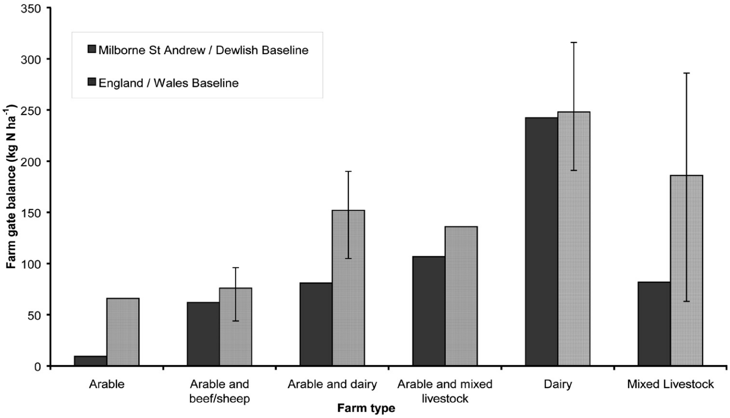


Figure 1: Baseline farm gate balance results compared with England and Wales baseline data (reference data from Chambers, 2006)

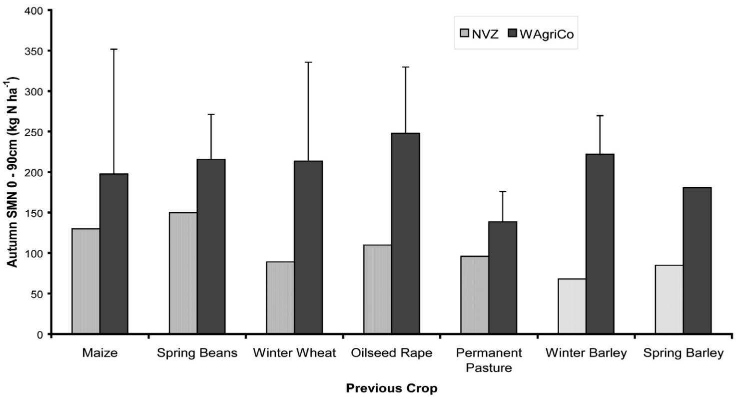


Figure 2: WAgriCo autumn 2006 SMN values compared to the national NVZ dataset (Lord *et al*., 2007) by crop

Programme of Measures

Out of the 74 farmers targeted by the project, a total of 51 farmers have agreed to participate (69%). The majority of the farmers use fertiliser recommendations provided by the advisers (75%) and, where appropriate, manure management plans will be developed (44%). About 38% of farms took advantage of a free fertiliser spreader calibration. Only 13% of farmers have decided to grow cover crops, but this is due to the predominant use of winter crop rotations which makes the use of cover crops inappropriate. Moving from autumn to spring application of manures has also had a very small uptake (4%) as many farmers do not have sufficient storage facilities to store manures over the winter. Farmer feed back on the measures are also being sought but the results are not available at this stage of the project.

Fertiliser Spreader Calibration

Results from those machines tested showed that 22% required static calibration for application rates to balance. Of these, the average coefficient of variation (CV) before calibration was 22%, following calibration 3%. The theoretical effects of poor spreading suggest that for cereal crops a relative low CV of 15% will increase nitrate leaching by 8% above a baseline loss of 57 kg ha^{-1}. This increases to approximately 13% at a CV of 30%. These results indicate that fertiliser spreader calibration is effective in reducing nitrate leaching as well as being beneficial to the farmer, providing £6-10 ha^{-1} in yield improvements.

N efficiencies based on farm gate nutrient budgets

Nitrogen efficiencies were calculated as a re-expression of the farm-gate N balance (N out/ N in x 100%). Unlike the farm-gate nutrient surpluses, N efficiencies show how effectively N inputs are being used (which may be a more meaningful expression for farmers). N efficiency is therefore the proportion of imported N that is exported as 'useful products'. Baseline N efficiencies were calculated for 26 farms within the Milborne St Andrew and Dewlish catchments based on farm data from 2001-2005 (Table 2). Efficiency will depend on farm type and farms with seemingly large surpluses may be more efficient than farms with smaller surpluses, as can be seen in Table 2.

