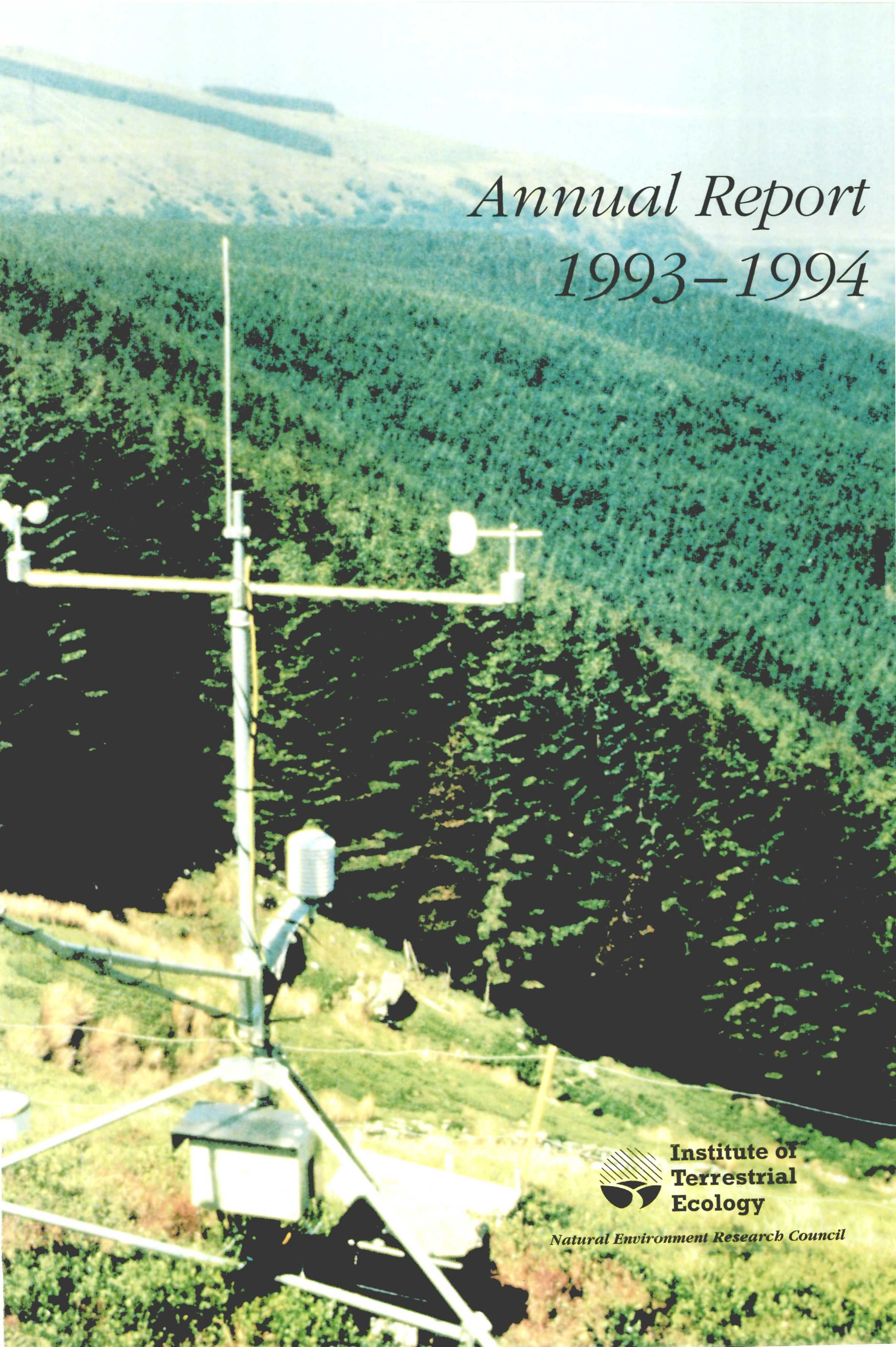

Annual Report 1993–1994

Institute of Terrestrial Ecology

Natural Environment Research Council

Foreword

The past year has been one of considerable change in the Natural Environment Research Council, both in the focus of its science and in its structures. The catalyst for these changes was the publication of the White Paper *Realising our potential: a strategy for science, engineering and technology* (Cm 2250). NERC was given a new mission for its science to embrace the concepts of meeting the needs of its user communities and contributing to wealth creation and the quality of life. We have, of course, always paid close attention to these objectives, but there is now a clear need for a sharper focus and better articulation of what we do in these areas. Basic science and long-term monitoring are also included in our mission, and due weight must be given to these when developing our science strategies.

The science directorates will cease to exist towards the end of 1994, and new structures will be put in place. TFSD Institutes are being regrouped as the Centre for Ecology and Hydrology, with a unified ITE under a single Director. However, the report of the Multi-Departmental Scrutiny of Public Sector Research Establishments is awaited, and decisions arising from this report may result in further organisational changes within NERC.

An important activity during the year has been the preparation of a new science and technology strategy for the terrestrial and freshwater sciences. Publication is expected in July, and a number of research areas will be identified for priority support over the next five years.

This is my second and final foreword. During my relatively short time with NERC, I have come to appreciate and value the breadth and strength of our work in the terrestrial, freshwater and hydrological sciences. Within ITE, for me, highlights of this year have been the report arising from Countryside Survey 1990 and the launch of the Land Cover Map. It is because of these, and very many other successes, that I am confident we can continue to produce high-quality and competitive science in the post-White Paper environment. Both ITE(North) and ITE(South) were visited by separate Science Management Audit Groups during the year. It is to the great credit of everyone involved in ITE that the reports of both groups recognised a continued advance in quality across the whole of the Institute over the five years since the previous visits.

Finally, I should like to state how much I have appreciated the friendships that I have established with so many members of our community. It is these that will be my most valued and lasting memories of NERC.

C Arme
Director of Terrestrial and Freshwater Sciences
Natural Environment Research Council

Report of the
Institute of Terrestrial Ecology
1993–1994

Natural Environment Research Council

CONTENTS - Science Reports 2

Global environmental change

Previous introductions to this Programme have drawn attention to the role of research in providing a sound basis for policies aimed at limiting the nature and extent of environmental change. Among other things, we need to identify and quantify the most important sources and sinks of agents of environmental change, such as the so-called 'greenhouse' gases (eg CO_2, CH_4, N_2O, O_3, CFCs). How sensitive are these sources and sinks to changes in the environment, and how amenable are they to modification or control by human actions? What are the feedbacks, and will action aimed at reducing emissions of one 'greenhouse' gas have more damaging 'knock-on' effects elsewhere? The policy-maker's aim is to direct available resources and effort where they will have their best effect. To underpin the decision-making, various types of research are needed, and each presents its own particular challenges. There may be technical difficulties of measurement and collection of data *in situ*, difficulties in the experimental manipulation of systems, or parts thereof, and problems of statistical analysis and modelling.

The first of this year's reports illustrates the value of having a good estimate of the biological pools and sinks of carbon in the UK. The large uncertainty about the size of the pool of organic carbon in Scottish peats pinpoints the need for more data in this area. At the same time, models of forest growth allow the potential of afforestation as a means of sequestering carbon to be viewed in the context of a national carbon budget. The experiment on responses of *Lolium perenne* to increases in CO_2 concentration and temperature (pp44–47) is one of several studies aimed at understanding and predicting the behaviour of a wide range of plant species in a changing environment. A particularly important response may be a reduction in the relative nitrogen content of litter from plants growing in CO_2-enriched air. Such litter tends to decompose more slowly, adding to the pool of organic carbon in soils.

The third of this year's reports illustrates how a technological development – a tunable diode laser (TDL) spectrometer – allowed the measurement of 'greenhouse' gas fluxes (Plate 27) over large areas of wetland in the flow country of Scotland. Associated groups are investigating the processes leading to methane emission and oxidation, from the genetic characteristics of the micro-organisms responsible to the environmental factors governing their activity. Scaling up from these laboratory and mesocosm systems to predict methane fluxes at patch,

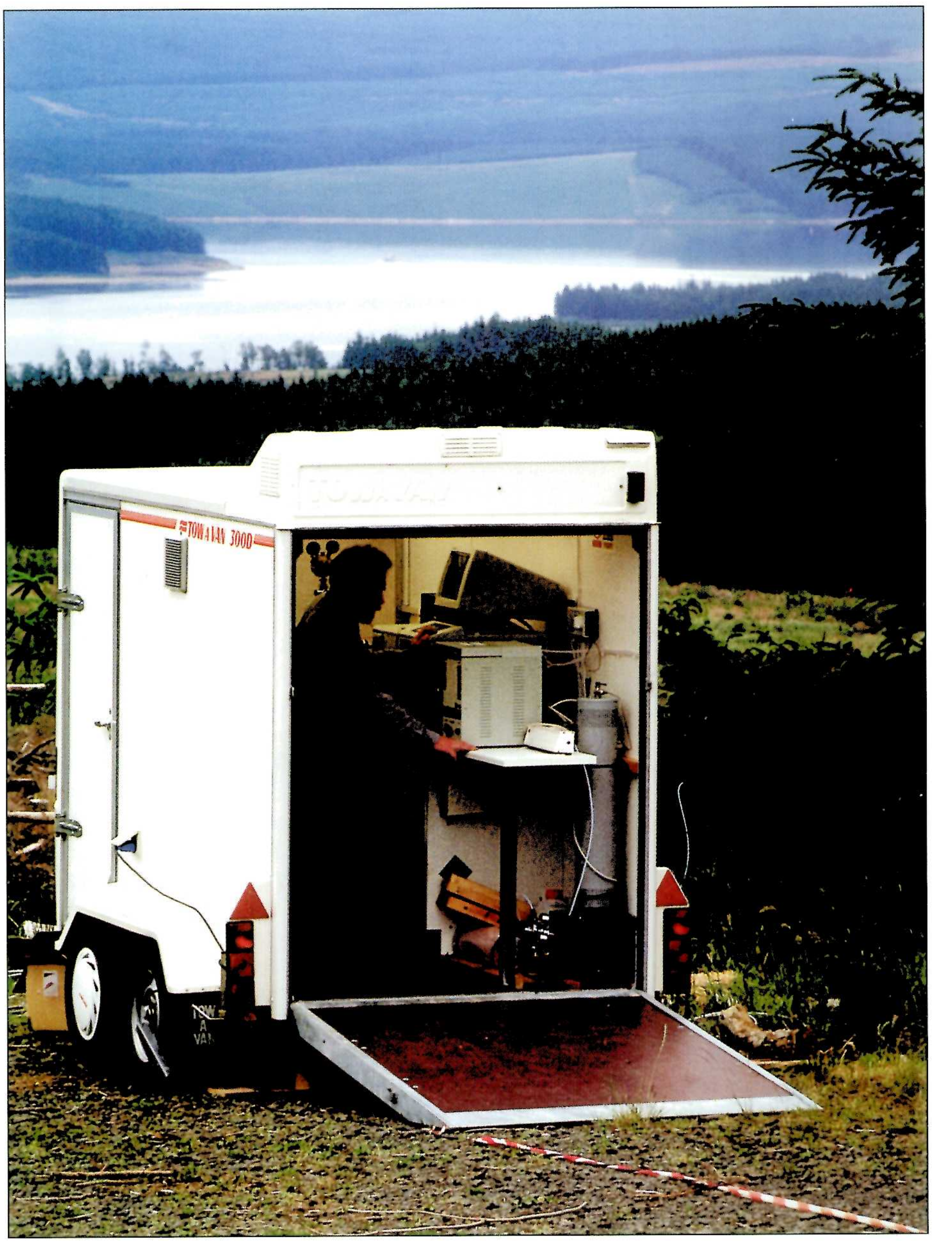

Plate 27. Mobile laboratory used by staff at ITE Merlewood for monitoring methane and nitrous oxide fluxes

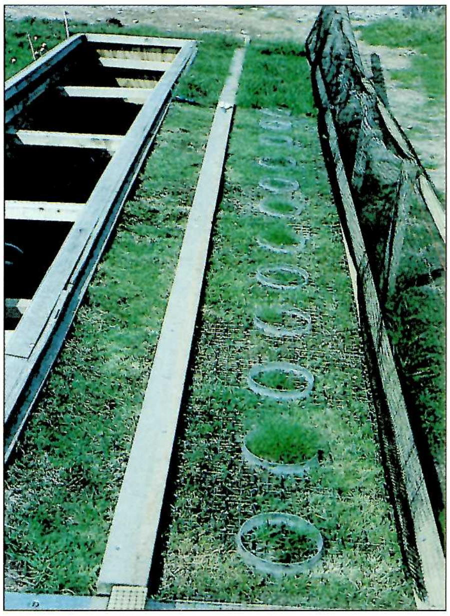

Plate 28. Soil warming experiment at Great Dun Fell, northern Pennines, Cumbria

landscape and even regional scales is a difficult task, because conditions are so variable in space and time. Being able to measure fluxes over a wide range of spatial scales in the field not only provides hard data on the current state and behaviour of the system, but also helps to identify the sources and influence of important environmental variables.

As the importance of interactions between the components and processes of ecosystems receives greater recognition, so there is an increasing use of manipulative experiments in the field. These experiments are often expensive

Plate 29. Soil core used in the transplant experiments at Great Dun Fell

and technically difficult to establish, but can provide unrivalled opportunities for multidisciplinary research. Examples in which ITE is involved include a soil warming experiment at Great Dun Fell in the northern Pennines, Cumbria (Plate 28), with associated reciprocal transplants of soil cores (Plate 29) along an altitudinal transect, and a free-air CO_2-enrichment (FACE) experiment at Lindau, in Switzerland. There is also a facility at ITE Monks Wood for imposing modulated increases in UV-B radiation in the field, simulating a reduction in stratospheric ozone. The manipulation of a gully wetland site to simulate drought (pp50–53) has already produced important insights into the production of methane by wetlands which might not have been revealed by laboratory studies. Once again, progress was aided by the specialist technical expertise of collaborators, in this case the Institute of Hydrology.

