Contents

CEH core strategic programmes

This core strategic programme comprises five distinct projects:

- ***acidifying pollutants,***
- ***organic pollutants,***
- ***toxic metals,***
- ***radionuclides, and***
- ***photochemical oxidants.***

Pollution assessment and control

Current risk assessment and risk management of chemicals rely on a variety of approaches ranging from multi-tiered, multi-media systems (pesticides), through pragmatic iterative risk management (atmospheric pollutants) to first-tier conservative systems of risk assessment (industrial chemicals). Each system operates with models of varying complexity and varying comprehensiveness; many such models lack field validation and none fully integrate fate and behaviour with bioavailability and biological effect. As a consequence, the dose–response relationships for pollutants are frequently precise (within a single model) but not accurate (reflecting the true field situation). The overall aim of this programme is to move closer to such accuracy by further integration.

The acidifying pollutants project continues to develop existing models of atmospheric transport and deposition and to integrate these models with studies of effects of deposited nitrogen and sulphur on terrestrial and aquatic ecosystems. The work allows the development of overall models of deposition, effect and recovery on a variety of scales.

The longer-term strategy is to move away from the current emphasis on emission control towards assessment of recovery of natural systems. The article by Emmett illustrates the approach and covers field validation of a catchment-scale nitrogen leaching model during development of forest stands of Sitka spruce on Welsh moorlands. This particular field study demonstrates good model agreement with observed change and that the system is already nitrogen saturated; increased nitrogen deposition will lead to leached nitrogen and acidification of waters drained from the area of the developing forest. The work involves a wide range of both national and international collaborators, reflecting global interest in the problem.

Work on organic pollutants and toxic metals continues to inform the regulation of chemicals, and Institute staff are involved in:

- monitoring for chemical residues in environmental media and biota,
- design of new ecotoxicological methods,
- hazard identification and risk characterisation both nationally and internationally.

There is increasing interest in the biological means of monitoring the extent of contamination and its biological significance through the use of biomarker techniques Two such methods are presented in the article by Meharg and colleagues, both are collaborative projects Contamination of land resulting from either waste disposal with inadequate containment or industrial accidents may pose a risk both to organisms in the environment and, ultimately, by direct or indirect exposure, to human health Rapid techniques which indicate both presence and bioavailability of released chemicals are needed as chemical measurement alone can give misleading results on true exposure Insertion of a gene for bioluminescence into a common soil bacterium gives a sensitive marker for effects on aerobic respiration Both laboratory and field experiments have demonstrated the utility of the organism in responding to benzene contamination, and its sensitivity to changes in bioavailability resulting from interaction of the solvent with soil particulates Similarly, two biomarkers in earthworms have shown utility in exposure assessment, preliminary results suggest correlation with ecologically relevant endpoints, the ultimate aim of biomarker studies

There has been increased interest in the rehabilitation of contaminated land through bioremediation and restoration techniques The article by Harrison and Daniels reports that oiled beach material readily degrades when incorporated into sandy soil, microbial populations develop degrading capability, there is low mobility of hydrocarbons to groundwater, and sites become rehabilitated with suitable vegetation The study, which is not yet complete, has also determined some of the conditions which facilitate degradation

Following on from previous studies which established models for uptake of deposited radionuclides from the Chernobyl accident in both the UK and the former Soviet Union, studies have been conducted on dietary radionuclides deriving from emissions to the atmosphere from the Sellafield complex in Cumbria A radiological surveillance programme at the Ministry of Agriculture, Fisheries and Food uses both monitoring of dietary components and computer simulations of dispersion and incorporation into foodstuffs A recent study selected adults and children living near Sellafield on the basis of their consumption of home or locally grown produce, and compared them to a control group Radiochemical analysis of diets for ^{239,240}Pu, ^{137}Cs, ^{90}Sr, ^{14}C and ^{129}I was conducted over a one-week period and extrapolated to annual intakes The results validated current models and demonstrated that doses are significantly less than recommended annual limits

Photochemical oxidants will continue to be major pollutants of the atmosphere, both urban and rural, for the foreseeable future As emission controls for nitrogen oxides and volatile organic compounds (VOCs) are implemented, ozone concentrations in cities may increase, though rural concentrations should fall There is considerable uncertainty about the extent and location of benefits of emission control, and there is a need to establish dose–response relationships for vegetation, materials and human health A consortium of university and Institute researchers have been involved in the ACSOE 'OXIdising Capacity of the Oceanic Atmosphere' (OXICOA) project, designed to investigate atmospheric chemistry in clean marine air and in polluted continental air in western Ireland Measurements of a wide range of trace gases, particles and meteorological data provided a framework for the modelling of highly reactive free-radicals such as OH (hydroxy) and RO_2 (peroxy) Direct measurement, using laser spectroscopy, was conducted for the first time by a UK group Models were run to predict gas-phase concentrations and generally good agreement was found for many chemical species In particular, modelled and measured free-radical concentrations gave credence to the models and the measurement techniques Clear evidence emerged of tropospheric ozone production in polluted air and ozone destruction in marine air arriving from the tropics There was also evidence of transport of trace gases across the Atlantic from North America

The pollution research programme within ITE has been augmented by funding under NERC's Environmental Diagnostics thematic programme Six research projects received awards, and ITE was also given responsibilty for both the programme and data management activities A major future development in pollution research will involve co-operation across Research Councils in the study of environmental exposure to chemicals and effects on human health

S Dobson

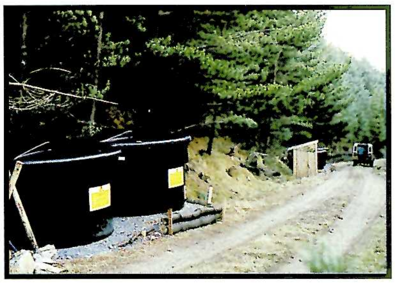

Plate 7. Large storage tanks at Aber forest to hold the water which is mixed with nitrogen-containing salts and applied weekly to the experimental plots

Modelling the ecosystem effects of nitrogen deposition in Welsh Sitka spruce stands

(The experimental work was funded by the National Power/Powergen Joint Environment Programme, the EC and the DETR. Model development was funded by the Norwegian Water Research Institute, Macaulay Land Use Research Institute, University of Amsterdam, NERC, a US EPA grant to the University of Virginia and a US Department of Energy grant to E&S Environmental Chemistry Inc)

In recent years the occurrence of nitrogen saturation, as indicated by increased nitrogen concentrations in natural waters, has increased, possibly due to greater atmospheric nitrogen deposition in many parts of North America and Europe. Enhanced nitrogen in natural waters can result in eutrophication and acidification in some sensitive areas. This can be detrimental to stream biota and may delay the recovery of some acidified waters in response to reductions of sulphur emissions. Concern in the UK has therefore recently focused on determining the present and future role of atmospheric nitrogen deposition in the acidification of freshwaters.

There are many processes which influence the flux of nitrogen in forested ecosystems. Feedbacks between plants and soils are of key importance, and a modelling approach enables these complex processes and feedbacks to be integrated. In addition, calibration of models to available data enables predictions for future streamwater nitrate concentrations and the consequences of a changing pollution climate or land use to be investigated.

ITE has been involved in testing and developing a new catchment-scale nitrogen leaching 'Model of Ecosystem Retention and Loss of Inorganic Nitrogen' (MERLIN) (Cosby *et al.* 1997), created by the University of Virginia in collaboration with the Institute of Hydrology, Norwegian Water Research Institute, Macaulay Land Use Research Institute, and the University of Amsterdam. Extensive datasets collected by ITE were used to test the model, and to assess the current fate of nitrogen in Sitka spruce (*Picea sitchensis*) stands in Wales and the likely contribution of nitrogen to acidification of streamwaters in the future (Emmett *et al.* 1997).

MERLIN is a catchment-scale mass balance model of linked carbon and nitrogen cycling (Figure 13). Fluxes of nitrogen are controlled by carbon productivity, by the C/N ratios of organic compartments, and by inorganic-N availability in soil solution. Inputs to the model are temporal sequences of carbon fluxes and pools, hydrological discharge, external nitrogen sources, and current nitrogen pools in the vegetation and soil. The number of inputs was intended to be compatible with those usually recorded in forest studies. Outputs available from the model are the concentrations and fluxes of inorganic nitrogen in runoff and carbon/nitrogen ratios of the plant and soil pools, and rates of fluxes through the soil microbial populations.

