Long term ecological change in British woodland (1971-2001)

A re-survey and analysis of change based on the 103 sites in
the Nature Conservancy 'Bunce 1971' woodland survey

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Main Report

ISSN 0967-876X

English Nature Research Reports
Number 653

Preface

The countryside has changed dramatically over the last 50 years in response to changing woodland management, general land-use practices and impacts such as air pollution and climate change. The effects of these changes on the richness and structure of our woodland have not always been apparent because they may be gradual or slow to show.

This report describes the results of a resurvey of 103 woods that were first looked at in 1971. The woods are spread across Britain and from the results we have been able to identify some of the ways in which woods have responded to potential drivers of change. We suggest some likely consequences of future changes. The data will be made available for further analyses.

The work was carried out largely by the Centre for Ecology and Hydrology and was funded by a consortium of government and non-government bodies.

The views expressed in this report are those of the authors and do not necessarily represent those of the organisations who funded the work.

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Acknowledgements

We thank the land owners and agents who gave permission to conduct surveys on their land. Without their cooperation this project would not have been possible. Various colleagues helped us to identify who owned which woods, or provided additional comments and material for the report. Particular thanks also go to:

- The funders - Countryside Council for Wales, Department of Environment, Food and Rural Affairs, English Nature, the Forestry Commission, Joint Nature Conservation Committee, Scottish Natural Heritage, the Woodland Trust.
- The steering group Sallie Bailley, Mark Crick, Jane Goodwin, Kate Holl, Deborah Jackson, Gordon Patterson, Simon Pryor, Mike Smith, Richard Smithers, Andy Stott, Ian Strachan, for advice and comment throughout.
- Rebecca Isted and Andy Wagstaff who ran the contract.
- The surveyors - Bob and Freda Bunce, Rachel Cowan, Lorna Edey, Henry Ford, Tim Goucher, Ruth Hadden, Veronica Howard, Miles King, Karl Krowther, Bruce Lascelles, Mike Prosser, Jude Smith, Hilary Wallace, Belinda Wheeler, Phil Wilson, Scott McG Wilson.
- Andy Scott for statistical advice and Caroline Hallam for data inputting, and other CEH staff – Dave Abel, Glen Havelock, Lindsay Maskell, Martin Rassall, Nick Ray, Diane Singleton, Claire Wood.

Executive Summary

1. The countryside and its woods have changed greatly over the last 50 years. This project explores and measures changes in woodland soils, tree and shrub layers and the ground flora through a re-survey of woods first studied 30 years ago.

2. In 1971 103 woods across Britain were selected as representative of a wider sample of 2,453 woods (> 4 ha) surveyed in the late 1960s. Within each of the 103 woods sixteen 200m^2 plots were located at random and records made of soils, tree and shrub composition, and ground flora with each plot.

3. Between 2000 and 2003 (the '2001' survey) the sites were revisited and the plots re-recorded on as close to the same point as possible. The original recording methods were used. The results from the 2001 survey were compared with the original data and changes identified at plot and site levels.

4. **Losses of plots and changes in vegetation types**

    a. Between 1971 and 2001 sixteen of the 103 sites were affected by major change in land use and thirty-eight plots (2.3%) out of a total 1,648 locations were lost, with about equal contributions from urban development and agricultural activities.

    b. The overall balance of plot types, classified by National Vegetation Classification and Countryside Vegetation System remained the same, but with some increases in plots of more fertile and more open conditions.

5. **Soil changes**

    a. Soil pH increased across sites, with a large reduction in the number with soil pH<5 and a corresponding increase in the number of sites with more alkali soil pH.
    b. The soil pH increase was more marked in organic than in mineral soil, and in non-calcareous than calcareous soils.
    c. The increase is consistent with other national trends from the Countryside Survey 2000 and the re-survey of the National Soil Inventory.
    d. There was no overall change in soil organic matter (SOM) although there were significant changes within 23 sites, of which 15 sites showed an increase.
    e. There was no increase in the mean level of plot SOM, but the number of plot with low levels of SOM (<10%)decreased.
    f. SOM increased for lowland soils and mineral and organo-mineral soils but declined in organic soils.
    g. There was a positive correlation between changes in modelled nitrogen deposition and increases in SOM.