Table 2: Comparison between baseline N surpluses and N efficiencies, by farm type

Farm type	Baseline surplus (kg N ha^{-1})	Baseline efficiency (%)
Arable	26	67
Arable and beef/sheep	73	74
Arable and dairy	81	47
Arable and mixed livestock	100	74
Dairy	272	38
Mixed Livestock	96	19

This approach assumes surplus nutrients are available for loss, and a reduction in this surplus and consequent improvement in N efficiency translates into a reduction in the nutrient load received by waterbodies. Therefore, farmers are being encouraged to improve their on farm N efficiency by focusing on key management areas:

- Fertilisers – e.g. use recommendations, account for all N in manures
- Manure – e.g. rapidly incorporate into soil to reduce ammonia loss
- Feed – e.g. look at diet formulations and feeding strategies
- Crops – e.g. look at pest/disease control strategies and soil structure to maximise crop uptake

Baseline results from this approach have been discussed with the farmers and feedback received suggested that this would be a useful and practical way to think about N management on farm.

The Future

There is still more to achieve. Farmer engagement and monitoring will continue. Due to the hydrological time lag within the catchments, it is unlikely that significant changes in nitrate concentrations at the boreholes will be identified during the project. Therefore, monitoring data is being used in conjunction with the farm audit data to construct field and catchment scale models for the assessment of mitigation measure impacts on groundwater. An economic assessment of the PoMs has also been undertaken to allow a detailed understanding of the economic implications of these actions at a range of levels; individual farm businesses, catchment-scale and national-scale.

CONCLUSIONS

Our assessment of these pilot areas is that no single approach will achieve large reductions in N losses to the environment, but a range of smaller changes and cost-effective N management measures may have a significant effect in reducing N losses. The catchment approach, the use of catchment advisors and farmer workshops has proven essential. Regular farmer engagement on a one-to-one basis has ensured farmers want to co-operate with the project. Feedback and discussions with farmers have indicated that advice is a key element in helping to tackle diffuse pollution. It has been clear that not all farmers understand the consequences of some of their actions but are, however, very keen to learn and use nutrients more efficiently on their farms. As the project continues, a key part will be understanding barriers to uptake of mitigation methods. Whilst this may simply be cost in some cases, other factors are also involved. It is necessary to understand these if we are to effectively implement change.

ACKNOWLEDGMENTS

This work was supported by EU LIFE Demonstration funding and Defra. We would also like to acknowledge the support of the principal project partners: UKWIR, NFU, Wessex Water and the Environment Agency.

REFERENCES

Chambers BJ (2006). Farm Nutrient Auditing: Support to PLANET (Benchmarking). Final report for Defra project ES0124. http://www.defra.gov.uk/science/default.htm

Goodlass G, Cottrill BR, Chambers BJ, Richardson SJ, Scholefield D, Goulding K and Oborn J (2006). Farm-gate nitrogen and phosphorus balances. Delivering sustainability within profitable farming systems – is it possible? Aspects of Applied Biology 80, 1-8.

Lord E, Shepherd M, Silgram M, Goodlass G, Gooday R, Anthony SG, Davison P and Hodgkinson R (2007). Investigating the effectiveness of NVZ Action Programme measures: Development of a strategy for England. Final report for Defra project WT03017. http://www.defra.gov.uk/science/default.htm

Oborn I, Edwards A C and Oenema O (2005). Nutrient balance; pros and cons. In: Proceedings of an International Workshop, ISPRA, June 28-29, 2005, Italy: Where do fertilisers go?. F Bouraoui, B Grizzetti, D Mulligan and L Galbiati (Eds.), pp. 9.1-9.15.