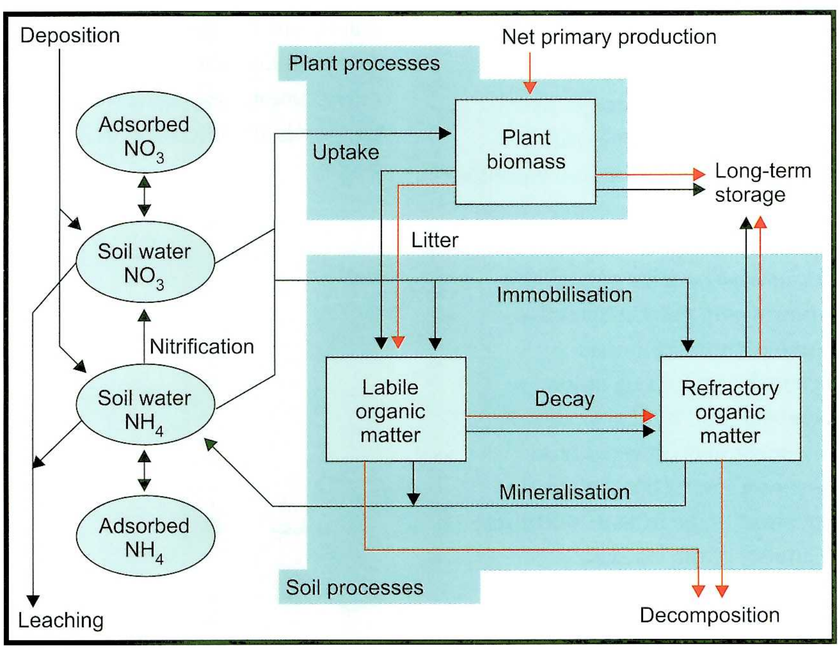

Figure 13. Pools and fluxes included in the MERLIN model (Cosby *et al.* 1997) (red arrows=carbon; black arrows=nitrogen)

Information collected by ITE across an age sequence of spruce stands through Wales (Figure 14), in combination with detailed information at an intensive experimental site in Aber forest, north Wales (Figure 15; Plate 7), provided information on the changes in carbon and nitrogen pools and fluxes as a Welsh moorland was converted to a mature forest stand. In the calibration procedure for application to the Aber site, the nitrogen uptake parameters were adjusted to match C/N ratios and N losses observed or inferred for the moorland ecosystem. The calibrated model was then used to simulate forest growth. A change in the uptake functions to a new 'forest' set of parameters was found necessary, indicating a change in the relative sensitivity of plants and soil microbes to internal and external C/N ratios in moorland and forest ecosystems.

An independent test of the model was carried out. The model's input was adjusted to account for an increase of 140% in nitrogen inputs and then run without further modification. This increase in deposition matched the experimental enhancement of nitrogen inputs at the Aber experimental site between 1990 and 1995. The output from the model was compared to the observed changes in nitrogen leaching losses and C/N ratios in soil and vegetation during this period.

MERLIN correctly predicted that most of the applied nitrogen was leached, with an immediate and significant increase in nitrogen leaching losses. This prediction indicates that the stand is already nitrogen saturated and any increase in nitrogen deposition may contribute to the acidification of waters draining from the stand. Little change in C/N ratios in vegetation, and the slow response in soil nitrogen transformations also agreed with the observed data. As a successful simulation was achieved without changes in uptake parameters, the changes in ecosystem functioning in response to enhanced nitrogen deposition are obviously small relative to the change following afforestation. It also suggests that MERLIN may be used to predict the response to future changes in deposition loadings. This will be a component of a new EC-funded project 'DYNAmic MOdels to predict and scale up the impact of environmental change on biogeochemical cycling' (DYNAMO), which will include further testing of the MERLIN model at many other forested experimental sites in Europe, and link output of this model to the well-established acidification model, MAGIC.

B A Emmett

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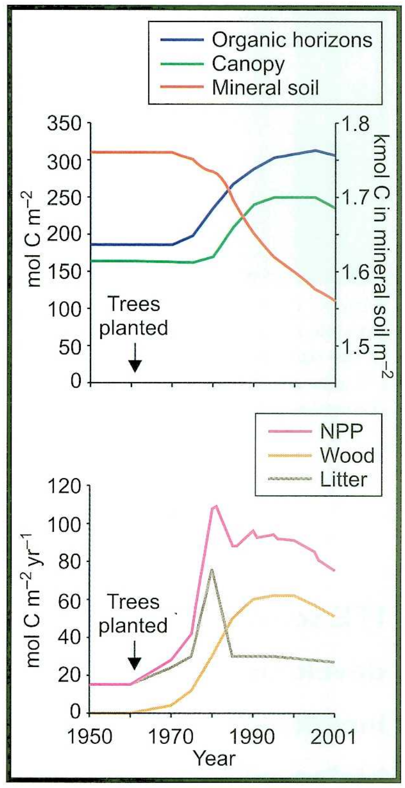

Figure 14. Changes in carbon pools and fluxes following afforestation of a moorland (NPP=net primary production)

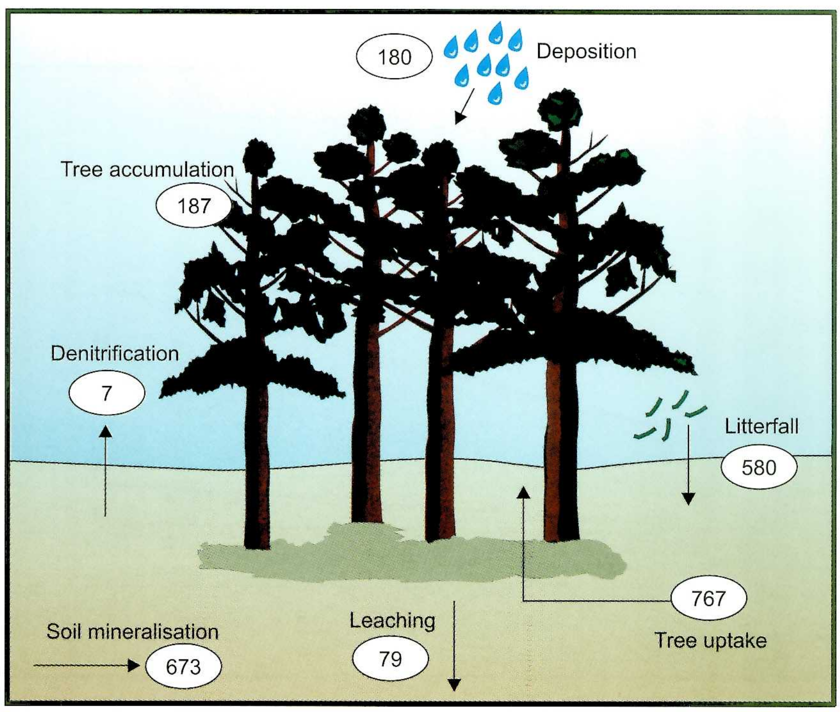

Figure 15. Nitrogen cycling at the Aber experimental site (mmol m^{-2} yr^{-1})

Plate 8. Poor understanding of the possible impacts of industrial practices in the past has, in some cases, produced a legacy of land and groundwater contamination

ITE scientists have been developing and validating inexpensive, rapid, biologically based tests to determine the toxicity and biological availability of chemicals in soils at a range of sites in the UK.

Rapid 'bioassays' that screen for chemical impacts on soil organisms

One legacy of industrial development can be the contamination of land and its associated groundwater resources with a wide range of chemicals, some of which are toxic or sufficiently mobile to move considerable distances off-site. Often, this contamination arises from past practices that were used because knowledge of potential impacts on the health of people and the environment was limited (Plate 8). It is now recognised that the level of contamination can be high enough to pose a risk. In these circumstances, expensive clean-up and reclamation costs may have to be borne by the landowner before a contaminated site can be used. Costs are often made even higher by the need to analyse for – and treat – very complex mixtures of chemicals. Hitherto, one real practical difficulty faced by site owners and operators was the absence of inexpensive, biologically based tests that could be used to determine rapidly the nature, extent and degree of contamination. Without such tests it may be difficult to know the biological significance of the contamination.