6. **Changes to the tree and shrub species**

    a. Most tree and shrub species remained stable in terms of their frequency of occurrence at plot and site levels, although 15 species (eight of these shrubs) declined, whilst five other species (four conifers) increased.
    b. There was a net loss of stems from the smallest size classes (particularly less than 10 cm dbh) with some smaller gains in the 30-60 cm classes. Stems greater than 60 cm remained scarce.
    c. Individual species showed distinct patterns: for example
        i. oak lost stems in the lowest size classes but gained in the larger ones;

    - ii. elm and beech lost stems across the size class range;
    - iii. holly increased substantially in the smallest size class;
    - iv. hazel lost small stems.
- d. Mean basal area of trees and shrubs increased both for individual plots and across most sites.
- e. Species richness amongst saplings (25-130cm high) decreased, but small increases in frequency were shown by some shade tolerant species including yew, beech and holly.
- f. Seedling (< 25 cm high) frequency declined for most species, but holly showed a notable increase.
- g. Open habitats (rides, glades etc) and some wet habitats (ditches, boggy patches) became less common.
- h. Grazing signs increased in the lowlands; across GB the biggest increase in recorded grazing signs was for 'Other deer' (i.e. species other than red deer).

7. Ground flora changes

- a. Overall species richness per plot and per 'site' (set of 16 plots) declined markedly by 36% and 12% respectively. Considerable variation in species richness occurred between sites: increases were more common in the south and east, particularly in woods affected by the 1987 storms.
- b. The surveys in 2001, being earlier in the season, were expected to detect more species, particularly in the vernal flora. The changes in species richness, cover and frequency reported here were significant after allowing for this survey date difference.
- c. Although species frequency in 2001 was strongly correlated with that in 1971, more species declined in frequency than increased at both site and plot levels. 'Woodland specialists' were more likely than other species to show decreases in frequency.
- d. Some widespread species increased in cover, with woodland specialists again disproportionately represented among those increasing.
- e. Increases in tree basal area were associated with species richness declines; other variables relating to disturbance (1987 storm damage, grazing, open habitats) were associated with increased richness.
- f. There was an overall shift across the data set towards more shaded assemblages of plants, but no change in mean Ellenberg light score. Reductions in the numbers of open habitats recorded per plot and increases in basal area were associated with declines in Ellenberg light scores. Species increasing in cover were more likely to be those associated with semi-shaded (rather than open) conditions.
- g. There was no overall shift in species towards more fertile/eutrophic assemblages and no change in mean Ellenberg fertility score. Increasing soil pH and high levels of intensive land surrounding the wood were however associated with increases in Ellenberg fertility scores. Species increasing in cover were more likely to be associated with high (rather than low) nutrient status conditions. Changes in abundance for some species were correlated with increases in modelled nitrogen deposition.
- h. There was evidence that the vegetation response to increasing soil pH and increased fertility could be partly uncoupled by increased shading.
- i. Stress-tolerator species scores declined and were negatively associated with changes in open habitats. Competitor and Ruderal species scores did not show any overall change, but the Ruderal changes were positively correlated with changes in open habitats and negatively with basal area change.
- j. 51 species out of 332 showed a significant relationship (in all but four cases positive) between spring temperature change and change in frequency within woodland plots. Other species changed in cover in relation to increased growing season length, with both increases and decreases being found.

k. Overall cover of the ground flora was positively correlated with increasing pH and negatively correlated with increasing basal area.

Conclusions and implications

8. The broad composition and structure of the whole suite of woods was not dramatically different in 2001 from that in 1971. Some of the results indicate some recovery from past damage – for example the increase in soil pH. Other changes, most particularly the decline in woodland specialist richness, represent deterioration in the quality of the woods.

9. We found effects on species distributions and abundance correlated with climate change over the last 30 years. Given the changes that are already being observed in the phenology of species, it seems likely that effects on woodland species abundance will become even more common in the next 50 years.

10. No simple woodland management response can be made to offset the future impacts of climate change, because we cannot predict with any certainty what the impacts will be in terms of species responses. However, in general larger populations (or meta-populations) are more likely to survive and spread than small ones. This supports the case for an increase in woodland area and for improving connectivity within the landscape, but also for addressing other causes of species decline that do have a clear management solution.

11. Signs of eutrophication in the ground flora were detected that were correlated with models of diffuse pollution and the management of adjacent land. Increased nitrogen inputs (and also spray drift effects) at the edges of woods from adjacent agriculture can be addressed in part through developing buffer strips next to woodland and by developing dense vegetation at the wood edge: in effect establishing a scrub/grassland ‘ecotone’.

12. The significance of diffuse pollution impacts for woodland species is becoming more widely appreciated. Impacts may be reduced by maintaining high shade levels – in effect ensuring that light levels rather than nutrients act as the limiting factor.

13. Given the generally young nature of most of the stands, then without deliberate management intervention broadleaved woods are, on average, likely to become older and darker in the next twenty years. This could benefit some species and communities – those of fallen dead wood and shade-loving conditions - but may lead to continuing decline in much of the ground flora and also other groups associated with open space and young growth.

