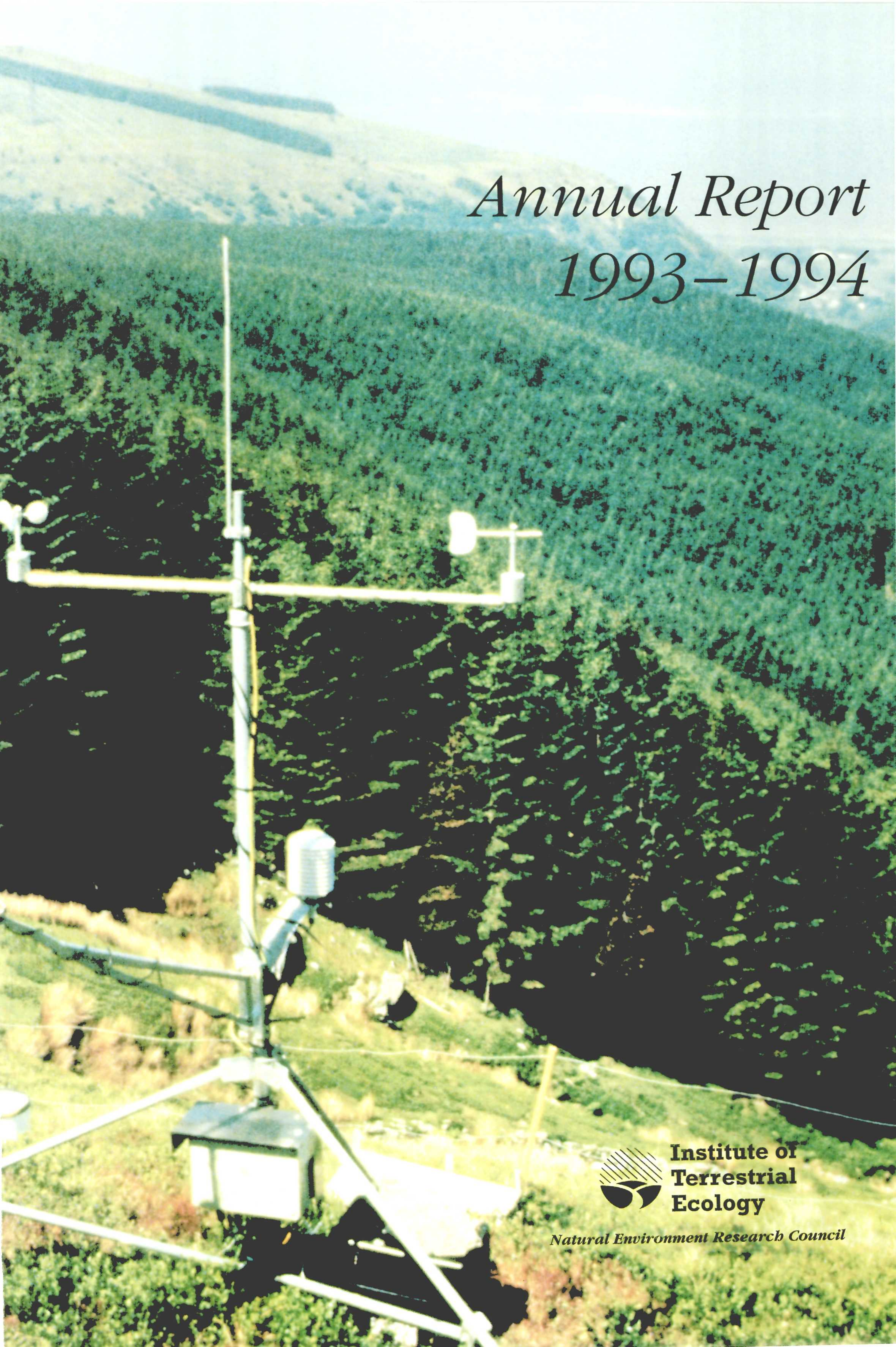
Annual Report 1993–1994



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 1

Forest science

Plate 9. A typical example of UK forestry. Plantations of Sitka spruce in the Borders, Dumfries, Scotland

Much of the work in forest science during the year has been related to global environmental change. Work on the effects of elevated CO_2 on spruce (*Picea* spp.) at ITE Edinburgh (Plate 9), and on oak (*Quercus* spp.) at ITE Bangor has shown, in both cases, that the foliage develops a high carbon/nitrogen ratio, which has two consequences. First, the litter tends to decompose slowly, in accordance with well-known relationships between C/N ratio and decomposition rate. And, second, the foliage tends to be nutritionally poor for leaf-chewing insects, which consequently tend to grow slowly in weight; however, leaf-sucking insects, such as aphids, appear to grow normally on trees grown in elevated CO_2.

The forest models developed in ITE have been deployed to answer questions concerning the effects of climate on forest, and the effects of forests on climate. A model of forest growth and dynamics called HYBRID (Friend, Shugart & Running 1993) that combines the strengths of the JABOWA/FORET-type forest gap models (Botkin, Janak & Wallis 1972) with those of process-based models such as FOREST-BGC (Running & Coughlan 1988), and includes a photosynthesis model, PGEN (Friend 1994), has been developed and deployed to examine questions at regional and global scales. The outline structure of HYBRID is given in Figure 10. When the PGEN part of the model was incorporated into a single-column model of the atmosphere, it was able to show how a decrease in stomatal conductance at elevated CO_2 might decrease transpiration over a rain forest, decreasing cloud formation and rainfall (Friend & Cox 1994). This result contrasts with those from current general circulation models which predict a general increase in rainfall in response to global warming.

Plate 10. A typical example of dryland agroforestry. Homegarden plot at Ndiery, northern Senegal. Fruit trees are interplanted with neem shade trees

Research on forests in the tropics has continued to deliver some surprising information, as evidenced by the report on *Acacia* in Senegal (Plate 10). In Costa Rica, it has been shown that alley cropping with *Erithryna* and *Gliricidia* benefits crop yields by increasing the amount of potassium available to the crop, as well as by fixing N_2. In Cameroon, the root/shoot ratio of large trees has been found to be much larger than previously thought, which may require a revision of biomass and carbon estimates for tropical forests. Also, following work on *Eucalyptus* in India, a model has been developed to predict the interception of precipitation by trees, taking into account rainfall intensity and droplet size.

At home, the disorder of Sitka spruce (*Picea sitchensis*) known as 'bent top', observed in south Wales, has finally been explained as the result of nitrogen and phosphorus deficiency combined with defoliation by spruce aphid. Work elsewhere in Wales has shown that,

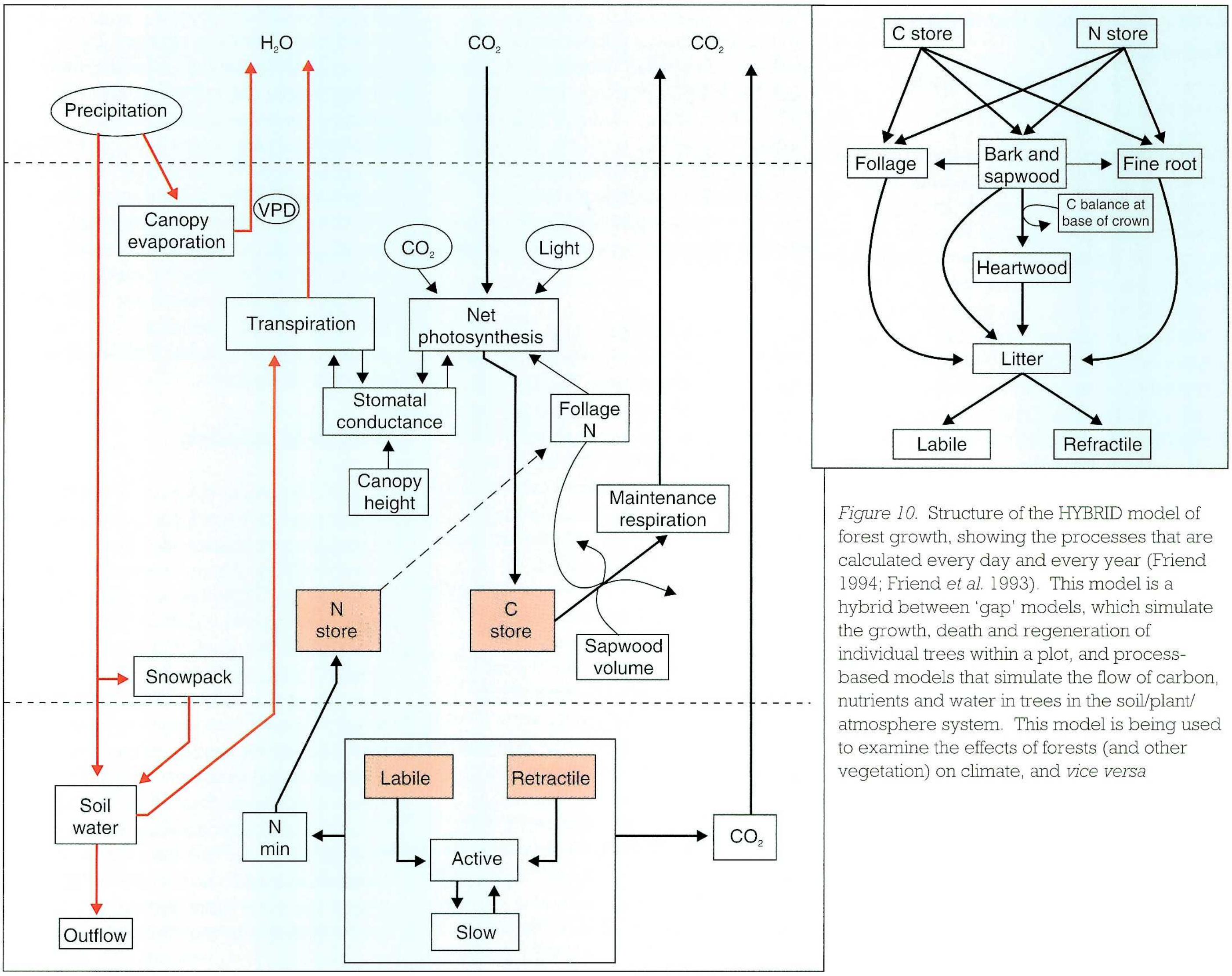


Figure 10. Structure of the HYBRID model of forest growth, showing the processes that are calculated every day and every year (Friend 1994; Friend *et al.* 1993). This model is a hybrid between 'gap' models, which simulate the growth, death and regeneration of individual trees within a plot, and process-based models that simulate the flow of carbon, nutrients and water in trees in the soil/plant/ atmosphere system. This model is being used to examine the effects of forests (and other vegetation) on climate, and *vice versa*

when ammonium is added to mature spruce forest, it is retained by the site (the trees and/or soils), whereas when nitrate is added much of it is released into the groundwater. That is, mature stands are nitrate-saturated but not ammonium-saturated. However, for many mature stands, the nitrate saturation may be due to P and K limitation, because additional N is taken up when P and K are added.

Work on tree health in the UK has shown that mature forests are visibly less damaged than seedlings by sulphur pollutants, because mature foliage does not take up so much sulphur. However, sulphur deposition does cause 'hidden' injury, and a map of the UK has been produced showing where 'critical levels' of sulphur deposition may occur on forests.

It may be noted that, during the year, two international organisations were formed to promote forestry research, namely the Centre for International Forestry Research (CIFOR) in Indonesia, and the European Forest Institute (EFI) in Finland. ITE has been involved in discussions with both of these organisations, and Dr Roger Leakey was seconded from ITE Edinburgh to be the Director of Research at the International Centre for Research on Agroforestry (ICRAF) in Nairobi, Kenya.

M G R Cannell

References

Botkin, D.B., Janak, J.F. & Wallis, J.R. 1972. Some ecological consequences of a computer model of forest growth. *Journal of Ecology*, **60**, 849–872.

Friend, A.D. 1994. PGEN: an integrated model of leaf photosynthesis, transpiration and conductance. *Ecological Modelling*. In press.

Friend, A.D. & Cox, P. 1994. Modelling the effects of atmospheric CO_2 on vegetation–atmospheric interactions. *Agricultural and Forest Meteorology*. In press.

Friend, A.D., Shugart, H.H. & Running, S.W. 1993. A physiology-based gap model of forest dynamics. *Ecology*, **74**, 792–797.

Running, S.W. & Coughlan, J.C. 1988. A general model of forest ecosystem processes for regional applications. 1. Hydrological balance, canopy gas exchange and primary production processes. *Ecological Modelling*, **42**, 125–154.

Deep beneath the trees in Senegal

(This work was funded by the Overseas Development Administration)

Population pressure in semi-arid areas has led to the abandonment of traditional land use practices, such as land fallowing, in favour of continuous agriculture. As successive crops are removed, the nutrient status of the land declines, with a concomitant reduction in crop yields. To maintain agricultural output, farming has expanded on to poorer soils originally covered by dry forest or savannah. Consequently, many arid zone soils have become badly degraded and are now characterised by poor nutrient status and meagre organic matter content, especially where all above-ground production is removed. Individual *Faidherbia albida* trees are frequently retained in fields and are also permitted to establish and grow in the belief that, because they fix nitrogen and are able to recycle deeply located nutrients to the surface via their deep roots, they can improve fertility in the upper layers of soil which crop roots exploit.

Unexpectedly, Edmunds (1990) found that deep interstitial and well water which had percolated through a degraded agricultural soil in the western Sahel in northern Senegal contained large amounts of NO_3–N in solution (20–50 mg l^{-1}). (The European limit in drinking water is 11 mg l^{-1}.) In this sandy soil, the largest concentrations of nitrate were found in a layer occurring between 10 m and 20 m beneath the soil surface (15–25 m above the water table). The nitrate nitrogen in solution in this part of the profile alone is equivalent to about 360 kg ha^{-1}. While this amount could pose a potential health risk if it entered water supplies, it also represents a considerable resource in an area where low farm incomes render the application of artificial fertilizers prohibitively expensive.

The existence of this layer of nitrate raises several questions.

- Where did it come from?
- Can it be brought to the surface by deep-rooted trees to help improve agricultural output?
- Is it moving down the profile and thus likely to pose health risks when it enters the water table?

ITE has begun to address these questions, in collaboration with the British Geological Survey; Department of Geology, University of Dakar; Department of Biological Sciences, University of Dundee; and the Soil Microbiology Laboratory of ORSTOM in Dakar. A small collaborative project is centred on the city of Louga (15°37'N, 16°13'W) in northern Senegal. Two short field campaigns timed to coincide with the end of the annual rains (October/November) have been made to date.

Because nitrogen inputs from the atmosphere in West Africa are very low, atmospheric deposition was immediately rejected as the potential source of nitrate. Similarly, from the recent history of the area, it seems unlikely that large-scale forest clearance could have provided a single substantial injection of nitrogen which had entered the soil over several years through decomposition. Nevertheless, references on the history of the vegetation of Senegal are being sought to check this supposition. Edmunds (1991) felt that biological nitrogen fixation was the most likely source of nitrate, and that it may subsequently have been concentrated by the evaporation of water in the arid conditions. To test this hypothesis, a number of holes were augered to the water table at two sites: (i) At Barale, 30 km north of Louga, where the water table was about 15 m deep; and (ii) at Ndiery, 8 km south of Louga, where the water table was about 32 m below the surface.

At each site, holes were augered under nitrogen-fixing trees about 0.5–0.75 m from stems, and, for comparison, in open fields at least 50 m from the nearest tree. Distributions of roots, nitrogen-fixing bacteria, mycorrhizas, organic matter, nitrate in solution, and other nutrient elements were examined at depth intervals varying between 0.5 m and 5 m, as appropriate. An additional hole was augered at Ndiery under a neem tree (*Azadirachta indica*), which is a fast-growing multipurpose tree species widely utilised within arid areas, not only to assess its rooting depth and hence its potential to 'mine' the nitrate, but also to examine nitrate amounts below a non-nitrogen-fixing tree.

Nitrate in solution

NO_3–N amounts in solution beneath *Acacia senegal* were largest, up to 180 mg l^{-1}, in surface horizons and also in a layer extending from about 15 m to 22 m depth (Figure 11i). While the existence of this layer of enhanced nitrate at depth confirms earlier work by Edmunds (1991), the concentrations found beneath this tree and the others examined to date greatly exceed those reported previously by Edmunds for treeless areas. Amounts of nitrate found in soil profiles at Barale were even bigger than those found at Ndiery, and concentrations outwith the trees were also larger than expected. Beneath a large *F. albida* tree at Barale, where the water table was only 15 m below ground, nitrate concentrations of up to 230 mg l^{-1} were found in a layer extending from 4 m to 10 m in depth. In the adjacent open field, concentrations up to 145 mg l^{-1} occurred in a layer extending from 3 m to 6 m deep. It is

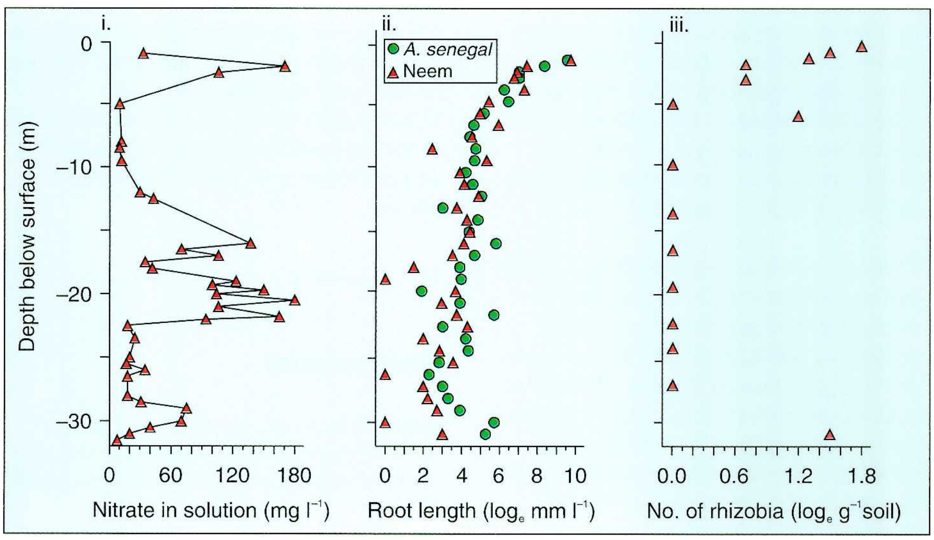


Figure 11. Distributions of (i) nitrate in solution beneath an *Acacia senegal* tree, (ii) fine root length of *A. senegal* and neem, and (iii) nitrogen-fixing rhizobia, between the soil surface and the water table at Ndiery in Senegal

possible that the large concentrations of nitrate found in the open at Barale come from peanuts, which also fix nitrogen and are frequently grown on the land. Such results tend to implicate nitrogen-fixing plants, and trees in particular, for increasing the amounts of nitrate in soils, but this will only be confirmed by examining further profiles taken from treeless non-agricultural ground and from beneath non-nitrogen-fixing trees.

Fine root distribution

Fine root concentrations of nine-year-old *A. senegal* and neem declined with increasing depth at Ndiery (Figure 11ii). However, there was a distinct increase in root amounts close to the water table. While this pattern was most pronounced for *A. senegal*, which was spatially separated from the neem by about 300 m, the water table beneath the neem was slightly deeper and overlain by a layer of dense sand which could not be penetrated with hand augering equipment. Thus, the trend of increasing fine root amount as the water table was approached may have been underestimated for neem. The increases in root amounts for *A. senegal* at about 17 m and 22 m coincided with the presence of live structural roots (2–4 mm diameter) at these depths. That *F. albida* growing at Barale also had roots which penetrated to the water table was not surprising. This species has a reversed phenology, carrying leaves in the dry season and being leafless during the rains. Accordingly, it must have access to deep water supplies to enable photosynthesis to take place during the dry season when surface soils lack physiologically available water. Clearly, both *A. senegal* and neem are also deep rooted enough to be able to access nutrients at depths below that from which crop plants can extract nutrients.

Soil microbiology

In the soil profile beneath the *A. senegal*, nitrogen-fixing species of *Rhizobium* capable of forming fully functioning nodules with the tree had a distribution which mirrored that of the fine roots, ie they were common close to the surface, infrequent from 7 m to 30 m, and their frequency increased close to the water table (Figure 11iii). This result is similar to that found in a laboratory study by Felker and Clark (1982), who utilised artificial deep soil profiles which were watered from below. They, too, found increased rhizobial activity in the wet conditions. While distributions of rhizobia found in profiles taken from the open field were similar to those found under N-fixing trees, bacterial frequency was about two orders of magnitude smaller than that found in profiles beneath trees.