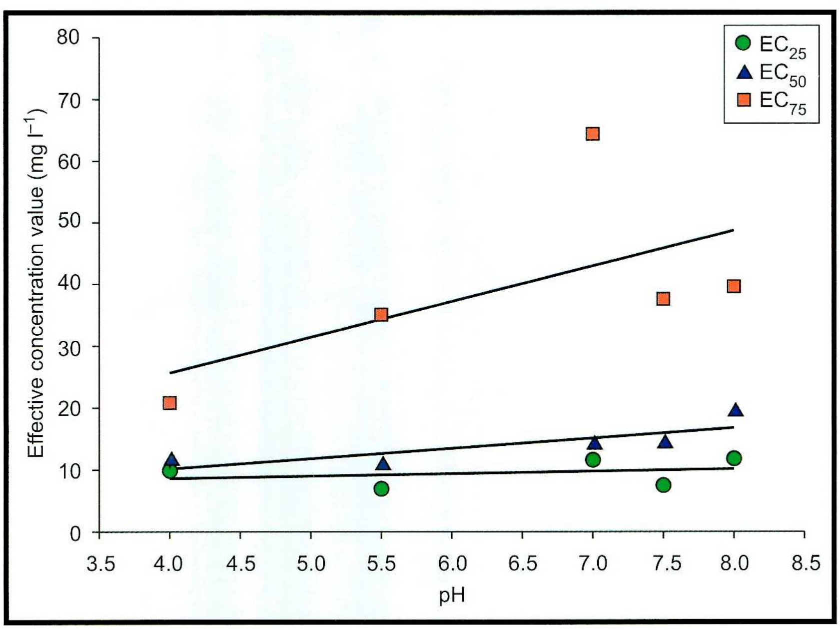

Figure 16. Effective concentrations, EC_{25}, EC_{50} and EC_{75} values, determined for the exposure of *lux*-marked *P. fluorescens* to benzene at increasing pH

In co-operation with researchers in universities, and also by building on the work of colleagues in other NERC Institutes, ITE scientists have been developing and validating inexpensive, rapid, biologically based tests to determine the toxicity and biological availability of chemicals in soils at a range of sites in the UK. Two examples are provided below. The first example is a bacterial biosensor. In this case, a genetically modified soil bacteria reacts to the presence of biologically available, organic, toxic chemicals by modifying light output. The second example concerns biomarker responses in earthworms (*Lumbricus castaneus*). Here, the vitality of cellular biochemistry is determined by the use of a red dye after worms become exposed to toxic material (eg cadmium), and, in a related test, immune competence is assessed.

Bacterial biosensors

A number of bacterial biosensor 'bioassays' have been developed at ITE Monks Wood in collaboration with the University of Aberdeen. They have been applied to studies of the toxicity of common aromatic pollutants (Boyd *et al.* 1997a, b). *Lux* genes cloned from bioluminescent marine bacteria have been inserted into the genome of a range of terrestrial bacteria. Bacterial bioluminescence is regulated by aerobic respiration. Any toxic chemical which interferes with this cell function will result in changes in luminescence. The assays utilise terrestrial bacteria, giving the tests ecological relevance to soils, groundwaters and freshwaters.

To illustrate the use of the *lux* gene in toxicity screening, *lux*-marked *Pseudomonas fluorescens* (a common soil bacterium) was used to assess the toxicity of benzene in solution and in soils. Figure 16 shows the concentrations of chemical that caused inhibition of luminescence after 20-minute exposure of *P. fluorescens* to benzene solutions buffered at different pHs (pH often influences biological activity). The effective concentrations producing 25% (EC_{25}) and 50% (EC_{50}) of the maximum response were relatively insensitive to a wide range of pHs, indicating that there is no toxicological interaction between pH and benzene with respect to *P. fluorescens*. However, the results for the response at 75% (EC_{75}) of maximum indicate that *P. fluorescens* may be more sensitive at low pHs for low levels of exposure. Benzene was toxic to *P. fluorescens,* with EC_{50} values around 10 mg l^{-1}.

To demonstrate how changing bioavailability attenuates the toxicity of chemicals in soils, the results of an *in situ* study utilising *lux*-marked *P. fluorescens* are illustrated in Figure 17. Air-dried soil was incubated for either 2 or 24 hours with benzene, and then hydrated to 80% of its water-holding capacity using a suspension of *lux*-marked *P. fluorescens*. The hydrated soil was then incubated for 20 minutes and a portion of the added cells removed by shaking the soil to give a suspension, followed by filtration. The filter retained soil particles but allowed the bacteria to pass through. The luminescence of the filtrate was then assayed. The benzene concentration of the soil was assayed at 2 and 24 hours, and luminescence was related to these concentrations. Three soils were used in the study; the soils were from the same site, but had been maintained at different pHs for over 30 years by the addition of lime or aluminium sulphate.

After 2 hours, benzene at high soil concentrations was toxic to *P. fluorescens* as assayed by luminescence, yet after 24 hours equivalent concentrations had no toxic effect at all, indicating considerable changes in bioavailability. These changes are thought to occur as a result of diffusion of benzene into inaccessible pores in the soil particles and partitioning into soil organic matter. While pH did not interact with benzene toxicity in solution (Figure 16), decreasing pH increased toxicity in soil at the 2-hour exposure period (Figure 17). It appears that the physiological condition of the bacteria is altered by the chemical environment in soil at different pHs (pH by itself does not affect toxicity; see Figure 16), resulting in enhanced toxicity of benzene at lower pHs.

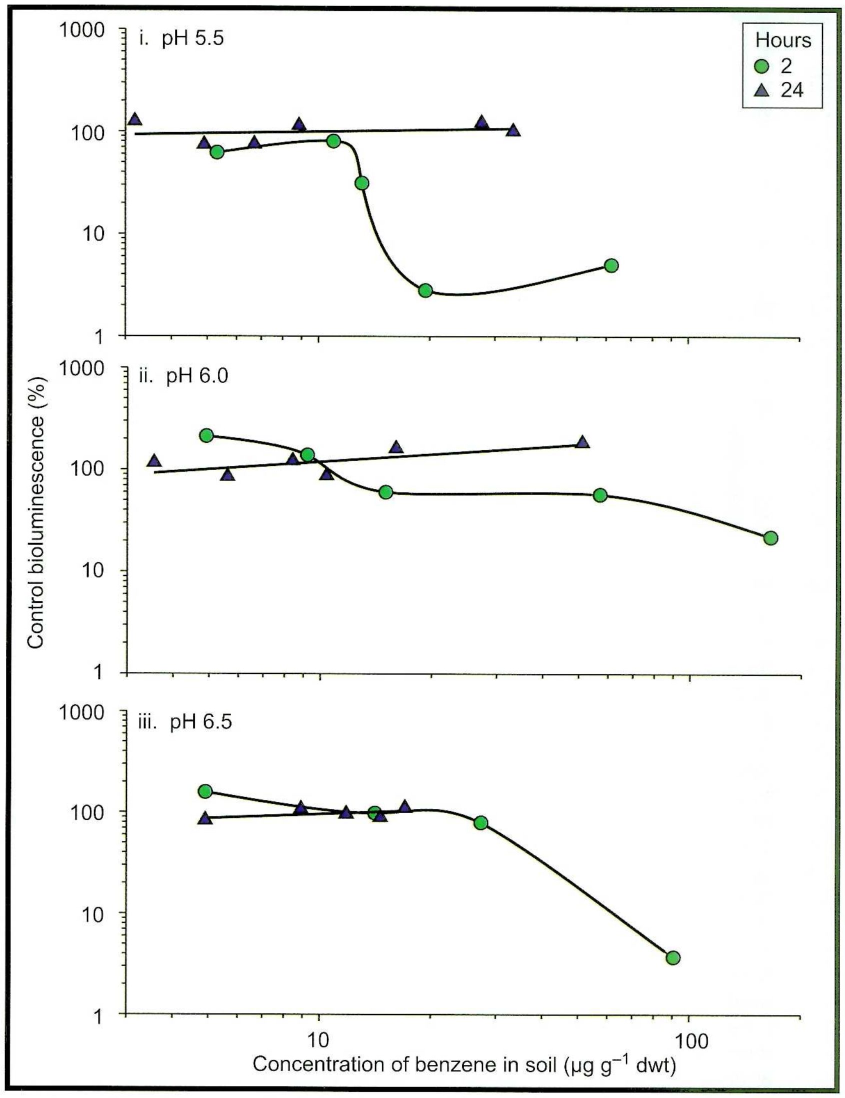

Figure 17. Exposure of *lux*-marked *P. fluorescens* to benzene in direct contact with soil adjusted to pH 5.5, 6.0 and 6.5 measured after 2 and 24 hours. Each point is the mean of three replicates and lines have been mathematically fitted

Bioavailability is crucial to understanding the toxicity of chemicals

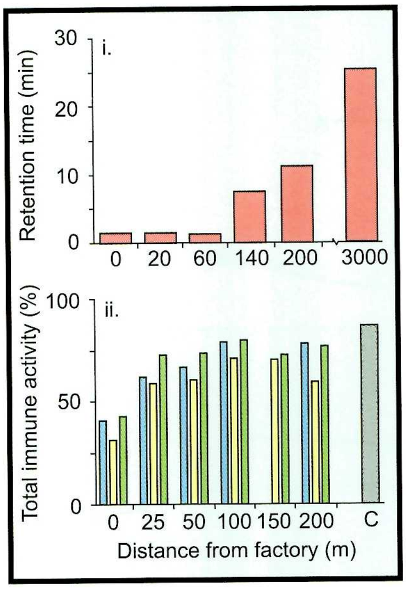

Figure 18. Responses of biomarkers at various distances from the site of an industrial accident. The site is contaminated with a wide range of materials whose total concentration in animals decreases as distance from the source of contamination increases. For example, total toxic metal concentrations in worms are highest (>800 mg kg^{-1} dry weight) at the factory perimeter (0 m) where few worms were found, and begin to approach UK background concentrations at 200 m

i. Mean neutral-red retention times (min) for earthworms (*Lumbricus castaneus*) sampled along a transect with distance from the factory perimeter
ii. Mean total immune system activity (as % active cells of total cells counted) for earthworms (*Lumbricus castaneus*) collected at the respective distances along three replicate transects together with a control (C) at Monks Wood

in the environment. This study showed the utility of *in situ* toxicity assays in soils to address questions concerning pollutant bioavailability.