14. Opening out the wood temporarily may increase the abundance of some species, which is desirable if existing woods are to act as sources for the colonisation of new woodland. In undertaking such management we must be aware of the increasing potential of interactions with and between other drivers (climate change, nitrogen deposition, deer grazing) to influence the outcome.

15. All long-term woodland studies gain in value with time: equally we are conscious that some of our analyses would be more useful if there were not such a long gap between the survey times. The data will be made available for further analyses. Consideration will be given to linking at least some of the sites surveyed into other woodland surveillance programmes.

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1. Introduction

1.1 Background to the 1971 – 2001 survey project

The countryside and its woods have changed greatly over the last 50 years and will continue to do so during the next half century. Farming and forestry practices have altered in response to socio-economic demands. There have been gradual responses in the composition and structure of woods as those felled in the 1940s have grown up again; species may have increased or decreased as a consequence of climate change and atmospheric pollution. Other woods have changed quite suddenly, for example following the Dutch elm disease outbreak of the late sixties and seventies or the 1987 storm in south-east England.

We cannot prepare for future changes unless we understand how woods have altered in the past. We need to be able to separate the effects of long-term cyclical processes from directional change, particularly because the latter may be of lower amplitude during the period being considered.

Change can be considered at three levels: alterations in the overall distribution and composition of woodland; changes in the composition of populations of woods in relation to particular major drivers (internal dynamics, climate change, etc); and the development of individual woods and stands.

Broad-scale surveys such as the Countryside Survey (Haines-Young and others 2000) and the National Inventory of Woodland and Trees (Forestry Commission 2003) are well documented. Woodland cover has increased over the last 50 years; there was much new conifer planting in the first half of this period, but more recently many new broadleaved woods have been created; stands of conifers on ancient woodland sites are being restored to native trees and shrubs (Goldberg 2003; Thompson and others 2003); and some recent conifer stands are being converted back to heath or bog.

At the other end of the monitoring spectrum are studies such as those of Peterken and Jones (1987) or Mountford (2004) who have documented the detailed changes in the tree and shrub populations of particular stands, illustrating responses to disease, storms, drought, competitive exclusion and grazing events (Table 1.1). Long-term studies of other parts of the woodland system are less well-documented in Britain, but include Barkham's (1992) work at Brigsteer Wood, Latham and Blackstock's study (1997) of a Welsh Alderwood, Ling's (2003) work on the Cotswold beechwoods, and Kirby and Thomas's (2000) work at Wytham Woods, Oxfordshire. The effects of long-term trends (deer and sheep grazing, air pollution) have been identified as well as responses to gradual or sudden changes in the nature of the tree and shrub layer.

However there is a dearth of long-term studies in between these two scales, that allow us to identify the relative importance of different factors across a range of woods. For example,

- have impacts from air pollution been more important than grazing over recent years;
- is climate change starting to have an effect on woodland species abundance nationally;
- to what extent do changes in the tree and shrub layers or in soil parameters determine changes in the ground flora;

- to what extent are the different elements of the woodland system responding independently of each other?

This report helps to fill this gap. It describes a re-survey of 103 woods spread across Britain and compares their current state with that in 1971. Standardised methods of describing the trees, shrubs, ground flora, soils and general habitats present were used (Bunce and Shaw 1973) that were then repeated in 1999-2003 (for convenience referred to subsequently as the '2001 survey'). The sample of 1648 plots spread through 103 woodland sites located across Britain makes it probably the most extensive quantitative ecological woodland survey undertaken in Britain; it is also notable for the range of sites that have been re-visited after such a long interval. The surveys provide a unique opportunity to explore the effects of a range of potential drivers of woodland change that operated between 1971 and 2001.

Table 1.1 Some suggested drivers of change identified in other long-term woodland studies in GB (examples only)

Potential drivers of change	
Phenological change	Sparks and Collinson 2003; Sparks and Smithers 2002.
1976 drought and other drought years	Peterken and Mountford 1998a; Ratcliffe 1984.
1987 storm and other wind-events	Kirby and Buckley 1994; Mountford 2004; Whitbread 1991.
Warmer winters	Hulme and Jenkins 1998.
NOx and ammonia deposition	Ackermann and Bobbink 2003; Ling 2003; NEGTAP 2001; Smart and others 2003, 2005.
Reduced acidification	NEGTAP 2001.
Dutch Elm disease	Kirby and Thomas 2001; Peterken and Mountford 1998b.
Changes in grazing by deer and sheep	Kirby and Thomas 2000; Latham and Blackstock 1997
Management for game	Firbank 1999.
Increased damage by grey squirrels	Mountford 2004
Inappropriate woodland management	Kirby and others 1998; Smart and others 2001
Stand development	Mountford 2004; Peterken and Jones 1987,1989.