The last of the studies reported in this section has developed from an international comparison of fluxes and processes in salt marshes. Although the coastal zone marks a boundary of ITE's remit for research, biogeochemical processes recognise no such constraints, and it is important to maintain a sense of this continuity. Apart from their intrinsic value as ecosystems and their function as habitat for marine life, salt marshes can provide an important buffer against the incursion of the seas on to low-lying land. Against a background of declining demand for agricultural land, and the threat of rising sea levels, the area of salt marsh in the UK may be set to increase substantially, and we need to understand its behaviour.

In many fields there is a growing need for groups from different disciplines to work together, both in conducting the research and in the interpretation and application of results. In no field is this more evident than in global environmental research. Within NERC, ITE's long-standing links with the Institute of Hydrology and the Institute of Freshwater Ecology will continue within the new Centre for Ecology and Hydrology. At the same time, more work is being undertaken in collaboration with Universities and other institutes. The consortium approach to multidisciplinary research has been a notable feature of NERC's Terrestrial Initiative in Global Environmental Research (TIGER) programme, and with TIGER nearing the end of its first phase of three years it is clear that this approach

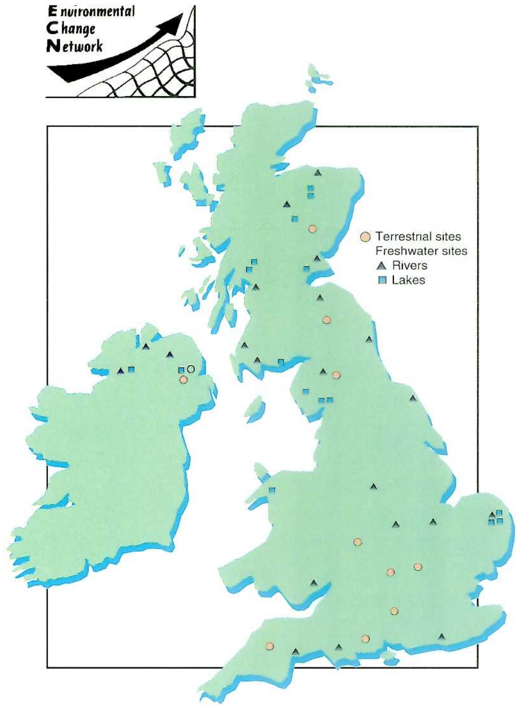

Figure 30. Map showing coverage of ECN sites in 1994

has brought significant progress on several fronts. The two TIGER flagship sites, at Wytham in Oxfordshire and at Moor House and Upper Teesdale in the Pennines, have become centres for a variety of multidisciplinary investigations and manipulative experiments. Both sites are part of the Environmental Change Network (ECN) (Figure 30), and ITE staff oversee the collection of data and co-ordination of research on the sites. As well as extending existing records of physical and biological conditions at these sites, detailed measurements provide a sound backdrop for studies of undisturbed and manipulated biological systems. While TIGER has a fixed lifespan of five years, the need for research on the processes underlying global environmental change will continue, and the value of comprehensive field experiments will undoubtedly grow.

C P Cummins and B G Bell

Carbon pools and sinks in British vegetation and soils

(This work was funded by the Department of the Environment)

The Framework Convention on Climate Change, signed by the UK at the United Nations Conference on Environment and Development (the 'Earth Summit') in Rio de Janeiro in June 1992 (ratified in 1993), commits all parties to prepare national inventories of emissions and sinks of 'greenhouse' gases, and to take measures to preserve and enhance the sinks of 'greenhouse' gases on their land surface. The work described below was sponsored by the UK Government to determine how much carbon is contained in UK vegetation and soils (the pools), and how rapidly carbon is being lost from, or gained by, UK vegetation and soils as a result of current and historic land use changes and natural processes (the sources and sinks). It will then be possible to explore potential policy options that will enhance carbon sequestration. Some of the salient findings were reported in the UK's follow-up report to the Climate Convention (Department of the Environment 1994).

Carbon pools

The conclusion of the study was that about 22 000 Mt C is present in British soils, peat and litter, and about 110 Mt C is present in living vegetation. However, the estimate that about 75% of soil carbon is contained in Scottish peats is highly dependent on the estimates of soil bulk density. For lower densities, the total British soil carbon may be as low as 10 000 Mt. Thus, soils contain 100–200 times as much carbon as vegetation. Only carbon in soil organic matter has been considered because mineral carbon (eg in calcium carbonate) is not subject to biological control.

The approach taken to estimate soil carbon contents was, first, to determine the dominant soil series in each 1 km^2 over Britain; second, to determine the dominant land cover in each km^2 (simplified to seven major classes, comparing cover types used by ITE, the Macaulay Land Use Research Institute (MLURI) and the Soil Survey and Land Use Research Centre (SSLRC); and, third, to derive a carbon density value (mass of carbon/area) for each combination of soil series and land cover. SSLRC and MLURI compiled the soil data, and SSLRC derived regression formulae to calculate bulk densities from other soil variables, which, combined with percentage organic matter contents, enabled carbon densities to be derived. Figure 31 presents the geographical distribution of carbon in British soils at a 20 km × 20 km resolution, showing the dominance of peatlands in Scotland. England and Wales have 155 314 1 km × 1 km squares, of which 140 049 have more than 50% soil cover, the remainder being under water or buildings. The total organic soil carbon content is 2773 Mt. Scotland has 84 929 1 km × 1 km squares, of which 82 420 have more than 50% soil cover. The total organic soil carbon is 19 011 Mt, 6.85 times the total organic carbon content of the soils of England and Wales. Thus, the total organic carbon content of the soils of Great Britain is estimated to be 21 784 Mt, of which 87% is in Scottish soils and 75% is in Scottish peats.

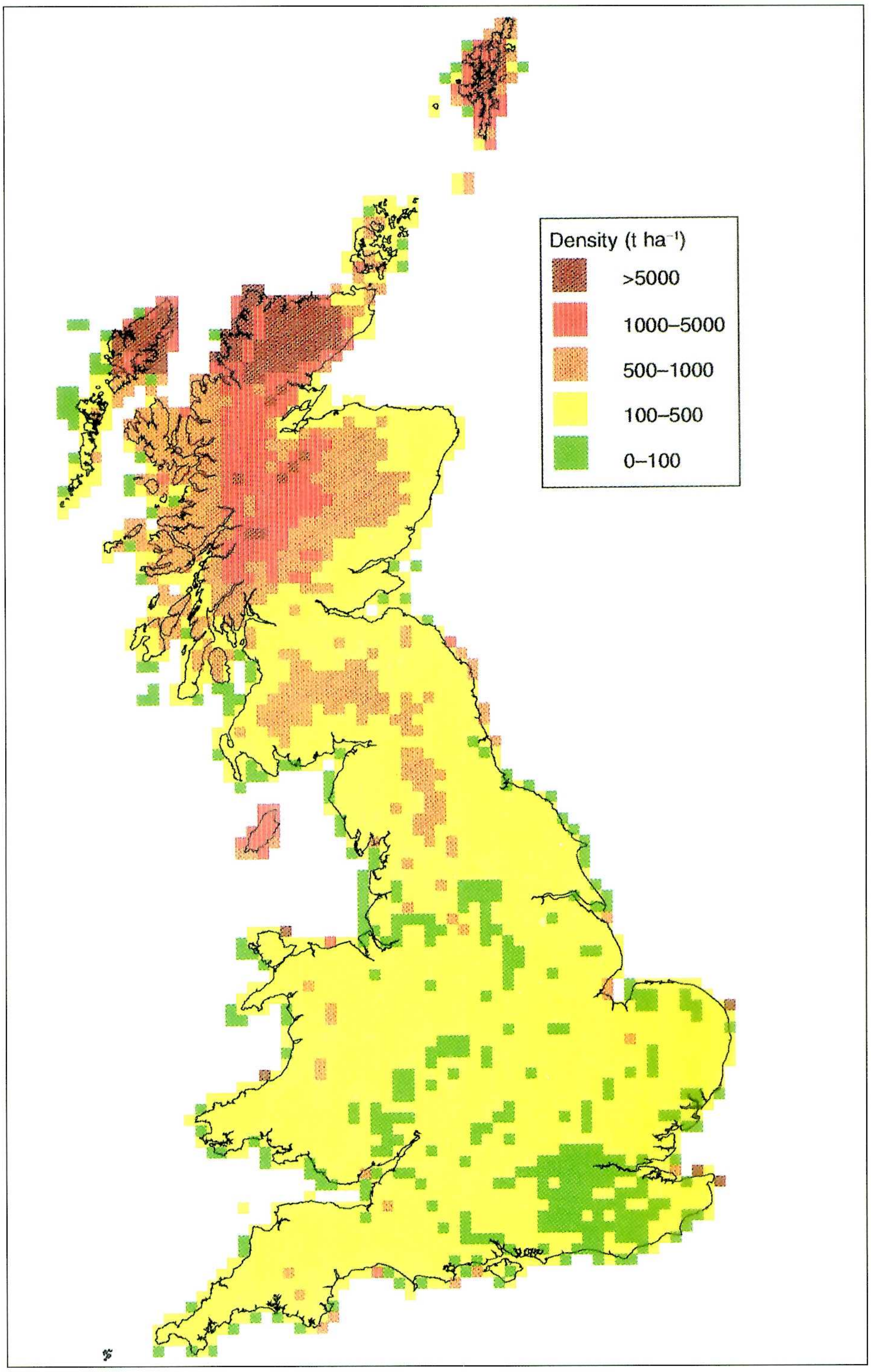

Figure 31. Carbon present in soils (including peats and litter) in Britain, shown as t C ha^{-1} (the carbon density) in 20 km × 20 km areas

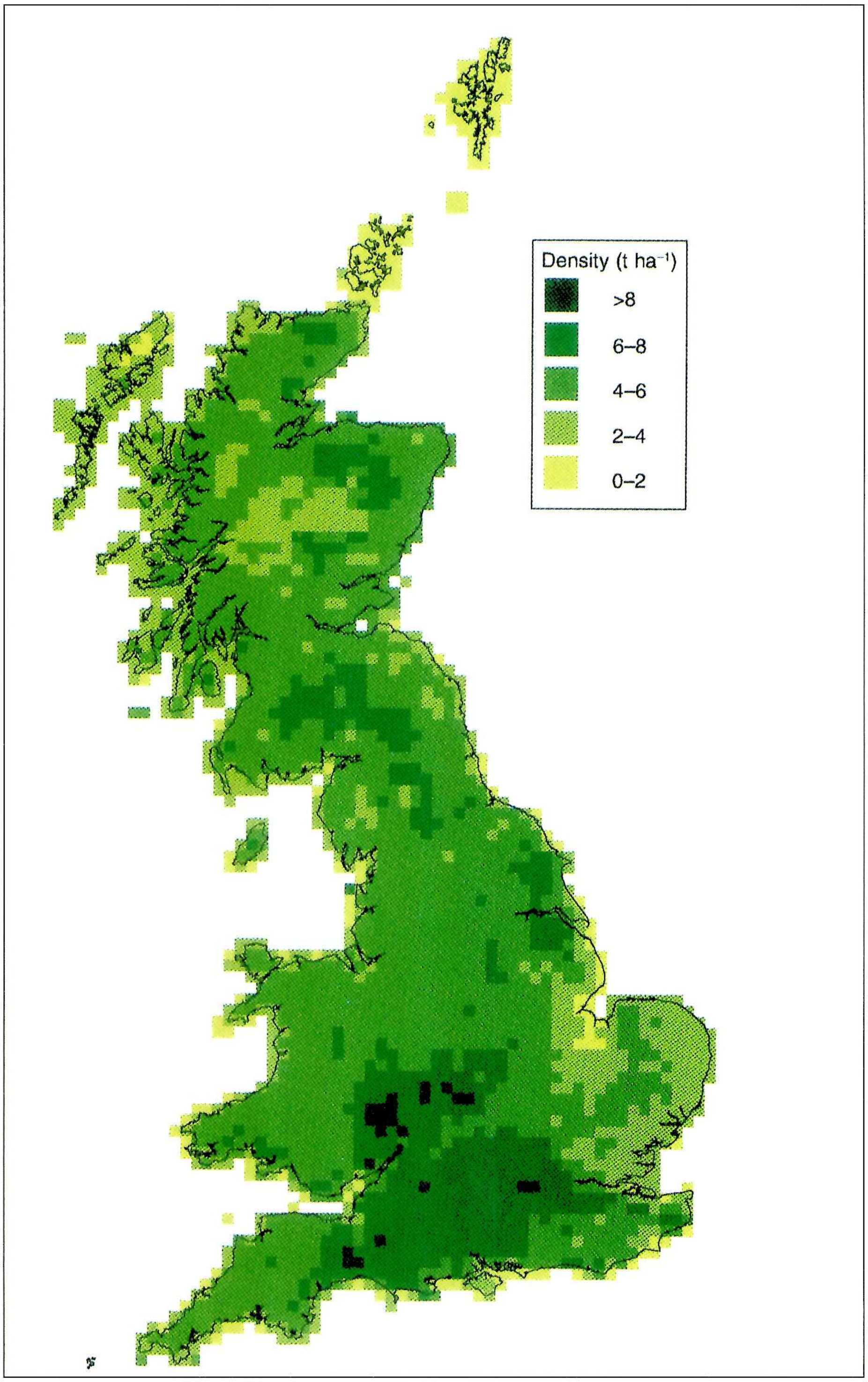

Figure 32. Carbon present in vegetation in Britain in 1990, shown as t C ha^{-1} (the carbon density) in 20 kmx 20 km areas

These estimates are based on bulk densities for Scottish peats of 0.3 g cc^{-1} at the soil surface to 0.5 g cc^{-1} at depth. There is a great deal of variation in, and uncertainty about, the bulk densities of different types of peats at different depths. If a range of densities of 0.10–0.2 g cc^{-1}, from surface to depth, is used for Scottish peats, then the total organic soil carbon content for Scotland is estimated to be about 7500 Mt. When added to the above estimated total for England and Wales, this gives a value of about 10 000 Mt for the total organic soil carbon content of Britain.