Earthworm coelomocyte bioassays

Two biomarkers that react to a range of toxic chemicals were developed in earlier work at the NERC Plymouth Marine Laboratory and in European universities. These have now been modified further by ITE for use in earthworms (*Lumbricus rubellus*) (eg Svendsen *et al.* 1996). One biomarker depends on the 'destabilisation' of membranes around lysomes (subcellular structures that are involved in the metabolism of waste and toxic material). In this 'bioassay' of the toxicity of contaminants in soil, the key measure is the ability of the lysosomes to retain a dye (neutral-red) using the technique of Weeks and Svendsen (1996). The second earthworm 'bioassay' provides a measure of immune function, and is based on incubating earthworm coelomocytes with mammalian red blood cells. Both 'bioassays' are relatively simple to undertake. In a practical test of these 'bioassays', earthworms were collected along a 200 m gradient of contamination resulting from an industrial accident. Samples of body fluid were taken from individual worms and lysosomal membrane stability was measured by assessing the time until dye leakage. The competence of the immune system was determined, after incubating worm cells with rabbit red blood cells, by recording the proportion of worm cells which exhibited any immune reaction.

The utility of these types of 'bioassays' would be considerably enhanced if it could be shown that changes in biomarker responses are related to ecological factors. In this case, the relationship between the 'bioassay' responses and earthworm population data was examined close to an industrial source of metals and organic contaminants. Results (Figure 18) showed that neutral-red retention times increased as one moved away from the source of contamination, and thus down the concentration gradient of the contamination. Decreased immune system activities were measured in earthworms collected very near to the source of the contamination. Effects on earthworm populations included reductions in adult numbers, cocoon production, and hence juvenile densities. It is clear that both assays can be successfully applied in the field, without the results being overshadowed by biotic and abiotic factors. The work on worm 'bioassays' currently forms the basis of a study with the Environment Agency, and is a key part of an EC-funded project investigating the toxicity of mixtures of chemicals.

A A Meharg, E M Boyd, C Svendsen, J Wright and J M Weeks

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Disposal of oiled beach material

(This work is being funded by the Marine Pollution Control Unit (now in the Department of Environment, Transport and the Regions)

This project is examining the potential for disposal of oiled beach material (OBM), derived from marine oil spills, by land farming and burial in suitable sandy coastal environments. The project is specifically addressing the potential for degradation of the weathered oil residue, hydrocarbon mobility to groundwater, and site rehabilitation, particularly revegetation. We are using three basic approaches:

- intensive trials at specific sites
- simpler studies at a range of sites around the UK coast
- manipulative studies in lysimeters at ITE Merlewood (see *1994–95 ITE Annual Report*, p50).

A 25 000 ton OBM deposit in dunes at Pendine Sands, Carmarthenshire, derived from a local oil spill, has been monitored since its formation in January 1994. Randomised block experiments to investigate the efficacy of land farming and burial disposal were established in early 1995 in dune/dune pasture environments at Eskmeals, Cumbria, using an artificially generated OBM containing *ca* 5% hydrocarbons derived from a Russian fuel oil. Most of the oil residues in the Pendine deposit have readily degraded, but there remains a small, more stable, slower-degrading component (Figure 19). Results from the Eskmeals experiments show that oil degradation in land farming plots is faster when OBM is incorporated in winter rather than summer. However, in the burial plots, degradation rates are very similar when incorporated in winter or summer. As a result of the experimental design, it has been found that degradation is faster in experimental blocks up-slope where the soil is drier compared to down-slope where the water table is slightly nearer the surface.

Plate 9. (Above) Planting marram grass (*Ammophila arenaria*) on the OBM deposit at Pendine Sands in November 1994. (Below) The plots and surrounding vegetation in October 1996 (reproduced by kind permission of the Defence Evaluation Research Agency, Pendine Sands)

Groundwater samples retrieved from piezometers situated adjacent to OBM in burial plots at Eskmeals indicate the low mobility of hydrocarbons from the weathered oil residues; polycyclic aromatic hydrocarbon (PAH) analyses of groundwaters showed no differences between control and

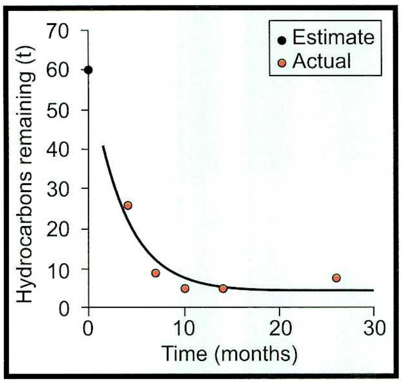


Figure 19. The degradation, expressed as hydrocarbons remaining, of weathered oil residues over time in the OBM deposit at Pendine Sands since January 1994

OBM plots. Revegetation both at Eskmeals and Pendine has generally been successful (Plate 9), though ground cover on the land farming plots at Eskmeals could be greater. Surface-dwelling invertebrate populations (eg beetles and spiders) appear to be recovering in the land farming experimental areas after two years; site disturbance, rather than the oil residues, is thought to have been responsible for the temporary declines in populations.

Variation in the rate of oil residue degradation has also been examined in artificial OBMs, containing *ca* 5% hydrocarbons derived from Russian fuel oil, when buried in 15 different dune pasture sites around the UK coast from Devon to the Shetland Isles. Site characteristics which may influence degradation rate (such as soil chemical and physical properties, soil microbial activity, vegetation cover, and temperature) have been recorded. Although this study is only part-way through, it appears that oil degradation is faster in OBMs buried in the autumn than in those buried in the spring; possible causes are being investigated.

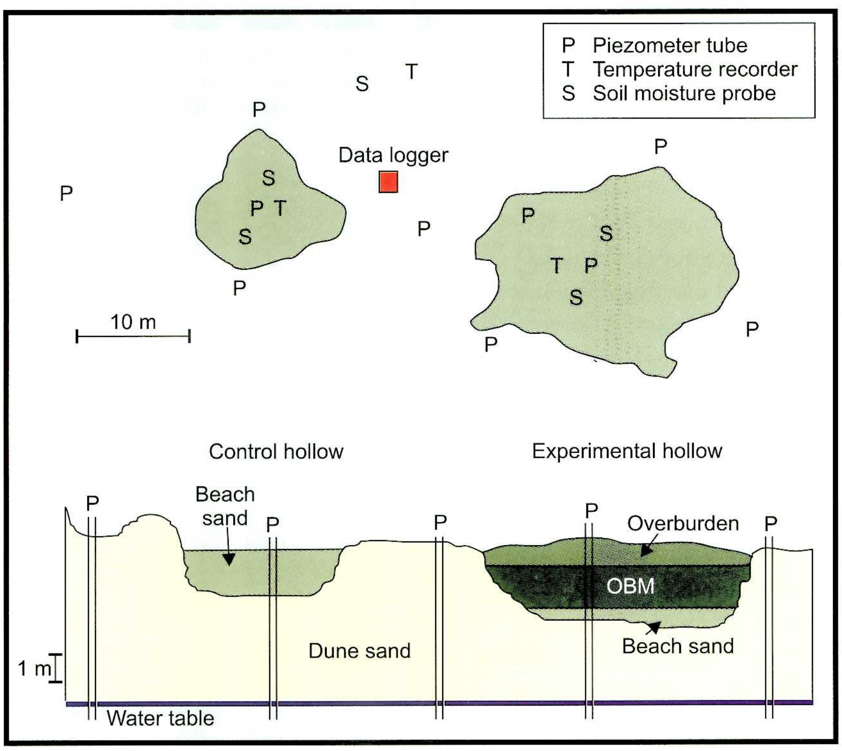


Figure 20. The design of experimental plots, constructed in two adjacent dune hollows at Pendine Sands, to test for hydrocarbon leaching to groundwater

Two manipulative lysimeter studies have been carried out. The first examined the rates of degradation and hydrocarbon mobility for OBMs derived from three different oils (Forties, Kuwait, and a medium fuel oil) given three different weathering treatments. The second examined the effects of different oil residue concentration on degradation rate. Both studies have confirmed that oil residues are readily degraded by natural microbial populations present in dune/dune pasture sands, and that degradation rate, as shown by reductions in hydrocarbon content and CO_2 emission rates, was in the order Forties>Kuwait>medium fuel oil. Weathering processes had the effect of reducing the hydrocarbon content of OBMs and making the residual fractions less degradable. The second lysimeter experiment indicated that there may be an optimum oil residue concentration around 7% for degradation processes; above this level, oil residues may block pore spaces within the OBM resulting in poorer aeration. The lysimeter studies also confirm that hydrocarbon mobility from weathered oil residues to groundwaters is very low.