This report covers the following:

- the background to the original survey, including how the sites were selected and surveyed;
- a summary of the main variation present in the 1971 data;
- a comparison of the results from the 2001 survey with those from 1971, looking particularly at the possible main drivers of change.

However further exploration of the data would certainly yield other significant results. Copies of some of the data are presented as appendices to this report. However we also aim to make the full dataset available under licence for future research purposes. Further lines of enquiry are suggested in the final sections of the report.

1.2 Derivation of the original 103 woods – the ‘Steele’ survey

In the late sixties the then head of the Nature Conservancy woodland section, R.C. Steele, organised a national series of woodland records as part of the data collection that provided background information for the *Nature Conservation Review* (Ratcliffe 1977).

> “The purpose of the survey is to obtain a record of the present day extent, locations and characteristics of woodlands of native tree species in Britain. To be of most use the survey must be completed quickly, say within the next three years, and to make this possible the information recorded must be kept as simple as possible. It is also for consideration whether the survey should be limited to woodlands of over (say) 10 acres in area. It is proposed that the survey be a joint operation, carried out by the County Naturalist’ Trusts and the Nature Conservancy, using such information and help as is available from the Forestry Commission and other organisations. It is intended to make use of the preliminary surveys and the more detailed studies [by the Nature Conservancy] to produce accounts of woodlands on a County or Regional basis or to describe a particular woodland type in Britain.” (Steele 1968).

A standard record card with instructions for how it was to be used were circulated (Appendix 6). Figure 1.1 illustrates the distribution of completed cards as at 31 December 1971. More returns were received from south-west England, South Wales and the Lake District than East Anglia and the south-east (although some of the gaps are partly explained by lack of woodland, eg the Fens and London). The west coast of Scotland was also well-represented. A total of 2,453 woods were eventually surveyed.