In contrast to the rhizobia, evidence of vesicular-arbuscular mycorrhizal activity was greatest in soil horizons close to the surface. For *A. senegal* and neem, mycorrhizal occurrence was most frequent at depths less than 5 m, where 45–75% of fine roots were colonised. At greater depths, evidence of fungal activity was rare. There was no evidence of mycorrhizal colonisation below 22 m. Nevertheless, it seems possible that mycorrhizas which extend the nutrient

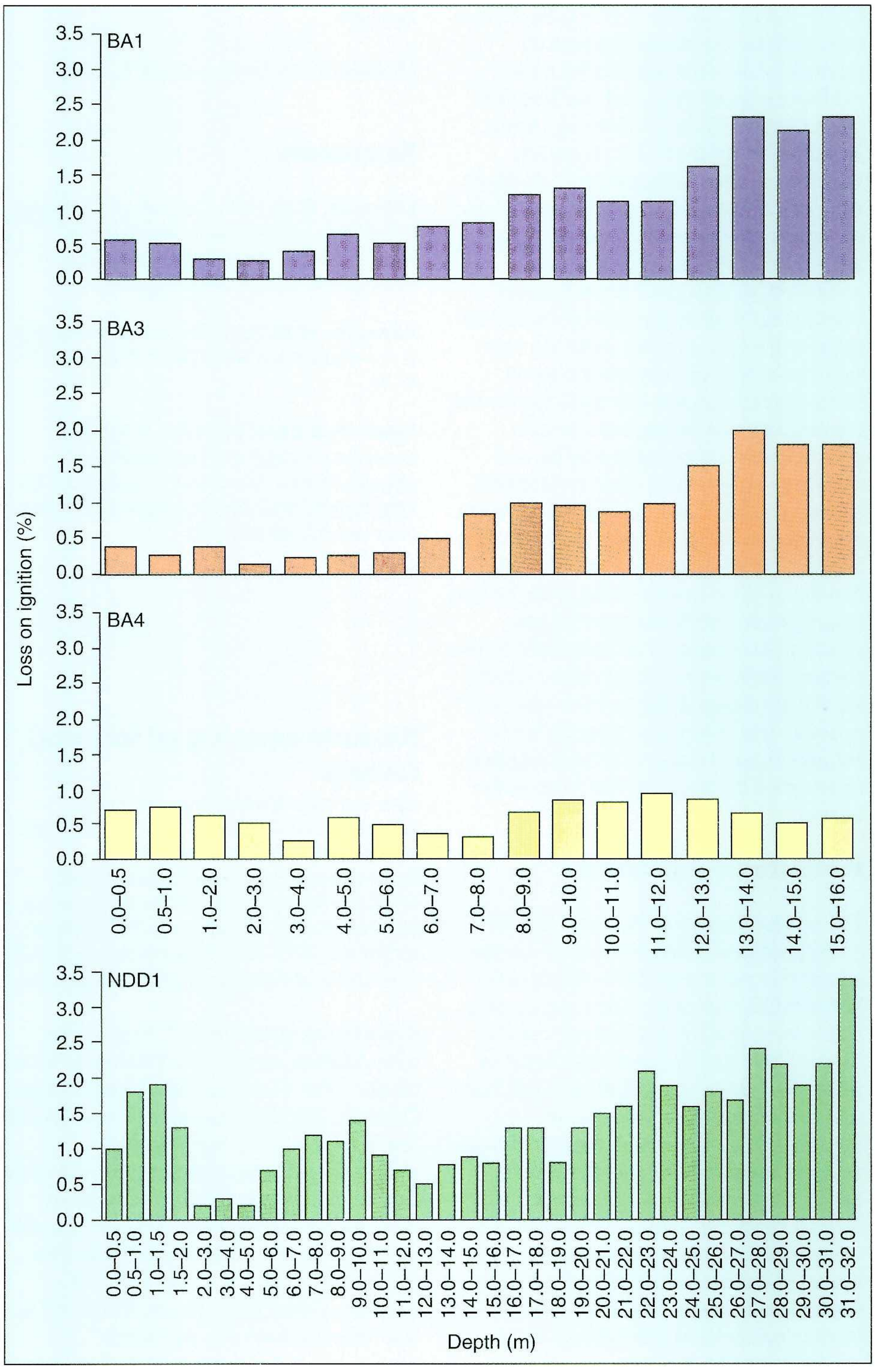


Figure 12. Loss on ignition (% dry weight) between the soil surface and the water table for soil samples taken either from an open field or beneath trees of *Faidherbia albida* at Barale, and beneath *Acacia senegal* at Ndiery in Senegal

and water uptake interface between the plant and soil could have a role to play in recycling mineral nutrients from intermediate depths

Soil organic matter

Because the major organic matter input to soil is from above- and below-ground plant litter, amounts of soil organic matter normally decline with increasing depth below the soil surface Soil organic matter distributions assessed from loss on ignition (LOI) are illustrated for three profiles beneath trees and one from the open field in Figure 12 Although there was a tendency for LOI to decrease between surface horizons and 4 m depth, without exception, largest concentrations of organic matter were found close to water tables As amounts of organic matter were two to four times greater under trees than in the open field, and the pattern of organic matter increase was more pronounced beneath the trees, these results suggest a strong influence of the trees on soil properties During sampling and processing of soils, and contrary to the results described above, large amounts of organic matter were not observed in samples taken close to water tables In consequence, it seems likely that the organic matter found at depth is in soluble form, rather than particulate material At present, it is impossible to be certain about why organic matter should be accumulating at depth However, it is possible that downward percolation of soluble organic material is being slowed or arrested by capillary rise from water tables

Preliminary conclusions

The consequences of the large NO_3–N concentrations observed may be serious for drinking water quality in the future Water infiltration rates in this region near Louga average about 12 cm yr^{-1}, and it seems likely that the interstitial water in which the nitrate is concentrated will also move downwards at this rate In consequence, concentrations of nitrate an order of magnitude above the World Health Organization recommended safe limit for drinking water could eventually enter water supplies, thus questioning the advisability of using nitrogen-fixing trees in such areas However, further work is required to substantiate these preliminary observations

In summary, it appears that nitrogen fixation is the most likely source of the nitrate in the soil profile and that non-nitrogen-fixing trees can root deeply enough to access it While it would seem sensible to utilise non-nitrogen-fixing trees which use large amounts of water (eg *Eucalyptus*) to bring the nitrate to the surface quickly, the matter requires careful consideration Water itself is a scarce resource in the semi-arid zone, and global circulation models are predicting even less rainfall for the Sahel in future

J D Deans, D K Lindley and R C Munro

References

Edmunds, W M 1990 *Groundwater recharge in Senegal* (NERC contract report to the Overseas Development Administration) Keyworth British Geological Survey

Edmunds, W M 1991 Groundwater recharge in the west African Sahel *NERC News*, no 17, 8–10

Felker, P & Clark, P R 1982 Position of mesquite (*Prosopis* spp) nodulation and nitrogen fixation (acetylene reduction) in 3 m long phraetophytically simulated soil columns *Plant and Soil*, **64**, 297–305

Remote sensing of tropical forests

(This work was funded under the British National Space Centre's ATSR-2 Special Topic, in joint collaboration between ITE's Environmental Information Centre (Dr B K Wyatt and Ms F Gerard) and the University of Leicester (Prof A Millington and Dr J Wellens), and by the NERC Terrestrial Initiative in Global Environmental Research (TIGER) programme)

Humid tropical forest (HTF) comprises over 3 billion hectares and occurs in three blocks – the Americas, Africa and SE Asia/ Oceania The importance of tropical forest lies in its role in the major global processes relating to carbon and water, in its rich biodiversity, as a source of raw material for many human populations, and as a constraint to land degradation The destruction of HTF is a major global concern Unlike other forest, when HTF is clearfelled it does not, on human timescales, regenerate as HTF According to the United Nations Environment Programme, the rate of annual loss during the 1980s was estimated at 15–20 million ha However, global and national figures do not reflect variations within countries which are a function of different tropical forest types and socio-economic factors, and there is no reliable, comprehensive, global database of changes in vegetative cover Earth observation by satellite remote sensing is a source of global data, with significant potential for the mapping and monitoring of tropical forest Earth observation provides the key to the extension of detailed local observations made by vital ground survey by scaling up to the regional or global context Within this overall framework, staff in the Remote Sensing Group of ITE's Environmental Information Centre at Monks Wood are actively involved in developing and applying satellite remote sensing, both multi-spectral scanning and radar, for the mapping of humid tropical forests

Multi-spectral scanning

In preparation for the late-1994 launch of the ERS-2 satellite with its British-manufactured Along Track Scanning Radiometer (ATSR-2), NERC is supporting a project to assess this new sensor's potential for identifying different HTF types on a continental scale The major goal is to examine how differences in (i) canopy roughness, (ii) gaps in the canopy, and (iii) phenology of different forest types are expressed in the ATSR-2 imagery The eventual result of this effort will be the development of new procedures and techniques for mapping HTF types The key characteristics of ATSR-2 which determine its utility for this particular application are

- the spatial resolution of 1 km, which has proven to be highly suited to vegetation monitoring over large areas,
- the improved spectral resolution of 20 nm, which implies greater sensitivity to differences in canopy composition and leaf colour,
- the simultaneous nadir and off-nadir (along-track) viewing, which allows improved correction of atmospheric haze effects and detection of forest structure through shadow texture, and
- the radiometric range in the three wavebands at visible and near-infrared wavelengths (from 545 nm to 875 nm), which considerably improves the potential for species differentiation and for detecting phenological change over comparable satellite systems

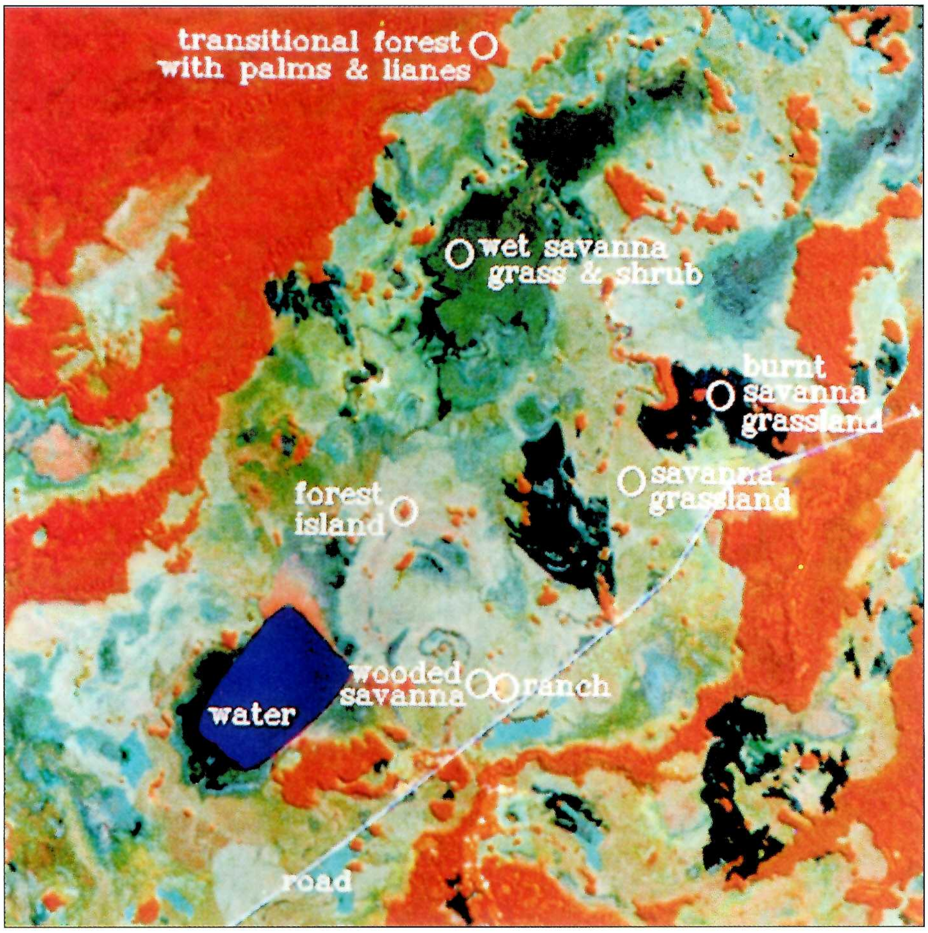


Plate 11. 12 km × 12 km subarea of a multi-spectral July 1989 Landsat Thematic Mapper scene, showing a part of the EBB reserve in Bolivia, with a range of tropical forest and other vegetation types

This study has adopted a 'bottom-up' approach, where detailed ground survey data sets are extrapolated to ATSR-2 images via intermediate medium-scale images (Landsat Thematic Mapper). Initially, NOAA Advanced Very High Resolution Radiometer (AVHRR) images will be used and ATSR-2 images will be simulated from Thematic Mapper data in order to identify possible problems (eg the degree of heterogeneity within the spatial resolution units of the data), and to allow for an early-stage solution.

Three field trips to Bolivia have been planned to procure detailed information about forest structure, canopy roughness and gaps for a range of forest types. The first field trip was successfully carried out during late-1993. The surveyed sites, ten in total, were located in the Estacion Biologica del Beni (EBB), a reserve in the Beni lowlands, and cover a range of savanna and forested areas (Plate 11). Survey methods were those of standard forest inventory (tree species, diameter at breast height, tree height, crown depth, crown area), hemispherical photography and measurement of the leaf area index (Plates 12 & 13). From these data sets it is possible to estimate the gap size distribution in the canopy and canopy density. With repeated surveys during different periods of the year, seasonality in the forest canopy will also be detected.

In addition to the site surveys, time was spent in reinforcing contacts previously made by Prof A Millington during an exploratory visit. Collaboration with several research groups (ABTEMA – Bolivian Remote Sensing Consortium, EBB and the US Missouri Botanical Gardens) was established for the future field work.

The surveyed forest types were mixtures of palm species, broadleaf species and lianas. The canopy density ranged from 1.8 to 2.6; large trees were rare. Although the forest types appeared very similar in structure, preliminary statistical analysis has shown subtle differences. The next step is to assess the gap size distribution of the canopies by:

- analysing hemispherical photographs, and
- three-dimensional simulations of the canopies to examine the gap size/ canopy structure relationship.

Radar imaging

The Remote Sensing Group is also involved in a number of research projects for developing radar remote sensing for general forest monitoring, with particular emphasis upon the HTF biome. This work is being undertaken in close collaboration with the NERC Remote Sensing Applications Development Unit.

Radar remote sensing is based upon the same principle as the navigation radar used in ships and aircraft (ie by the transmission and reception of pulses of microwave radiation), with measurement of the strength and time-delay of the returning pulses. Forward motion of the radar platform enables a two-dimensional image of the microwave response of the earth's surface to be created (Groom 1991). Radar remote sensing has considerable potential for earth observation by virtue of (i) its greater independence of cloud cover, compared with optical remote sensing (ie visible and infrared), and (ii) the sensitivity of the microwave response to moisture content and geometric characteristics of the imaged surface. Furthermore, the ability

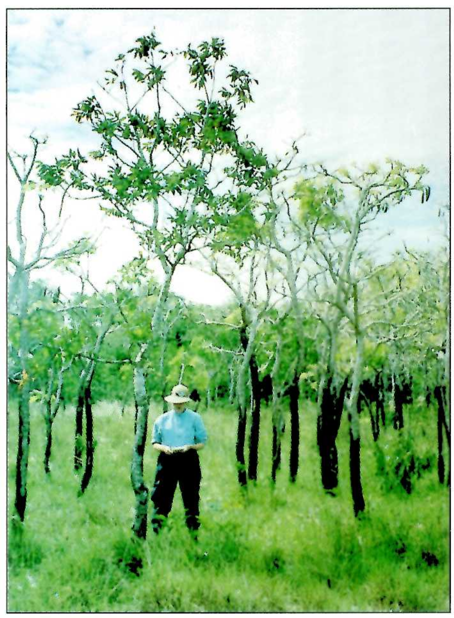


Plate 12. Project scientist making biometric measurements for forest inventory within an area of wooded savanna in the EBB reserve, Bolivia

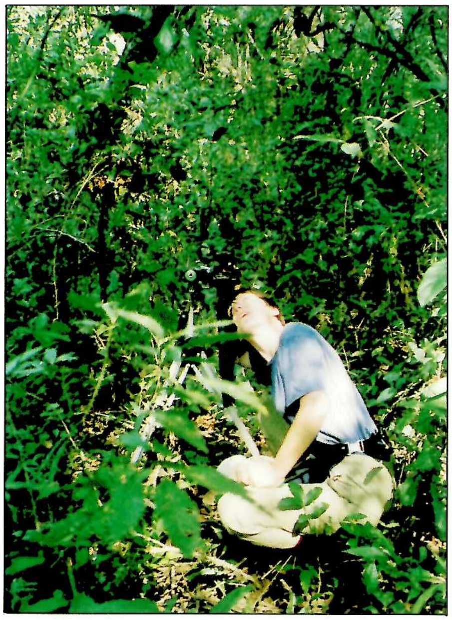


Plate 13. ITE scientist undertaking hemispherical photography of the forest canopy at a transitional forest (palm and liane) field site in the EBB reserve, Bolivia

of microwave radiation to probe beyond the outermost leaf layer of vegetation canopies, and thereby to provide information upon internal canopy structure, gives it major significance for the study of forests, with the potential for direct estimation of the woody biomass (Baker *et al.* 1994). However, routine radar remote sensing from satellites is relatively recent (1992), with limited image data sets. Fundamental differences between radar data and optical data demand the development and use of particular analytical techniques.

A major aim of the radar programme at ITE Monks Wood is to support research within the NERC Terrestrial Initiative in Global Environmental Research (TIGER) programme in estimating the carbon flux in tropical forests on a continental scale. This project sees the integration of radar remote sensing with ground-based measurements of the carbon flux, micro-meteorology, vegetation characteristics, and optical remote sensing. The radar remote sensing component has focused upon the estimation of biomass in uncut forest, cut areas, and regenerative forest stages, with particular attention upon the latter, where very rapid regrowth by trees such as *Cecropia* spp. and *Vismia* spp. is associated with high rates of carbon fixation. Several test areas, in the Brazilian Amazon and West Africa, have been identified and sets of radar image data, representing significant radar system parameters (wavelength, polarisation, spatial resolution, and time series), have been acquired.

Maps of forest age derived from cloud-free Landsat (optical) imagery (Lucas *et al.* 1993) have provided a basis for initial analysis of the Rio Tapajós (Brazil) radar data. Textural information within fine spatial resolution (6 m pixel) airborne radar data discriminates between uncut primary forest, a regrowth category and cleared areas (Plate 14), even where only a single set of short wavelength (5 cm) microwave radiation is available (Groom *et al.* 1994). However, the very limited canopy penetration at these frequencies does not permit adequate discrimination between more mature regrowth stages, and in coarser spatial resolution (12 m) satellite data many of the textural distinctions are lost. Examination of multi-temporal sets of satellite data of longer wavelengths (eg 20 cm), with greater penetration of the forest leaf canopy, has demonstrated the potential of such data to determine the multi-season microwave signatures of the various forest types, and to quantify the progressive changes in the forest cover.