The approach adopted to estimate the carbon content of vegetation was to use the ITE land classification, available at 1 km^2, and the ITE 1984 and 1990 Countryside Survey data, to provide estimates of the area of 58 land cover types. The areas were then divided into non-woodland and woodland. Non-woodland areas were assigned carbon densities of 0 (fallow), 1 (eg cereals and pasture) or 2 (eg shrub and heath) t ha^{-1}. Woodland areas were divided into the areas covered by 15 dominant species, as estimated in the Countryside Surveys. The carbon densities of each of these species was then estimated using:

- relationships between stand age, timber volume and carbon mass, and
- the age distribution of each species determined in the 1984 census of woodlands conducted by the Forestry Commission.

The carbon density of all vegetation in each ITE land class was estimated as the average density for each vegetation type weighted by the area covered by each type in each land class. Figure 32 presents the geographical distribution of carbon in British vegetation at a 20 km^2 resolution in 1990, the higher values reflecting the dominance of broadleaved woodland in southern England, and of coniferous plantations in the Borders.

Details of the carbon contained in six major groups of vegetation cover are given in Table 6. It is seen that about 80% of the carbon is in woodland, covering about 11% of the land area of Britain. Coniferous plantations, being relatively young, contain only about a third of the carbon in woodlands.

Carbon sources and sinks: forests

Dewar (1991) wrote a model which describes the fluxes of carbon in even-aged, single-species plantations that are clearfelled at the time of maximum mean annual increment and then replanted. This model was used to estimate:

- carbon storage in plantations differing in yield class (m^3 ha^{-1} yr^{-1}), thinning regime and species characteristics (Dewar & Cannell 1992);

Table 6. Carbon contents and areas of six major groups of vegetation in Britain in 1990

	Carbon (Mt)	% in Britain % carbon	% area
Non-vegetated	0	0	10
Agricultural	11	9	49
Semi-natural	11	10	30
Conifer	29	25	6
Broadleaf	53	47	4
Mixed woodland	10	9	1
Total	114	100	100

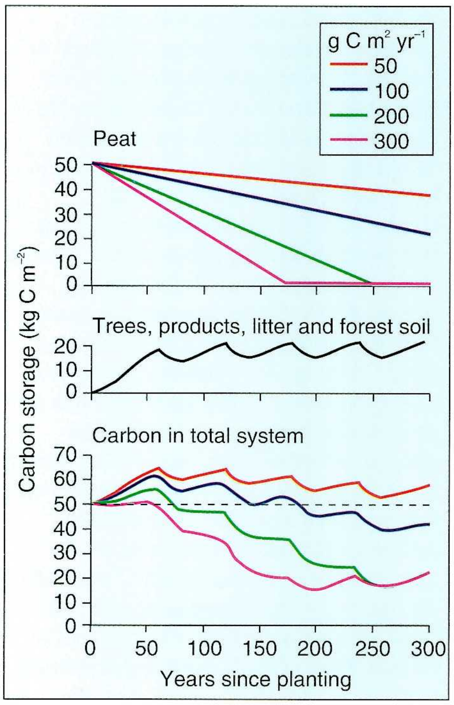

Figure 33. Possible changes in the carbon stored in peat (top graph), trees, products, litter and forest soil when growing Sitka spruce yield class 12 (middle graph), and the total system (bottom graph), assuming constant carbon loss rates from the peat (after drainage) of 50, 100, 200 and 300 g C m^{-2} yr^{-1}, with an initial carbon content of 50 kg C m^{-2}

- the effect of planting forests on peatlands on the net flux of carbon over time (Cannell, Dewar & Pyatt 1993); and
- the size of the carbon sink currently provided by forest plantations in Britain (Cannell & Dewar 1994).

For most plantation forest types, the maximum average amount of carbon that can be stored (at equilibrium) is in the range 40–80 Mg C ha^{-1} in the trees, 15–25 Mg C ha^{-1} in litter, 70–90 Mg C ha^{-1} in soil organic matter and 20–40 Mg C ha^{-1} in wood products (assuming that all products decay over a period equal to the length of a rotation). Increasing the yield class increases the maximum carbon storage, and the rate of sequestration. An analysis of species characteristics led to the conclusion that poplar (*Populus*) plantations growing on fertile land would both store carbon rapidly in the short term and achieve a high carbon storage in the long term. But, if the objective were to achieve high carbon storage in the medium term (over 50 years) without regard to the initial rate of storage, then plantations of any conifer species with above-average yield class would suffice.

Undisturbed peatlands emit methane but accumulate CO_2-derived carbon. The net 'greenhouse effect' may be near zero. Peatland drainage for forestry lowers the water table such that virtually all the methane is oxidised in the surface aerobic layers. Drainage also increases CO_2 loss by aerobic decomposition, but this loss is offset by CO_2-derived carbon accumulated in the trees. Figure 33 shows the results of an analysis to determine the net CO_2-carbon change in the whole system when conifers with yield class 12 m^3 ha^{-1} yr^{-1} are planted. The loss of carbon from the peatland is shown at the top of the Figure, assuming that the peat contained 50 kg C m^{-2} to start with, and drainage resulted in a constant loss of 50, 100, 200 or 300 g C m^2 yr^{-1}. The gain in the amount of carbon from afforestation over five rotations (in the trees, litter, forest soil, and tree products) is shown in the middle of the Figure. At the bottom is shown the net effect on carbon storage. If CO_2 loss rates from drained peats are in the range 50–100 g C m^2 yr^{-1}, there will be increased carbon storage in the whole system for at least three rotations, but, if CO_2 loss rates are 200–300 g C m^2 yr^{-1}, then increased storage will be restricted to the first rotation, after which there will be a net loss of carbon. At present, the published data on peat decomposition rates span the range 50–300 g C m^2 yr^{-1}; it is an objective of future work to lessen uncertainty about the rate of this process in the UK. Regardless of this fact, however, the system is predicted to suffer a net loss of carbon in the long term.

The past, current and future rates of carbon accumulation in forest plantations in Britain were estimated using the Dewar (1991) model and the record of new forest areas planted each year since 1925. It was assumed that all forests planted so far have the carbon storage characteristics of Sitka spruce (*Picea sitchensis*) yield class 14, and that there has been an increase in surface litter but no net change in soil organic matter. Figure 34 shows the estimated rate of increase in carbon storage (Mt C yr^{-1}) in trees, litter and wood products from 1925 to 1990, and from 1990 to 2050 assuming that the forest area is increased by 0, 10 000, 20 000, 30 000 or 40 000 ha yr^{-1}. Because the current forest estate is relatively young, it was estimated to be accumulating about 2.5 Mt C yr^{-1} in 1990. In order to maintain this rate of carbon removal from the atmosphere, planting would need to continue at a rate of 20 000–30 000 ha of conifers per year. It should be noted that 2.5 Mt C represents only 1.5% of UK annual emissions of carbon from fossil fuels.

Carbon sources and sinks: non-forest areas

A number of natural and human factors are causing losses or gains in the amount of carbon stored in non-forest areas, principally in the soils. These include a gain in carbon in undisturbed wetlands (owing to slow decomposition in anaerobic conditions), which may be occurring at a rate of 40–70 g C m^2 yr^{-1} over an area of 3 million hectares, totalling 1.2–2.3 Mt C yr^{-1}. Also, about

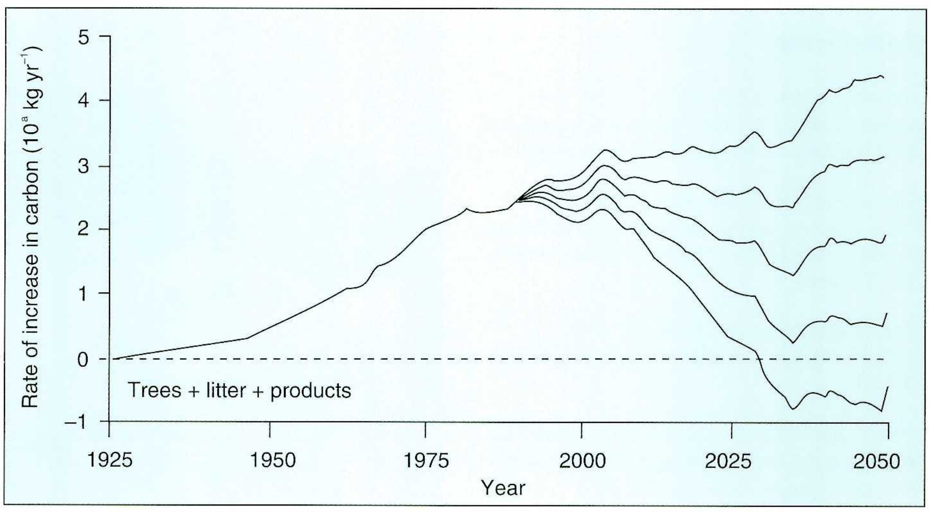

Figure 34. Estimated rate of increase per year in the amount of carbon stored in trees, litter and forest products as a result of afforestation in Britain since 1925, and as a result of continued planting at rates of 0, 10 000, 20 000, 30 000 and 40 000 ha yr^{-1} (of new forest area assuming full restocking)

70 000 ha of lowland fens may be losing carbon at a rate of about 20 g C m^2 yr^{-1} (owing to drainage and aerobic oxidation), totalling a loss of about 0.01 Mt C yr^{-1}. Additionally, we might include a CO_2 and nitrogen fertilization effect, which may be increasing carbon storage in Britain at over 1 Mt C yr^{-1} (by increasing litter input to soils), and a loss of carbon to the sea of over 1 Mt C yr^{-1}.

When permanent pasture is converted to arable land, there is a loss of carbon, which has been modelled using the Hurley Pasture Model, and by fitting exponential decay functions to data. Data from Rothamsted showed that an old pasture that originally contained 93 t C ha^{-1} lost 35 t C ha^{-1} during the first 25 years after it was ploughed up and continuously cropped, half of this loss occurring in the first 3.5 years.

Conversely, laying arable land down to grass usually increases the soil organic content, and hence carbon storage. The Hurley Pasture Model and Rothamsted data show that the rate of carbon accumulation is very slow, taking over 200 years to reach a steady state. Rothamsted data suggested that 49 t C ha^{-1} might be added over 275 years, half of which would be gained in the first 38 years.

The current challenge is to reconcile and use the available matrices of historic land use change in Britain to determine the current net flux of carbon to or from the non-forest areas.

M G R Cannell, R Milne, R C Dewar and P J A Howard

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Responses of grasses to changes in CO_2 and temperature

(This work was funded by the NERC Terrestrial Initiative in Global Environmental Research (TIGER) programme and the CEC 3rd Framework Programme)

Since the Industrial Revolution, the combination of fossil fuel combustion and deforestation has led to an increase of 26% in atmospheric carbon dioxide (CO_2) concentrations and a rise in global mean surface air temperature of 0.3–0.6°C (Houghton, Jenkins & Ephraums 1990). Future predictions of global climate change vary, but a popular scenario is that there will be a doubling of present atmospheric CO_2 concentrations (from 340 ppm to 680 ppm) coupled with a 2–5°C increase in mean air temperatures within the next 100 years (Cannell 1990). These changes in climate are likely to have profound effects on crop production and semi-natural plant communities. At ITE Bangor, we are involved in several collaborative programmes, investigating the effects of elevated CO_2 and/or temperature on the growth and physiology of grasses. Here, we report on data from a European Community-funded project aiming to determine the impact of changing climate on the productivity of a major agricultural grass *Lolium perenne*.

The investigation was conducted in the new Solardome facility at ITE Bangor (Plate 30). This provides one of the most sophisticated large-scale exposure systems for climate change research in Europe. It is comprised of eight hemi-spherical glasshouses (Solardomes) in which atmospheres can be controlled, with a high degree of precision, both in terms of CO_2 concentration and temperature. Each Solardome is fitted with special 'Sanalux' glass which extends light transmission into the ultraviolet region, thus maintaining more realistic radiation conditions than under normal glass. A computer system controls the operation of the facility and maintains a continuous record of environmental conditions in the Solardomes. The facility is set up as a factorial design experiment, with two levels of CO_2 (ambient and ambient+340 ppm), two levels of temperature (ambient and 3°C tracked continuously above ambient) and two replicates for each $CO_2 \times$ temperature combination. The desired experimental treatments are effectively controlled throughout the year (Figure 35). Some of the deviations from mean target levels can be attributed to watering and mains power failures. The dip in concentration for the elevated CO_2 Solardomes around week 29 is an example of power supply interruptions due to stormy weather.