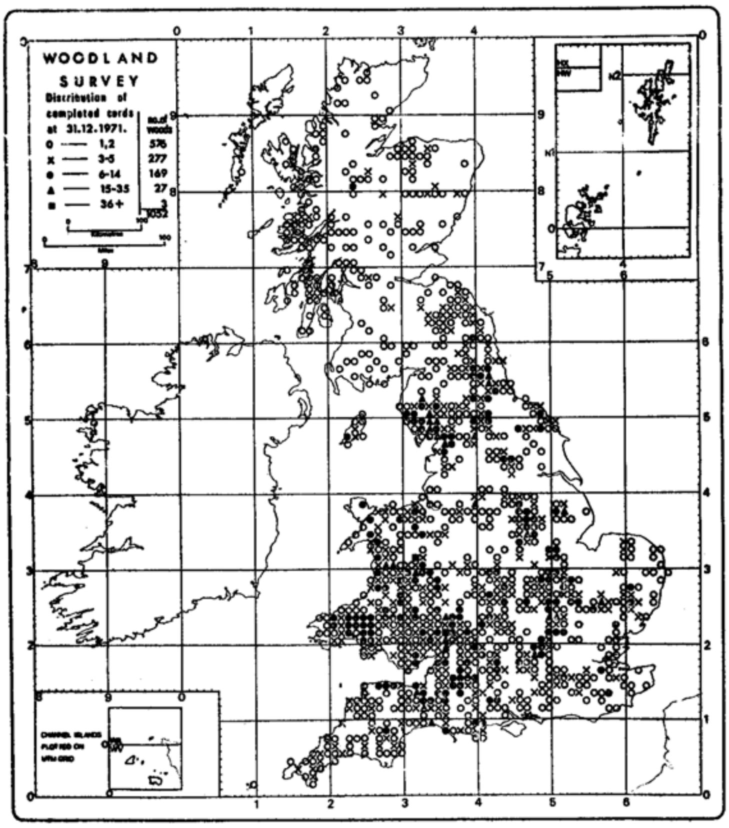
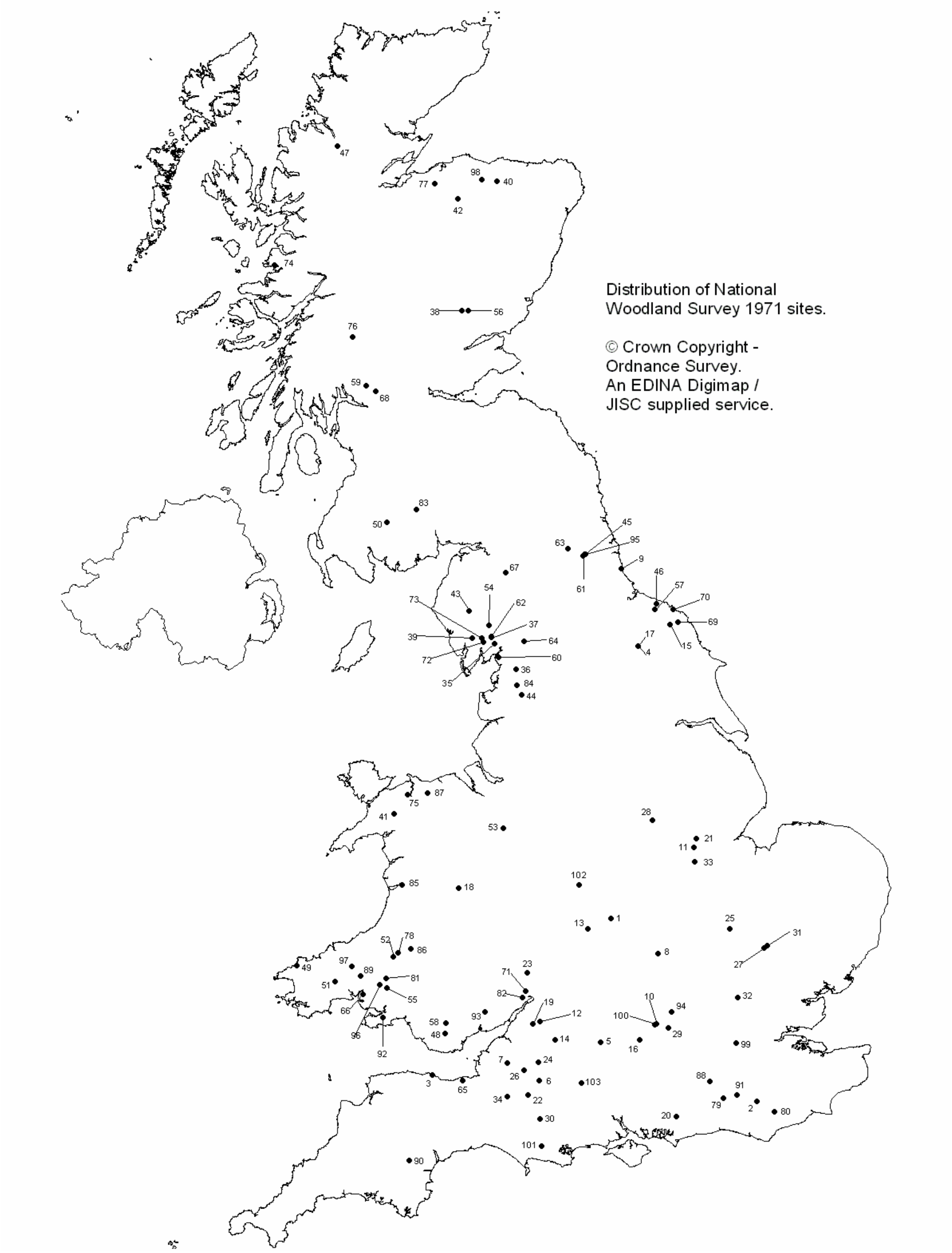


Figure 1.1 The distribution of records from the National Woodland Survey as at 31 December 1971

The original cards have been transferred to the Biological Record Centre at Monks Wood (Cambridgeshire) but a copy set is held by English Nature at Northminster House.

The woods used in the more detailed 1971 survey were a sub-set derived from the 2453 by Association Analysis (Williams and Lambert 1969) and other numerical techniques that, at the time, were still novel and undergoing rapid development (Bunce and Shaw 1973; Hill and others 1975; Bunce and Jeffers 1979). These analyses put the woods into 103 groups according to the similarity of their plant species composition. The wood that was nearest to the centre of each group in terms of the ordination space, ie most typical of that group, was then selected for detailed survey (Figure 1.2).



Distribution of National
Woodland Survey 1971 sites.

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Figure 1.2 Woods surveyed in 1971 and 2001

Surveys using similar procedures were also carried out in the native pinewoods defined by Steven and Carlisle (1959). Two of these sites were re-recorded in 2000 to test the feasibility of repeating that survey (Hill and others 1975; Smart and others 2001), but those results are not considered further in this report.