Multi-wavelength, fully polarimetric radar, and ground data sets of temperate forest 'test sites' have provided the foundation for model-based development of the estimation of tropical forest biomass, relating increases in the radar wavelength to increases in the range of biomass densities that can be measured (Baker *et al.* 1994). These studies indicate that, with radar wavelengths of 70 cm, the relationship between microwave backscatter and biomass saturates at a level high enough to provide a basis for mapping and biomass estimation of a significant range of tropical forest regrowth stages. This summer, the Remote Sensing Group will participate in the field survey and forest mensuration

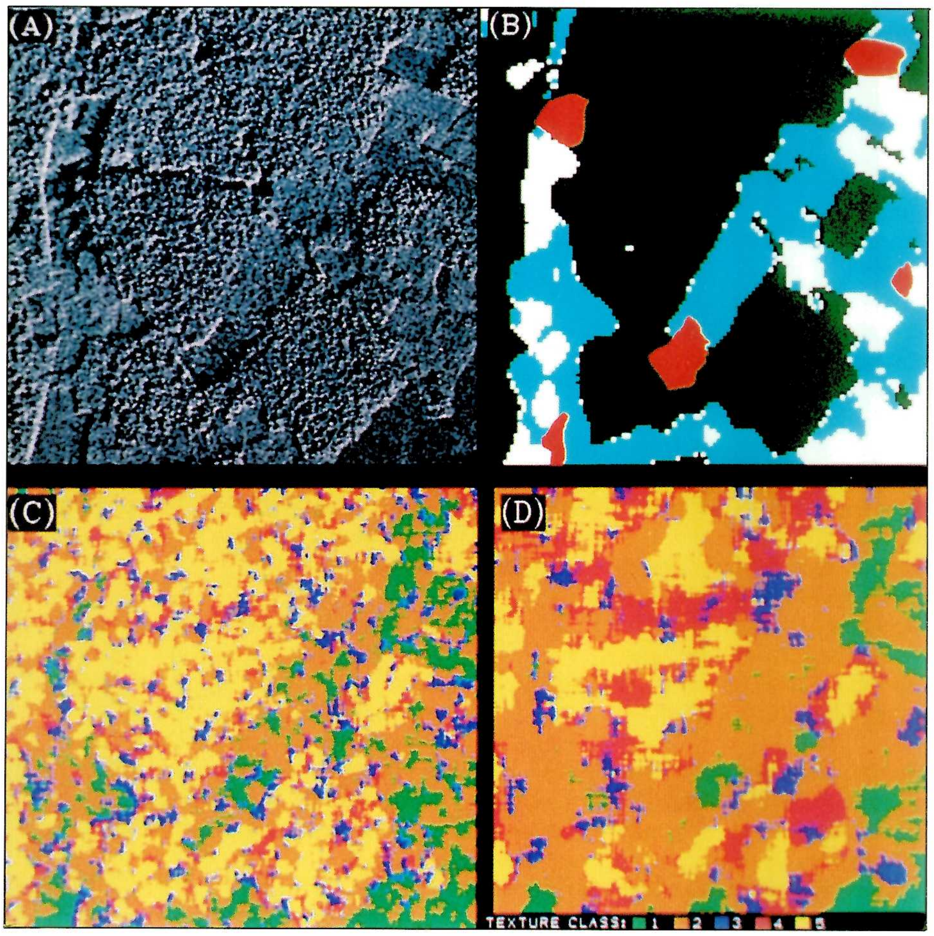


Plate 14. A 3 km × 3 km subarea of the Rio Tapajós radar test area, Brazil. (A) shows the 5 cm wavelength, 6 m spatial resolution airborne radar image, with both tonal and textural differences between forest blocks. In (B) image tone patterns in Landsat Thematic Mapper and the radar data have been used to construct a reference map of areas of uncut forest (dark green), forest regrowth (light green) and clearcut vegetation (red). In (C) and (D), a texture expression algorithm, with differing spatial parameters, has been applied to the radar data in (A), giving correspondence of five nominal texture classes to the reference map

campaign of the Brazilian Space Research Agency at the Rio Tapajos test area These ground data sets will provide a basis for the extension of these models to tropical forests NASA Space Shuttle missions during mid-1994 will see the acquisition of the first multi-wavelength (5 cm, 20 cm, and 70 cm), fully polarimetric radar data from space, areas being imaged by these limited duration missions include the Tapajos and other Amazon test sites These data, representative of satellite radars planned for the late 1990s, should significantly advance our ability to map tropical forest biomass from space

G Groom and F Gerard

References

Baker, J R , Mitchell, P L , Cordey, R A , Groom, G B , Settle, J J & Stileman, M R 1994 Relationships between physical characteristics and polarimetric radar backscatter for Corsican pine stands in Thetford Forest, UK *International Journal of Remote Sensing* In press

Groom, G B 1991 Future applications of radar remote sensing in terrestrial ecology *Annual Report of the Institute of Terrestrial Ecology 1990–91*, 63–66

Groom, G B , Luckman, A J , Baker, J R & Cordey, R A 1994 SAR image texture and spatial resolution for tropical forest mapping In *South American Radar Experiment SAREX-92 Workshop Proceedings, 1993,* 95–100 (ESA WPP-76) Paris European Space Agency

Lucas, R M , Honzak, M , Foody, G M , Curran, P J & Corves, C 1993 Characterising tropical secondary forests using multi-temporal Landsat sensor imagery *International Journal of Remote Sensing*, **14**, 3061–3067

Land use, agriculture and the environment

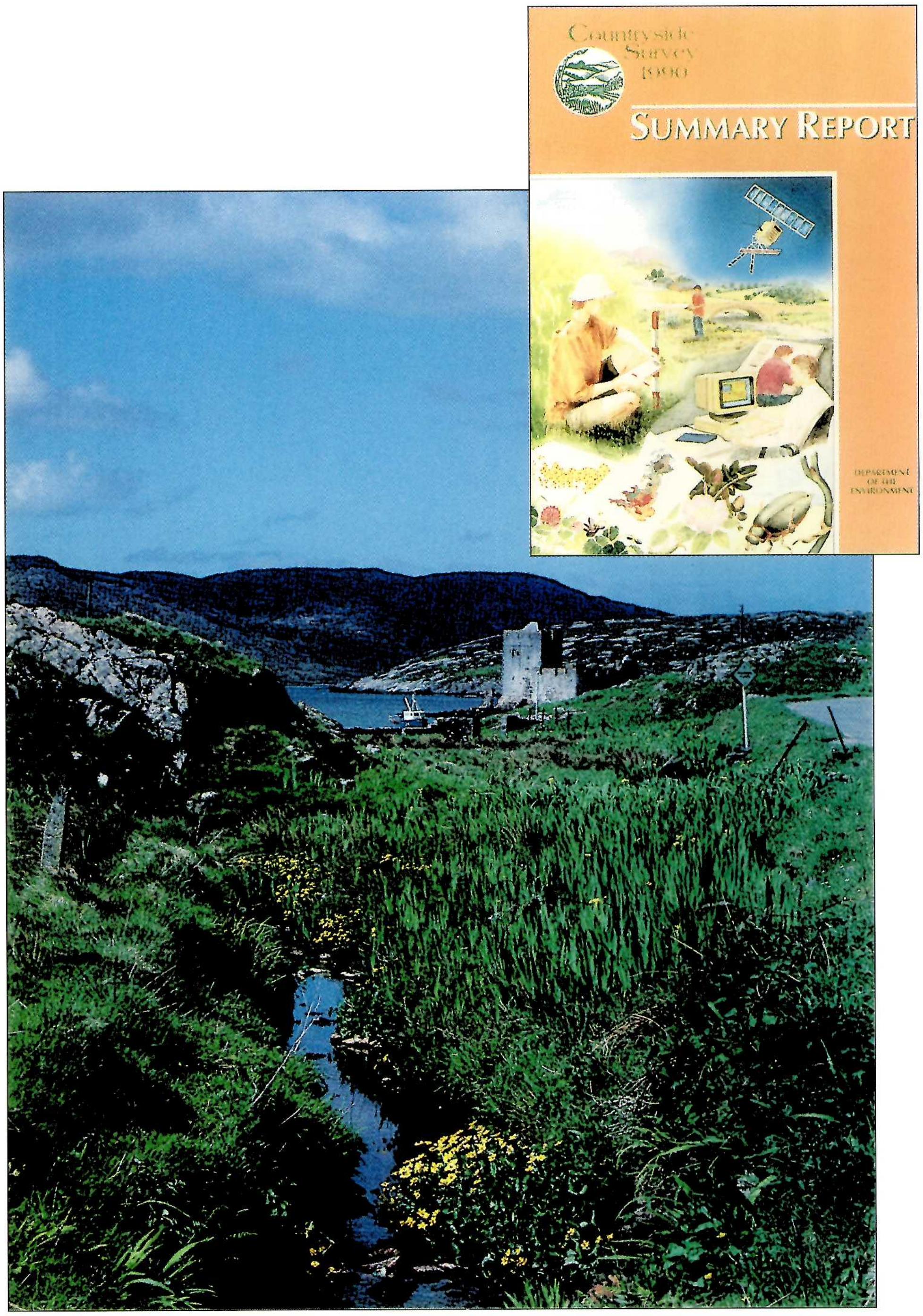


Plate 15. Species number decreased along stream banks

Reforms in the Common Agricultural Policy of the European Commission, aimed at controlling costs and moving towards more sustainable farming systems which take greater account of environmental policy objectives. It is, therefore, appropriate that monitoring changes in land use and determining their effects on the environment continue to be essential core activities in ITE. The value of the outputs from such research in support of policy-making is indicated by comments made by the Secretary of State for the Environment, John Gummer, at the publication in October 1993 of the reports of the Countryside Survey 1990. In his speech, the Minister stressed the value of the survey as an essential baseline against which future change could be assessed and the success of countryside policies determined. He pledged Government support for a further survey in the year 2000 to build on those done in 1978, 1984 and 1990.

Summarising the results of a survey which involved visits to 508 1 km squares throughout Britain inevitably involves oversimplification and, even in the report which follows, it is not possible to give more than a flavour of the findings. In the space available here it is only possible to note a few of the major conclusions. Thus, while the proportion of the main semi-natural vegetation types in the British countryside has remained constant through the late 1980s, the quality of that vegetation, especially as measured by plant species diversity, has declined. Losses of species and of habitats (notably those associated with linear features such as hedges, verges and watersides), have generally been greater in the lowlands than the uplands (Plates 15 & 16). Similarly, the quality of freshwater habitats, also measured by species diversity, was lower in the lowlands. It is the aim of ITE to make the information from this and other surveys available in easily usable form so that it may be widely disseminated. With this in mind, a Countryside Information System has been devised for use on personal computers, which will contain information on the land classes used to provide the sampling framework, the data contained in the main report, plus further data at the land class level. Due to be launched in autumn 1994, the System will allow the user to integrate, combine or overlay the constituent data sets and to output derived results in tabular or graphical formats. Further developments arising from the Countryside Survey 1990 include research to reveal the factors (including socio-economic constraints) determining the susceptibility of particular types of land to change, also the ease with which change can be reversed, with the ultimate objective of being able to predict more accurately the effects of future policy changes.

While loss of habitats is of obvious concern to those concerned with the maintenance of diversity in the countryside, the breaking up of remaining habitats also demands the attention of ecologists as fragmentation is one of the major causes of a worldwide and accelerating reduction in biodiversity. This is a serious concern for those involved in setting conservation policy in lowland Britain, where loss of semi-natural habitat has often been followed by fragmentation of what remains. This is perhaps particularly noticeable in the case of woodland; few remaining lowland woods exceed 100 ha and most are less

Plate 16. Yellow flag iris (*Iris pseudacorus*), a species of wet places

than 10 ha in extent. In a study of the effects of such fragmentation on breeding birds in Cambridgeshire, little evidence was found for strict minimum area requirements, although some species, mainly 'woodland specialists', nested in the smaller woods only infrequently and then in small numbers, making them liable to local extinction. In order to ensure that such extinctions are only temporary, woodland planting at the landscape scale should aim for a mix of small and large woods, linked where possible by hedgerows and shelterbelts to provide cover for birds travelling between the woods.

Also described is a related study in which the relationship between the nature of field boundaries and birds was observed in an area of East Anglia where such linear features are scarce. Clear differences were found between the types of field boundaries separating arable as compared with pasture crops, which were reflected in differences in bird species richness. The greater bird species diversity in pasture field hedgerows was partly due to their more complex structure, involving greater diversity of woody plants and higher frequency of hedgerow trees. The project has enabled an objective assessment of the habitat requirements of birds in farmland which, in turn, has facilitated improved advice to farmers and wildlife conservationists on field boundary management to aid bird conservation.

Like small woods and hedgerows, unimproved lowland pastures have declined markedly in lowland Britain this century. Agricultural improvement led to the conversion of much of this unimproved pasture to arable, most of the remaining land being either ploughed up and reseeded or, less drastic, improved by fertilizer application. In either case, nitrogen was added as it was known to stimulate the growth of coarse grasses, which provide the chief feed value of such pastures for domestic livestock. While it was assumed that this increase in grass biomass was at the expense of reduced diversity of less vigorous plant species, little experimental work had been done to test this hypothesis, at least in wet grasslands. This situation has been rectified with a long-term study at Tadham Moor in Somerset, where exceptionally rich meadows occurred on low-lying peaty soils. The report on the study describes the effects of nitrogen fertilizer addition and the 'reversion' of the vegetation following cessation of fertilization. Species richness decreased both with increasing nitrogen addition and with time. After seven years of nitrogen addition, species richness was significantly reduced even in the plots receiving least nitrogen (25 kg ha^{-1} yr^{-1}). When nitrogen addition ceased, there was a gradual increase in species richness but the rate of recovery varied depending on the plant community type and the amount of nitrogen added, being slower in the more heavily fertilized plots. As reversion takes several years, it was felt that it would be useful to develop a model for predicting the effects of rate of nitrogen addition on reversion time. Such a model has been derived, based on a range of vegetation variables, which allows the reversion times for different plant communities to be estimated following a range of previous nitrogen fertilization regimes. Preliminary plot results suggest that the prediction of two years for reversion in the 25 kg ha^{-1} yr^{-1} experimental treatment in the Tadham trial is accurate. However, there is need for caution in assuming that the estimates for the higher levels of nitrogen addition will be similarly precise, because the effect of heavy nitrogen treatment in eliminating rather than reducing some species introduces a greater level of unpredictability in estimating the rate of their reinvasion.

One of the practical advantages of gaining a good ecological understanding of the way ecosystems function is that it is easier to provide advice on their reinstatement following damage. This is by no means a one-way process, however, as there is no better way to try and understand how a complex natural system works than to try and 'repair' it when damaged. Thus, restoration ecology is an intellectually satisfying and rewarding science, as well as providing a sound basis for reinstatement of damaged ecosystems. ITE has been active in this field since its formation, and continues fundamental research underpinning restoration as well as giving advice to engineers, developers and planners on the most appropriate solutions to problems associated with particular reinstatement schemes. In doing so, it maintains and develops links with industry of the type which are increasingly required if applied research in Research Council institutes is to make the contribution to industrial advancement which the Government demanded in its 1993 White Paper *Realising our potential: a strategy for science, engineering and technology*. The first article on this theme gives an overview of the type of work that has been done at ITE Banchory, near Aberdeen. Outlines are given of reinstatement schemes on trunk roads, at high elevation around ski areas and footpaths, and along long-distance pipelines. The importance of quality control, both in devising reinstatement prescriptions by ITE scientists in co-operation with those responsible for the work, and in carrying out those prescriptions, is emphasised. In the second article, an example is given in much more detail of one scheme involving installation of an oil pipeline through an area of high nature conservation interest, potential archaeological importance and landscape quality, all within a region where tourism is an important component of the local economy. That this scheme was successful owed no small part to the fact that the customer, BP Exploration, realised from the outset that, for the installation to be successfully completed, there would be a need for high-quality, impartial advice from a number of organisations with different fields of expertise. Thus, ITE was involved in assessing the ecological implications of all proposals made and worked closely with the other experts and the customer throughout. For the customer, this allowed appropriate decisions to be taken which allowed the development to take place in the most ecologically sensitive manner. For ITE, the chance was presented for accumulating data relevant to a number of current studies of ecosystem dynamics and processes.

J E G Good

Countryside Survey 1990

(This work was funded by the Department of the Environment, the Department of Trade and Industry, the former Nature Conservancy Council and NERC)

Background

Change has always been a feature of the British countryside. As the economic pressures on land continue to change and the value placed on our landscape and wildlife increases, policy decisions which may affect the rural environment need to be informed by knowledge of what is happening to the fields, woodlands, hedgerows, verges and streams that make up the fabric of the British countryside.

Countryside Survey 1990 set out to provide some of this information in the form of an overview of the countryside of Great Britain. Its main objectives were:

- to record the stock of countryside features in 1990, including information on land cover, landscape features, habitats and species;
- to determine change by reference to earlier surveys in 1978 and 1984;
- to provide a firm baseline, in the form of a database of countryside information, against which future changes could be assessed.

It was a survey of the countryside in its widest sense, which concentrated on the common features and habitats which are most likely to influence the public's perception of rural Britain. Unlike most other surveys of the countryside, it was not aimed at any particular sectoral interest, such as agriculture, forestry or nature conservation.

A general aim of Countryside Survey 1990 was to provide a common and reliable set of data as a contribution to a wider debate on the causes, consequences and directions of countryside change. In line with that aim, this report is factual. It summarises, but does not interpret or evaluate, the main results from the survey.

Countryside policy

Countryside Survey 1990's focus on the wider countryside came at a time when reform of the European Community's Common Agricultural Policy was presaging rapid changes in British agriculture, and when there was an increasing awareness of the value of the countryside as a national resource for recreation and wildlife. Although it was a survey aimed at common features, it was also understood that the conservation of rare species and habitats in protected reserves should not be set in isolation from the surrounding countryside.

The importance of the wider countryside was highlighted in the 1990 Environment White Paper *This common inheritance,* which spelled out the Government's plans for protecting the countryside. The White Paper also emphasised the Government's commitment to providing a statistical report on the state of the environment and to a process of decision-making in which policies are based on well-established facts. The publication of the report on Countryside Survey 1990 and the full survey results on which it is based form part of this process. The results will also contribute to the Biodiversity Action Plan to be produced by the Department of the Environment as part of the UK commitment to the International Biodiversity Convention agreed at the United Nations Conference on Environment and Development (the 'Earth Summit') in Rio de Janeiro in 1992. The Plan aims to turn international concern about the loss of biodiversity into practical action in the UK.

Surveying the countryside

There have been many other surveys which have looked at the British countryside, either in part or as a whole. Apart from being the most recent of these surveys, two things set Countryside Survey 1990 apart from the rest. First, its breadth of coverage and, second, its generality.

Countryside Survey 1990 is the first study to integrate satellite mapping with detailed field surveys of vegetation, soils and freshwater at a national scale. Satellite mapping was used to give an overview of land cover of the whole country, and field survey was used to describe a sample of the countryside, including freshwaters, in much more detail. In combination, these two approaches provided the best available data about the total resource of the countryside.

Landscape types

Each 1 km square of land in Great Britain has been allocated to one of 32 distinct land classes on the basis of environmental characteristics, such as geology, altitude and climate derived from maps. This land classification formed the sampling structure for Countryside Survey 1990 and provided the framework by which the sample survey of 508 1 km squares was used to make national and regional estimates. In this report, the 32 land classes have been grouped into four major landscape types: two in the lowlands – arable and pastural landscapes; and two in the uplands – marginal upland and upland landscapes.

Satellite mapping

The satellite mapping was based upon cloud-free images from the Landsat Thematic Mapper satellite between 1988 and 1991, but using 1990 images whenever possible. Summer and winter satellite data were combined to enhance the seasonal differences in the various cover types.

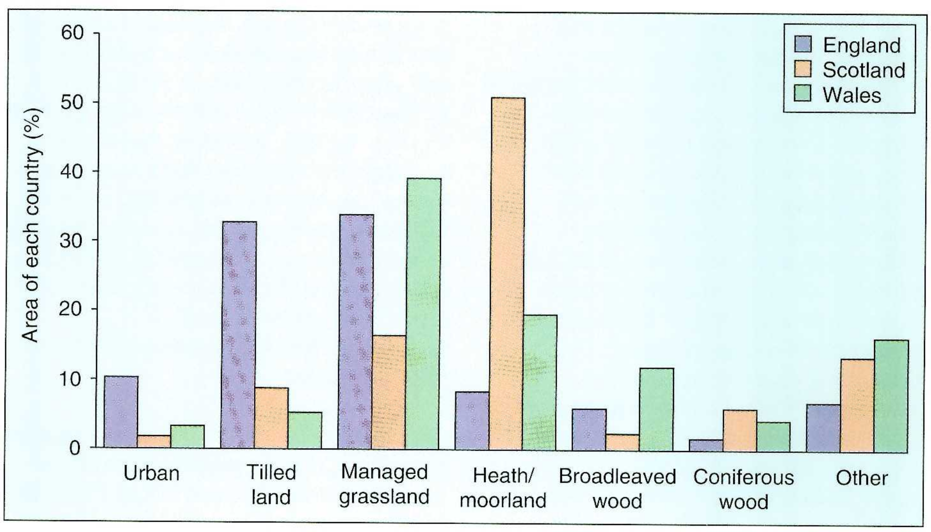


Figure 13. Land cover of England, Scotland, Wales and Great Britain from satellite

Field survey

A stratified random sample of 508 1 km squares was visited by survey teams. The squares were taken from all rural areas of Great Britain, excluding any square with over 75% urban land. Within each square, survey teams:

- mapped land cover on a field-by-field, or patch, basis;
- recorded landscape features, such as walls, hedges and individual trees;
- recorded plant species in random plots located in fields, woodlands and linear features (hedgerows, roadside verges and stream banks); and
- sampled freshwater animals (macro-invertebrates) in streams and watercourses.