More recently, a second experiment has been set up at Pendine Sands to test more critically the potential for transfer of hydrocarbons from weathered OBM to groundwater. Two dune hollows have been established for monitoring, one filled with beach sand to act as a control and the other filled with OBM containing a nominal 12% weathered oil residues The OBM was artificially prepared from 23 tonnes of medium fuel oil emulsified with an equal volume of sea water and spread on local beach sand, left for two days to weather, and then ploughed into the top layer of the sand The experimental design is shown as a diagram of the plan and vertical cross-section (Figure 20) The site has been instrumented to automatically record rainfall, fluctuations in water table and in soil moisture, and changes in temperature at different depths within sand and OBM placed in the two hollows Groundwater samples taken from a series of piezometer tubes in and around the hollows, and core samples of OBM and underlying sand, are being collected at regular intervals and analysed to track the fate of hydrocarbons

The preliminary conclusions from the project so far are that

- weathered oil residues in OBMs readily degrade when incorporated into sandy soils,
- natural microbial populations rapidly develop hydrocarbon-degrading capabilities,
- there is low mobility of hydrocarbons to groundwater from weathered oil residues, and
- sites can become rehabilitated with suitable vegetation

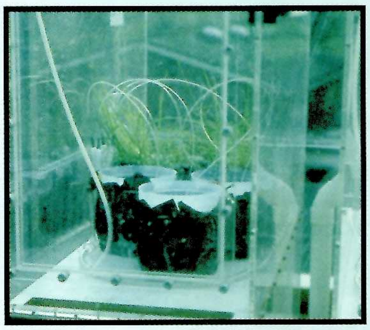
Weathered oil residues in OBMs readily degrade when incorporated into sandy soils ... and sites can become rehabilitated with suitable vegetation.

Acknowledgements

We wish to thank the Defence Evaluation and Research Agency at Eskmeals, Cumbria, and at Pendine Sands, Carmarthenshire, for its valued assistance in providing sites for these studies

A F Harrison and R E Daniels

The quantitative estimation of the risks to ecological systems from a range of hazards (such as storm, flood, drought, freeze, wildfire, landslide and the release of invasive organisms) is problematic.



Environmental risks

There is an underlying difficulty in assessing hazards, because existing methods do not adequately represent the spatial and temporal structure in forcing environmental variables. Infrequent, large perturbations to ecological systems may have overwhelming effects on their structure and function and may cause long-term or permanent changes. That said, forecasting of hazards and the environmental risks associated with them presents an opportunity to mitigate damage through improved decision-making and management. The study of environmental risks involves measurement of both the harm caused to ecological systems by such hazards and the likelihood of their occurrence. We must be clear what harm is caused, but also whether and how often the hazard will occur. Research at ITE into two types of environmental risk emphasises the need to consider both elements.

Extreme events

Physical extremes experienced by ecological systems include fires, droughts, floods, landslides and freeze. The periodicity of extreme events is well studied, though there is a need to classify them according to their causes and temporal scale. The consequences of extreme events are much less well understood. An extreme event is only significant if it perturbs an ecological system. The two studies described here provide valuable measures of how ecological systems respond to measurable physical extremes. Webb, Rose and Clarke consider responses to drought in terms of changes in plant community structure, while Hughes and colleagues describe how the soil microbial and biochemical process of methanogenesis in an upland mire is changed by summer droughts. Similar messages come from both studies. It is important to study the effects of changes in the periodicity of extreme events as well as the responses to individual events. Thus, Dorset wet heathland and peatland communities have contracted possibly as a consequence of successive dry summers, and the response of the Welsh mire to drought has evolved over several years of diminished flow. This research also shows that it is vital to carry out long-term field studies of responses to extreme events by monitoring responses to natural events or using experimental manipulation. A final lesson is that the state of an ecological system is very important in determining its responses to extreme events. Exposure to previous droughts allowed acclimation of the methanogenic community to subsequent droughts, and the lack of management of the Dorset heathlands may have made them more susceptible to droughts

The study of extreme events is being expanded through a collaborative programme of work between the Institute of Hydrology (IH), Institute of Freshwater Ecology (IFE) and ITE This work will link IH data and models of climatic extremes and the occurrence of physical extremes (such as drought, fire or storms) to ecological studies of the responses of populations, communities and ecosystems Models will be developed to simulate changes in event return-times and severity, and will allow examination of the stability, persistence and resilience of systems and the planning of actions to ameliorate the effects of extreme events

Genetically modified organisms (GMOs)

Another form of hazard comes from direct changes to the biological component of an ecological system These changes could be through introduction of a new species, or, in the study presented by Raybould and colleagues, through the release of GMOs In this case, the consequences of GMO introductions have been much discussed Many publications suggest a wide range of consequences, including the possibilities that

- GM plants may become crop weeds or show increased ability to invade semi-natural communities,
- pesticidal plants may cause population crashes in non-target herbivores,
- GM micro-organisms may be pathogenic to non-target hosts,
- GM fish may show increased competitive ability

Such consequences are possible, but there is very little information on the likelihood of their occurrence This has created the situation where the risks from GMO use are being assessed in some quarters by opinion rather than on the basis of scientific evidence The research at ITE aims to address this problem of probability The importance of viruses and herbivores in populations of crop and wild plants is being assessed in order to determine whether virus-resistant or pesticidal GMOs will have a substantially changed ecology such that they can invade crop or semi-natural communities One interesting point made is that the effects of genetic modification must be considered in relation to the ecology of the species and system under study It is too simplistic to make general statements about the risks from a certain type of modification

Future research on risks

ITE's research on environmental risks is interdisciplinary in nature because it requires study of a wide range of aspects of ecological systems, such as hydrology or the virus community, and therefore involves collaboration with other CEH Institutes the Institute of Virology and Environmental Microbiology (IVEM), IH and IFE The aim is to couple field studies of the full complexity of responses to hazards with realistic models of effects on populations, communities or ecosystems The result will be an ability to predict both the likelihoods and consequences of extreme events and harm from GMO release, and thus allow the development of programmes to reduce risks

Forecasting of extreme events and the environmental risks associated with them presents an opportunity to mitigate damage through improved decision-making and management.

J M Bullock

Plate 10. Cross-leaved heath (*Erica tetralix*), one of the characteristic species of wet heath. In Dorset wet heath has declined by 45% possibly due to drier conditions

Table 4. Characteristics of the principal heathland vegetation types

Dry heath	Dominated by heather (*Calluna vulgaris*), this type occurs on free-draining soils
Humid heath	Co-dominated by heather and cross-leaved heath (*Erica tetralix*), this vegetation type is found where drainage is impeded
Wet heath	Cross-leaved heath is usually the most abundant dwarf shrub and *Sphagnum compactum* and *S. tenellum* are usually present. The water table is within 100 mm of the soil surface
Peatland	Often referred to as valley mire this vegetation type is characterised by the abundance of *Sphagnum* species. The soils are organic and are waterlogged for much of the year

Losses of Dorset heathland between 1987 and 1996

(This work was partly funded by the Royal Society for the Protection of Birds, with support from English Nature and Dorset County Council)

There has been a steady decline in the area of heathland in Dorset for more than 100 years, and concerns about the ecological and conservation consequences of these losses were first voiced by Moore (1962). The Dorset Heathland Survey was initiated by ITE in 1978 (Webb & Haskins 1980) and was designed as a large-scale, repeatable survey which provided a baseline for monitoring change. The survey was repeated in 1987 (Webb 1990) and 1996.