There is potential for bias in the selection because of the different intensities of recording across the country in the Steele Survey (Figure 1.1) – for example the lack of samples from East Anglia versus the high numbers of samples from south Wales and the Lake District. However even though some geographic areas may appear to be under-represented this does not mean that that woodland type was not represented in the derived sample (Figure 1.2) because a wood of that type somewhere else may have been selected as ‘more typical’, through the cluster analysis. Further discussion on the degree to which the sample reflects broadleaved woodland across the country is given in a later section.

Table 1.2 Relationships between woodland surveys from 1968 to the present.

1968	The 'Steele' Nature Conservancy 2453 woods	
	Classification of 2453 woods and selection of 103 representative type sites	
1971	'Bunce' woodland survey	}
1998	Partial re-survey of 12 sites in North-west England (Bunce and Hurst 2000)	ANALYSIS OF CHANGE
2000	Pilot re-survey of 14 sites plus two from the 1971 Native Pinewood Survey of Scotland (Smart and others 2001).	}
2002-3	Re-survey of the remaining 89 sites	
2002-3	Quality Assurance survey of eight plots on each of six sites	

2. Methods

2.1 Field survey

The 1971 survey of 103 sites was planned by Bob Bunce and Wally Shaw (Bunce and Shaw 1972; Hill and others 1975; Bunce and Jeffers 1979). Fourteen sites were visited in 2000 as part of a pilot survey to assess the logistical and analytical implications of trying to carry out a re-survey (Smart and others 2001). No surveys were carried out in 2001 because of the Foot and Mouth outbreak, but 56 sites were surveyed in summer 2002 and the remainder in 2003. Site names, grid references and current designation status (SSSI and Ancient Woodland) are given in Appendix 1.

The pilot survey in 2000 was carried out by Bob Bunce (who also did much of the 1971 fieldwork) with help from Simon Smart, Freda Bunce, Nick Ray and Rebecca Isted. The surveys in 2002 and 2003 were carried out by consultant ecologists (see Acknowledgements). Prior to each survey a two-day training course was held at CEH Merlewood to go through the detailed field protocols (Appendix 2).

The methods used in 1971 are described in detail in Appendix 2 (Bunce and Shaw 1973). Sixteen plots were randomly positioned within each site in 1971 and the location of each marked on a 1:25000 map. Each plot was 14.1 x 14.1 m (200m^2) (Figure 2.1). In the field, plots were located by pacing from the nearest relocatable feature. Data were then collected on ground flora, tree and shrub layers, soils and habitat characteristics for the plot (Table 2.1). A habitat sheet for the whole wood was also compiled.

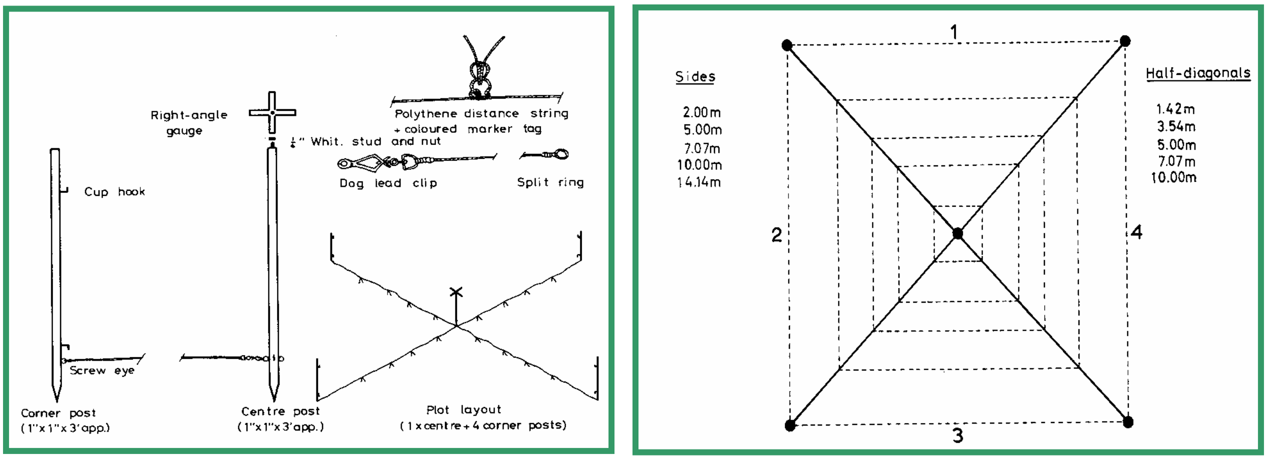


Figure 2.1 Quadrat design and layout for the 1971 woodland survey.

Table 2.1 Summary of the data collected (see Appendix 2 for more detail).