Soil surveyors from the Soil Survey and Land Research Centre and the Macaulay Land Use Research Institute also mapped the soil types in each square.

The field survey repeated 256 1 km squares previously surveyed by ITE in 1977–78, and 384 squares surveyed in 1984. These earlier surveys form the basis for the assessment of countryside change given in this report. As the Land Cover Map of Great Britain and the freshwater component of the field survey were done for the first time in 1990, these could not be used to assess change, but form an important baseline against which future change can be assessed.

Land cover

Satellite-based mapping

The Land Cover Map (p90) shows the dominant land cover for each 25 m × 25 m area (pixel) of Great Britain. Land cover was classified into 17 key types, but the data can be examined for subdivisions of these types. The Map shows the predominantly agricultural nature of the British countryside, with 49% of the land being tilled or managed grass (Figure 13). Large areas of the country, particularly in upland and marginal upland areas, predominantly in Wales and Scotland, are covered by semi-natural land cover types. Thus, heath, moorland and bog make up over 50% of Scotland, 20% of Wales and 10% of England (Figure 13). In total, 8% of Great Britain is wooded – predominantly broadleaved and mixed woodland in England and Wales, but mostly coniferous woodland in Scotland.

COVER TYPE	AREA 1990 ('000 km^2)
TILLED LAND	
Wheat	22.3
Barley	11.5
Oats	0.9
Other cereals	0.3
Maize	0.4
Turnips/swedes	0.7
Kale	0.5
Oil-seed rape	4.1
Other crucifer crops	0.3
Peas	1.1
Field beans	1.0
Other legumes	0.0
Sugarbeet	2.2
Other root crops	0.1
Potatoes	1.4
Other field crops	1.0
Horticulture	0.4
Subtotal	**48.1**
MANAGED GRASS	
Recreational grass	2.5
Recently sown grass	7.1
Pure rye-grass	20.3
Well-managed grass	19.3
Weedy swards	9.9
Unimproved grassland	2.0
Calcareous grass	0.7
Upland grass	6.1
Maritime vegetation	0.3
Subtotal	**68.2**
ROUGH GRASS/MARSH	
Ploughed/fallow	3.5
Tall herb	2.7
Felled woodland	0.4
Wetland	3.7
Waste land	0.4
Dense bracken	3.7
Purple moor-grass	3.7
Other moorland grass	8.1
Dune	0.2
Subtotal	**26.4**
HEATH/BOG	
Wet heath and bog	16.6
Open-canopy heath	8.2
Berry-bush heath	1.2
Drier northern bogs	5.2
Dense heath	4.5
Subtotal	**35.7**
BROADLEAVED WOOD	
Perennial crops	0.7
Mixed woodland	2.2
Broadleaved woodland	9.2
Shrub	0.9
Subtotal	**13.0**
CONIFER WOODLAND	
Subtotal	**13.7**
BUILT UP	
Railway	0.4
Roads	4.4
Agriculture buildings	1.4
Residential buildings	6.8
Other buildings	3.0
Unsurveyed urban land	4.8
Subtotal	**20.8**
OTHER	
Inland water	2.9
Coastal unvegetated	1.8
Saltmarsh	0.4
Rocks/quarries	0.6
Subtotal	**5.7**
TOTAL	**231.8**

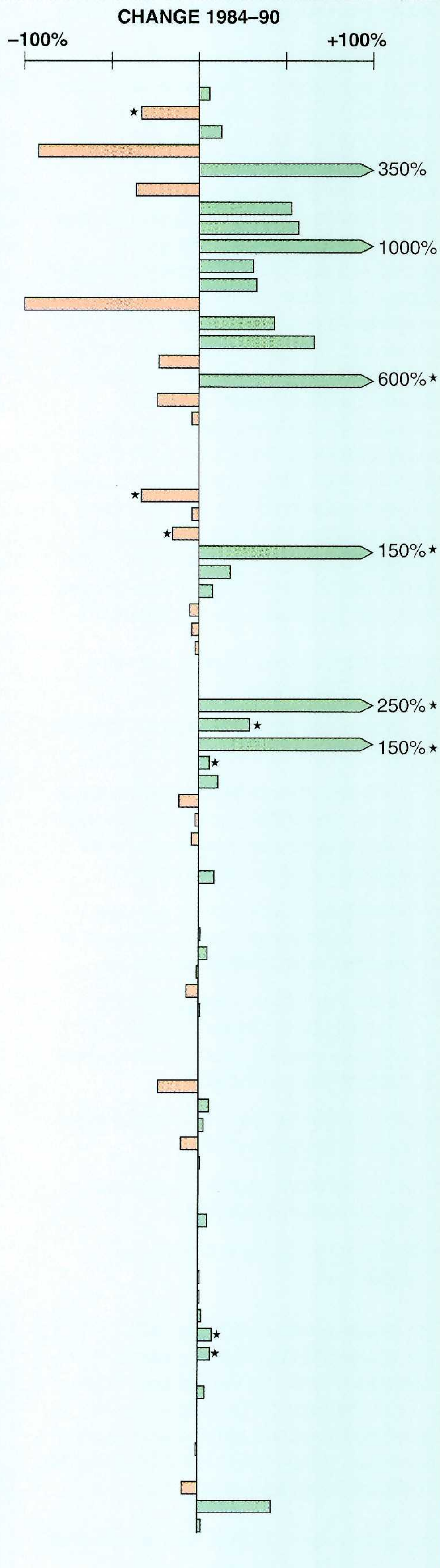


Figure 14. Land cover stock in 1990 and change between 1984 and 1990 (* changes significant at a national level). The 1990 stock figures are based on all 508 1 km squares surveyed in 1990. The 1984–90 change statistics are based on 381 1 km squares which were surveyed in both years

Land cover change from field survey

The field survey gave a more detailed picture of land cover on the ground and provided the opportunity, using the 381 km squares surveyed in both 1984 and 1990, to examine changes in land cover over that six-year period. The field survey results are summarised in 58 land cover types (Figure 14). The use of identical squares allowed measurement of changes between different cover and vegetation types, giving matrices of from/to information. The net changes in each land cover type were made up of the balance between losses and gains. Overall, 87% of GB stayed in the same broad land cover type, and most of the large changes were typical of agricultural rotations between crops and grassland. In many semi-natural vegetation types, including broadleaved woodlands, there was a rough balance between losses and gains and the totals remain unchanged.

In summary, the net changes between 1984 and 1990 were:

- an increase in urban land of 10 500 ha yr^{-1};
- a 4% decrease in tilled land, mostly in barley which decreased by 33%, but there was also an increase in minor crops such as maize and linseed;
- a decrease of intensively managed types of grassland and an increase in weedier, unmanaged grasslands;
- a doubling of non-cropped arable land (which would be the type of land typically resulting from land set-aside from cereal production);
- little change in the area of many semi-natural vegetation types;
- a 4% net increase of built-up land, mostly in the countryside;
- a 5% net increase in coniferous woodland.

The results contrast with the well-documented losses of semi-natural habitats in Britain during the post-War period. Instead of a wholesale loss of these habitats in the wider countryside, there was, in the period 1984–90, more of a balance between loss and gain.

The grouping of the field data into 58 land cover categories provided a general overview of the GB countryside, but the Countryside Survey database allows greater flexibility for more detailed analyses of particular habitats, landscape features and species, eg ponds, heather (*Calluna vulgaris*), and non-native species.

Field boundaries

The linear features that criss-cross the countryside form an important part of the British landscape. Many of these features are field boundaries, such as hedges, lines of trees, walls, fences, banks, or just grass strips. In 1990 the total estimated length of field boundaries was nearly 1.5 million km. On average, there were about 6.3 km of field boundaries in each 1 km square. Many of these boundaries (30%) contained more than one type of element, such as a hedge with a fence. Over two-thirds of boundaries (72%) contained a fence, and almost a third (31%) contained a hedge. Between 1984 and 1990, the total length of boundaries declined slightly but individual boundary types showed substantial losses or gains (Figure 15). Thus, hedges and walls decreased in total length, while fences and relict hedges (woody boundaries that had once been hedges) became more common.

- **Hedges** – 31% of all boundaries in Great Britain contain hedges; 81% of these are in England, with 51% in the pastural landscape areas. Most hedges are found in combination with other boundary types, particularly fences in arable landscapes and banks in pastural landscapes. Hedges are less common in marginal upland landscapes and absent from the uplands.

  In net terms, 23% of hedges recorded in 1984 in GB had changed by 1990. Most of this change was due to changes to different boundary types (eg a hedge becoming a line of trees), probably because of changes in hedge management regimes. The 23% net loss of hedges represents the balance between gains from new hedge planting and natural hedge regeneration (+4%), and losses from boundary removal (–11%), together with changes to (–20%) and from (+5%) other types of boundary.

- **Relict hedges** are boundaries that are recognisable as having once been hedges, but have become, for example, rows of trees or lines of shrubs and are no longer stock-proof boundaries. They form parts of 6% of all boundaries (75% of these in England). The length of relict hedges increased by 55% between 1984 and 1990, mainly as a result of hedges becoming lines of trees.

- **Walls** – 13% of boundaries contain walls; 47% of these are in Scotland and 39% in England. Most walls are found in the marginal upland landscapes but, perhaps surprisingly, there are more walls in arable landscape types (19%) than in the uplands (16%) where they are a more noticeable feature of the landscape.

  The total length of walls decreased by 10% between 1984 and 1990, although in Scotland there was a small increase. Walls next to fences were twice as likely to be lost than walls on their own, perhaps because the former were already in decline. The greatest length of walls was lost in the marginal uplands, where the initial 1984 stock was highest, but a higher proportion of walls was lost in the arable landscape.

- **Fences** are the most widespread and common boundary component, occurring in 72% of all boundaries and being the most common single-element boundary type in all landscape types. Nearly half of all boundaries (46%) were formed by

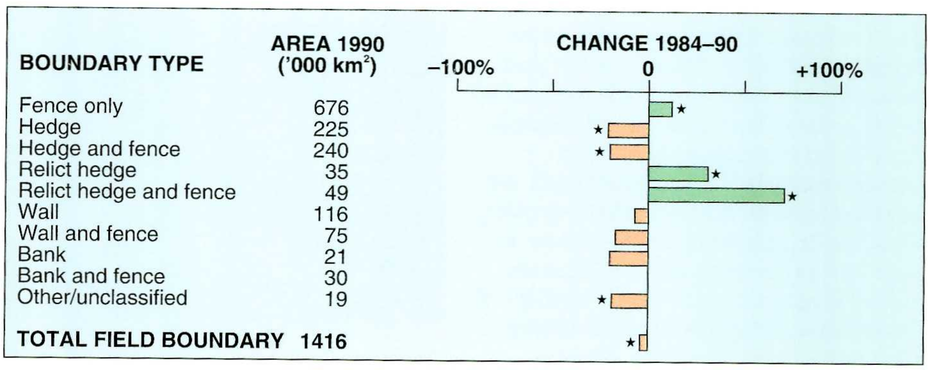


Figure 15. Length of field boundaries in 1990 and change from 1984 to 1990 for Great Britain (* changes significant at a national level; hedges include hedges besides walls and/or banks)

fences alone. Fences were also the most stable boundary type, with almost two-thirds remaining as fences over the survey period.

The length of fences increased by 11% between 1984 and 1990. Almost half of these new fences were built in pastural landscapes, with relatively few in the arable and upland landscape types.

Plant diversity

In addition to major and obvious step-wise shifts from one type of land cover, habitat or linear feature to another, change in the countryside can take the form of more gradual, subtle changes in the balance of species within habitats. The loss or gain of flowering plants in meadows is a typical example. Using the detailed records of species composition from the same plots in 1978 and 1990, it was possible to distinguish between those losses and gains in plant diversity due to shifts *between* vegetation types and those due to changes in the quality, ie the species composition, *within* vegetation types.

The vegetation plots have been classified using TWINSPAN into vegetation types, characteristic of open fields, woods or moors, hedges, roadside verges and stream banks.

Changes within vegetation types

The vegetation plots from open fields, woods and moors were grouped into six types of vegetation – arable fields, improved grassland, semi-improved grassland, woodland, upland grass and moorland. In Britain as a whole, three of these six major vegetation types (arable fields, semi-improved grass and woodland) showed significant losses of species between 1978 and 1990. Only one vegetation type, moorland, showed a significant increase in species diversity but this vegetation type is inherently species poor (Figure 16).

All the four landscapes showed a change in species composition with an increase of species more characteristic of intensively managed vegetation, and this change was most pronounced in the lowlands. In the arable landscape, there was a net shift towards more intensive types of vegetation in 29% of all plots. In pastural, marginal upland and upland landscapes, the shifts in the same direction were 27%, 11% and 9% respectively.

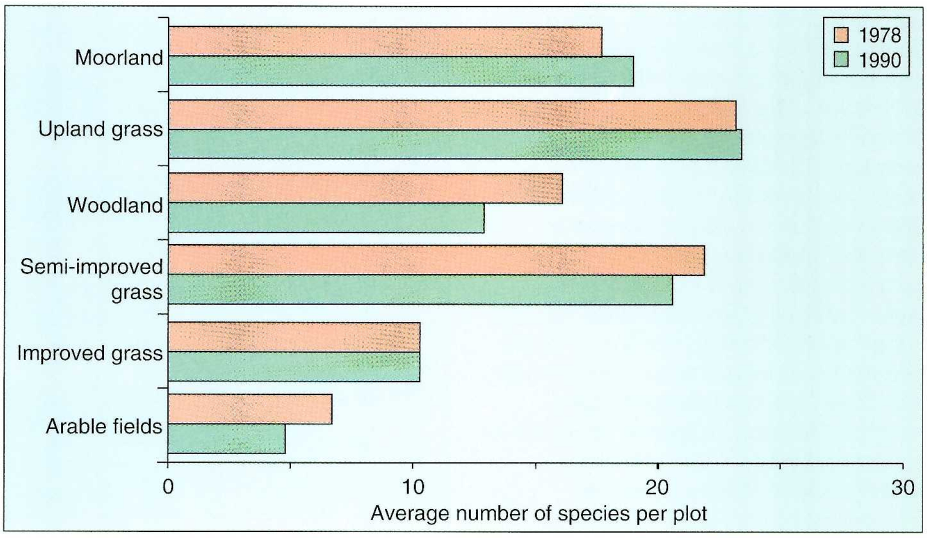


Figure 16. Change in plant diversity within vegetation types, 1978–90

The main changes in six main types of vegetation in Great Britain are summarised below.

- **Arable fields**
  The loss of species in arable fields, particularly in arable landscape types, was associated with a shift towards groups of species typical of more intensive use. This reflected a decrease in broadleaved weeds and an increase in grass weeds within cereal crops, particularly in arable landscape types where arable fields now have 25% fewer species than those in pastural landscapes.

  The decline in annual and perennial weed species, especially the broadleaved species, may have some implications for invertebrate and bird species, but, from a botanical point of view, the species in decline are found elsewhere in the landscape, on disturbed ground. Rare species associated with arable fields, such as the corncockle (*Agrostemma githago*), had already disappeared from the vast majority of fields by 1978.

- **Improved lowland grassland**
  No significant changes were recorded in plots of this vegetation type.

- **Semi-improved grassland**
  In arable and pastural landscapes, plots from semi-improved grasslands lost diversity and shifted towards more intensely managed vegetation types. Plots in pastural landscapes had 14% fewer species in 1990 than in 1978. The most pronounced decline has been in species associated with 'unimproved meadows', which include many of the rarer grassland species. Plots from fields in all three landscapes had very similar numbers of species in 1978, but by 1990 those in marginal upland landscapes were more diverse than those in other areas. These data indicate that at least some of the grassland types in the marginal upland landscapes were being less intensively managed in 1990 than in 1978.

- **Woodlands**
  Woodlands in all landscape types except arable have shown a significant loss of species. Most species groups have decreased but species more characteristic of disturbed and grassy habitats within woodlands are increasing, indicating that at least some woodlands are becoming more open and grassy.

- **Upland grassland**
  No significant changes were recorded.

- **Moorland**
  In moorland plots from marginal upland and upland landscapes, there was an increase in species number, in contrast to the pastural landscapes where the species number had declined (because of a loss of bog/moorland species). Moorland habitats are inherently species poor, so increases in species diversity might indicate invasion by non-moorland species.

Plant diversity of hedgerows

Countryside Survey 1990 recorded 40 species of woody shrub and 270 herbaceous species in 1000 hedge plots. These plots have been grouped into seven hedgerow types and four ground flora types. The results confirm that British hedges are dominated by hawthorn (*Crataegus monogyna*) – a consequence of the planting of hedges to enclose fields around 200 years ago. However, they also contain many other plant species. The two most diverse types, in terms of tree and shrub species, were the mixed hazel (*Coryllus avellana*) hedges found mainly in the west of Britain and mixed hawthorn hedges, found mainly in the east. Hedgerow ground flora was most diverse in the west of the country where hedges are more frequently adjacent to woodland and grassland, rather than arable fields.

Between 1978 and 1990 there was:

- no change in the woody species composition of hedge plots in any landscape type, despite the reduction in the total length of hedges;
- no significant change in the species richness of hedge plots in arable landscape types, although there was a shift towards species more characteristic of arable fields;
- a decrease in the number of species in hedges in pastural landscape types, particularly in the ground flora of hedges bordering grassland where meadow and chalk grassland species decreased;
- no significant change in the number of species of hedge plots in marginal upland landscapes.

Plant diversity in road verges

Verges are susceptible to a number of factors which influence their species composition. They are directly affected by changes in management, eg mowing regimes, road salting, use of herbicides and growth retardants. Verges are also vulnerable to disturbance from road works, ditch clearance and vehicles, all of which may create bare patches which allow the invasion of colonising species.

Almost 2000 verge plots were surveyed in 1990 and grouped into eight types of verge flora. Species diversity decreased significantly in road verges in arable landscapes, but not elsewhere. However, in all areas, there was some loss of characteristic meadow species and, with the exception of the uplands, where verges were often grazed, there was a trend towards an increase in overgrown verge types, often dominated by tussocky grasses and tall herbs, and sometimes scrub.

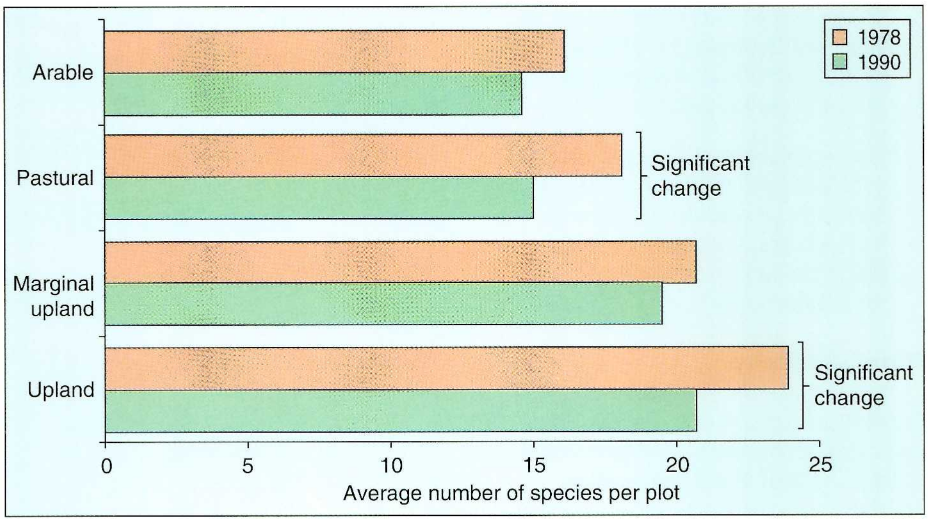


Figure 17. Number of species in stream bank plots, 1978–90

Plant diversity in watersides

Over 2000 plots were surveyed beside ditches, streams and rivers. They were grouped into seven types of stream bank flora. The plots showed a loss of species in all four landscapes between 1978 and 1990, the losses being significant in pastural and upland landscapes (Figure 17).