THE IMPACTS ON WATER QUALITY AND RESOURCES OF REVERTING ARABLE LAND TO GRASSLAND

JR Williams[1], E Sagoo[1], BJ Chambers[2], RB Cross[1], J Short[1] and RA Hodgkinson[2]

[1]ADAS Boxworth, Battlegate Road, Boxworth, Cambridge, CB23 4NN, UK, E-mail: john.williams@adas.co.uk; [2]ADAS Gleadthorpe, Meden Vale, Mansfield, Notts, NG20 9PF, UK

SUMMARY

The effects of reverting long-term arable land to grassland on drainage water volumes, nitrate leaching losses and topsoil total carbon and nitrogen storage were studied on a drained clay soil in Oxfordshire. Nitrate leaching losses from arable reversion grassland were *c.*8-fold lower than from land maintained in arable production. However, drainage volumes from arable reversion grassland (mean volume = 74 mm) were on average 40% lower than from arable land (mean volume = 121 mm). Topsoil (0-15 cm) total organic carbon content increased ($P<0.001$) by 24% and total nitrogen content ($P<0.001$) by 17% after 6 years of arable reversion to grassland. The increase in topsoil organic carbon storage was equivalent to 15 t/ha (i.e. 2.5 t/ha/year) and total nitrogen 1.4 t/ha (i.e. 0.23 t/ha/year). This work has provided valuable quantitative data on the effects of land use change on water quality and resources, and soil carbon/nitrogen storage, which will be valuable in assessing the environmental impacts of land use change.

INTRODUCTION

Nitrate losses from agricultural land are estimated to account for *c.*60% of nitrate entering water systems in England and Wales (Defra, 2004). Arable crop production, which covers *c.*5 million ha of agricultural land in the UK, is particularly nitrate 'leaky' (Goulding, 2000). Autumn cultivation stimulates the mineralisation of soil organic nitrogen (Silgram and Shepherd, 1999), which coupled with low crop N uptake before the onset of winter drainage, leaves a large pool of soil nitrate vulnerable to over-winter loss by leaching. Reverting arable land to long-term (permanent) grassland has the potential to reduce nitrate leaching by retaining nitrogen in accumulated soil N reserves. In addition, grass N uptake in the autumn may limit the amount of soil N at risk of loss by nitrate leaching. Arable reversion to (permanent) grassland also has other potential environmental benefits, such as increasing soil carbon storage and the presence of permanent grass crop cover during the autumn/winter period may reduce soil erosion. However, increased evapo-transpiration losses and canopy interception of rainwater on grassland compared with winter cereal cropped arable land, may reduce over-winter drainage volumes and thereby deplete aquifer recharge and surface water supplies.

MATERIALS AND METHODS

The experiment was undertaken at the Faringdon experimental site in Oxfordshire on a heavy clay soil (54% clay) of the Denchworth Association. The site consists of 18 hydrologically isolated plots (48 m x 40 m) which were in continuous arable production for more than 20 years until autumn 2001, when grass was established on 9 plots. All plots are drained with pipes at 90 cm depth and 48 m spacing, with gravel backfill to within 30 cm of the surface. The drainage system is supplemented with mole drains at 50 cm depth and 2 m spacing.

Winter wheat was established on the arable plots in the autumn of each study year and grass on the arable reversion grassland plots was cut for silage in late May/ early June of each year and grazed with sheep in late summer 2003 and 2004. Cattle slurry was applied in autumn, winter and spring to both the arable and arable reversion grassland plots in harvest years 2003, 2004, 2005 and 2006, using a 11 m^3 Joskin slurry tanker fitted with a 12m wide trailing hose boom. The mean cattle slurry application rate was 45 m^3/ha supplying a mean of 124 kg/ha total N, 63 kg/ ha ammonium-N and 1.72 t/ha of organic carbon (C). Inorganic fertiliser N was applied to all plots at standard recommended rates (Anon, 2000), taking account of the readily available N supplied by the slurry applications, to ensure that crop growth was similar on all the winter wheat and grass plots and representative of good commercial practice. The mean inorganic fertiliser N rate on the winter wheat crops was 170 kg/ha N and on the grassland plots 198 kg/ha N.

Drainage (and surface runoff) volumes were measured continuously using V-notch weirs from all plots over-winter 2003/04, 2004/05 and 2005/06. Drainage water samples were collected on a flow proportional basis using automatic water samplers from all the plots and analysed for nitrate-N and sediment. Soil mineral nitrogen samples (0-90 cm depth) were taken from all the treatment plots in autumns 2003, 2004 and 2005. Topsoil samples (0-15 cm) were taken from all the treatment plots in autumn 2001 and autumn 2007 and analysed for total organic C and total N.