	1971 survey	**2001 survey**
Ground flora	Species present in the plot % cover/ abundance estimates Bryophyte collection	As 1971 except that only most common bryophytes recorded. (Bryophytes were excluded from most analyses.)
Trees	DBH (diameter at breast height) and species recorded from all four quadrants	As 1971
Shrubs and saplings	DBH and species recorded from diagonally opposite quarters	As 1971
Seedlings	Included with the ground flora records	As 1971
Plot description and habitats	Tick card features (broad categories) Tree management Regeneration Dead trees Epiphytes Rock habitats Aquatic habitats Open habitats Human elements Vegetation structure Animal signs	As 1971
Soil data	Tick card description from small pit and augur boring in the centre of the plot Composite soil sample from top 10-15 cm.	Composite soil sample from top 10-15 cm.
Whole wood description	Tick card features (broad categories) As for plot, plus adjacent land use and boundary type	As 1971, plus surveyors were asked to make a summary report for the whole site.

In the 2001 survey the original maps were used to find the same position as accurately as possible. Processing and analysis of the 1648 plot records taken in 1971 and 2001 assumed that the records could be treated as paired data (ie relocation error was not significant). The advantage of paired data is that derived variables, such as species richness, can be reduced to differences for purposes of statistical testing. The total variation across time and sites will be less than if two completely random samples were collected in each year and the power of tests is thereby increased. Some relocation error was however inevitable given the limited information available and potential risks in this approach are discussed in section 2.5.

2.2 Soil sample processing and analyses

Soil samples were taken from every accessible plot in every woodland, using the same protocols as used during the 1971 woodland survey (Appendix 7). A single soil sample was taken from each plot, at the centre of the vegetation quadrat, using a trowel. Samples (*c*1 kg) were taken to a depth of 15 cm and placed in a labelled plastic bag. On return to CEH Merlewood, all soil samples were stored at 4^{o}C prior to processing and analyses.

Soil samples from the 2001 survey were sieved using a 2 mm automatic sieving machine. A pH reading was taken on a representative sub-sample from each soil sample before air-drying at 20^{o}C after which another pH reading was taken using another representative sub-sample. The air-dried pH measurement can be used to further validate methods in any future analyses of change (see section 2.3). Another sub-sample was then taken to determine loss-on-ignition, as a measure of soil organic matter content.

All analyses were carried out under the supervision of the Environmental Chemistry Section, CEH Merlewood, following standard methodologies (eg Allen 1974) and quality control procedures, including the analyses of certified standard reference samples within batches.

Analytical methods were the same as those subsequently used for Countryside Survey 2000. Protocols for processing the soils in 2001 (i.e. handling and storage of samples between collection and analysis) were adapted from those used in CS2000. Completed analyses were entered into Microsoft Excel and validated before entering into the project database in Microsoft Access. Unless otherwise stated, soil pH values reported are from the soil sample ('fresh') prior to air-drying.

The original soil samples from 1971, stored as air-dried samples in bags under dry conditions, were retrieved from long-term storage at CEH Merlewood. Each sample was transferred into a new air-tight labelled container. All remaining soil samples from the original 1971 survey and the 2001 survey are also now stored in labelled, air-tight plastic pots at CEH Lancaster.

2.3 Validation of soil analytical methods

During the 2001 survey, we used the same soil analysis protocols as in the 1971 survey but the equipment was different. Changes in analytical precision since 1971, due to modifications in technical equipment , could influence the significance of the results obtained from both pH and loss-on-ignition (LOI). Therefore repeat analyses of LOI on the 1971 samples and comparisons between fresh and air-dry soil samples from 1971 and 2001 were used to check the comparability of analytical methods between the two surveys. If analytical precision had improved significantly over the last 29 years then this should be reflected in a significant variation in the difference between fresh (1971) and air dry pH values (1971samples measured in 2001) as compared to the difference between fresh and air-dried samples from the 2001 samples.

A representative number, *c* 20%, of soil samples from 1971 were analysed for pH and LOI using the same procedures and equipment as for the 2001 survey. These results are presented in Tables 2.2 and 2.3 along with results from the same soil sample in 1971 and for the 2001 survey soil samples from the same location. The difference between the fresh and air-dried soil samples are similar for both means and medians, with equivalent standard deviations (Table 2.2). A t-test for dependent samples did not detect any significant difference between 1971 and the 2001 survey air-dried and fresh samples (Table 2.3). We conclude that there is no detectable effect of changes in methods over the intervening years.

Table 2.2 Summary statistics for pH from fresh and air-dried soil from the same samples taken in 1971 and 2001.