Throughout the lowlands there was a loss of species typical of wet meadows and moist woodlands, all of which require damp conditions. In the pastural and upland landscapes, there was a loss of species in most species groups, but particularly traditional meadow species.

In the arable landscape there is some indication that removal of grazing and/or the cutting of watercourse banks has led to the development of coarse grassland or tall herb vegetation, and possibly scrub invasion. Some losses imply that the habitat has dried out. For example, species typical of wet meadows and aquatic margins decreased in all four landscapes, though more so in the lowlands. These changes may reflect the fact that 1990 was a drought year in southern parts of Great Britain. However, changes were also recorded in the unaffected upland regions, and most of the species that decreased were long-lived perennials which are unlikely to be lost because of short-term changes in water levels.

Biodiversity in the wider countryside

Hedges, verges and watersides contained many species which were absent or rare in the surrounding landscape and represented an important reservoir of botanical diversity. This was particularly true in arable landscapes in which most types of species were more frequently found in the linear features than in the fields or woodlands (Figure 18).

Although meadow species were declining in hedgerows, verges and watersides, these linear features still contained more of the total resource of meadow species than was found in the open countryside. Linear features are important for biodiversity not only because of their contribution to overall plant diversity in the countryside, but also because they can act as a source of locally native seed. In years to come, given the right conditions, it may be possible for species-rich habitats to regenerate from these seed banks. Any further loss of linear features, or the meadow species they often contain, may limit the scope to conserve biodiversity in the lowlands.

Freshwater animals and water quality

The Institute of Freshwater Ecology identified a total of 479 freshwater species or groups of species in the samples collected from 361 sites. Although species diversity was greater in lowland watercourses than in the uplands, the reverse was true for water quality. In arable landscapes, 60% of watercourses were 'good' quality compared to 88% in the uplands.

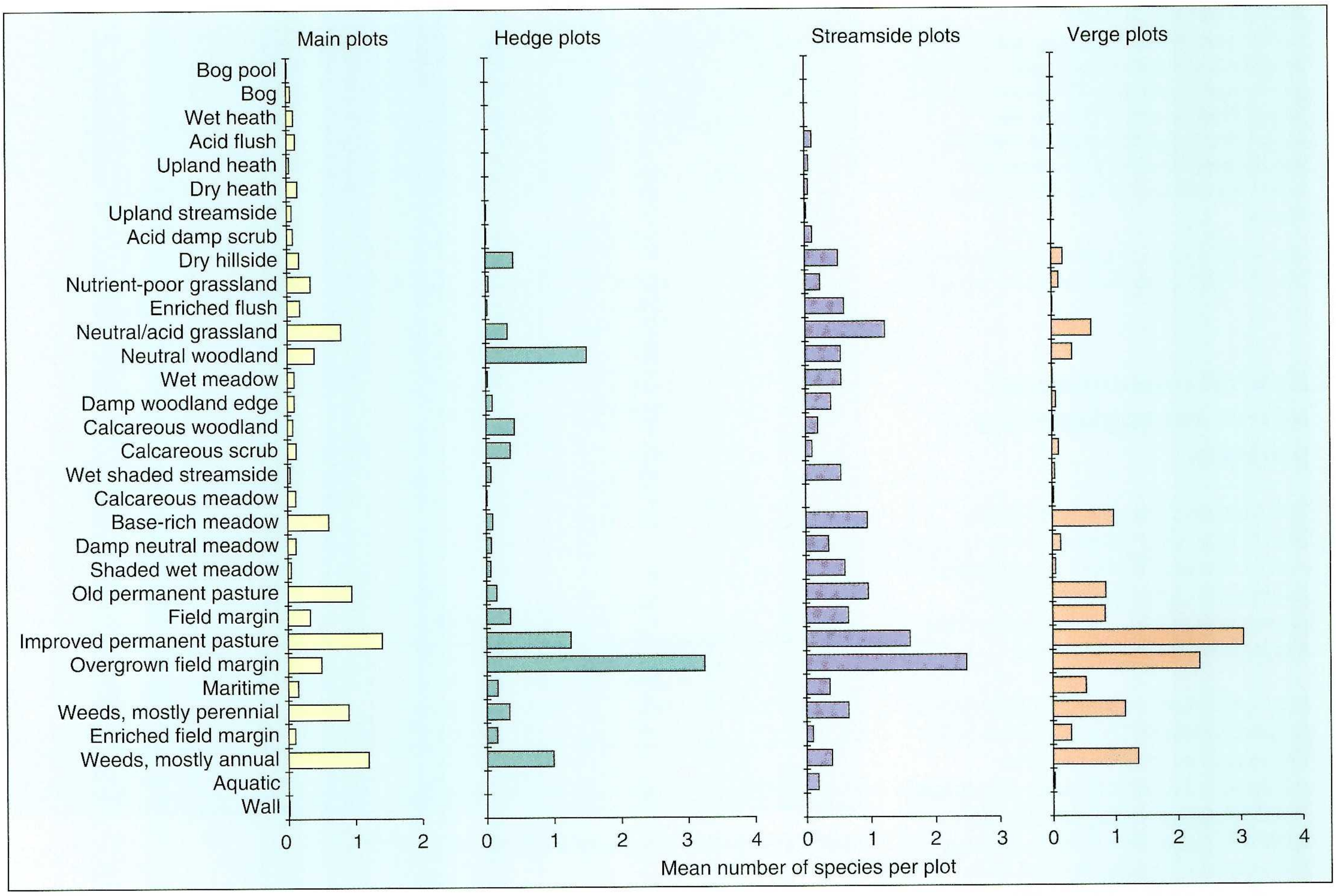


Figure 18. Comparison between the mean number of species in the species groups, recorded in the main plots and linear plots in 1990, in arable landscapes

Main points from Countryside Survey 1990

It is not easy to summarise the results from a project as complex as Countryside Survey 1990 without a risk of oversimplification. However, the following general conclusions are supported by the results.

- The proportion of the main semi-natural vegetation types in the British countryside has remained constant throughout the late 1980s, but the quality of the vegetation in areas of semi-natural and agricultural land cover has declined.

- Loss of habitats and species diversity has also occurred in linear features (hedges, verges and watersides) but these features were still important reservoirs of plant species, particularly in the lowlands.

- Loss of species and decreases in the quality of vegetation were greater in the lowlands than the uplands. The quality of freshwater habitats, as reflected by the invertebrate species they contained in 1990, was also lower in the lowlands.

Data availability

This report has, inevitably, provided only a glimpse of the wealth of data collected during Countryside Survey 1990. A more detailed presentation of the methodology, the stock and change data on land cover, landscape features and vegetation at the national and landscape scale, and of the information on freshwater biology, is provided by Barr *et al.* (1993). Information on the land classes used to provide the sampling framework, the data contained in the main report, plus further data at the land class level, will soon be available in a Countryside Information System for use on personal computers. Due to be launched in autumn 1994, the System will allow the user to integrate, combine or overlay the constituent data sets and output derived results in tabular or graphical form.

The future

Change in the countryside is a continuous process and the three countryside surveys provide snapshots at given times. At the launch of the Countryside Survey 1990 reports in October 1992, John Gummer, the Secretary of State for the Environment, stressed that the 1990 Survey provided an essential baseline against which future change could be assessed and against which the success of countryside policies could be judged, and he gave a commitment to Government support for a further survey in the year 2000.

C J Barr, R G H Bunce and T W Parr

References

Barr, C.J., Bunce, R.G.H., Clarke, R.T., Fuller, R.M., Furse, M.T., Gillespie, M.K., Groom, G.B., Hallam, C.J., Hornung, M., Howard, D.C. & Ness, M. 1993. *Countryside Survey 1990: main report.* London: Department of the Environment.

Acknowledgements
Countryside Survey 1990 required a large integrated, multidisciplinary team The main teams were drawn from ITE Merlewood, EIC Monks Wood and the IFE Wareham Laboratory, but staff from all the other ITE stations were involved plus a team of surveyors recruited to help with the field survey

Habitat reinstatement advice for engineering projects

The Applied Ecology Group at ITE Banchory is often asked to give advice on reinstatement of habitats to engineers, developers and planners Types of project concerned and the approaches adopted are described below

Replacing destroyed vegetation cover is not particularly difficult anywhere in the UK (except on toxic or unstable substrates) if a uniform grass sward is all that is required In the last decade, however, there have been increasing pressures for developers and land managers to reinstate with more appropriate vegetation where the ground affected is of some conservation or landscape value Pressures have come from the public sector, planning authorities, and from statutory nature conservation agencies Often the necessary advice can be provided by landscape architects or by ecological consultants ITE has generally become involved when the work is part of a larger environmental impact assessment, when the habitat is difficult to restore, or when there is a research element The Institute has a particular advantage in this field with an ongoing research programme testing new techniques, coupled with extensive experience of the management of natural habitats Examples of projects in which there has been ITE involvement include road and bridge schemes, pipeline routes and at ski resorts The input usually covers not only the initial establishment of vegetation but also its subsequent management and any related monitoring requirements

Wildflowers for trunk roads and motorways

Revegetation of road verges and slopes was for many years dictated by the Department of Transport standard grass mixture A contract to ITE (Monks Wood and Banchory) from the Department resulted in a more flexible set of guidelines, the *Wildflower handbook* (DOT 1993), recognising that, although there are sites where a simple grass mixture is appropriate, there are many situations where wildflower mixtures would be better The *Handbook* uses a series of keys to help identify the appropriate approach for different types of site, taking into account features such as landscape setting, soil type, and previous vegetation An example of one of the keys is given in Figure 19 Guidelines are also given for subsequent management, which is particularly crucial for successful wildflower swards

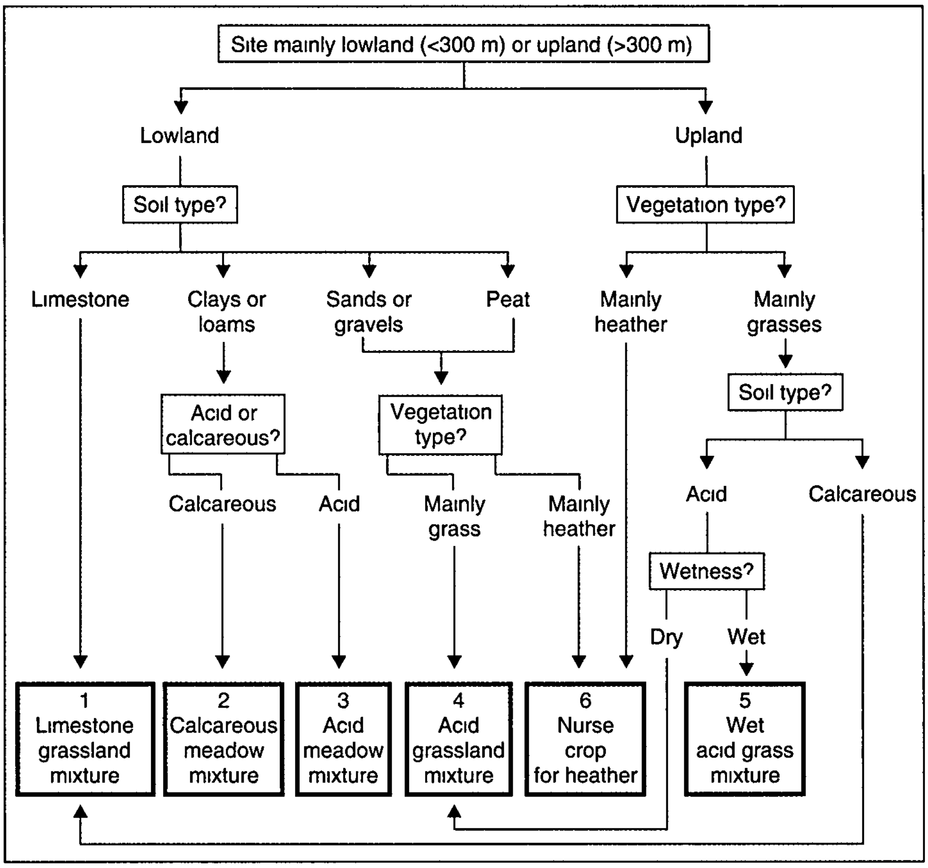


Figure 19 Key to the selection of wildflower mixtures (DOT 1993)

Glen Coe arched bridge competition

In 1993, the Scottish Office ran a design competition for the replacement of an arched bridge in an area of high scenic value (Glen Coe) The entry from Ove Arup, Laings and ITE won Environmental protection aspects of the design were the responsibility of ITE Reinstatement prescriptions were based on the conservation of existing vegetation resources Turf from the site was categorised into a small number of basic types (Figure 20), for stripping, storage and replacement after the bridge was completed The landscaping design incorporated small areas of scree to blend with surrounding similar ground Supplementary procedures were adopted to ensure that the stream was not contaminated by demolition or construction material The bridge has now been completed but monitoring of vegetation establishment will continue for two years

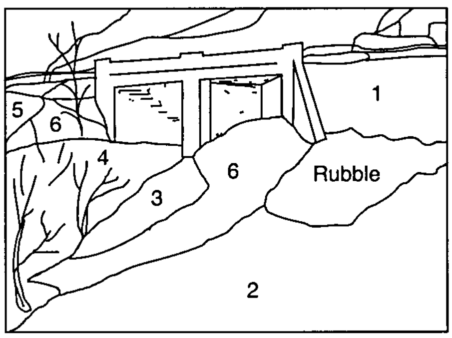


Figure 20 Vegetation categories below the original Glen Coe bridge prior to turf stripping and storage
1, embankment acid grassland, 2, wet flush grassland, 3, bracken, 4, rock ledges, 5, calcareous grassland, 6, acid grassland

Plate 17. Part of a pipeline route showing excellent re-establishment of vegetation on mineral soils (grassed area) and poor establishment on peat (foreground)

Pipelines

In devising prescriptions for reinstating long-distance pipelines, ITE has mainly been involved in advising on habitats of conservation interest. The information required to produce prescriptions includes details of the initial vegetation, soil characteristics, drainage and buried viable seed populations (Plate 17). The work involves both pre-construction survey and post-development monitoring.

In most situations, the most effective method for reinstatement would be returfing with the original vegetation. However, this technique is not generally cost-effective on a large scale, and is usually confined to high-value sites or communities such as bogs and small species-rich pastures. Instead, a variety of seeding and transplanting methods are used which are substantially cheaper to implement, although generally less effective.

High-altitude reinstatement

Bare ground on ski areas and along high-altitude footpaths is particularly difficult to reinstate because of severe climate, and infertile soils (Plate 18). It is, of course, essential to minimise further disturbance of the ground, either by fencing it off or by canalising use along hardened routes. A programme of trials is underway to develop methods of re-establishing mountain plant communities, particularly those which are rich in moss. This work has been sponsored by Scottish Natural Heritage and other agencies. The technical difficulties are considerable, although the use of surface mulches of gravel to improve establishment is a promising approach. Where feasible, however, the most rapid and reliable methods involve the re-use of existing material, either transplanted from intact ground or stripped in advance of any construction activities and relaid afterwards.

Quality control

ITE provides reinstatement advice based as far as possible on documented site information. The success of a prescription depends on many factors, including weather, timing of operations, and how well the prescription is followed. Unfortunately, there are many possible reasons for failure. Reinstatement work is generally the last stage of a development, and as a result prone to many setbacks. Delayed implementation is a major problem, particularly when turfing or other live transfer methods are being used. Failure to follow the specification is also a common failing. Deficiencies in the materials used are yet another problem. It is not unusual to find that seed mixtures supplied do not meet the specifications; species substitution or contamination can result in inappropriate vegetation. To reduce these problems and possible blame being placed on the prescription, it is useful to have tightly worded prescriptions, and close supervision of the reinstatement work. The prescription should also cover subsequent management of the site. Whenever feasible, a monitoring programme is advised to check on actual performance.

These quality assurance procedures aim to ensure that the advice given by the Institute is seen to be appropriate and that any failings can be identified and remedied without delay.

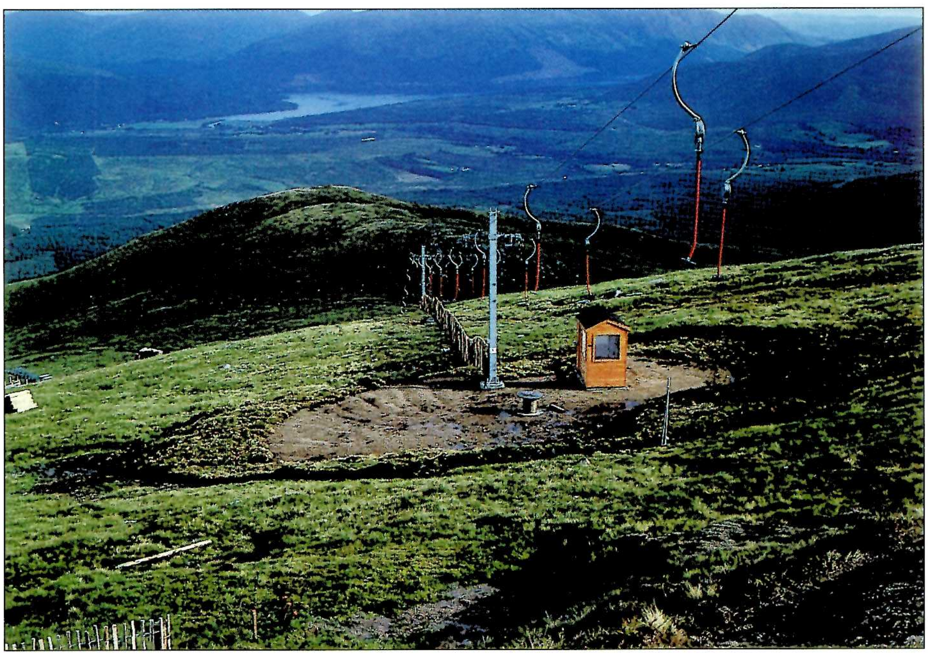


Plate 18. Turfing around a ski lift pylon on Aonach Mor. ITE has been advising on reinstatement and environmental monitoring at this resort since it opened in 1990

The work outlined here combines information from basic research on successional processes and screening of individual species with practical experience of reinstatement techniques to create a dialogue with engineers, landscapers and planners.

N G Bayfield

Reference

Department of Transport. 1993. *The wildflower handbook.* London: Department of Transport.

The Wytch Farm oilfield development – an example of ITE/industry partnership

Extraction of oil from deposits beneath Poole Harbour (an area of recognised high nature conservation interest, potential archaeological importance and known landscape quality, all within a region where tourism is an important component of the local economy) needed great sensitivity and care. At the outset, BP Exploration recognised that there would be a need for high-quality, impartial, advice from a number of organisations having different fields of expertise. ITE Furzebrook was able to provide the range of specialists needed to assess the ecological implications of any proposals made. At the same time, the opportunity was presented for the collection of population data and the testing of ideas on community development. The result was a close working relationship which provided benefits for both industry and research.

The assessment of environmental impacts of the development involved a number of stages, working from large-scale overall effects at one extreme to details of construction work or operating systems at the other. In the case of the export pipeline carrying oil from Wytch Farm (Plate 19) to the Hamble Oil Terminal on Southampton Water, a preliminary route was defined and advice was sought to determine the types of semi-natural communities that would be affected, and the scale of disruption to those communities. On the basis of these discussions, a more positive route was defined. Despite attempts to route the pipeline through agricultural land wherever possible, disturbance of some areas of heathland, grassland, woodland and marsh was unavoidable.

The next stage in the procedure was a series of surveys of the semi-natural sites. These allowed inventories and maps to be drawn up. Different plant and animal communities vary in their susceptibility to specific impacts (eg modification of drainage patterns) and in their rates of recovery from disturbance. Surveys revealed the extent and disposition of communities so that assessments could be made of the potential short-term impacts of specific aspects of development. The surveys were also the basis for guidance on the possible long-term consequences – resulting in either re-establishment of similar communities in the same location or the development of different vegetation types. At the same time, any species of special interest – because of their distribution and rarity – could be identified and similar assessments of development impacts could be made. These surveys were important in helping to point the way to appropriate construction procedures designed to cause least disruption and to ensure effective site restoration or management during and after construction.