RESULTS AND DISCUSSION

(i) Drainage Water Volumes

Drainage water volumes in all three seasons were relatively low, reflecting drier than average over-winter rainfall volumes in each year. Mean drainflow volumes from the arable plots (Figure 1) were 130 mm in 2003/04, 125 mm in 2004/05 and 109 mm in 2005/06 (mean = 121 mm). Mean drainflow volumes from the arable reversion grassland plots (Figure 1) were 77 mm in 2003/04, 91 mm in 2004/05 and 55 mm in 2005/06 (mean = 74 mm). Over the three drainage seasons, drainflow volumes from the arable reversion grassland plots was on average 47 mm (c.40%) lower than from the arable plots. Drainflow from the arable reversion grassland plots began 1-2 weeks later than the start of drainage from the arable plots. The lower drainage volumes and later start of drainage on the arable reversion grassland plots reflected greater evapo-transpiration losses and associated soil moisture deficit compared with the arable plots.

In each study year, surface runoff volumes were low reflecting the relatively low amount of over-winter rainfall. Surface runoff was only generated following large (typically >10 mm) rainfall events when soils were 'wet' in winter and early spring. Mean surface runoff volumes were similar from both the arable and arable reversion grassland plots at 5 mm in 2003/04, 7 mm in 2004/05 and 4 mm in 2005/06.

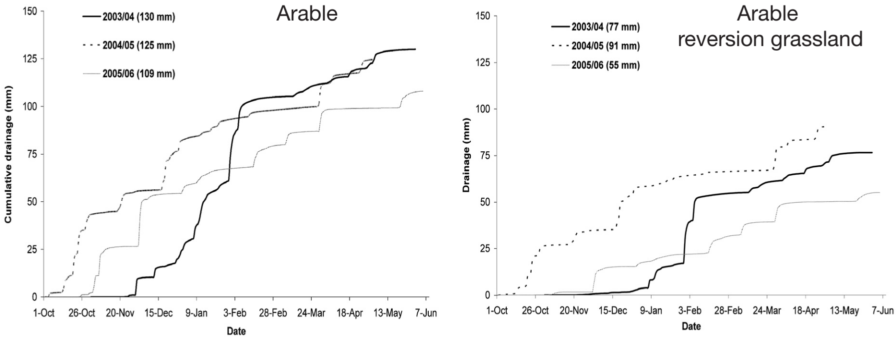


Figure 1: Over-winter drainage volumes from arable and arable reversion grassland plots (2003/04 – 2005/06)

(ii) Nitrate Leaching Losses

In all three study years, drainage water nitrate concentrations were highest (P<0.05) from the arable plots with mean peak nitrate concentrations at the start of drainage in the autumn/winter period ranging between *c.*60 and 130 mg/l NO_3-N, compared with peak concentrations of 4-15 mg/l NO_3N from the arable reversion grassland plots (Figure 2).

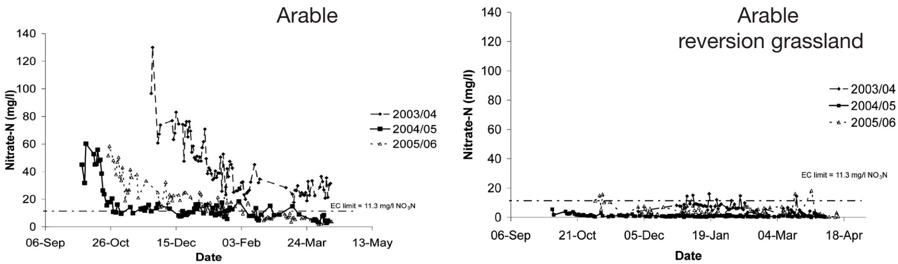


Figure 2: Nitrate concentrations in drainage waters from arable and arable reversion grassland plots (2003/04 – 2005/06)

Total mean overwinter nitrate leaching losses from the arable plots at 42 kg/ha N in 2003/04, 16 kg/ha N in 2004/05 and 19 kg/ha N in 2005/06 were greater (P<0.05) than from the arable reversion grassland plots at 6, 1 and 2 kg/ha N in 2003/04, 2004/05 and 2005/06, respectively (Figure 3). On average, nitrate leaching losses from the arable plots were *c.*8-fold greater than from the arable reversion grassland plots. Annual sediment losses in drainage water from the arable plots ranged between 82 and 259 kg/ha and were not different (P>0.05) to sediment losses from the arable reversion grassland plots (range 90-290 kg/ha).