Of the area of heathland surveyed in 1978, 425 ha (5%) had been lost by 1987. Although some of this loss was caused by conversion to agriculture or to development, a considerable area was lost because of the encroachment of scrub and trees (succession) on to open heathland. Scrub vegetation on open heathland increased by 15% during the period between the two surveys. The invasion of heathland by scrub was the main evidence that enabled funding to be obtained by conservation organisations for programmes of recovery which began in the late 1980s.

By 1996 the heathland on the area surveyed in 1978 had declined by a further 706 ha (9%). Almost all of this loss can be attributed to succession and very little to development or conversion to agriculture. The rate of succession of open heathland to scrub and trees has continued at an annual rate of about 1.7%, equivalent to a rate of about 15% over the nine years. Conservation management has done no more than contain the rate which prevailed between the two earlier surveys. The continuing growth of scrub and trees on open heathland can be attributed to the lack of fires over the last 20 years. Previously, the open heaths were burnt regularly, either for management or accidentally, but after severe fires in 1976 (Bullock & Webb 1995) stringent control measures were introduced. As a consequence, management by burning hardly occurs whilst no adequate alternative has been substituted. Today, these heaths carry extensive areas of mature heathland and increasing amounts of scrub. This vegetation provides a greater quantity of fuel and, when accidental fires occur, they now burn more fiercely, cover larger areas, and are more difficult to control. The increased temperatures attained during such fires may impede the recovery of heathland vegetation and result in the establishment of an increased number of invasive species.

Heathland is composed of a number of closely related vegetation types (Plate 10) which have been defined by Chapman, Clarke and Webb (1989). The most important are dry heath, humid heath, wet heath and valley mire. The last was called peatland by Chapman *et al.* (1989). The occurrence of each type of vegetation is dependent on the topographic and hydrological conditions (Table 4). Over the period 1978–87 there was almost no change in the extent of wet heath and peatland vegetation; however, over the period 1987–96 wet heath declined by 378 ha (45%) and peatland vegetation by 151 ha (25%) (Figure 21). Without a more detailed analysis, one can only speculate on what these two types of vegetation have become. It is likely that peatland vegetation has changed in composition to become wet heath, and wet heath has become humid heath. However, the changes are more complex, as the preliminary results do not show comparable gains in the extent of humid heath and dry heath. This is because there has been a succession of these types of vegetation to scrub and woodland, which may exceed any gains from the wetter heathland communities.

The composition of heathland vegetation is determined by topography and hydrology, and the dramatic changes in the composition which the 1996 survey has shown suggest that the heathlands have become drier. Three factors may be responsible for this change in hydrology. First, the heathlands carry a larger standing crop of vegetation which is of greater structural diversity. As a consequence, both interception of precipitation and evapotranspiration will have increased and less water will reach the wet heath and peatland communities. Second, rainfall in Dorset has shown a downward trend, particularly in the summer months, during the last decade (Paxman 1992). Third, land use changes, such as the growth of mature coniferous plantations, outside of the heathlands yet within the heathland catchment, may affect groundwater levels and lateral flows that are essential to the maintenance of wet and valley mire communities.

These hypotheses will be investigated by further analyses, in collaboration with the Institute of Hydrology, in which the detailed changes within the heathland vegetation types will be linked to hydrological models. If climate change has been responsible for determining the flux between the heathland vegetation types, it is interesting to note that the responses are both fairly rapid and detectable at the landscape scale.

The change in composition of the principal heathland vegetation types illustrates the dynamic nature of vegetation when viewed at a landscape scale. There is a tendency to see more or less fixed proportions of the various vegetation types over the heathland landscape, a view which is enforced once the distribution of the types is represented as a map at a fixed point in time. Yet, when observed at various points over a long period of time, the dynamics of the heathland vegetation are revealed. Succession, management, climate change and rare events such as wildfires are the principal factors affecting each of the heathland vegetation types.

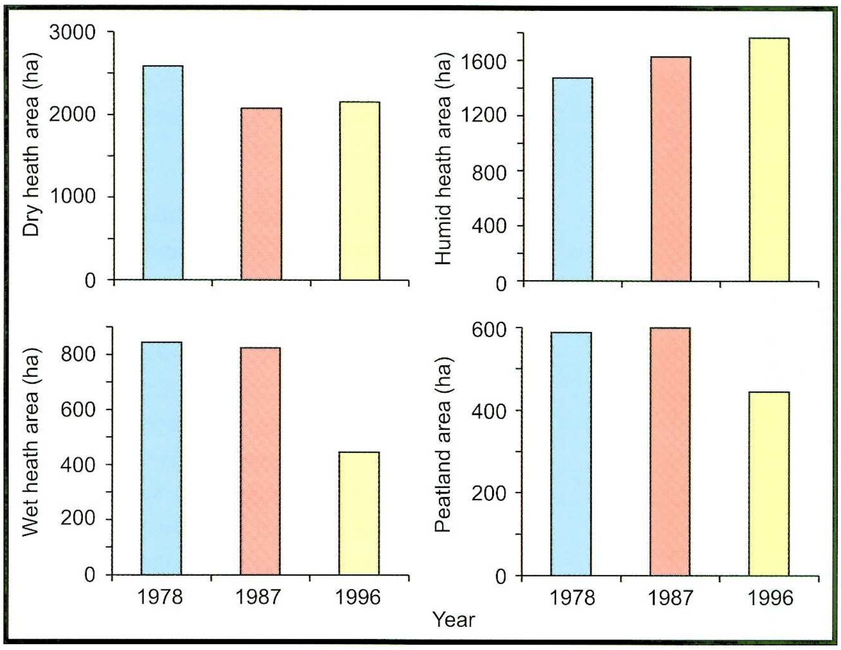


Figure 21. The areas of dry heath, humid heath, wet heath and peatland in Dorset in 1978, 1987 and 1996

The composition of heathland vegetation is determined by topography and hydrology, and the dramatic changes in the composition which the 1996 survey has shown suggest that the heathlands have become drier.

N R Webb, R J Rose and R T Clarke

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Possible acclimation of methane emissions from a gully mire to consecutive drought

(This work was partly funded by the Welsh Office)

Wetlands, including blanket bogs, raised bogs, flushes and valley bottom mires, form an important component of many upland areas in Wales and in the rest of western Britain. At the global level, much scientific interest in wetlands has been directed at the role of these ecosystems as sources/sinks for greenhouse gases. Methane emissions have received particular attention because wetlands are a major contributor to the global methane (CH_4) budget, with some estimates attributing up to 20% of the natural global emissions of this greenhouse gas to wetlands (Cicerone & Oremland 1988).

Of much recent concern is the possibility that drier summers predicted by climate models (Mitchell & Warrilow 1987) may alter the CH_4 flux from northern wetlands, with potential feedbacks to climate change (Bridgham *et al.* 1995). Based on reports from short-term laboratory and field studies (Bridgham *et al.* 1995; Moore & Roulet 1993; Sebacher *et al.* 1986), the consensus identifies water table drawdown as restraining CH_4 emissions from wetlands – an assumption that needs to be tested over the longer term.

Our early investigations of the potential effects of drier conditions within wetlands were carried out through laboratory experiments in collaboration with the University of Wales, Bangor (Freeman, Lock & Reynolds 1993). We have followed up these pilot studies with a full-scale field experiment designed to simulate realistically drier conditions within a small gully mire site in mid-Wales, in wider collaboration with the Institute of Hydrology (Plynlimon).