Another important function of the surveys was to provide baselines against which speed and success in restoration could be measured. Clearly, within the one woodland site traversed, there could not be a quick return to the pre-construction vegetation type of coppice hazel (*Corylus avellana*) with scattered standard oak (*Quercus*) trees. For other habitat types it was possible to design 'method statements' for construction work in which the aim was to reconstitute a vegetation cover with the essential characteristics of that present before work began and to do so within the aftercare period of five years. The precision of these prescriptions varied with the types of vegetation through which the pipeline passed. In the case of heathland, for example, three different techniques were used: natural and supplemented reseeding (where communities were reconstituted on bared soil), through the use of 'clodding' (in which blocks of soil and the vegetation supported by them were removed and replaced roughly in position), to turfing (where more care was exercised to restore the complete original surface vegetation to its pre-construction position). Although these techniques were used on different parts of the pipeline, some comparisons of the differences produced were possible. Where no additional seed was added to dry heathland sites, there was a tendency for dense populations of bristle bent (*Agrostis curtisii*) to develop, whilst turfing produced a more rapid return to closed vegetation cover, provided droughting did not follow turf replacement. Clodding seemed more appropriate where the soil contained a high proportion of organic matter and where water was present

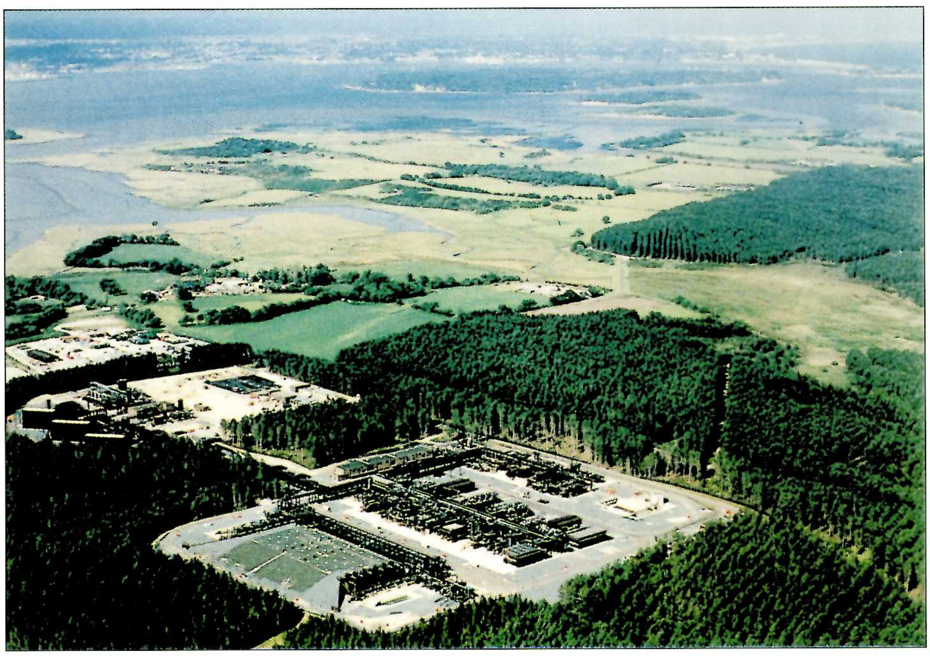


Plate 19. Wytch Farm Gathering Station and Poole Harbour from the air (reproduced with permission from BP Exploration)

Plate 20. Use of the clodding technique: reinstated heathland at Barnsfield Heath

close to the soil surface. An example of successful clodding at Barnsfield Heath is shown in Plate 20.

The greatest care of all was needed in the restoration of saltmarsh vegetation, where working conditions were not always easy (Plate 21). Here, it was important to cut and lift turves precisely and to return them, following pipeline installation, in the correct locations so that creek lines and small-scale topographic variations were reinstated to their pre-construction configuration, thus reducing the potential for erosion within the marsh and maintaining floristic gradients. Built into all these procedures were contingency plans which could be adopted where unforeseen circumstances prevented complete adherence to method statements. These plans allowed appropriate remedial action to be taken.

An essential feature of any ecological assessment programme is monitoring. This may be undertaken for a number of reasons:

- to confirm that biological communities react in the manner expected;
- to ensure that unpredicted effects can be detected and any remedial action taken;
- to refine the predictive process by assessing rates or patterns of change with time; and
- to provide experience leading to possible procedural modification in future operations.

The relevance and success of the restoration methods recommended and adopted were measured by regular follow-up monitoring of sites to assess the rate of colonisation and the floristic composition of post-construction vegetation. For the Purbeck/Southampton pipeline, this was done by assessing the cover of different species within fixed transects established before construction began. Vegetation is the key to whole-biotope rehabilitation, as both its structure and species composition are important in providing suitable habitat conditions for different animal species (though these may not necessarily be early colonists of restored sites). In specific cases, monitoring of animal populations may be appropriate. In heathland, census studies were made to determine the rate of reoccupation of disturbed areas by reptiles which had been captured and translocated prior to construction. For example, as the vegetation cover of the pipeline route across the reseeded Wareham Forest section of the pipeline became more complete and taller, numbers of smooth snake (*Coronella austriaca*) and sand lizard (*Lacerta agilis*) increased (Figure 21). Immediately following construction, the broad area of bared soil lacked the appropriate balance of bare ground and vegetation to allow regular occupation, and it appeared to form at least a partial barrier to the movement of reptiles. However, as early as one year after construction, once vegetation had begun developing, the effectiveness of the pipeline as a barrier was much reduced: the same animals were recaptured on both sides of the pipeline.

The possible effects of other construction and operational activities were examined in a series of sharply focused monitoring studies, which also provided data of more general interest. For example, observations on the overwintering shorebird populations over three years at a number of locations in Poole Harbour not only showed that

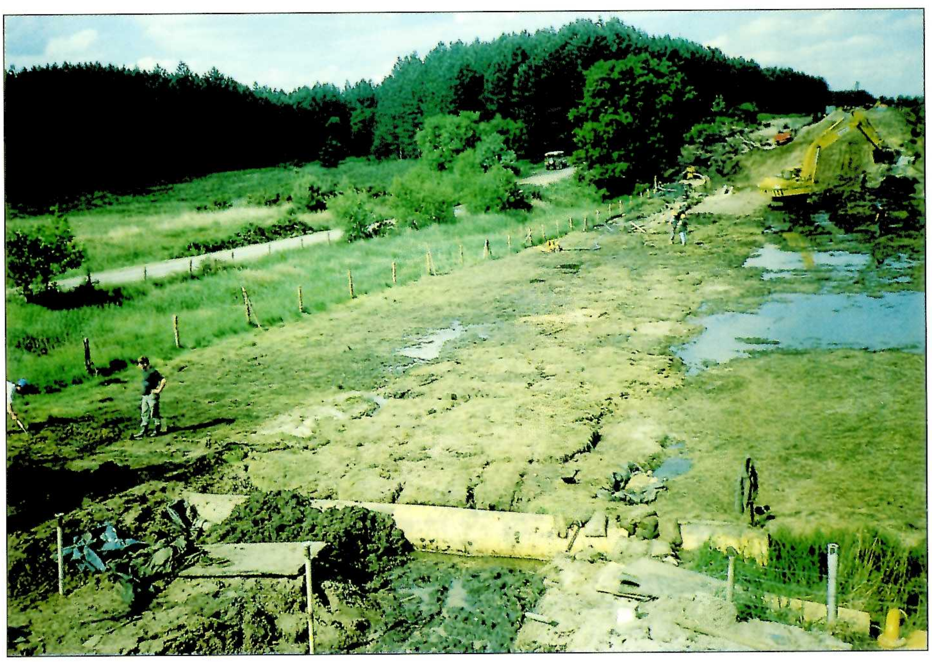


Plate 21. Reinstatement of saltmarsh turves at Shotover Moor (reproduced with permission from BP Exploration)

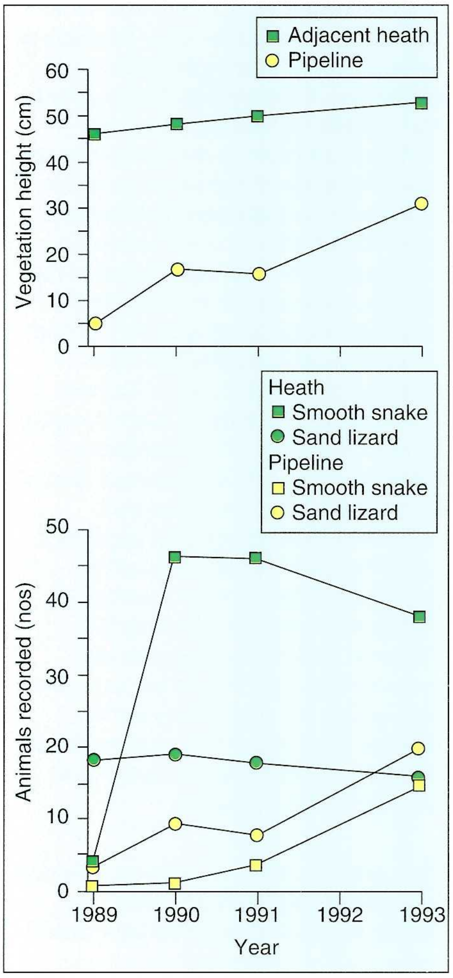


Figure 21. Heath vegetation height and the recolonisation of developing heathland by reptiles (Wareham Forest section of export pipeline)

construction work had no effect on feeding and roosting patterns, but added to existing knowledge of variations in distribution of birds in both space and time. Red squirrels (*Sciurus vulgaris*) were introduced to Furzey Island from Cannock Chase in 1977 and a population free from interference by grey squirrels (*Sciurus carolinensis*) has developed there in the pine (*Pinus*) woodland. ITE made a first assessment of population size in 1984 and has continued to make annual counts during development of the Wytch Farm oilfield. These counts have shown the population density to be especially high compared with observations made elsewhere in Britain, and that loss of coniferous woodland resulting from the construction of two wellsites on the Island had a minimal effect on the number of red squirrels present. More important factors which controlled breeding success, and hence population size, were the weather and the supply of pine cones – a major food source for this species.

Studies of the process of development and recession in different saltmarshes around the coast of Furzey Island were undertaken in order to assess underlying trends in accretion or erosion, and to determine any effects of additional works associated with, for example, jetty construction. Techniques employed included the use of levelled transects and Kestner cores to measure rates of surface deposition or erosion (Plate 22). The results provided guidance to BP in a specific, local, context, whilst at the same time providing information to build into more general models of saltmarsh dynamics. The Furzey saltmarshes show characteristics typical of others in southern Britain: a decrease in the extent of *Spartina anglica* and a steepening of the shore profile, resulting from increased erosion at the bottom of the marsh coupled with increased deposition at the top.

Plate 22. Measuring Kestner core in Furzey Island saltmarsh (reproduced with permission from BP Exploration)

Collaboration on this project showed the value of early involvement in the planning stages of the development and the benefits of continued involvement at progressive scales of refinement of site selection and site development procedures. For the industrial partner, appropriate decisions could be made which allowed development in the most ecologically sensitive manner, with follow-up surveys which provided early warning of any remedial work needed. For ITE, the chance was presented for accumulating additional data relevant to a number of on-going studies of ecosystem dynamics and processes.

R E Daniels

References

Moy, I.L. & Daniels, R.E. 1993. *Wytch Farm development. Purbeck–Southampton pipeline: restoration monitoring.* (NERC contract report to BP Exploration.) Wareham: Institute of Terrestrial Ecology.

Reading, C.J. 1993. *Purbeck–Southampton export pipeline: herpetological monitoring.* (NERC contract report to BP Exploration.) Wareham: Institute of Terrestrial Ecology.

Relationships between field boundaries and birds

(This work was funded jointly by NERC and the Ministry of Agriculture, Fisheries and Food (MAFF). The project was carried out in collaboration with the Soil and Water Research Centre of the Agricultural Development and Advisory Service)

East Anglia is one of the most intensively farmed areas in Britain, with arable activity playing the dominant role. Within this framework there remain small pockets of less intensively managed land. One such area was the floodplain of the River Great Ouse where 17th century drainage permitted pastural activity but regular flooding prevented arable cultivation.

Around the village of Swavesey, in Cambridgeshire, areas below the 5 m contour were subject to regular flooding, and cultivation was concentrated on slightly higher ground to the south. The parish boundary encompassed a range of farming activity, from low-intensity grazing to intensive arable production.

In 1986 a new flood protection and pump drainage scheme was installed (Plate 23) which had the effect of protecting about half of the remaining flood meadows in Swavesey. This scheme permitted conversion of pasture to cultivation, and an intensification of management where effective underdrainage became possible. A collaborative study to examine the direct and indirect effects of drainage of the Swavesey fens, the Swavesey project, lasted from 1985 to 1991 and generated a vast amount of ecological data on the fields, field margins and drainage courses of the area.

A major part of the project involved surveys over a number of years of 200 m long transects of the field boundaries, with an emphasis on recording bird populations (but also included surveys of both ground flora and butterflies). These boundaries varied greatly in appearance, from simple post-and-wire fences through annually trimmed hedges to large unmanaged hedges. In addition, some boundaries included ditches, trees and grass verges.

Bird surveys were repeated at approximately monthly intervals in winter and summer months and comprised 131 transects. The resulting large data set was used to model the relationships of bird species richness and abundance with a large number of botanical and structural attributes of the field boundary transects, as well as information about land use on either side. Between 44 and 60 bird species were observed in each of the six seasons in which surveys were carried out; 77 species were observed in total. With a large number of bird species richness and abundance variables, and the data extending over summers and winters of several years, the regression modelling was extensive – the Swavesey data required an investigation of about 200 models. The majority achieved high levels of statistical significance, with the models explaining a large proportion of

Plate 23. Part of the new drainage scheme in the northern half of Cow Fen which enabled conversion to arable production

the variation in bird species variables (Parish, Lakhani & Sparks 1993, 1994a, b).

Significance of linear features

There were clear differences between pasture and arable field boundaries in the species richness of both birds and butterflies. In addition, there were substantial differences in plant species composition, even though this was not reflected in species richness (Table 1). These differences can be partly, but not totally, explained by the structure and size of hedges and other field boundary features. For example, arable boundaries tended to contain smaller hedgerows and fewer trees. After eliminating field boundary effects, there still existed some unexplained but negative effects of arable production on the species richness of bird populations.

Regression models incorporated crop type (arable, pasture or intermediate), both as a categorical variable and as an interaction with other variables. To illustrate the modelling done, the model for blackbirds (*Turdus merula*) (summer 1991) related mean observed bird number, Y, in transect j of crop type i to hedgerow size (length × width), X1, number of trees, X2, and the number of woody plant species, X3, and had the form:

$$Y_{ij} = A_i + B1_i X1_{ij} + B2_i X2_{ij} + B3_i X3_{ij} + e_{ij}$$

R^2=53.6%; P<0.001

Such a model can be described as hyperplanes in many dimensions, and was typical of most bird variables.

For many species, the volume of hedgerow (and particularly height) was important, eg wrens (*Troglodytes troglodytes*) and robins (*Erithacus rubecula*) (Figure 22), where hedgerow vegetation provides cover for nesting and foraging, and, in the case of robins, perches from which potential prey items can be seen. Field boundaries containing trees and a diversity of woody plant species also enhanced bird species richness (Plate 24).

Table 1. Mean species richness ± 1 SE of birds, butterflies and flora in boundaries of pasture and arable fields, in two different years

	1986		1991	
	Pasture	Arable	Pasture	Arable
Breeding bird species	4.4±0.36	2.2±0.37	3.9±0.41	1.5±0.28
Butterfly species	4.4±0.35	2.6±0.38	6.0±0.33	3.2±0.34
Dicot flora species	15.1±0.87	17.0±0.92	20.2±0.76	21.0±1.02

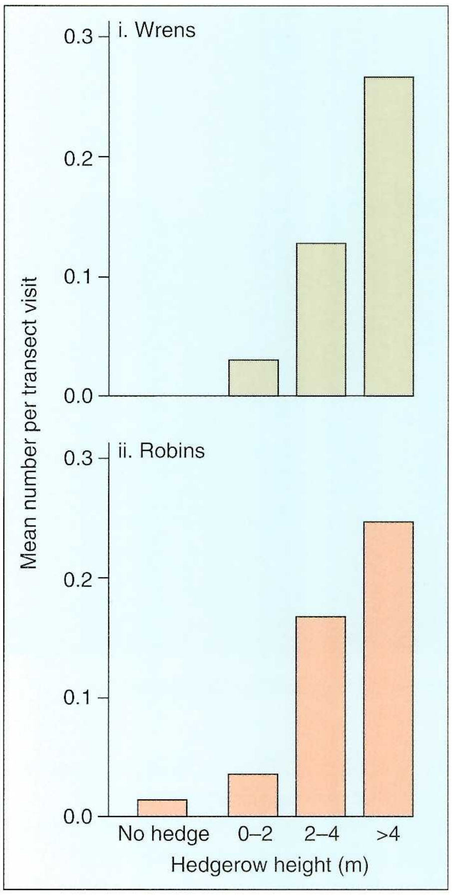


Figure 22. Average number of (i) wrens and (ii) robins observed per transect visit for hedgeless transects and for those with hedges in three height categories. Data are means of summer 1986 and summer 1991 observations

Effects of drainage

Within the study area, the northern half of Cow Fen experienced most change following drainage. This area was previously susceptible to frequent flooding, but became suitable for arable production. The southern half, being on slightly higher ground, was inundated only in the worst floods. Some engineering had taken place in Cow Fen before the first bird survey, but it is believed that the northern half would have been similar in bird community to Middle Fen, an area adjacent to the River Great Ouse. Highfield, a gravity-drained area to the south of Cow Fen, had been dominated by intensive production for some decades and was unaffected by flooding.

An examination of the changes in bird populations between 1986 and 1991 showed a decline in breeding species richness in all study areas, possibly as a result of successive drought years (Parish, Sparks & Lakhani 1994). Figure 23 confirms that Cow Fen 'north' may have been very similar to Middle Fen. However, this area was most affected by agricultural improvement and, in terms of breeding species, declined rapidly to a more typical arable situation.

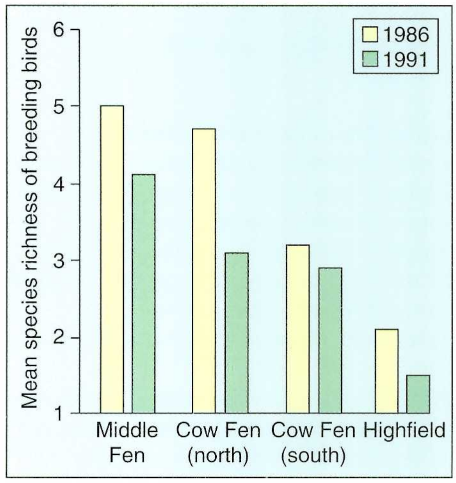


Figure 23. Species richness of breeding birds in 1986 and 1991 for four areas in the Swavesey fens. The northern half of Cow Fen was undergoing most agricultural change

Practical applications

The birds of the Swavesey fens are not rare either nationally or locally. Management for conservation of species is, therefore, likely to concentrate on enhancing species richness. The Swavesey fens are a woodless environment and hence maintenance of woodland bird populations will depend on the continued provision of larger hedgerows, as both breeding sites and as corridors for dispersal. Ditches, ditch banks and verges also play a crucial role here. A relaxation of current hedge management would be expected to yield benefits in increased nesting sites and food supply for birds. Rotational cutting on a two-year or longer cycle and cutting towards the limit of reach would also reduce costs whilst having only a moderate effect on yield at the field edge. Grass verges provide a buffer between cultivated field and hedge bottom and can be expected to be an important habitat resource in their own right. Grass verges may have economic benefits also, in reducing weed encroachment from headlands and allowing access to machinery without damage to the crop (Plate 25). Cutting different sides of ditch banks in alternate years would ensure that some cover was always provided.

Plate 24. Large hedgerows of diverse composition are a valuable habitat for birds

Plate 25. Grass verges and hedgerow trees can enhance bird populations. Allowing such a hedge to become taller would be of even greater benefit

Information from the Swavesey project has been used to provide advice to the Farming and Wildlife Advisory Group, MAFF, and other interested bodies and has been incorporated into relevant literature. The Swavesey project has enabled an objective assessment of the habitat requirements of birds in farmland, and has suggested suitable field boundary management to aid bird conservation.