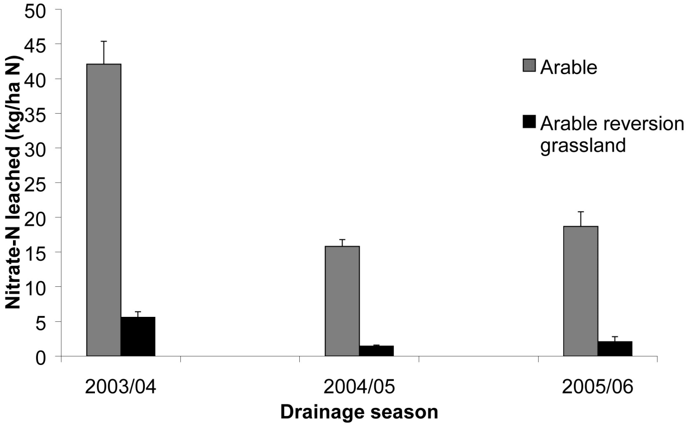


Figure 3: Nitrate-N leached from arable and arable reversion grassland plots (2003/04-2005/06)

The higher nitrate concentrations and nitrate leaching losses from the arable plots reflected their higher autumn soil mineral nitrogen content compared with the arable reversion grassland plots (Table 1). The lower soil mineral N content was most probably a reflection of N immobilisation in accumulating soil organic matter on the arable reversion grassland plots, along with greater N uptake (*c.*20 kg/ha N) by the grass compared with the immature winter wheat crop (<5 kg/ha N) on the arable plots during the autumn period before the start of over-winter drainage.

Table 1: Autumn soil mineral nitrogen content (0-90 cm depth)

Landuse	2003	Date 2004 kg/ha N	2005
Arable	130 (10.9)	59 (7.4)	50 (7.5)
Arable reversion grassland	85 (10.4)	42 (7.4)	21 (2.1)

Figures in brackets ( ) are standard errors of the mean

(iii) Topsoil Total N and Total Organic C Content

In autumn 2001, when the arable reversion grassland plots were established, the overall site mean topsoil (0-15 cm) total N content was 0.34% (Figure 4). In autumn 2007, 6 years after the grass had been established, the mean topsoil total N content on the arable reversion grassland plots was 0.41%, which was 17% higher (P<0.001) than on the arable plots (0.34%). The increase in topsoil total N was equivalent to 1.4 t/ha (i.e. 0.23 t/ha/year) assuming a soil bulk density of 1.33 t/m^3.

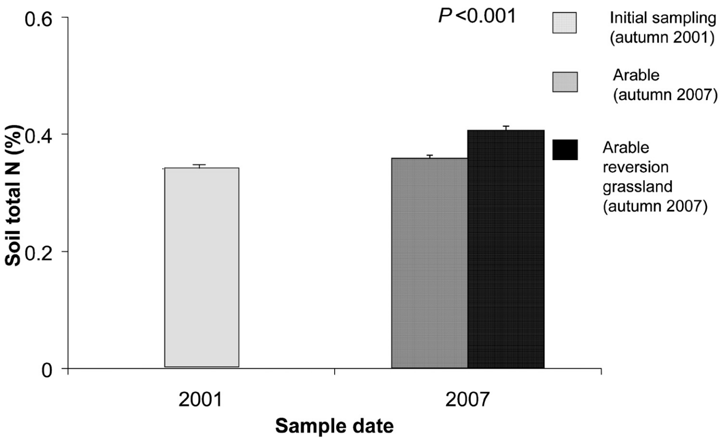


Figure 4: Changes in total topsoil N content on arable and arable reversion grassland plots between 2001 and 2007