The catchment chosen for study, Cerrig-yr-Wyn in the Upper Wye

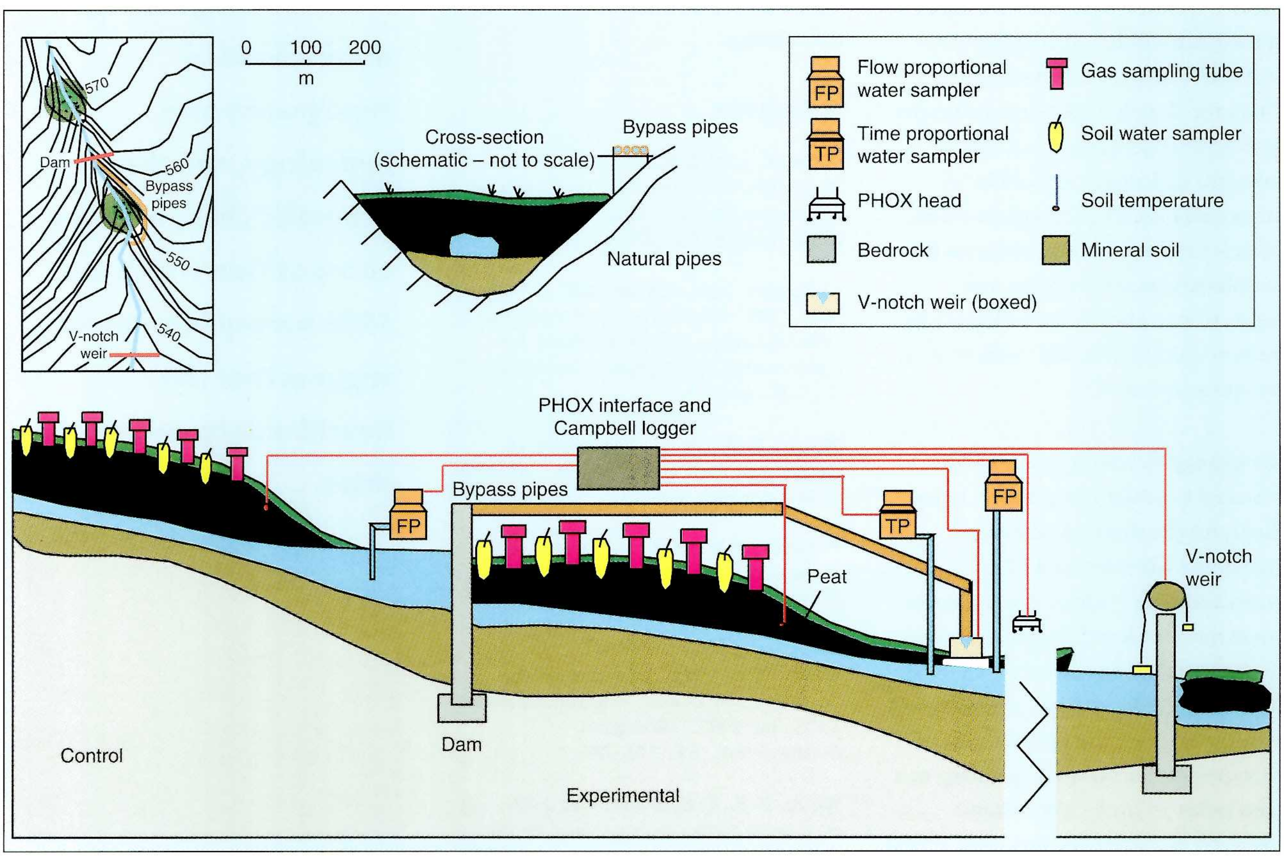

Figure 22. Design and instrumentation of the experimental field site (PHOX=water quality monitor)

catchment on Plynlimon, is typical of many in the uplands of Wales in which flush wetlands have developed as discontinuous serial cascade systems. The gully mire (peat pH range 4.0–4.8) is characterised by *Sphagnum* and rush (*Juncus*) communities. In this type of wetland, the main source of flushing water is catchment-derived streamflow rather than direct rainfall, and thus the effects of summer drought can be simulated by preventing streamwater from recharging the mire.

A flow manipulation system was set up (Figure 22) with a control wetland at the head of the system fed by rainfall and streamflow. Below the control wetland, a dam was constructed to allow diversion of streamflow through 150 mm pipes around an 'experimental' wetland to simulate summer droughts. Over a three-year period (1992–94), the wetland was subjected to simulated drought for varying lengths of time between late spring and early autumn. Rewetting after each drought simulation was achieved by distributing surface recharge using streamflow through 150 mm pipes across the head of the wetland. Methane emissions were then regularly monitored from the control and experimental wetlands during this period and subsequently following a natural summer drought which affected the wetlands in the summer of 1995.

The effects of successive periods of summer drought on methane emissions revealed some particularly unexpected results, which are summarised in Figure 23. The bivariate plot of methane fluxes and water table height in the control and experimental wetlands shows that simulated drought lowered the water table levels by 5–10 cm in the experimental site, relative to the control. The response of the experimental wetland to consecutive drought simulations changed over time, with an initial increase in CH_4

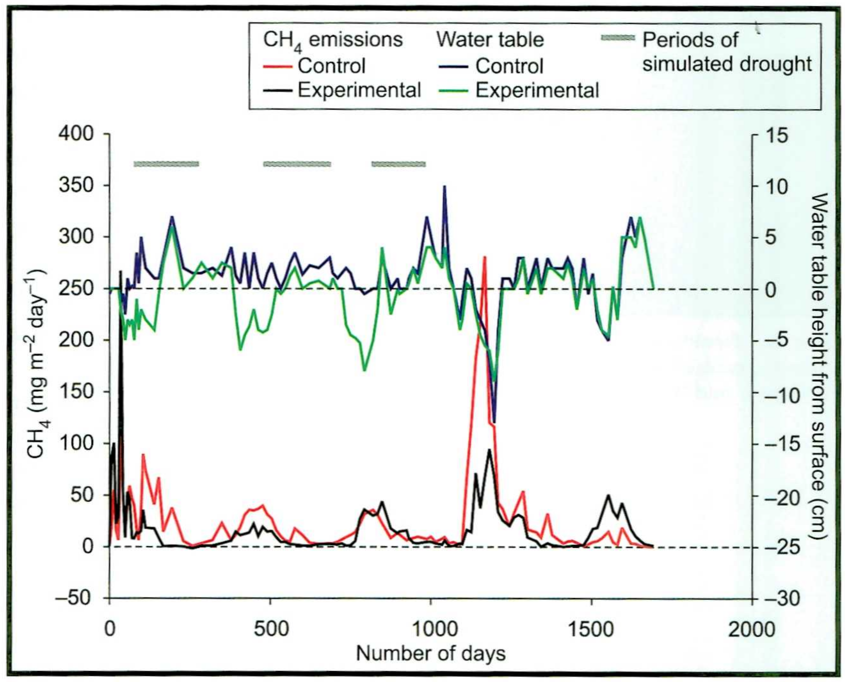

Figure 23. A time-series bivariate plot of CH_4 emissions from the control and experimental wetland sites and water table height in the control and experimental wetland sites. Solid grey bars denote periods of simulated drought

emissions then reduction relative to the control, which continued through the second year. A recovery in CH_4 emissions occurred during the third year. Following the natural drought in the summer of 1995, an increase in CH_4 emissions was observed from both sites, especially the control. Peak emissions (280 mg CH_4 m^{-2} day^{-1}) from the control wetland during the natural drought were approximately seven times the 1992–94 seasonal average. Peak emissions (90 mg CH_4 m^{-2} day^{-1}) from the experimental wetland increased less, but were still twice those of the previous year. During the summer of 1996, CH_4 emissions from the control wetland were greatly diminished and much lower than the seasonal average, while emissions from the experimental wetland were close to the seasonal average.

During the summer of 1996, CH_4 emissions from the control wetland were greatly diminished and much lower than the seasonal average.

The trend for the control wetland following natural drought mirrors that found for the experimental wetland (Plate 11) following simulated drought. An initial increase in CH_4 emissions occurs, followed by much lower emissions than normal during the following summer (Figure 23): an

Plate 11. Pipeline diverting water around a manipulated wetland at Plynlimon, mid-Wales

example of a 'boom and bust' cycle. Initial increase in CH_4 emissions from peatlands in response to a falling water table has been reported in other studies (Moore & Roulet 1993) and is often attributed to a falling water table releasing entrapped CH_4 from the peat. However, the extent and temporal scale (months) of the increase in CH_4 emissions from our control site suggest that an increase in methanogenesis had occurred. We believe that drought initially stimulates methanogenesis in these flush wetlands by cutting off the supply of surface streamflow, which recharges the mire. Stagnation and oxygen consumption by micro-organisms could be expected to render the peat anaerobic, providing conditions conducive to methanogenesis. Thereafter, as a falling water table gradually uncovers the surface peat, aerobic conditions are induced which are harmful for the methanogens, the secondary effects of which are not fully evident until the following summer.

Continuous monitoring of CH_4 emissions from the experimental wetland over the past five years indicates acclimation to consecutive, low-intensity summer droughts.

Continuous monitoring of CH_4 emissions from the experimental wetland over the past five years indicates acclimation to consecutive, low-intensity summer droughts. Methane emissions from the experimental wetland returned to control values during the third (and final) year of summer drought simulations in 1994, and the trend is continuing following the natural drought in the summer of 1995 (Figure 23). As described above, the control wetland exhibited a 'boom' (in 1995) and 'bust' (in 1996) cyclical response to natural drought, replicating the initial response of the experimental wetland to simulated summer drought. By contrast, CH_4 emissions from the experimental wetland were neither increased substantially to the same extent as the control during the natural drought in the summer of 1995, nor markedly suppressed during the summer of 1996. This suggests the wetland may have become acclimatised to the more frequent droughts. These findings have important implications as wetland CH_4 flux models (Cao, Marshall & Gregson 1996) have yet to consider the possibility of such acclimation.