T H Sparks, K H Lakhani and T Parish

References

Parish, T., Lakhani, K.H. & Sparks, T.H. 1993. *The assessment of the effects of field boundary attributes and of the adjacent land use upon bird population variables using multiple regression models: Huntingdon 1983–1985 and Swavesey 1985–1991.* (NERC contract report to the Ministry of Agriculture, Fisheries and Food.) Abbots Ripton: Institute of Terrestrial Ecology.

Parish, T., Lakhani, K.H. & Sparks, T.H. 1994a. Modelling the relation between bird population variables and other field margin attributes. I. Species richness of winter, summer and breeding birds. *Journal of Applied Ecology*. In press.

Parish, T., Lakhani, K.H. & Sparks, T.H. 1994b. Modelling the relation between bird population variables and other field margin attributes. II. Abundance of individual species and of groups of similar species. *Journal of Applied Ecology*. In press.

Parish, T., Sparks, T.H. & Lakhani, K.H. 1994. Models relating bird species diversity and abundance to field boundary characteristics. In: *Hedgerow management and nature conservation*, edited by T.A. Watt & G.P. Buckley. Wye: Wye College. In press.

Habitat fragmentation and the occurrence of breeding birds in small woodlands

(This work was funded partly by the NERC Joint Agriculture and Environment Programme (JAEP) and partly by the Joint Nature Conservation Committee (JNCC))

The fragmentation and concomitant loss of contiguous habitat are two of the major causes of a worldwide, and accelerating, reduction in biodiversity (Wilson 1992). Thus, the ability of species to persist in fragmented habitats and the relationships between their occurrence and fragment size are major concerns of conservation policy. In lowland Britain, deforestation was extensive as long ago as the 14th century; by 1350, woodland is thought to have occupied only about 10% of the area of England (Rackham 1986). In Cambridgeshire (where this work is based), with the exception of new conifer plantations, less than 10% of woodland exists in patches of more than 100 ha and about 50% is in patches of 10 ha or less. In consequence, many birds of broadleaved woodland now live in a network of scattered, and often small, habitat patches.

During 1990–92, the bird species breeding in 151 broadleaved woods in Cambridgeshire and south Lincolnshire were recorded. The woods ranged in size from 0.02 ha to 30 ha, but the majority (78%) were of 2 ha or less (Plate 26). With these data, species' minimum area requirements were examined and the probabilities of species breeding were investigated in relation to woodland area using logistic regression analysis.

Minimum area requirements

For the 31 species examined, there was little evidence for strict minimum area requirements (Table 2). Eight species bred in woods down to a minimum size of 0.02 ha in at least one of the three years of the study. A further 20 species bred in woods down to a minimum size of 0.5 ha, again in at least one year, and two species – chiffchaff (*Phylloscopus collybita*) and marsh tit (*Parus palustris*) – bred in woods down to 0.5–1.0 ha. Only the nightingale (*Luscinia megarhynchos*) failed to breed in woods of less than 1.0 ha in any of the three years. These results differ from those of other studies, particularly in the eastern deciduous forest of the United States (Robbins, Dawson & Dowell 1989), where forest interior species disappeared from forest fragments which were still large (100 ha or larger) by British standards. This difference may be due to the much earlier deforestation of Britain and to the earlier disappearance of area-sensitive species unable to adapt to forest fragmentation. The present decline of the capercaillie (*Tetrao urogallus*) in Scotland and elsewhere in Europe has been linked to the fragmentation and degradation of its pine (*Pinus*) forest habitat (Rolstad & Wegge 1989).

Plate 26. Typical small farm woodland, surrounded by cereal crops

The size of the wood in which a nest is found may be misleading in terms of a species' minimum area requirement. Some species readily cross gaps between woods and thus may meet their area requirements by using groups of small woods, providing these are close enough together. In this study, great-spotted woodpeckers (*Dendrocopos major*) bred in individual woods as small as 0.26 ha, but travelled routinely to other nearby woods. One such bird was seen to fly nearly 0.5 km between woods, pausing briefly about half-way across in an isolated group of trees. This behaviour makes it difficult to determine a 'genuine' minimum area requirement, but it may be considerably larger than the size of the single wood containing the nest site. The complex of woodlands supporting the

Table 2. Smallest wood in which each species bred for the years 1990–92. Minimum size of wood studied was 0.02 ha

Species	Smallest wood (ha)	Species	Smallest wood (ha)	Species	Smallest wood (ha)
Woodpigeon	0.02	Spotted flycatcher	0.10	Sparrowhawk	0.25
Turtle dove	0.02	Willow warbler	0.10	Great-spotted woodpecker	0.26
Blackbird	0.02	Carrion crow	0.12	Long-tailed tit	0.27
Dunnock	0.02	Goldfinch	0.12	Jay	0.32
Blue tit	0.02	Tree sparrow	0.12	Garden warbler	0.33
Great tit	0.02	Song thrush	0.12	Treecreeper	0.39
Chaffinch	0.02	Stock dove	0.15	Bullfinch	0.44
Whitethroat	0.02	Starling	0.16	Chiffchaff	0.65
Yellowhammer	0.03	Blackcap	0.20	Marsh tit	0.96
Robin	0.10	Greenfinch	0.25	Nightingale	1.76
Wren	0.10				

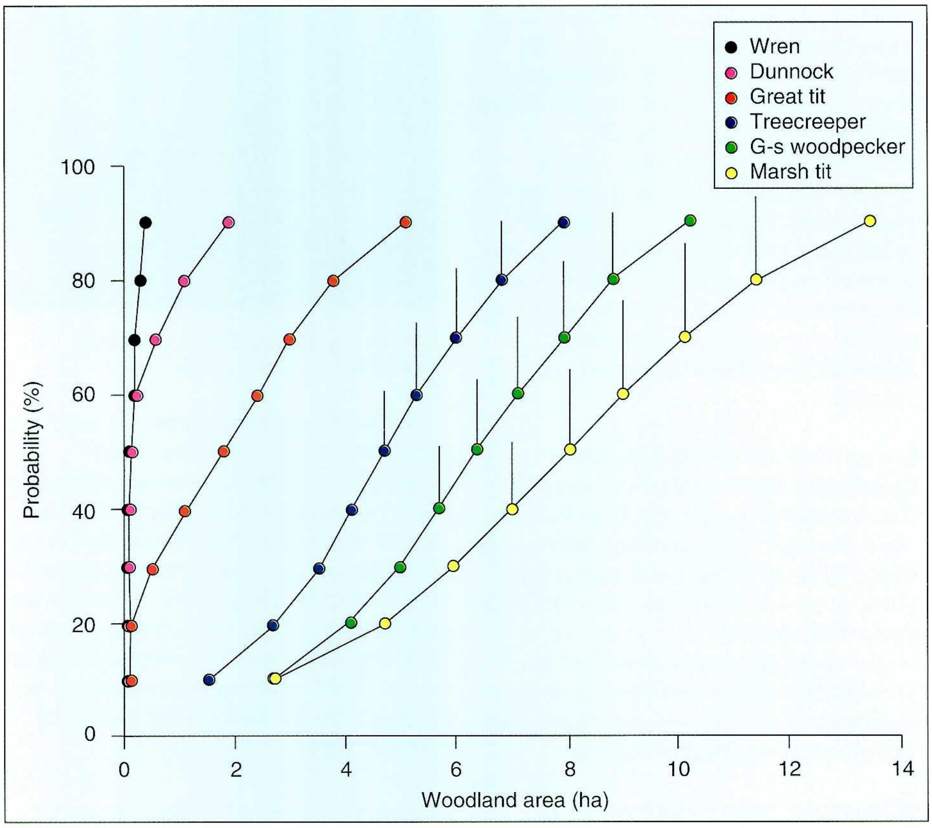


Figure 24. Relationship between the probability of breeding and woodland area for different species. All relationships are for 1990, except for marsh tit which is for 1991. Vertical bars represent 1 SE (those with a value <10% not shown)

woodpeckers described above comprised 15.5 ha (within a total area of about 2 km^2) and contained at most 2–3 pairs, giving an area of woodland per pair of 5.2–7.8 ha.

For species unwilling to cross open country, the option of using groups of woods is not available. Thus, as fragmentation of once continuous habitat proceeds, the stage at which the populations of the fragments become isolated from each other will be different for different species. For a given degree of fragmentation, the most mobile species may still move freely between woods, while more sedentary ones are already isolated with little chance of recolonisation in the event of extinction. Within the study area, nuthatches (*Sitta europaea*) were rare, even in the larger woods (100+ ha). Nuthatches have short dispersal distances (especially across open country) and their absence from suitable habitat suggests that these woods were sufficiently isolated from potential source populations to prevent recolonisation (Verboom *et al.* 1991).

Breeding and woodland size

Analyses of species' occurrences in relation to woodland area showed that some species bred in woods of all sizes, while others bred in small woods only infrequently (Figure 24). For widespread and common woodland species, the probability of breeding increased rapidly with increasing woodland area, whereas for less common species it usually increased more slowly. In general, for woodland species which are not commonly associated with other woody habitats, such as gardens and hedgerows, eg marsh tit, long-tailed tit (*Aegithalos caudatus*), treecreeper (*Certhia familiaris*) and great-spotted woodpecker, the probability of breeding did not approach 100% until woodland size was approximately 10 ha or larger (Table 3). Thus, species which could be termed 'woodland specialists' have a low probability of breeding in small woods. When small woods are used, the number of pairs present is liable to be small, making the species population vulnerable to random events which cause local extinction. Nightingales bred in too few of our woods for a reliable relationship with area to be calculated, but their absence from woods smaller than 1.0 ha indicated that they are also unlikely to breed in small woods.

Table 3. Woodland area required for a 90% probability of breeding for examples of widespread and common species and those more likely to be restricted to woodland habitats

	Woodland area required for a 90% probability of breeding						
	Widespread species				Woodland 'specialists'		
Species	1990	1991	1992	Species	1990	1991	1992
Blackbird	0.6	1.2	0.8	Long-tailed tit	8.2	9.1	8.3
Wren	0.4	3.0	0.8	Treecreeper	7.9	16.0	17.5
Chaffinch	1.1	0.8	0.9	Great-spotted woodpecker	10.2	27.2	10.9
Blue tit	1.8	2.0	1.9	Marsh tit	*	13.4	13.0

*In 1990, the relationship with area for marsh tit was insufficient to calculate probabilities

Annual variation

Repeating the analyses for each of the three years showed a large variation in the relationship between the probability of breeding and woodland area for some species, whereas for others it was consistent between years (Figure 25). When variation occurred, it usually took the form of a reduction in the probability of the species breeding in small woods; the probability/area curve either shifted to the right (with the slope largely unchanged) or the slope of the curve decreased substantially.

In February 1991, much of Britain, including East Anglia, suffered a period of severe weather with ice-glazing on vegetation and snow cover which resulted in a decrease in the size of many bird species populations, at least in eastern England (Marchant & Musty 1992). The loss of species from the study woods coincided with these generally lower breeding population levels in 1991 compared to the previous year. In 1992, some of these species recolonised small woods, their probability/area relationships returning to the patterns seen in 1990, but others showed no signs of recovery (Figure 25).

The underlying causes of these changes in the probability/area relationships are complex and differ between species. A species' distribution within an area will depend, amongst other factors, on its

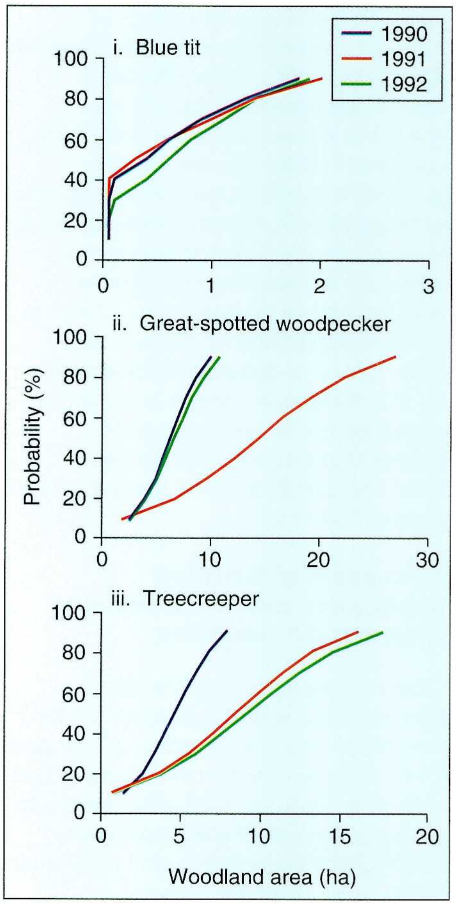


Figure 25. Relationship between the probability of breeding and woodland area for three species showing different patterns of variation between years

overall abundance, habitat requirements and dispersal abilities. We are carrying out further analyses to determine the relative importance of these various factors for different species. Although the majority of species in our study did breed, at least occasionally, in woods of 1.0 ha or less, this was not a common occurrence for the more specialised woodland species. Even for woods as large as 10 ha, their presence was not assured, as indicated by the large variation between years found over even the relatively short timespan of three years. Thus, the presence of many bird species in small woods depends not only on the size of the woods, but also on the distance to other woodland in which the species in question is present, and on the nature of the intervening landscape.

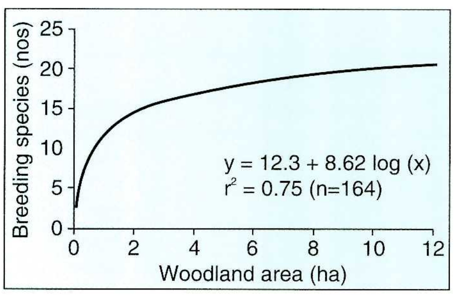


Figure 26. Relationship between the number of breeding bird species and woodland area

This study builds on previous work in which woodland area was found to be the most important factor influencing the total number of breeding species present in small woods (Figure 26).

Changes between years in species composition were found to be common, with the species most likely to disappear being those with small population sizes. Although small woods may support common woodland species in numbers large enough to survive regularly from year to year, this may not be true for the more specialised woodland birds which usually occur at lower densities. Thus, to ensure that the vulnerable small populations of these woodland 'specialists' are replaced in the event of a local extinction, woodland planting at the landscape scale should aim for a mix of small and large woods. The latter should support populations of woodland 'specialist' species large enough to avoid regular random extinction and may be able to supply a surplus of individuals to repopulate 'empty' small woods. To facilitate recolonisation, small woods and hedgerows could be used to provide cover for birds travelling between woods.

S A Hinsley, P E Bellamy and I Newton

References

Marchant, J. & Musty, L. 1992. Common birds census: 1990–91 index report. *BTO News,* no. 182, 9–12.

Rackham, O. 1986. *The history of the countryside.* London: Dent.

Robbins, C.S., Dawson, D.K. & Dowell, B.A. 1989. Habitat area requirements of breeding forest birds of the Middle Atlantic states. *Wildlife Monographs,* **103**, 1–34.

Rolstad, J. & Wegge, P. 1989. Capercaillie *Tetrao urogallus* populations and modern forestry – a case for landscape ecological studies. *Finnish Game Research,* **46**, 43–52.

Verboom, J., Schotman, A., Opdam, P. & Metz, J.A.J. 1991. European nuthatch metapopulations in a fragmented agricultural landscape. *Oikos,* **61**, 149–156.

Wilson, E.O. 1992. *The diversity of life.* London: Allen Lane.

Effects of nitrogen fertilizer application on sward composition and recovery following cessation of input

(This work was funded by the Ministry of Agriculture, Fisheries and Food, English Nature and the Department of the Environment. It was conducted in collaboration with the BBSRC Institute of Grassland and Environmental Research)

The Tadham Moor experiment

In the mid-1980s, the extent of unimproved pasture in Britain was only 3% of that present prior to World War II (Fuller 1987). Fertilizer application was known to stimulate sown grasses at the expense of indigenous species, and hence affect sward composition. However, experimental data on the effects of fertilizer on the botanical composition of semi-natural grasslands were almost absent, particularly for wet grasslands. More recently, there has been a growing interest in habitat recreation and rehabilitation, and in the effects of less intensive fertilizer use. In 1986, the Ministry of Agriculture, Fisheries and Food, the Nature Conservancy Council (now English Nature) and the Department of the Environment commissioned an experimental study of the effects of a wide range of fertilizer applications at Tadham Moor in Somerset, where exceptionally rich meadows occurred on low-lying peaty soils. The grassland at Tadham had been mown for hay for many years, with stock grazing the aftermath, but there was no history of fertilizer use. The fields supported a mosaic of grassland types, with two communities of the National Vegetation Classification particularly prominent: MG5 (Centaureo-Cynosuretum cristati) and MG8 (Senecioni-Brometum racemosi) (Rodwell 1992).

During Phase I (1986–90), a randomised block experiment (three blocks) studied the effects of five nitrogen treatments (0, 25, 50, 100 and 200 kg ha^{-1} yr^{-1}) on a variety of botanical variables. They included percentage cover of individual species, and four measures of species diversity (vegetation community variables):

- mean number of plant species per 1 m^2 quadrat (M),
- total number of species recorded in the quadrats of each subplot (S),

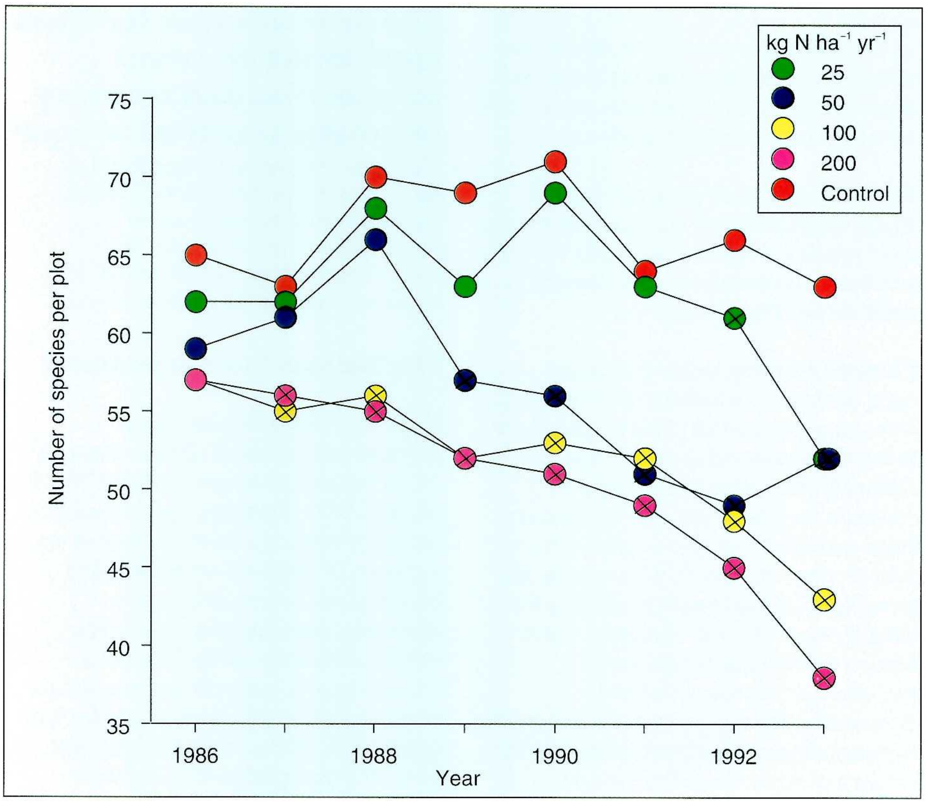


Figure 27. Effects of five treatments (0, 25, 50, 100 and 200 kg N ha^{-1} yr^{-1}) on S (total number of species per plot). Observed treatment means are joined by lines to guide the eye, and those which are significantly different from the control mean are indicated with ×

- total number of species in flower at the time of observation (S'), and
- Simpson's index of diversity (I).

All plots were cut for hay and the aftermath was grazed by steers until mid-October. In nitrogen-treated plots, replacement levels of phosphorus and potassium (determined from cut herbage) were applied. In 1990, each of the 15 plots was divided into two subplots: nitrogen treatments were continued in one of the subplots (N$^+$) but discontinued in the other (N$^-$).

Seven years of nitrogen fertilizer application (N$^+$ subplots)

Combining the Phase I data with further observations from those subplots which continued to receive nitrogen treatments enabled the assessment of the effects of nitrogen input for seven years. Fertilizer addition over this period produced a coarser sward composed mainly of a few aggressive grasses (perennial rye-grass (*Lolium perenne*), Yorkshire-fog (*Holcus lanatus*), Timothy (*Phleum pratense*) and soft-brome (*Bromus hordeaceus*) (Mountford, Lakhani & Kirkham 1993; Mountford, Lakhani & Holland 1994). Other species (including all sedges, rushes, bryophytes and the majority of forbs) were most common where no nitrogen was applied and rarest where high N levels were applied. Species diversity decreased with increased nitrogen – this effect became increasingly significant with time. By 1992 and 1993, species richness was significantly reduced even under the low rate (25 kg ha^{-1} yr^{-1}) of nitrogen application (Figure 27).