The reversion of arable land to grassland also led to increases in topsoil organic C content (Figure 5). The mean topsoil (0-15 cm depth) organic C content over all the plots in autumn 2001, when the grass was established on the arable reversion plots was 2.88%. In 2007, the topsoil organic carbon content on the arable reversion grassland plots was 3.78%, which was 24% higher (P<0.001) than on the arable plots (3.04%). The increase in topsoil carbon storage on the arable reversion grassland plots was equivalent to 15 t/ha (i.e. 2.5 t/ha/year) assuming a soil bulk density of 1.33 t/m^3, and was *c.*4-fold greater than the annual net accumulation (*c.*0.6 t/ha/year) estimated following farm manure applications supplying 250 kg/ha total N (Chambers *et al.,* 2008). If the increase in soil carbon could be 'credited' based on a social cost of carbon of £70/tonne (Clarkson and Deys, 2002), the C stored as a result of arable reversion to grassland would be worth *c.*£175/ha/annum.

The increased topsoil total N and C contents under arable reversion grassland were probably due to a combination of the lack of cultivation coupled with the build up of organic matter under permanent grassland management, which allowed N and C to accumulate in the soil biomass. In contrast, annual cultivation of the arable plots would have stimulated the oxidation and breakdown of soil organic matter. Moreover, N accumulation in soil organic matter reserves decreased soil mineral N contents (Table 1) and over-winter nitrate leaching losses (Figures 2 and 3)

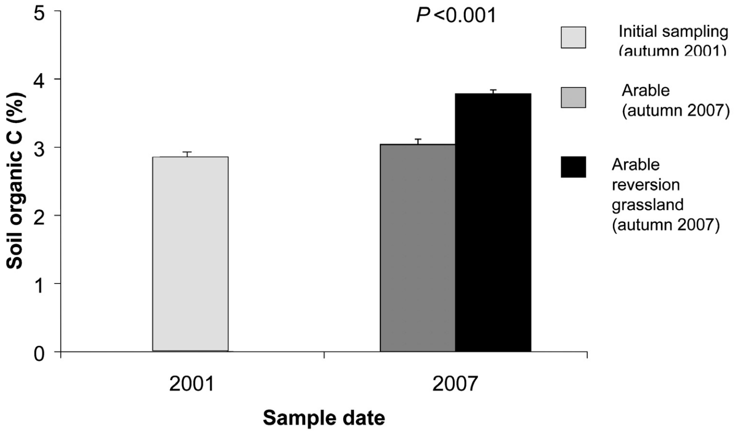


Figure 5: Changes in total topsoil organic carbon on arable and arable reversion grassland plots between 2001 and 2007

CONCLUSIONS

The results from this study show that reverting arable land to (permanent) grassland will reduce nitrate leaching losses. In addition, arable reversion can bring environmental benefits in terms of increased soil C storage (mitigating carbon dioxide emissions to the atmosphere) and N storage (mitigating nitrate leaching losses). However, drainage volumes from arable reversion grassland were on average 40% lower than from arable land. The results of this work provide quanitative data on the implications of reverting arable land to permanent grassland which can be used to assess the environmental impacts of land use change, for example, as part of environmental stewardship, groundwater protection zone schemes, etc.

ACKNOWLEDGEMENTS

Funding for this work from the Department for Environment, Food and Rural Affairs (Defra) is gratefully acknowledged.

REFERENCES

Anon (2000). Fertiliser Recommendations for Agricultural and Horticultural Crops. Defra Reference Book 209 (Seventh Edition). The Stationery Office, Norwich.

Clarkson R and Deyes K (2002). Estimating the Social Cost of Carbon Emissions. GES Working Paper 140, London, HM Treasury. http://hm.treasury.gov.uk/Documents/Taxation_Work__and_Welfare/Tax_and_the_environment/tax_environ_index.cfm

Chambers BJ, Bhogal A, Whitmore A and Powlson DS (2008). The potential to increase carbon storage in agricultural soils (this volume).

Defra (2004). Updating an Estimate of the Source Apportionment of Nitrogen to Waters in England and Wales - Phase 2 Report to the Department for Environment, Food and Rural Affairs. London, pp. 34.

Goulding K (2000). Nitrate leaching from arable and horticultural land. Soil Use and Management 16, 145-151.

Silgram M and Shepherd M (1999). The effect of cultivation on soil nitrogen mineralization. Advances in Agronomy 65, 267-311.

NOTES

NOTES

www.sac.ac.uk
www.sepa.org.uk

ISBN 1 85482 867 3

£35.00