Swards of *Lolium perenne* cv. Melle were raised in an unheated glasshouse in July 1992. John Innes no. 2 potting compost was used as a growing medium and seedlings were cut to a height of 6 cm at two and three weeks after sowing to ensure a closed sward. Six experimental

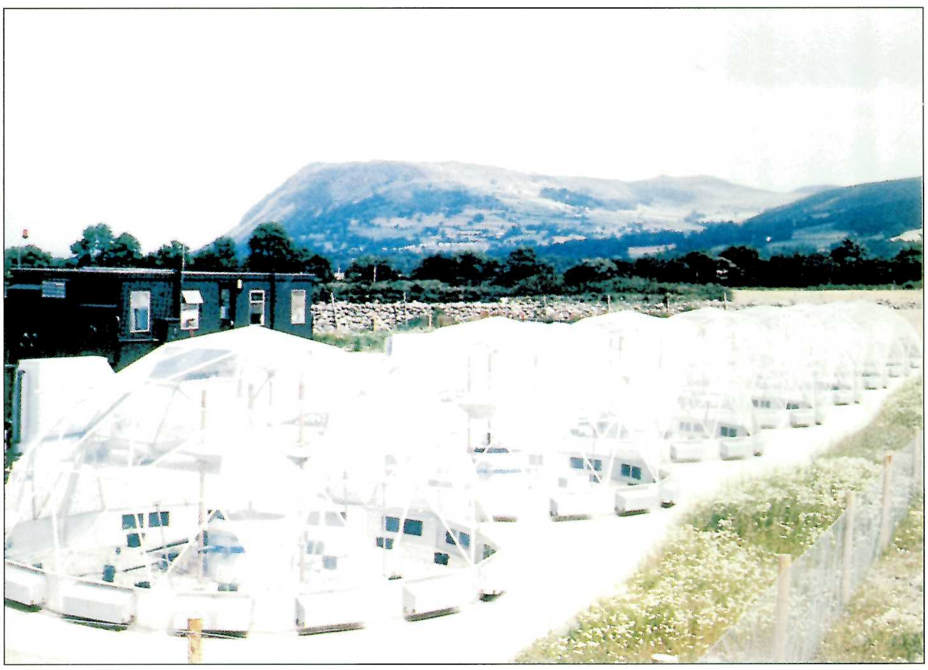

Plate 30. Solardome climate change facility

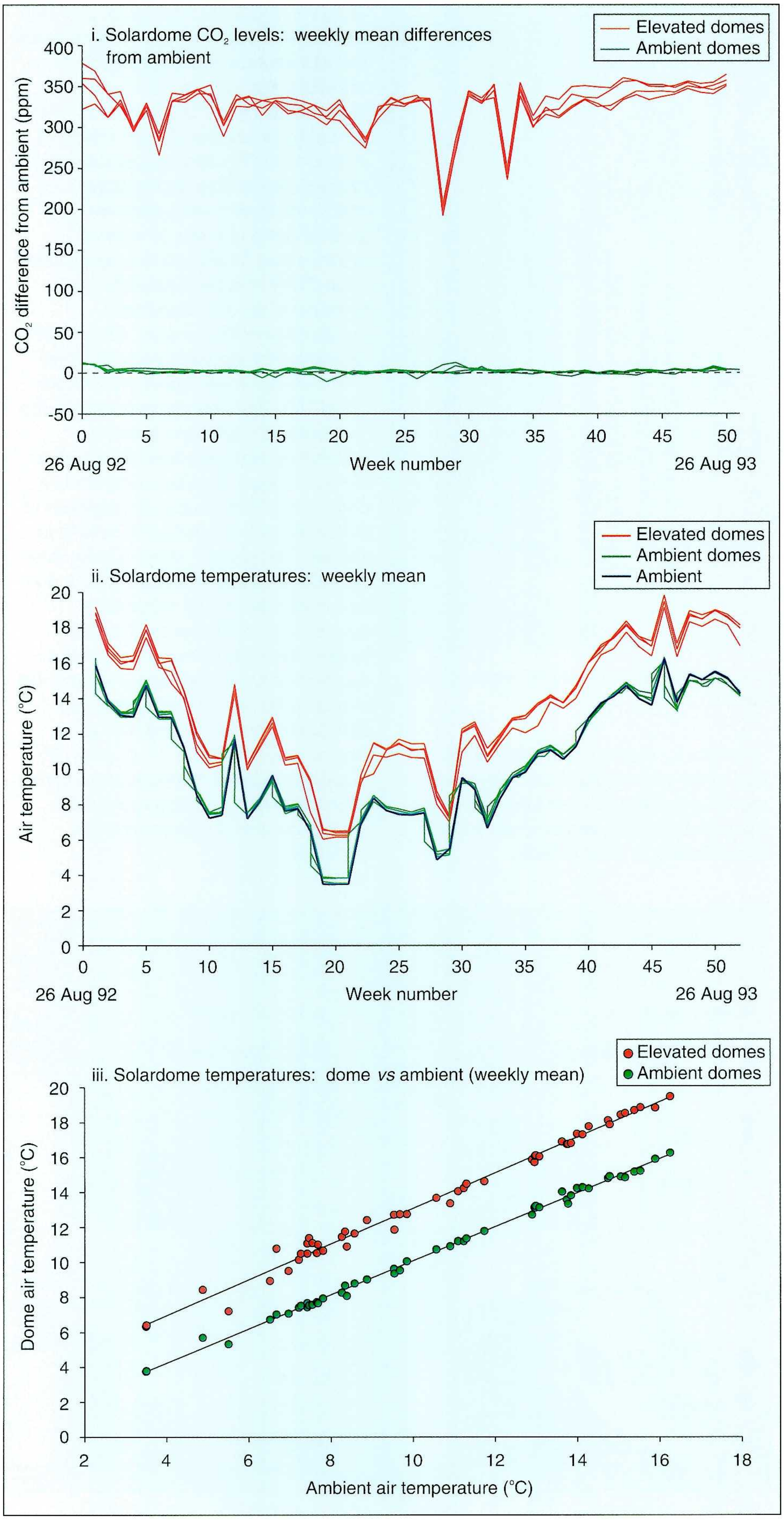

Figure 35. Summary of treatments maintained within Solardomes, August 1992/August 1993 (i) Differences from ambient CO_2 concentrations; (ii) and (iii) Weekly mean temperatures

swards were transferred to each of the eight Solardome exposure glasshouses and cut back to 6 cm in mid-August (day 0) for the start of experimental treatments. The swards remained in the Solardomes for 391 days during which they were maintained with non-limiting supplies of nutrients and water. At frequent intervals, swards were cut back to a height of 6 cm and clippings collected from a 20 cm × 20 cm quadrat. Total dry weight and leaf area were determined at each harvest.

Both elevated CO_2 and elevated temperature resulted in a 10–15% increase in accumulated dry matter production of *Lolium perenne* over the 13 months of the investigation (Figure 36). However, this was not increased further in response to combined elevated CO_2 + elevated temperature. These increases are lower than predicted from many other studies where a general 30% increase in yield has been reported for a doubling of CO_2 concentration (Kimball

Table 7. Summary of significant differences shown by analysis of variance of dry weights of clippings of *Lolium perenne* for individual harvests (* $P \leq 0.05$; ** $P \leq 0.01$; *** $P \leq 0.001$)

Time of harvest (days)	CO_2	Temperature	CO_2 × temp
8	***	*	NS
15	*	NS	NS
22	NS	NS	NS
29	NS	NS	NS
37	NS	NS	NS
50	NS	NS	NS
64	NS	NS	NS
78	NS	***	NS
105	NS	***	NS
134	***	NS	NS
162	*	**	NS
190	***	**	*
226	***	NS	***
252	NS	NS	***
267	*	NS	NS
280	**	NS	NS
296	NS	NS	NS
309	NS	NS	NS
323	NS	NS	NS
336	NS	NS	NS
351	*	NS	NS
364	*	NS	NS
379	*	NS	NS
391	NS	**	NS

The experimental treatments began on 17 August 1992 (day 0). Critical periods for growth responses were days 0–15 (17 August/ 1 September 1992), days 64–280 (20 October 1992/24 May 1993) and days 351–391 (2 August/13 September 1993)

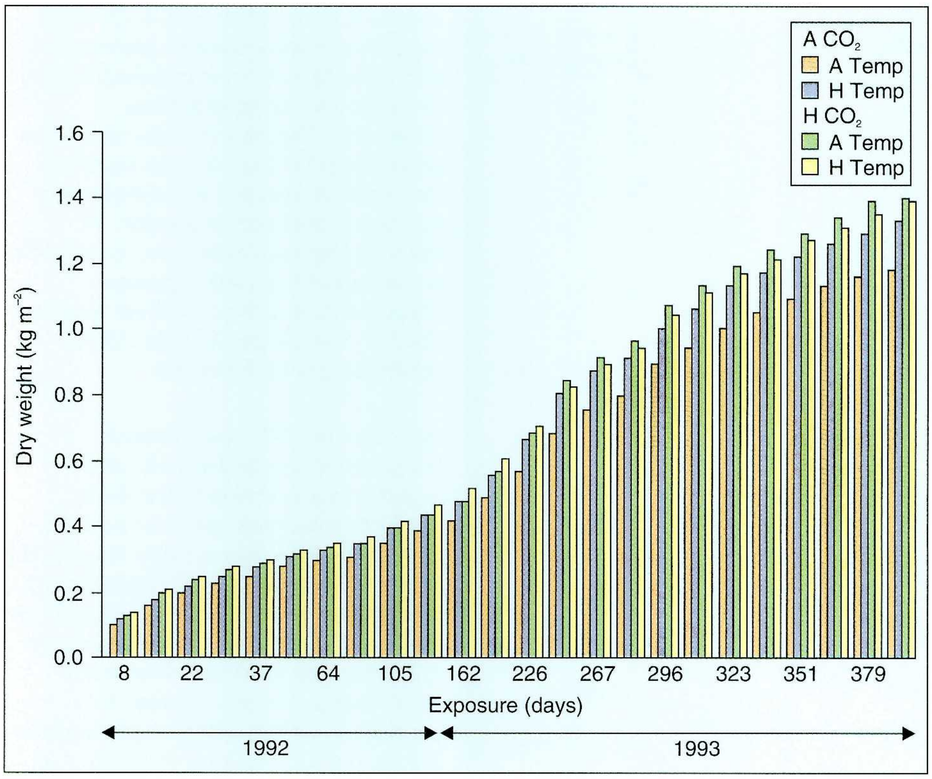

Figure 36. Effects of $CO_2 \times$ temperature regimes on cumulative dry weight of leaf clippings from swards of *Lolium perenne* (A ambient; H elevated (high))

1983). However, previous workers have also reported only a moderate growth response of *Lolium perenne* to elevated CO_2, and it has been suggested that potential growth is severely restricted by the absence of CO_2 effects on tiller number (Ryle, Powell & Tewson 1992).

There were substantial differences in biomass production between harvests which largely reflected time of year. The highest rates of growth occurred at the start of the experiment, and then in March/April (days 190–267) of 1993. There was a second flush of growth in late May/early June (days 280–296). Differences between treatments were not consistent for successive cuts, the highest yields being found for all four treatments at different individual harvests (Table 7). However, there was a greater frequency of occasions when the lowest yield was recorded for the ambient $CO_2 \times$ ambient temperature treatment. Exposure to elevated CO_2 resulted in higher dry weight yields ($P \leq 0.05$) on 11 of the 24 harvests. These effects occurred in distinct phases and were not linked to flushes in growth. They occurred at the start of the exposure period (days 0–15), during winter/spring (days 105–280) and, again, in late summer (days 336–379). At a smaller number of harvests, there was increased dry matter production in response to temperature, again at the start of the exposure period (days 0–8) and in late autumn/winter (days 64–190), when ambient temperatures were most likely limiting to plant growth.

Where a significant interaction between elevated CO_2 and elevated temperature was observed in early spring (days 162–252), the effect was to reduce productivity below that expected from the independent stimulatory effects of increases in CO_2 and temperature. Factors responsible for treatment effects on growth were analysed in greater detail in terms of relative growth rates (RGR) and net assimilation rates (NAR), the latter providing a measure of the photosynthetic productivity of the swards. Shoot RGRs calculated for the initial harvest (days 0–8) and one in late January/February (days 162–190) clearly demonstrate the shift from an additive to less-than-additive growth response of swards to elevated CO_2 + elevated temperature (Figure 37). This change in response of RGR with time was reflected in NAR of swards. The greater effect of increases in CO_2 than of temperature on RGR was due to effects on NAR rather than increases in the leafiness of swards. Indeed, throughout the course of the investigation, the leafiness of swards (as indicated by the leaf area ratio) was significantly reduced ($P \leq 0.05$) in response to elevated CO_2 (Figure 38). In contrast, elevated temperature in the absence of CO_2 resulted in a notable stimulation in leaf area production ($P \leq 0.001$).