An attempt was made to predict the response of groups of species to seven years of nitrogen application. The species were grouped partly by their higher taxonomic units (grasses, forbs, bryophytes, sedges and rushes) and partly by the Ellenberg ecological indicator values for the individual species (Ellenberg 1988). In the Ellenberg system, the nitrogen value (*N*) of a species is a nine-point scale, reflecting occurrence of the species along a gradient of available nitrogen during the growing period. Species were allocated to seven groups using an *N* indicator value of 6 as a threshold to separate species thought likely to respond well to fertilizer application ($N \geq 6$) from those more typical of nitrogen-deficient soils ($N \leq 5$). All species groups behaved as predicted (other than those forbs expected to increase with N), and the trend was significant in all groups except sedges (Table 4).

Four years of fertilizer application and three years of recovery (N$^+$ subplots)

Observations from subplots where nitrogen input was discontinued showed the effects of past applications of nitrogen. Thus, even though the nitrogen input had ceased in 1990, the 1993 observations for all four measures of species diversity showed significant trends with past levels of nitrogen. However, when N$^-$ and N$^+$ subplots were directly compared, there were indications that the vegetation in the former was becoming more diverse, whilst diversity further declined where fertilizer treatment continued (Table 5). During the three years after the four-year period of fertilizer application, while grasses tended to decline and forbs generally increased, species remained affected by the previous nitrogen input. Hence, they showed either a positive or negative trend with past levels of nitrogen.

Species showing a positive linear trend with past nitrogen use

In 1986, the original sward was dominated by common bent (*Agrostis*

Table 4. Summary of 1993 ANOVA results (after seven years' application of fertilizer) for species groups. Groups defined by their Ellenberg N indicator value – a functional index of the distribution of species in relation to N apply

Species group	Sign of linear trend with nitrogen	Significance
Grasses predicted to increase ($N \geq 6$)	+	**
Grasses predicted to decrease ($N \leq 5$)	–	*
Forbs predicted to increase ($N \geq 6$)	–	NS
Forbs predicted to decrease ($N \leq 5$)	–	*
Sedges and rushes (all $N \leq 5$)	–	NS
Bryophytes (not allocated an N value)	–	***

NS not significant; * P<0.05; ** P<0.01; *** P<0.001

Table 5. Summary of 1993 ANOVA results for community variables (M, S, S' and I), showing: (i) the significance of the (negative) linear trend; and (ii) those treatment means (m25, m50, m100, m200) which are significantly different from the control mean. Results are for N^+ plots receiving the same management as 1986–90 and N^- plots receiving no fertilizer application after 1989

		M	S	S'	I
N^+	(i)	***	***	**	***
	(ii)	m50, m100, m200	m25, m50 m100, m200	m50, m100 m200	m200
N^-	(i)	**	***	*	**
	(ii)	m200	m50, m100 m200	None	None

Significance as in Table 4

capillaris), sweet vernal-grass (*Anthoxanthum odoratum*), crested dog's-tail (*Cynosurus cristatus*) and red fescue (*Festuca rubra*), but these grasses subsequently declined where fertilizer was applied. In 1993, three years after the cessation of fertilizer use, four grasses and one forb were significantly commoner in those plots which had received high levels of nitrogen up to 1990: common bent, sweet vernal-grass,Yorkshire-fog, perennial rye-grass and common sorrel (*Rumex acetosa*).

In Phase I, Yorkshire-fog and perennial rye-grass had been stimulated by N application, and were co-dominant in 1990 at high levels of N. Although the cover of these species decreased after 1990, they remained significantly commoner in the N^{200-} than in the control plots. Between 1990 and 1993, the cover of common bent and sweet vernal-grass increased, partially replacing perennial rye-grass.

Species showing a negative linear trend with past nitrogen use

In 1993, 11 species remained significantly more common in the control subplots than where fertilizer had been applied during Phase I: daisy (*Bellis perennis*), brown sedge (*Carex disticha*), glaucous sedge (*C. flacca*), hairy sedge (*C. hirta*), southern marsh-orchid (*Dactylorhiza praetermissa*), red fescue, ragged-robin (*Lychnis flos-cuculi*), creeping-jenny (*Lysimachia nummularia*), changing forget-me-not (*Myosotis discolor*), *Climacium dendroides* and *Eurhynchium praelongum*. Though fewer species showed a significant negative linear trend in 1993 than in 1990, the total number of negative trends was almost identical in 1993 and in 1990, or when the N^- subplots were compared with the N^- for 1993.

Estimation of time required for the grassland to recover

If the nitrogen input is discontinued, the vegetation in the plots which were previously subject to N_k (k=25,50,100, 200) treatments may revert to a composition similar to that in the N_0 control plots. It is reasonable to suppose that the time taken to achieve this reversion will be different for the vegetation in the plots which had received different levels of nitrogen input. If these reversion times are denoted by R_{25}, R_{50}, R_{100} and R_{200}, it is likely that $R_{25}<R_{50}<R_{100}<R_{200}$, and, by definition, $R_0=0$.

The field vegetation may be quantified via a large number of variables, and we may attempt to obtain estimates of the time to reversion for each of these vegetation variables. However, these estimates might vary substantially for the different variables. This might be particularly true for the percentage cover variables (p_i) because of what might be called the problem of possible 'multiplicity of equilibria'. The problem arises from the possibility that, if the percentage cover values $p_1,p_2,...,p_n$ of n species are modified by fertilizer treatment to $p'_1,p'_2,...,p'_n$, then upon the discontinuing of the nitrogen treatment the p'_i values may not all change to exactly the original p'_i values. The problem may be less serious if the vegetation is characterised in broad terms using derived variables based on all species, or groups of species. We used three types of variables.

- Percentage cover of *individual species* were included, despite the likelihood of possibly unsatisfactory estimates.
- *Community variables* – M, S, S' and I – represent important attributes of the entire plant community and were expected to yield meaningful estimates.
- Percentage cover of *groups of species*, chosen deliberately for the purpose of estimating R_k, were also expected to provide robust estimates. The species were grouped partly by higher taxonomic units and partly by the ecological indicator value for nitrogen (Ellenberg 1988).

To obtain the estimates of the reversion times for a particular variable, simple straight-line models were fitted separately to the N^- observations under different treatments. Thence, for a given nitrogen treatment, an estimate of the reversion time is given by noting the time point when the trends fitted to the data under that treatment and the control treatment meet. Smoothed estimates were obtained by using the inequality $R_0<R_{25}<R_{50}<R_{100}<R_{200}$, ie by plotting R_k values against the nitrogen treatment levels, and fitting a monotonic increasing model.

Figure 28 shows estimates of the reversion times based upon the community variables. Similar estimates were also obtained using percentage cover of groups of species and of individual species. They were used to obtain average estimates under the three types of variable. These three sets of average estimates of R_k values were very similar (Mountford *et al.* 1994). Overall

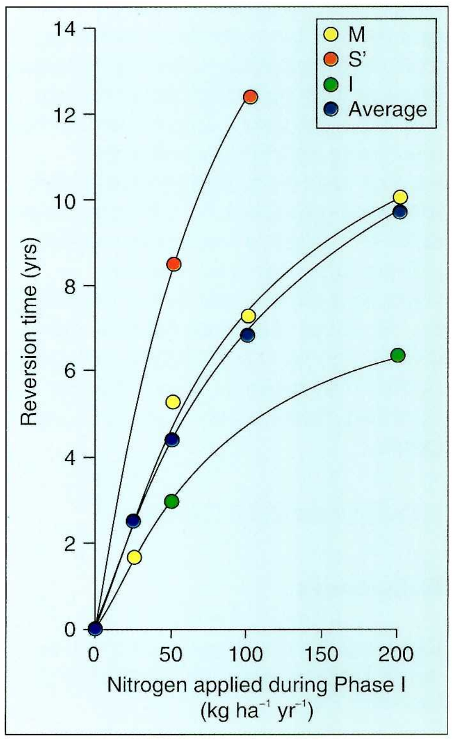


Figure 28. Composite plot showing estimates of reversion time based on community variables, M, S' and I

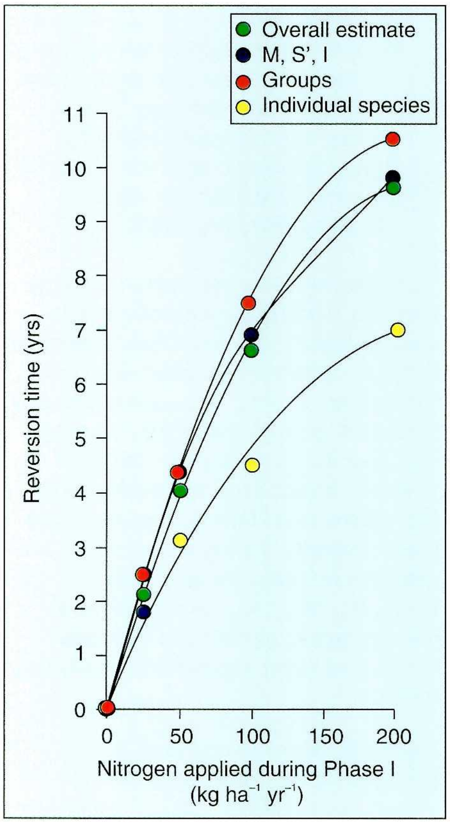


Figure 29. Composite plot showing overall estimates of reversion time based on all variables

estimates of the reversion times for the Phase I nitrogen treatments were obtained from these average estimates (Figure 29), suggesting that the working estimates for the reversion times (from 1990) corresponding to the four nitrogen treatments are approximately two, four, seven and ten years. Despite the fact that there were only three years of post-fertilizer data (requiring extrapolation to obtain the estimates), by 1993 vegetation in the N^{25-} subplots was not significantly different from that in the controls, supporting the estimate of two years for R_{25}. However, it must be borne in mind that few species were entirely eliminated by the N^+ treatments. Where this has occurred, reversion times may be much longer.

J O Mountford and K H Lakhani

References

Ellenberg, H. 1988. *Vegetation ecology of central Europe.* Cambridge: Cambridge University Press.

Fuller, R.M. 1987. The changing extent and conservation interest of lowland grasslands in England and Wales: a review of grassland surveys 1930–1984. *Biological Conservation*, **40**, 281–300.

Mountford, J.O., Lakhani, K.H. & Holland, R.J. 1994. *The effects of nitrogen on species diversity and agricultural production on the Somerset Moors, Phase II: (a) after seven years of fertilizer application; (b) after cessation of fertilizer input for three years.* (NERC contract report to the Institute of Grassland and Environmental Research.) Abbots Ripton: Institute of Terrestrial Ecology.

Mountford, J.O., Lakhani, K.H. & Kirkham, F.W. 1993. Experimental assessment of the effects of nitrogen addition under hay-cutting and aftermath grazing on the vegetation of meadows on a Somerset peat moor. *Journal of Applied Ecology*, **30**, 321–332.

Rodwell, J.S., ed. 1992. *British plant communities. Vol. 3: Grasslands and montane communities.* Cambridge: Cambridge University Press.

Forest science

Plate 9. A typical example of UK forestry. Plantations of Sitka spruce in the Borders, Dumfries, Scotland

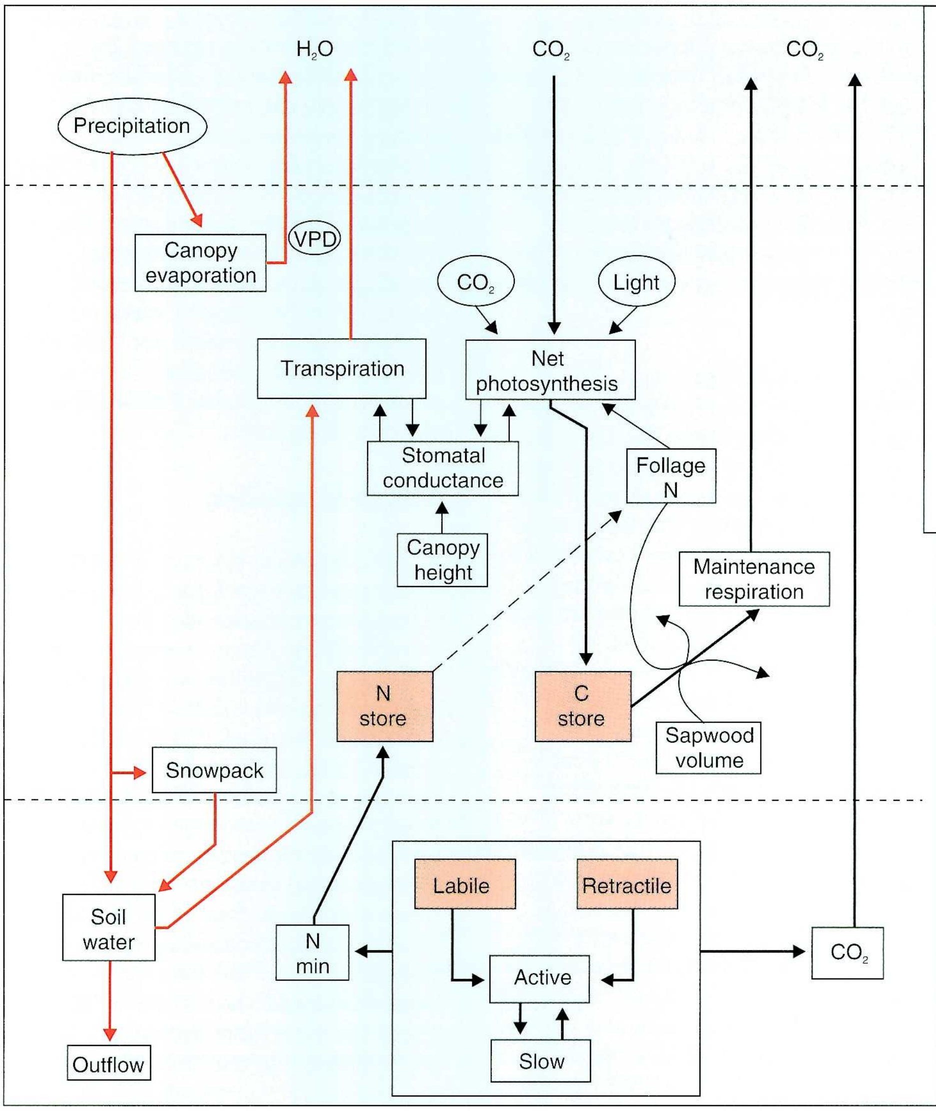
Much of the work in forest science during the year has been related to global environmental change. Work on the effects of elevated CO_2 on spruce (*Picea* spp.) at ITE Edinburgh (Plate 9), and on oak (*Quercus* spp.) at ITE Bangor has shown, in both cases, that the foliage develops a high carbon/nitrogen ratio, which has two consequences. First, the litter tends to decompose slowly, in accordance with well-known relationships between C/N ratio and decomposition rate. And, second, the foliage tends to be nutritionally poor for leaf-chewing insects, which consequently tend to grow slowly in weight; however, leaf-sucking insects, such as aphids, appear to grow normally on trees grown in elevated CO_2.

The forest models developed in ITE have been deployed to answer questions concerning the effects of climate on forest, and the effects of forests on climate. A model of forest growth and dynamics called HYBRID (Friend, Shugart & Running 1993) that combines the strengths of the JABOWA/FORET-type forest gap models (Botkin, Janak & Wallis 1972) with those of process-based models such as FOREST-BGC (Running & Coughlan 1988), and includes a photosynthesis model, PGEN (Friend 1994), has been developed and deployed to examine questions at regional and global scales. The outline structure of HYBRID is given in Figure 10. When the PGEN part of the model was incorporated into a single-column model of the atmosphere, it was able to show how a decrease in stomatal conductance at elevated CO_2 might decrease transpiration over a rain forest, decreasing cloud formation and rainfall (Friend & Cox 1994). This result contrasts with those from current general circulation models which predict a general increase in rainfall in response to global warming.

Plate 10. A typical example of dryland agroforestry. Homegarden plot at Ndiery, northern Senegal. Fruit trees are interplanted with neem shade trees

Research on forests in the tropics has continued to deliver some surprising information, as evidenced by the report on *Acacia* in Senegal (Plate 10). In Costa Rica, it has been shown that alley cropping with *Erithryna* and *Gliricidia* benefits crop yields by increasing the amount of potassium available to the crop, as well as by fixing N_2. In Cameroon, the root/shoot ratio of large trees has been found to be much larger than previously thought, which may require a revision of biomass and carbon estimates for tropical forests. Also, following work on *Eucalyptus* in India, a model has been developed to predict the interception of precipitation by trees, taking into account rainfall intensity and droplet size.

At home, the disorder of Sitka spruce (*Picea sitchensis*) known as 'bent top', observed in south Wales, has finally been explained as the result of nitrogen and phosphorus deficiency combined with defoliation by spruce aphid. Work elsewhere in Wales has shown that,



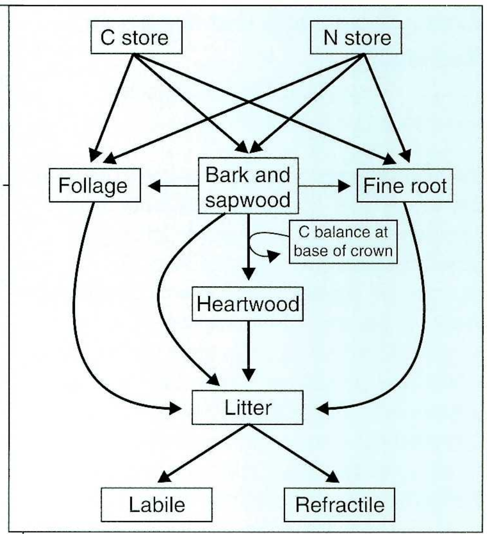


Figure 10. Structure of the HYBRID model of forest growth, showing the processes that are calculated every day and every year (Friend 1994; Friend *et al.* 1993). This model is a hybrid between 'gap' models, which simulate the growth, death and regeneration of individual trees within a plot, and process-based models that simulate the flow of carbon, nutrients and water in trees in the soil/plant/atmosphere system. This model is being used to examine the effects of forests (and other vegetation) on climate, and *vice versa*

when ammonium is added to mature spruce forest, it is retained by the site (the trees and/or soils), whereas when nitrate is added much of it is released into the groundwater. That is, mature stands are nitrate-saturated but not ammonium-saturated. However, for many mature stands, the nitrate saturation may be due to P and K limitation, because additional N is taken up when P and K are added.

Work on tree health in the UK has shown that mature forests are visibly less damaged than seedlings by sulphur pollutants, because mature foliage does not take up so much sulphur. However, sulphur deposition does cause 'hidden' injury, and a map of the UK has been produced showing where 'critical levels' of sulphur deposition may occur on forests.

It may be noted that, during the year, two international organisations were formed to promote forestry research, namely the Centre for International Forestry Research (CIFOR) in Indonesia, and the European Forest Institute (EFI) in Finland. ITE has been involved in discussions with both of these organisations, and Dr Roger Leakey was seconded from ITE Edinburgh to be the Director of Research at the International Centre for Research on Agroforestry (ICRAF) in Nairobi, Kenya.

M G R Cannell

References

Botkin, D.B., Janak, J.F. & Wallis, J.R. 1972. Some ecological consequences of a computer model of forest growth. *Journal of Ecology*, **60**, 849–872.

Friend, A.D. 1994. PGEN: an integrated model of leaf photosynthesis, transpiration and conductance. *Ecological Modelling*. In press.

Friend, A.D. & Cox, P. 1994. Modelling the effects of atmospheric CO_2 on vegetation–atmospheric interactions. *Agricultural and Forest Meteorology*. In press.

Friend, A.D., Shugart, H.H. & Running, S.W. 1993. A physiology-based gap model of forest dynamics. *Ecology*, **74**, 792–797.

Running, S.W. & Coughlan, J.C. 1988. A general model of forest ecosystem processes for regional applications. 1. Hydrological balance, canopy gas exchange and primary production processes. *Ecological Modelling*, **42**, 125–154.