It is important to discover whether the observed acclimation is retained or lost following a prolonged period (ie two or three years) without drought conditions. If acclimation is lost, this may once again result in a 'boom and bust' type of response to any subsequent summer drought event.

S Hughes, B Reynolds, C Freeman[1], D J Dowrick[1] and J A Hudson[2]
[1]School of Biological Sciences, University of Wales, Bangor
[2]Institute of Hydrology, Plynlimon

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Predicting the ecological impacts of pest- and disease-resistant genetically modified crops

(This work was funded by the Department of Environment, Transport and the Regions and the CEH Integrating Fund, and involved collaboration with the Institute of Virology and Environmental Microbiology and the University of Liverpool)

Genetic modification of pest and disease resistance offers new methods of increasing crop yields and reducing chemical inputs. Crops that have been genetically modified to resist insect herbivores and viral diseases are now available for commercial use. In 1996, 1.8 million hectares of cotton containing a bacterial gene for an insecticidal protein (13% of the crop) were grown in the United States. The GM cotton controlled pests sufficiently to reduce insecticide application by 250 000 gallons.

There is concern, however, that genetically modified crops with pest and disease resistance may also have undesirable impacts on the environment. If diseases or insects control the population growth rate of feral crops (crops that grow in non-agricultural land), or wild plants that can hybridise with crops, the release of GM-resistant crop varieties could result in plants with increased weediness. To provide information for risk assessments prior to the commercial use of GM pest and disease crops in the UK, ITE is working on several projects which aim to increase our understanding of the role of insects and diseases in the dynamics of feral crops and wild crop relatives.

Insect-resistant feral oilseed rape

Feral oilseed rape (*Brassica napus*) can establish readily in disturbed ground, but is rapidly replaced by perennial plants (Crawley & Brown 1995). To investigate whether insects affect the persistence of rape, we established experimental 'feral' populations in a freshly cultivated field. Plots were regularly treated with insecticide or left untreated (controls), and rabbits (*Oryctolagus cuniculus*) and deer (mainly *Capreolus capreolus*) were excluded.

Plate 12. Variation in flowering time among plots of oilseed rape. Plots in flower were treated with insecticide

Plants in the treated plots performed better at all stages of development. There were more seedlings per square metre, the plants flowered earlier (Plate 12) and seed production was much higher (about 230 000 seeds m^{-2} compared with about 70 000 m^{-2}). However, in the second year of the trial, when rabbits and deer were not excluded and there was no weed control, seedling establishment was much reduced, despite continued application of insecticide. Therefore, while insect resistance appears to be advantageous during colonisation of disturbed ground, it may not result in greater persistence of feral rape populations.

While insect resistance appears to be advantageous during colonisation of disturbed ground, it may not result in greater persistence of feral rape populations.

Insect and virus resistance in wild cabbage

To assess the potential consequences of the spread of GM

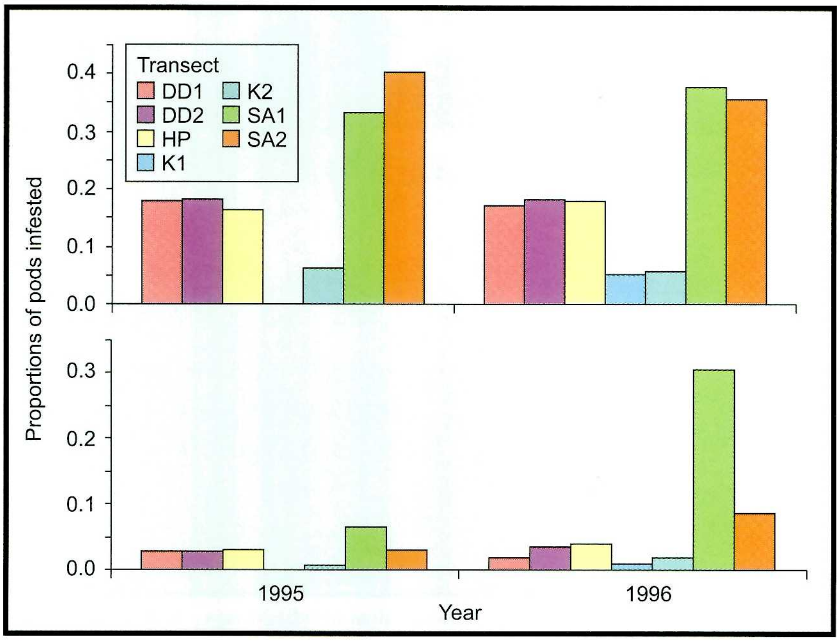

Figure 24. The relationship between seed weevil (above) and brassica pod midge (below) infestations in seven populations of wild cabbage on the Dorset coast (DD=Durdle Door, HP=Handfast Point, K=Kimmeridge and SA=St Aldhelm's Head)

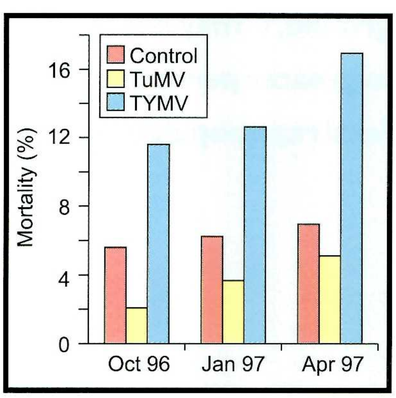

Figure 25. The relationship between virus infection and mortality of wild cabbage plants in a field trial near the Dorset coast. Plants were inoculated in April 1996 and planted in June 1996 (TuMV=turnip mosaic virus; TYMV=turnip yellow mosaic virus)

insect resistance to a crop relative, we are studying insect damage to wild cabbage (*Brassica oleracea*) on the Dorset coast. Rather than applying insecticide to manipulate insect numbers, we use natural variation in the prevalence of insect herbivores as a means of measuring their effects. The distribution of insects may, in part, be determined by differences among the cabbages in the type and concentration of chemicals (glucosinolates), which attract some herbivores and deter others (Mithen, Raybould & Giamoustaris 1995).

The seed weevil (*Ceutorhynchus assimilis*) is a common herbivore on wild cabbage. Our results show that weevil infestation of seed pods significantly reduces the number of viable seeds per pod, because seeds are eaten by weevil larvae and because fungi can more easily infect damaged pods. In addition, the weevil interacts with another herbivore, the brassica pod midge (*Dasineura brassicae*). The midge cannot penetrate intact pods, but can lay its eggs through the feeding and egg-laying holes of seed weevils. Midge larvae therefore tend to occur where weevil larvae are prevalent (Figure 24). Midge damage causes the seed pod to shatter, spilling the seed before ripening, and so increasing the detrimental effect of weevil infestation.

The weevil data suggest that plants with insect-resistance genes would be at an advantage in wild cabbage populations. However, plants that are least attractive to specialist herbivores like weevils tend to be most attractive to slugs, which can cause high mortality among seedlings (Moyes, Collin & Raybould 1997). Thus, while insect-resistant plants might produce more seed than non-resistant plants, they may not necessarily leave more offspring.

As well as being damaged by herbivores, a large proportion of cabbage plants have multiple virus infections. To investigate the extent to which these infections are deleterious, we raised wild cabbage seedlings in a glasshouse, and inoculated them with either turnip mosaic virus (TuMV), turnip yellow mosaic virus (TYMV) or sterile water (controls). When the plants were two months old they were planted in a field close to a large natural population of wild cabbage. Plants infected with TYMV died at a higher rate than the controls, whereas plants with TuMV seemed to survive better than the controls (Figure 25). Among the surviving plants there was little difference in height, leaf number and the proportion that flowered between the controls and TuMV-infected plants. However, the plants with TYMV were smaller, had fewer leaves and flowered less frequently. Work is in progress to establish whether seed output is also lower.

The effects of weevils and viruses on cabbages show that GM-resistance genes could be advantageous to plants in wild populations Nevertheless, as with rape, we need to be careful in assuming that resistant plants will be 'weedier' Although resistant plants may produce more seed, population size may not change if the resistant types are selected against at a different part of the species' life cycle (eg seedling establishment), or if the resistant types are less fit in the absence of patchily distributed insects or viruses Another consideration is whether seed production is a limiting factor to population growth Further work will investigate these questions

A F Raybould, C L Moyes, L C Maskell, G W Elmes, J C Wardlaw, Z Randle, W E Rispin, J I Cooper[1], M-L Edwards[1], D McCall[1] and A J Gray
[1]Institute of Virology and Environmental Microbiology

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