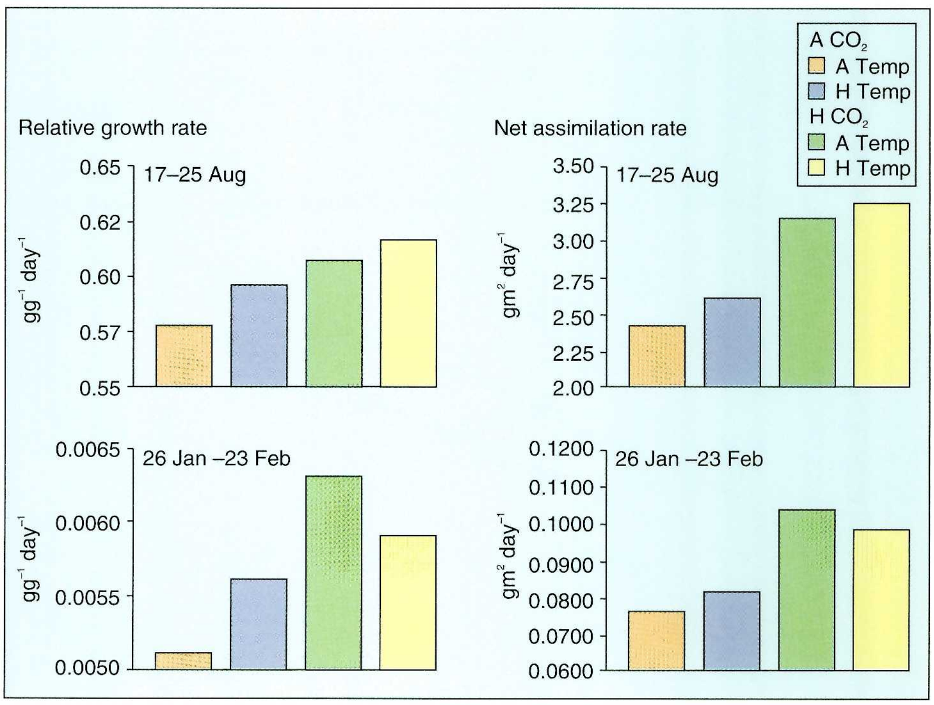

Figure 37. Effects of $CO_2 \times$ temperature regimes on relative growth rate and net assimilation rate of *Lolium perenne* swards at two stages of the season: 17–25 August 1992 (days 0–8) and 26 January/23 February 1993 (days 162–190) (A ambient; H elevated (high))

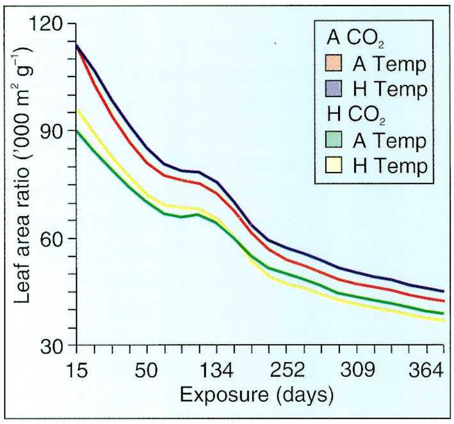

Figure 38. Effects of CO_2 × temperature regimes on leaf area ratio of *Lolium perenne* swards (A ambient; H elevated (high))

A major fraction of the extra yield produced under elevated CO_2 and elevated temperature was derived from increases in RGR and NAR during the first eight days after exposure to treatments (Figure 37). These results are consistent with the widespread observation that initial increases in photosynthesis with increases in CO_2 concentration do not persist in the long term. It may be that the poor plasticity of growth of *Lolium perenne* at elevated CO_2 results in acclimation of key physiological processes which restrict the continued utilisation of the extra assimilates for the regrowth of shoots.

Data collected from other countries within our CEC programme suggest that grassland production in temperate Europe will be generally 10–20% higher than present, with a doubling of CO_2 concentration. Research on the effects of changing climate on the growth of these swards of *Lolium perenne* will be continued for another growing season to determine if growth responses to the treatments persist. In the next phase of our research we will undertake more detailed studies of the physiological processes underpinning the growth responses observed in a wider range of species of ecological significance in semi-natural pasture.

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Methane emissions from wetlands at the landscape scale

(This work was funded by the NERC Terrestrial Initiative in Global Environmental Research (TIGER) programme)

Methane (CH_4) is known to play an important role in global warming, contributing around 15% of the radiative forcing due to trace gases (which comprise CO_2, CH_4, N_2O and CFCs). Although there are many sources of CH_4, the main global source is believed to be natural wetlands, estimated to produce around 115 Tg CH_4 yr^{-1}. There are, however, large uncertainties in the estimates of emission rates from wetlands and the processes controlling them. Most published measurements of CH_4 fluxes have been obtained using enclosure (or cuvette) methods, over 0.1–1.0 m^2 of the surface, and the huge spatial variability in fluxes (orders of magnitude over 0.5 m) has made it very difficult to extrapolate from these measurements to estimate emissions at the landscape scale.

Recently, two techniques have been applied by ITE Edinburgh to overcome this problem and to obtain a direct measure of landscape scale fluxes using micrometeorology and boundary-layer methods. Both methods were employed during a joint field experiment with the University of Manchester Institute of Science and Technology (UMIST) and the University of Edinburgh in summer 1993 over the flow country of northern Scotland – a large area of blanket bog typical of many northern wetlands. The site chosen was near Loch More in Caithness, situated in a large, relatively level region of blanket bog.

Micrometeorology

The micrometeorological approach provides landscape scale (10^4–10^6 m^2) average fluxes over periods of 30–60 minutes by making measurements of gas concentration and windspeed at a single point several metres above the surface. The technique chosen was the eddy correlation method, where the flux of any trace gas is given by

$$F = \overline{w'\chi'}$$

where F is the flux (μg m^{-2} sec^{-1}), w' is the instantaneous deviation of vertical windspeed from the mean (m sec^{-1}), and χ' the instantaneous deviation of gas concentration from the mean (μg m^{-3}). To make eddy correlation measurements, a

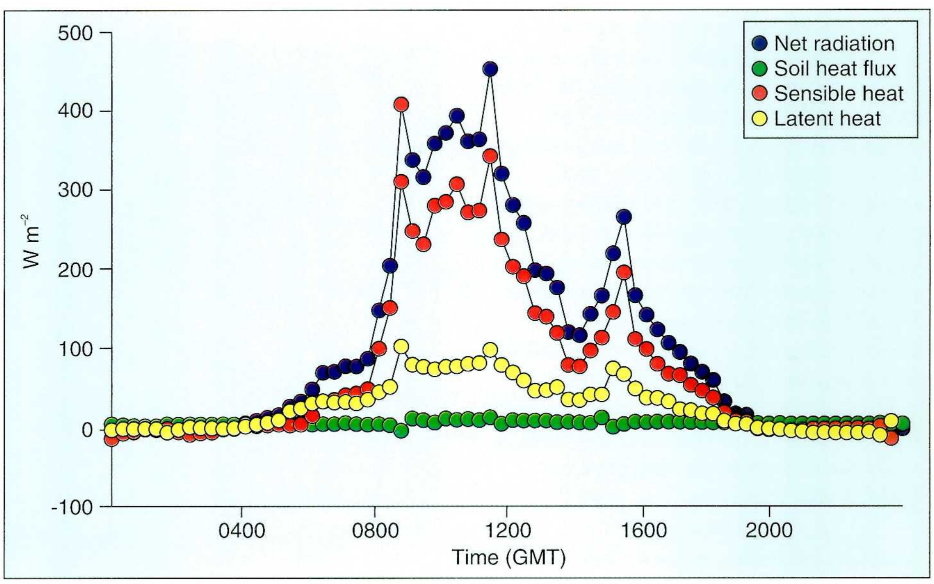

Figure 39. Loch More energy balance – 29 May 1993

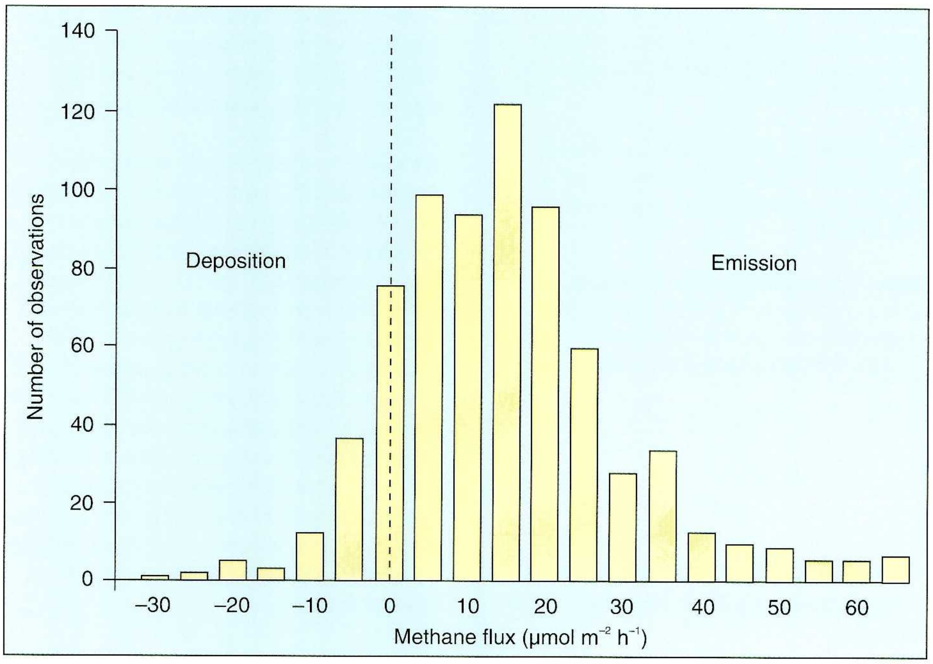

Figure 40. Frequency distribution of methane fluxes over Loch More during three weeks in May/June 1993

fast-response gas analyser and an ultrasonic anemometer were required, and these were obtained with funding from the NERC TIGER programme. The choice of ultrasonic anemometer was fairly easy, a suitable type being available commercially. However, the only instrument currently suitable for making fast-response measurements of methane is a tunable diode laser spectrometer (TDL), and this was not commercially available. Enquiries were made with several companies capable of manufacturing a suitable TDL, and a machine was subsequently ordered from Aerodyne Research in the USA.

The instrument is constructed on a 2' × 2'4" optical bench and comprises a liquid nitrogen-cooled dewar housing the diode lasers and infrared detectors, a fully adjustable optical path, a monochromator for characterisation of diodes, and a modified Herriott flow cell with an optical path of 36 m and a volume of 0.3 l. A high-capacity vacuum pump provides a low pressure in the flow cell (approximately 5.3 kPa) to minimise pressure broadening, and a response time of 0.25 sec. Laser temperature and current control are performed by Laser Photonics Inc instruments, and the whole system is controlled by software running on a 50 MHz 486 computer.

Methane concentrations are determined by scanning the laser frequency over two or three wavenumbers around a strong infrared methane absorption line at 3017 cm^{-1}. The area of the peak is determined in real-time using a non-linear least squares fit and the methane concentration is calculated by comparison with the peak area obtained using cylinder standards. The output from the instrument is available at rates up to 20 Hz – the maximum rate being determined by the speed of the computer. At a data rate of 20 Hz, the TDL has a typical noise of 0.5% (9 ppb CH_4 at ambient concentrations). By reducing the data rate to 1 Hz and averaging over 2500 spectra, the typical noise is reduced to 0.05% (1 ppb CH_4).

After a period of laboratory testing and familiarisation with the operation of the TDL, equipment was prepared for use in the field and the TDL was installed in a mobile laboratory (Tow-a-Van). Eddy correlation measurements of CH_4 exchange were carried out over a three-week period on a near-continuous basis. Wind profiles, energy balance, carbon dioxide and ozone fluxes were also measured to provide background information. Figure 39 shows a typical energy balance for the site on an unusually warm day for this area, with high levels of net radiation. Most of the available energy was partitioned into sensible (dry) heat, despite quite wet conditions underfoot and extensive areas of pools. The small latent heat flux (evaporation and transpiration) results from the slow-growing moorland vegetation, predominantly heather (*Calluna*) and cottongrass (*Eriophorum*), which shows a large minimum canopy resistance to water vapour. A frequency distribution of approximately 350 hours of semi-continuous methane fluxes taken over three weeks is shown in Figure 40. This reveals an apparently normally distributed emission rate with a mode of 15 µmol m^{-2} h^{-1} at a bin size of 5 µmol m^{-2} h^{-1}. This value is in close agreement with results obtained by scientists at the University of Edinburgh using a conditional sampling (relaxed eddy accumulation) technique. The peat wetland fetch used for the measurements, with a water table ranging from a few cm to 30 cm below the surface, provided emission fluxes most of the time with values as large as 50 µmol m^{-2} h^{-1}. However, methane deposition (oxidation by micro-organisms) was also observed

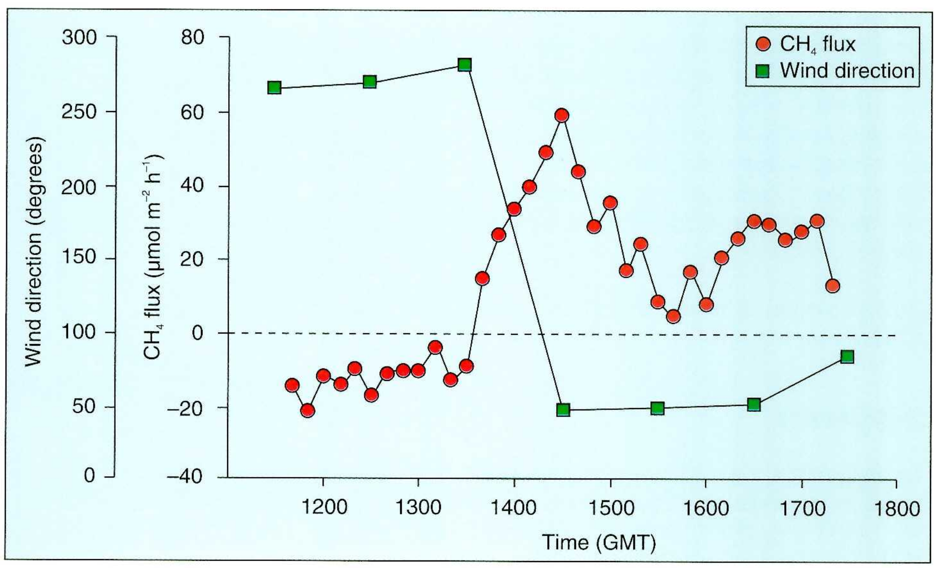

Figure 41. An apparent change in methane flux resulting from a change in wind direction

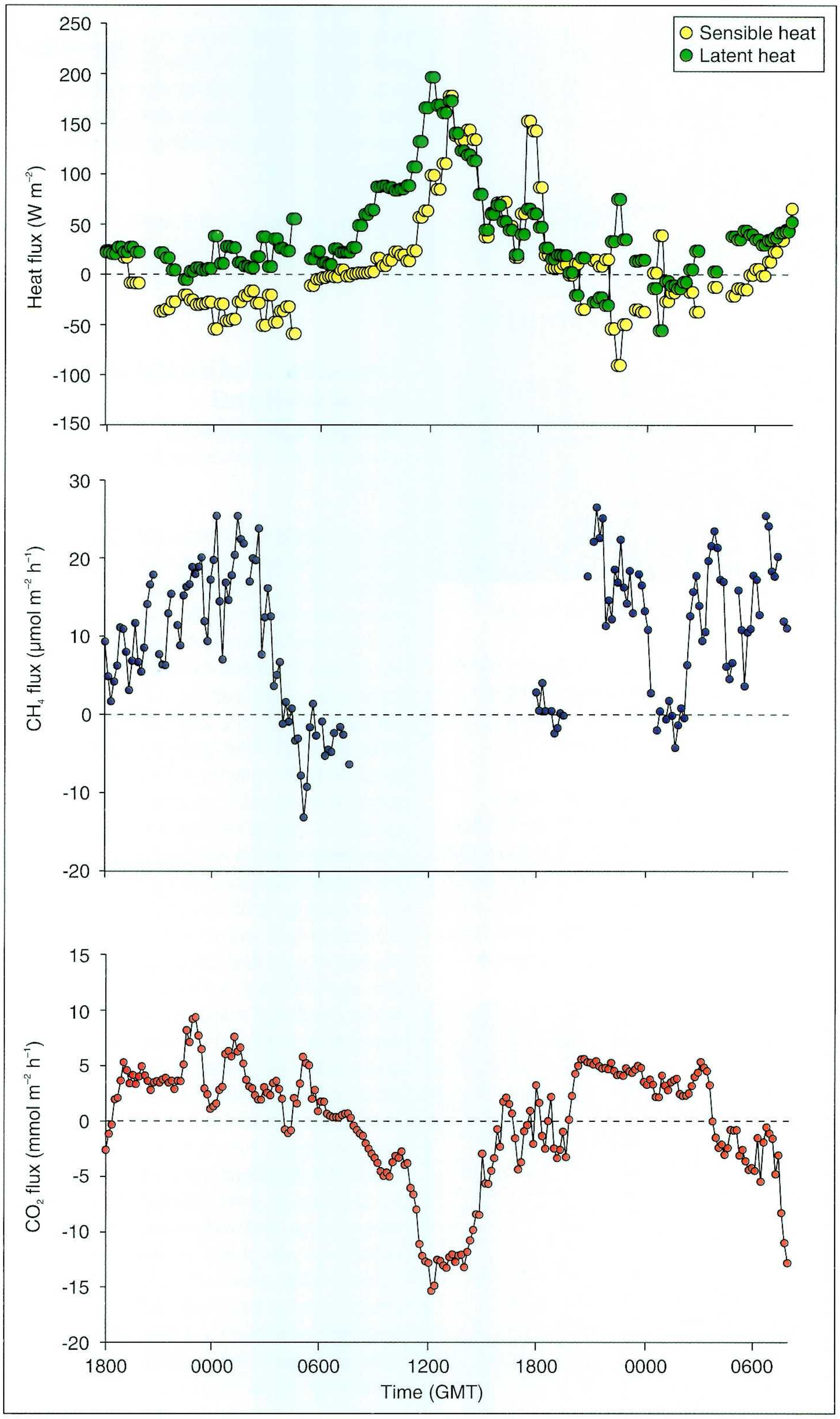

Figure 42. Example of carbon balance at Loch More, 5–7 June 1993

about 8% of the time. Closer inspection of a period towards the end of the campaign (Figure 41) reveals a dependence of flux direction on wind direction. In the period up to 1330 GMT, the wind was westerly over an area of bog consisting mostly of dry hummocks, and during this period fluxes were negative showing CH_4 oxidation; however, the rapid change in wind direction to north-easterly (with air advected across a very wet, low-lying area with pools) changed the methane flux from deposition to emission. This dependence of flux direction on water status has been demonstrated in open-top chamber studies at ITE Edinburgh, although in this case the magnitude of the deposition flux is much smaller.

Figure 42 illustrates a period of 38 hours on 5–7 June, where CH_4 and CO_2 fluxes were measured simultaneously. CH_4 was emitted for most of the period, except for a short period around dawn on 6 June when small rates of deposition were seen. This was thought to be a result of a change in wind direction. Nocturnal CO_2 respiration was around 5 mmol m^{-2} h^{-1} and rates of photosynthesis were relatively small, peaking at 15 mmol m^{-2} h^{-1}. The methane flux thus represented around 0.2% of the net carbon balance.

Boundary-layer measurements

In addition to the ground-based measurements, estimates of the CH_4 emission from a large area of Caithness and Sutherland were made by examining the changes in CH_4 concentration within and above the daytime boundary-layer in collaboration with UMIST. These data, together with measurements of boundary-layer depth and upper air windspeed and direction, enable a large region to be treated as a 'big box' within which emissions of CH_4 from the surface are trapped. The Meteorological Office Research Flight C130 Hercules was used on four separate days during the Loch More Experiment, with ITE staff on board the aircraft filling 5 l Tedlar sample bags with air (Plate 31). These samples were returned to the site and analysed using the TDL. Aircraft-borne instrumentation provided the data for boundary-layer height and windspeed. Wind direction data were obtained from measurements made on the ground at Loch More, and by studying back trajectories. Figure 43 illustrates the concentration data from one flight during a period of south-easterly airflow. A methane plume is apparent off the north coast of Scotland and is consistent with a mean emission flux of 270 µmol m^{-2} h^{-1}, averaged over 2500 km^2 of Caithness. This figure is larger than the ground-based measurements, but could be explained by the presence of very wet but inaccessible areas within the region (eg Knockfin Heights) which are believed to be the large CH_4 emission areas.

Conclusions

The use of TDL spectroscopy to provide a high-sensitivity, fast-response methane sensor has allowed estimates of landscape-scale emissions to be

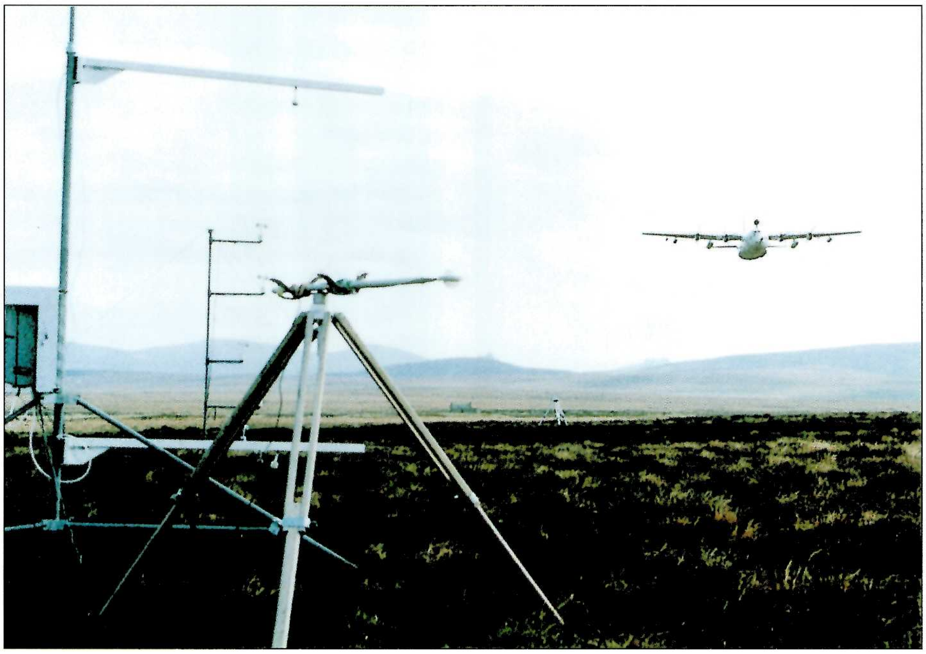

Plate 31. The Meteorological Office Research Flight C130 Hercules over the Loch More site

measured directly for the first time in the UK. The development of boundary-layer budget methods in collaboration with UMIST has enabled the scale of the measurements to be extended to regions (in which micrometeorology would be unsuitable) through use of the aircraft. The discrepancy between the two measurement techniques has not yet been adequately explained, although the use of a common instrument to measure CH_4 concentrations should eliminate measurement problems as the source of the error. It is most likely that spatial variability at the *landscape* scale is responsible for this difference.

The fluxes span the range of values observed in North America (Alaska) but are generally larger. They also show that the average fluxes over very large areas are closely coupled with water table and temperature of the upwind fetch and with full analysis will provide the functional dependence of methane emission and deposition to peatlands in a range of atmospheric and surface conditions. This may then be applied across regions to compute the net source strength and the response to climate change over the next three decades.

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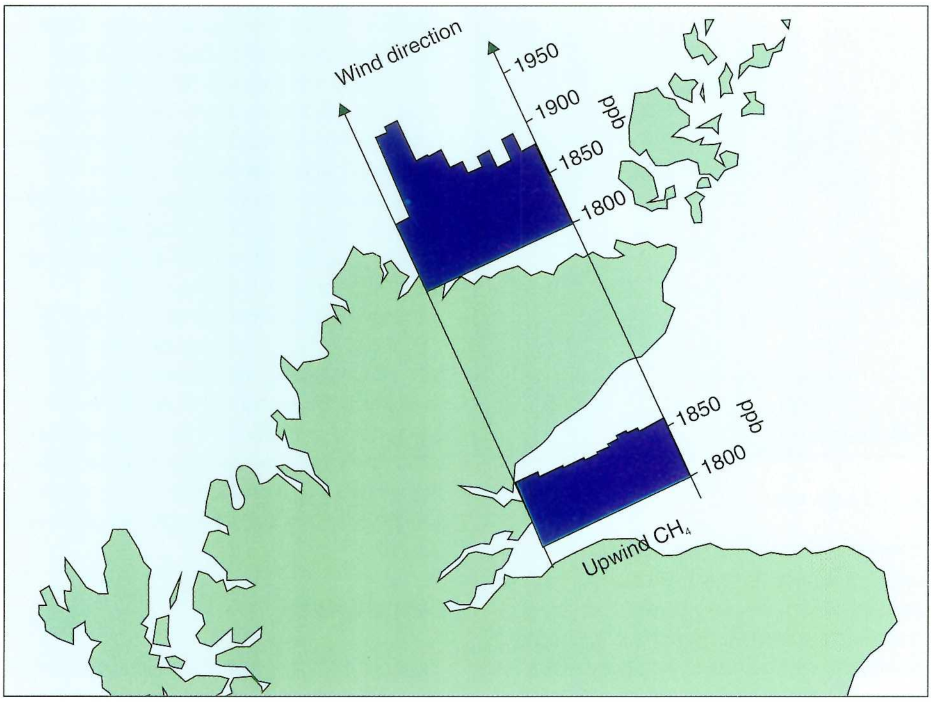

Figure 43. Boundary layer budget – average emission flux over 50 km of Caithness/Sutherland of 270 μmol CH_4 m^{-2} h^{-1}

Impacts of climate change upon wetland biogeochemistry

(This work was funded partly by the Welsh Office)

Peat-accumulating wetlands occur on land which experiences waterlogging due to high rainfall and poor drainage. They cover almost 10% of Britain and Ireland, and are essentially unbalanced ecosystems, where the rate of peat production exceeds that of decomposition. It is now widely acknowledged that wetlands are important for a variety of reasons that are expressed at local, regional and even global scales. At the local level, they are of immense value not only from a recreational perspective because of the great beauty of this habitat, but also from a conservation viewpoint, due to the variety of rare and unusual plant species that are unique to wetland sites. At a wider 'regional' scale, it has been noted that wetlands lying between terrestrial and aquatic ecosystems have the ability to 'filter out' potentially detrimental chemicals (of terrestrial origin) before they reach the recipient aquatic systems. Thus, they prevent riverine transport of such materials over large distances. At a global level, wetlands contribute significantly to global emissions of 'greenhouse' gases. Methane emissions are currently receiving considerable scientific attention for it has been estimated that a substantial 20% of global emissions of this gas originate from wetlands (Cicerone & Ormeland 1988).

Because these properties are unique to wetland systems, it seems likely that wetland functioning is largely dictated by the waterlogging, for this is the one factor that sets wetlands apart from other ecosystems. Their dependence upon waterlogging for stability is of interest because it has been proposed that the 'greenhouse effect' could increase the

Plate 32. The Cerrig yr Wyn field site, showing the diversion of inflowing waters through a system of pipes in order to simulate drought in the lower wetland. The intact control wetland is visible in the background

frequency of high-pressure areas (anticyclones) that we experience (Meteorological Office 1989). This could lead to an increased number of droughts because of a weakening of the rain-bearing Westerlies which give the UK and western Europe a characteristically wet climate. The consequent drying of our wetlands could radically alter their biogeochemical functioning.

Our early investigations of the potential effects of drier conditions within wetlands were carried out through laboratory experiments at the School of Biological Sciences, University of Wales, Bangor (Freeman, Lock & Reynolds 1993). We are now following up these pilot studies with a full-scale field experiment which attempts to simulate realistically drier conditions within a gully wetland site in mid-Wales. The work involves collaborative research with the University of Wales (Bangor) and Institute of Hydrology, Plynlimon.

In the case of gully mire wetlands, the main source of flushing water is catchment-derived streamflow rather than rainfall, and thus the effects of drought can be induced by preventing streamwater from recharging the mire. The gully chosen for this experiment, Cerrig yr Wyn (Plate 32), has stepped flush areas in series, characterised by *Sphagnum* and rush (*Juncus*) communities, and is connected hydrologically by natural pipes flowing at the interface between the peat

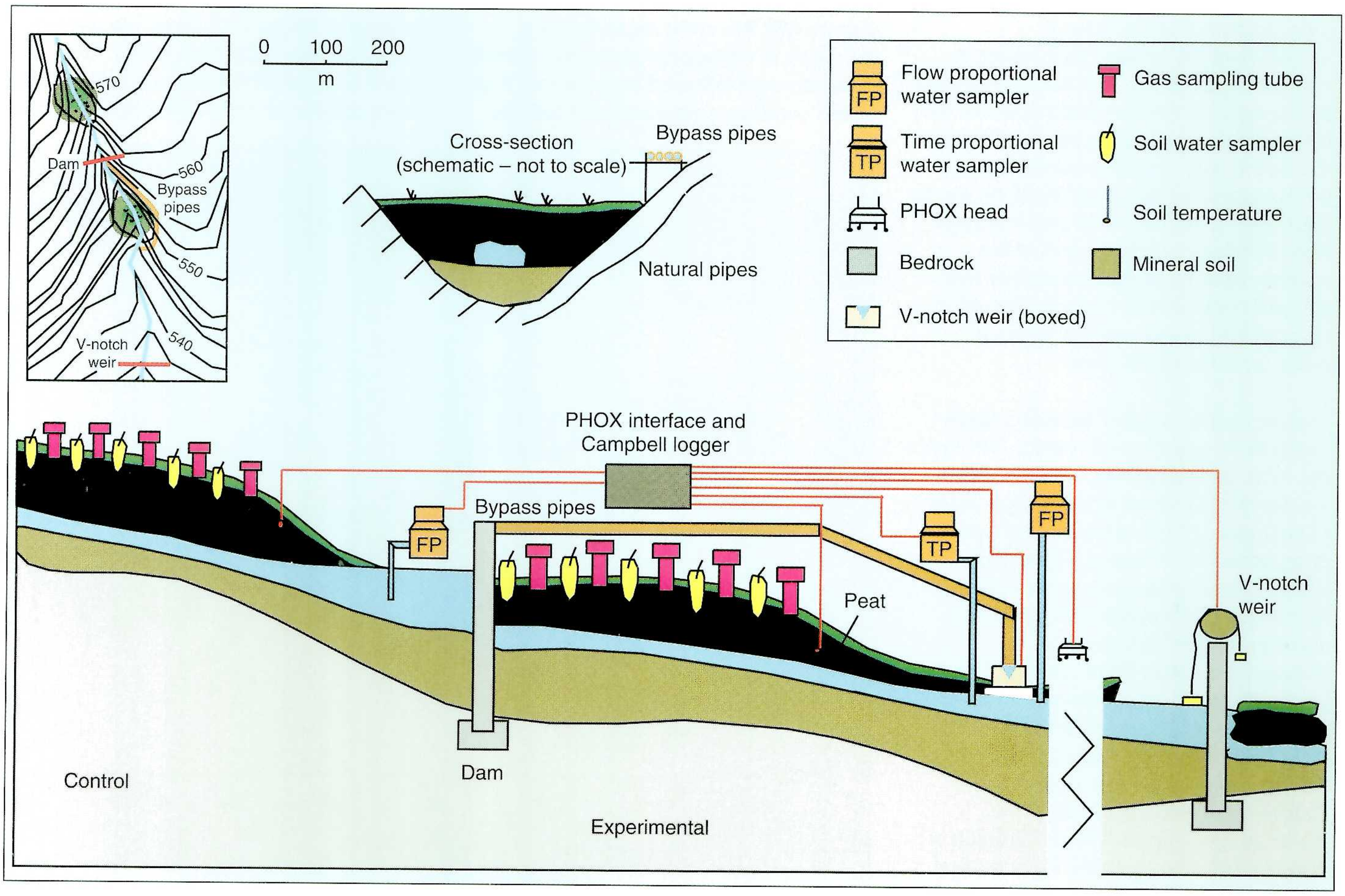

Figure 44. Design and instrumentation of the experimental field site

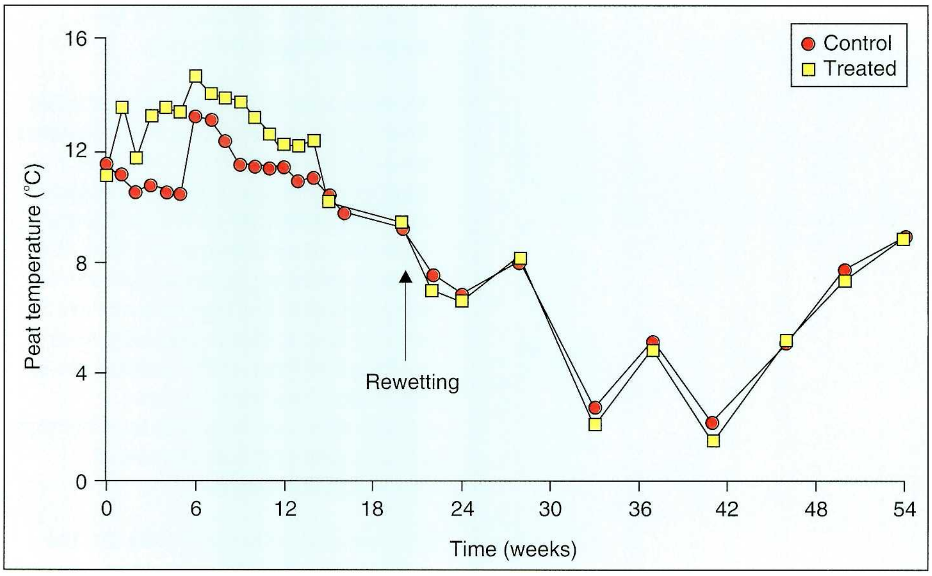

Figure 45. Effects of the simulated drought upon peat temperature in the upper 10 cm. Drought was simulated between weeks 0 and 20, after which normal water flows were re-established

and mineral deposits in the bottom of the gully. The stepped nature made it practical to select and delineate control and experimental areas.

At Cerrig yr Wyn (British National Grid Reference SN 820 866), the delineation has been achieved by excavating the gully upstream of the area of 'experimentally treated' flush, down to impervious bedrock, and installing a dam which seals off the flow and bypasses the flush using a plastic pipe system (Figure 44). In this way, only severe storms, which exceeded the capacity of the pipes, were able to recharge the experimental flush. At the downstream end of the experimental flush, the streamflow was allowed to discharge back into the main channel and to mix with the drainage water from the study area.

Our early observations revealed some particularly unexpected results. Perhaps the most surprising was the rapid onset of a significant increase in the temperature of the upper 10 cm of the treated area compared to the control (Figure 45). This rise in temperature only occurs during the period of simulated drought. If this should prove to be a general phenomenon, then there are significant implications for (temperature-dependent) microbial activity, and thus the biogeochemistry of wetlands. Microbial activity governs the rate of peat degradation, and thus there is the potential for a substantial mobilisation of materials which would normally remain sequestered within the peat. This hypothesis is supported by Figure 46, where it can be seen that concentration of magnesium in waters released from the drought-impacted wetland increased by approximately 50% during a drought simulation.

Another unexpected finding was the effect of the simulation upon methane emissions (Figure 47). We were expecting a reduction in emissions during the period of the drought followed by a recovery as wetter conditions returned. Instead, we found that there was little response during the simulated drought, and, most surprisingly of all, a suppression of emissions during the period of rewetting. It is possible that the inhibitory effects of drier conditions on methane emissions may have been mitigated by the stimulatory effects of warmer peat temperatures (Figure 45). Following the drought, methane emissions correlated inversely with the concentration of sulphate within the peat ($r=-0.61, P<0.01$). This apparent indication of a role for sulphate in the post-drought suppression of methane fluxes is supported by work showing that (i) sulphate abundance increases following droughts (eg Braekke 1981), and (ii) sulphates inhibit methanogenesis (Ormeland 1988). However, the most interesting aspect to the findings is that they suggest that the post-drought suppression of methane fluxes may have more significance in the global regulation of methane emissions than the effects of the drought itself. The annual flux of methane from the wetland experiencing the simulated drought was almost halved, compared with that of the control. If, following climate change, this occurred across all northern peatlands, then global methane emissions could potentially fall by *c* 23 Tg yr^{-1} (based on flux estimates of Gorham 1991). This figure compares with the current decadal mean imbalance between global sources

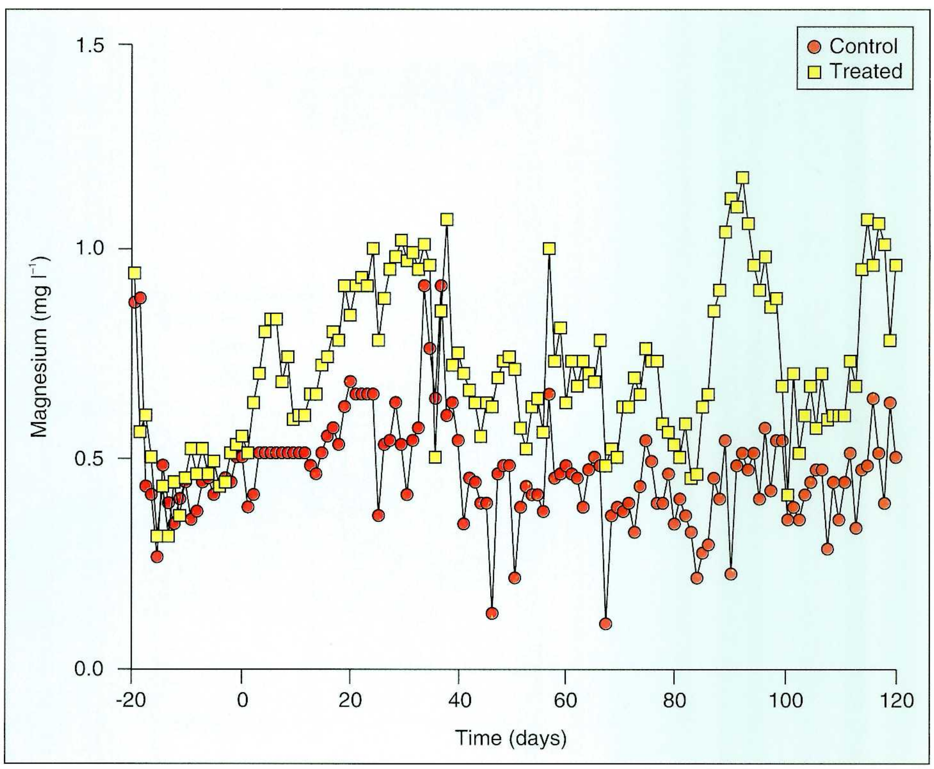

Figure 46. Effects of the simulated drought upon the release of magnesium from the two wetlands. Samples were collected daily using an automated sampler

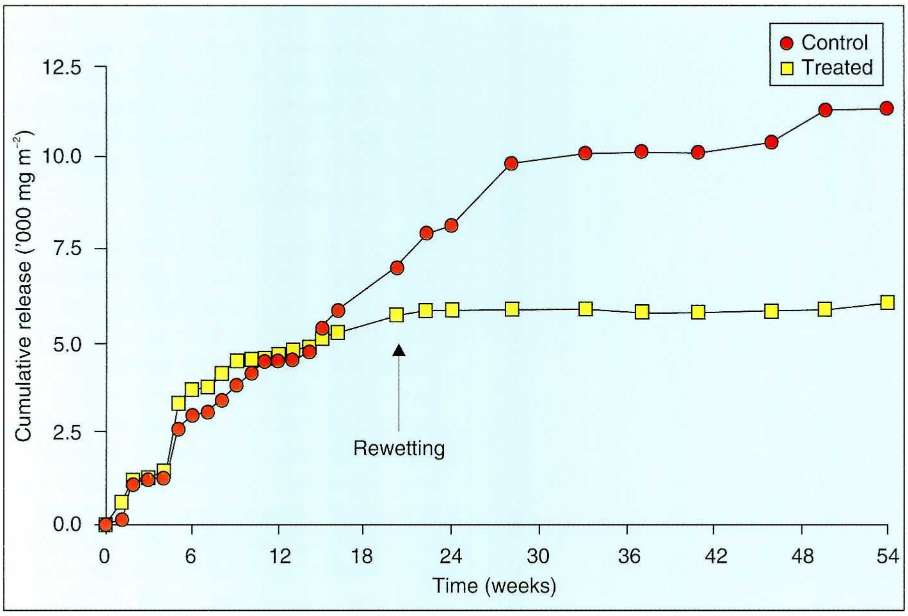

Figure 47. Effects of the simulated drought upon methane emissions from the two wetlands. Drought was simulated between weeks 0 and 20, after which normal water flows were re-established

and sinks of methane of 31 Tg yr^{-1}. Thus, an increased frequency of droughts (and hence post–drought periods) following a change of climate could substantially reduce the rate at which methane is accumulating in the atmosphere.

Even at this early stage, our experiment has given us a new insight into the possible effects of a change to a drier climate, and we anticipate that, as further results emerge from the study, our understanding of the factors regulating wetland biogeochemistry will be greatly enhanced.

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Salt marsh processes in four European countries

(This work was funded partly by the European Union and by the Ministry of Agriculture, Fisheries and Food)

Salt marshes are intertidal areas of fine sediments brought in by the sea and stabilised by the growth of vegetation. The species and communities making up the vegetation are in distinct zones that are related to the duration of submergence during periods of high tide. In Britain salt marshes are particularly extensive in East Anglia where they cover nearly 9000 hectares, but there are extensive marshes along most of the coasts of north-west Europe. Salt marshes are important for the wide range of distinctive plants and animals they support (Plate 33). As they occur near the level of high tide, they also provide protection for the sea walls. In many areas the walls are plain earth banks which are particularly vulnerable to wave attack. In addition to their value for sea defence, the salt marsh plant communities are particularly productive and some of this productivity is transferred to adjacent coastal waters where it can support marine food chains.

Salt marshes have declined in total area over the years because they have been reclaimed extensively to provide land for agriculture and for industrial and residential development. Nevertheless, up to the 1970s the marshes of East Anglia generally showed an overall positive balance, with local erosion, even severe erosion, being outweighed by accretion elsewhere. More recently, however, the balance has changed towards erosion and a general loss of the marsh area. This change has been particularly marked in Essex (Plate 34), with annual losses in area of 0.75–1.4% (Boorman 1992). Actual losses are probably higher as the estimates do not include increases in the proportion of bare ground within the marsh itself, and an annual loss of 2% is probably a conservative estimate. Moreover, the evidence suggests that the rates of loss are increasing steadily.

The reasons for these losses are not entirely clear and various theories have been put forward to account for them:

- the effects of pollution, including eutrophication;
- local reductions in sediment supply;
- increases in wave action and other erosive forces as a result of the sinking of the land relative to the sea;
- increases in the maturity of the marsh system;

  or, most probably,
- various combinations of some or all of these factors.

If recent projections of a rise in sea levels as a result of global warming are realised, these are certain to exacerbate existing erosive trends, with little compensation in the form of new accretion as sea walls prevent the eroding marshes from migrating landwards and re-forming at a higher level. Without any positive steps to help the creation of new marshes it is estimated that there could be a loss of over 40% of the Essex marshes over the next 60 years (Boorman 1990, 1992).

Plate 33. Typical salt marsh vegetation with the blue-flowered sea lavender (*Limonium vulgare*) and sea meadow-grass (*Puccinellia maritima*) at Tollesbury, Essex

European studies of salt marsh fluxes

The effective management of salt marshes demands an understanding of functional processes, and of the effects of environmental factors. In 1990 a collaborative project was set up with funding from the European Union to study salt marsh productivity and related fluxes of sediments, organic matter, and nutrients (Figure 48; Plate 35) (Boorman *et al.* 1994). This study involved scientists in England, France, The Netherlands and Portugal, with a study site in each country. In England the site was at Tollesbury, Essex. Under a further grant in 1993, the study was broadened to include an assessment of the effect of environmental change on the fluxes and to include at least two sites in each country.

There were major differences between the four sites in terms of the sediment fluxes. The developing nature of the French site in the Baie de Mont St Michel, Brittany, was reflected in the very high imports of suspended sediment, the sediment that provided the material for rapid salt marsh growth and expansion. The Portuguese marsh also imported quite large amounts of suspended sediment, although these imports were an order of magnitude less than in France. The Dutch marsh imported suspended sediment, but again by an order of magnitude less than the Portuguese marsh (Figure 49).

The English marshes, however, differed from all the others in the low amounts of suspended sediment measured in the creek flow. This result is in accordance with the static, even locally eroding, nature of the marsh. Just as the sediment imports in the French marsh were confirmed by the use of sediment traps set in the creek bed, the English sediment trap data suggested that, although the concentration of sediment in the creek generally was low, there was a significant export of sediments at the level of the creek bed. The loss of sediment from the English marshes by this mechanism fitted in with the measured rates of sheet erosion from the vegetated surface taken with the observed creek-head erosion which produces large areas of bare mud that can rapidly be eroded.

There is not necessarily a direct relationship between the chronological age of a salt marsh and the stage of development that it has reached. The study assessed whether functional processes were also related to plant productivity and the spread of vegetation cover. Immature marshes of whatever age generally have a low productivity in the pioneer zones that are developing and extending seaward. Mature marshes tend to show higher productivities in the low-level communities because these have had more time to develop. Over-mature marshes, on the other hand, have lower productivity as the whole system begins to degenerate and release both organic matter and nutrients.

In contrast to the very high import of sediment, the French marsh was overall an export-dominated system, releasing both nutrients and particulate organic matter (with the notable exception of dissolved organic carbon which was imported). This release was associated with a high productivity in the main marsh (Figures 50 & 51).

The Portuguese site also showed large exports of both dissolved organic carbon

Plate 34. Eroding salt marsh at Tollesbury, Essex, characterised by bare mud mounds at levels suitable for salt marsh vegetation

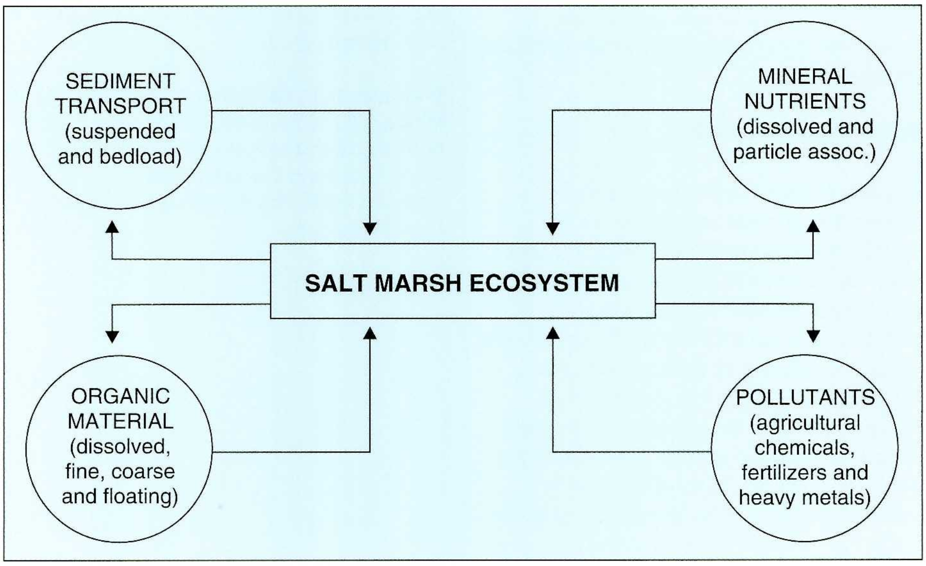

Figure 48. Salt marsh processes

and coarse organic material on an annual basis. This statement conceals, however, the complex situation regarding coarse organic material. Normally there was a modest import but this could be reversed by a single winter storm tide. The situation regarding nutrients was very different with only small, probably insignificant, differences for most components, with the exception of nitrate–nitrogen which is exported. As in France, this export was associated with a high productivity, particularly in the main marsh.

The Netherlands also showed an export of dissolved organic carbon, although the quantities were less. As in Portugal, there was usually an import of coarse organic material but, similarly, this process could be temporarily reversed by winter storm tides. The nutrient situation in The Netherlands was also similar to that in Portugal with only small budgetary differences. Productivity was lower than in the other sites, but this can largely be attributed to the shorter growing season in the most northern of the sites.

In the English site there was an export of coarse organic material, despite the fact that no storm tides were studied. Regarding nutrients, the English site was more dynamic than either The Netherlands or Portugal. There was an export of both nitrate–nitrogen and total dissolved nitrogen, as in France, while in contrast to France there was a small import of phosphorus. Productivity was low in the main marsh but high in the pioneer marsh.

With regard to the basic question of whether salt marshes are sources or sinks of nutrients and organic matter, the present studies suggest that there is no simple answer. European marshes appear to be a sink for all kinds of particulate matter of which a considerable part is organic, but the import/export balance is related to the maturity of the marsh in a developmental sense. Young marshes appear to be the most efficient sinks as they build and develop. Older marshes, on the other hand, tend to become sources particularly as they become over-mature and degenerate. European salt marshes of all stages of development are sources of dissolved organic matter. Nutrient release is important both in developing marshes and in over-mature marshes that are beginning to degenerate. What is very clear is that salt marshes function as processors of organic material of estuarine or coastal origin.

It must be emphasised that these are only tentative conclusions drawn from the four sites originally studied, but they have provided a basis for the development of hypotheses and the planning of a second phase of studies involving at least two sites in each of the four participating countries. The detailed studies are producing an increasing body of data relating to salt marsh function, further augmented in England by the inter-related Land/Ocean Interaction Study

Plate 35. Naturally regenerated salt marsh at Barrow Hill, Mersea Island, Essex. This area was flooded as a result of storm surges, with the final major breach occurring during the 1953 storm surge

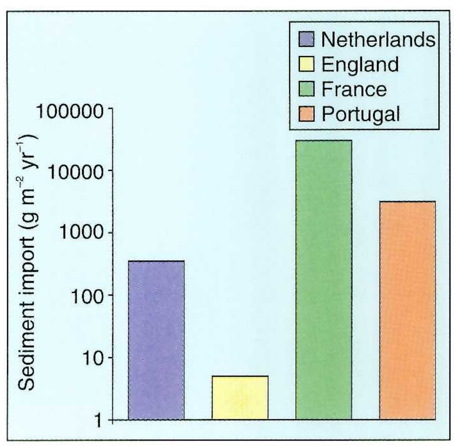

Figure 49. Histogram of sediment flux

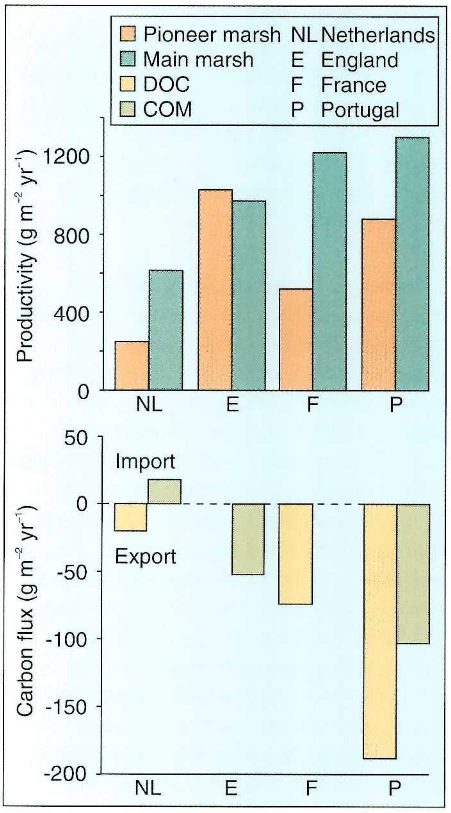

Figure 50. Histogram of productivity and organic matter flux

(LOIS) project. The data collected are being used to construct a model of salt marsh function which will integrate information derived from airborne remote sensing, the productivity of different plant communities, and the behaviour of sediment in water above the marsh surface. The fluxes and fates of carbon and nutrients will also form part of the processes modelled. The model is of the cellular automaton type, as the spatial distribution of plant communities with regard to marsh surface height and

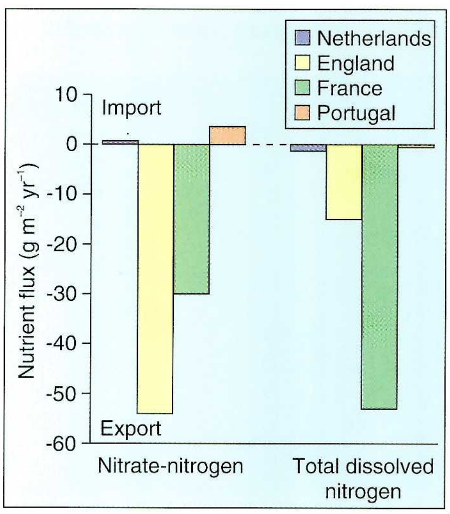

Figure 51. Histograms of nutrient fluxes

distance from sediment sources is important with regard to the deposition of sediment.

Managed setback

In parts of East Anglia, especially in Essex, the present sea defences are reaching the end of their useful life and, with the loss of salt marshes accelerating as a result of sea level rise, very expensive armoured walls will be needed if the present line of defence is to be maintained. There are a number of instances where the replacement of sea defences on their present line cannot be justified economically, technically or environmentally. The alternative is to set back the defence landward and allow the development of salt marsh on land which has been reclaimed. In certain areas, some salt marsh regeneration has occurred naturally following the failure of the sea wall during a severe storm (Plate 35).

The development of new salt marsh is crucial to the success of this 'set-back' policy, and the Ministry of Agriculture, Fisheries and Food has commissioned ITE to carry out research on a large-scale trial of the managed set-back of the sea defence at Tollesbury Creek in Essex. This trial will set back the sea defence line on 21 ha of low-lying agricultural land acquired by English Nature for the purpose. A low embankment will be constructed along the landward boundary to maintain the standards of flood defence to adjacent areas. The existing wall will then be breached in a controlled manner to encourage silt accumulation and the long-term establishment of salt marsh vegetation. Within the site, different techniques for encouraging siltation and for establishing vegetation will be investigated using trial plots. In addition, the effects of creating the new marsh on the existing marshes in the estuary will be carefully monitored.

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