Survey	Sample	analysed	N	Mean	Median	Min.	Max.	S.D.	s.e.
1971	fresh soil	1971	306	5.03	4.80	3.40	7.90	1.118	0.064
	air-dried soil	2001	306	4.78	4.44	3.01	8.35	1.199	0.069
2001	fresh soil	2001	304	5.30	4.96	3.60	8.80	1.186	0.068
	air-dried soil	2001	306	5.09	4.70	3.16	8.60	1.140	0.065

Table 2.3 Results from t-test of differences in pH values from fresh and air-dried samples from 1971 and the 2001.

Difference between fresh and air-dried pH	Mean	s.d.	N	d.f	P
1971	-0.245	0.461			
2001	-0.192	0.374	304	303	0.125

Loss-on-ignition (LOI) was carried out on all soil samples in 1971. Therefore re-analyses of these soil samples gives a direct comparison of the two methodologies. There was no significant deviation of LOI from the re-analyses with the original data (Figure 2.2). There was no significant difference between these data, from a t-test for dependent samples ($p>0.05$). Three re-analysed soil samples (circled) were much lower in LOI than the 1971 values, and than the LOI values obtained in 2001, which were similar to the original 1971 values. The most likely explanation is that these three values were obtained from mis-labelled soil samples.

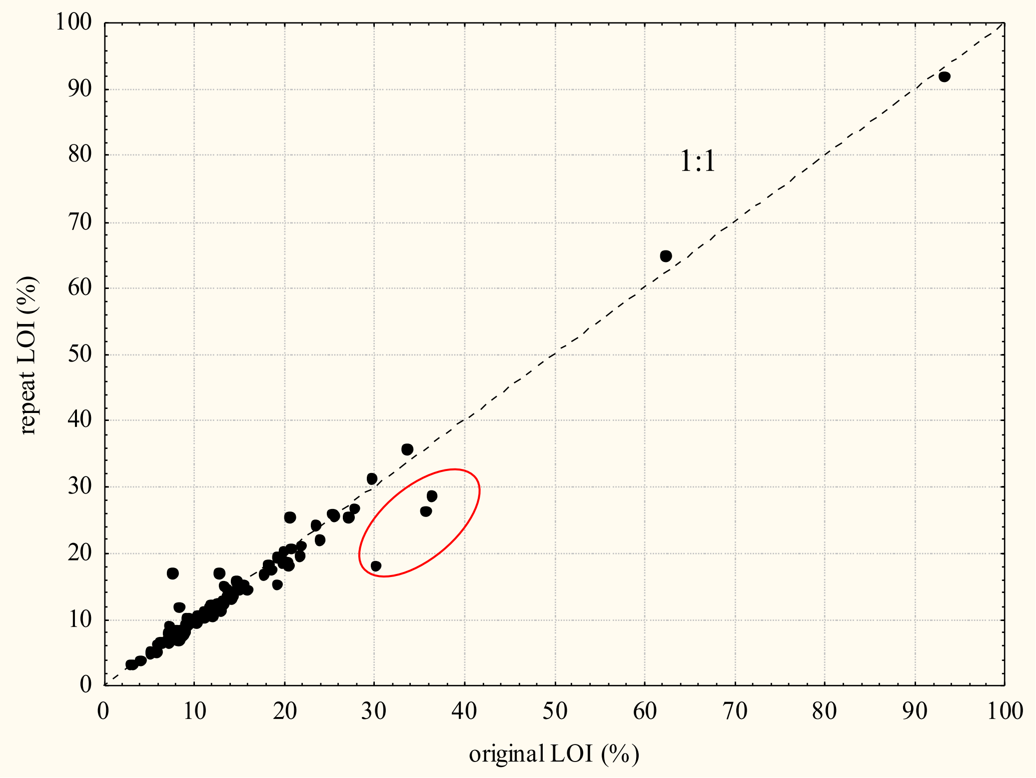


Figure 2.2 Scatterplot of loss-on-ignition values from re-analysis in 2001 of 1971 soil samples plotted against original 1971 values.

2.4 Vegetation and habitat data processing

The ground flora component of the 1971 survey was already available on an Access database. However, other 1971 datasets had to be transferred from the original field sheets to spreadsheets. This was carried out by a consultant who had previously been sub-contracted by CEH to input Countryside Survey 2000 plot data. The newly entered 1971 data were double-punched and then corrected to produce a final validated copy. Transfer of the data from the soil analyses to spreadsheets was done by staff of the Soil Ecology and Environmental Chemistry Sections and validated using standard protocols.

Field recorders in the 2001 survey were asked to ensure that all data were corrected and validated prior to transfer in electronic form to CEH Merlewood. Some errors were expected in the final data-set, just as there may have been some errors in the location and recording of the plots on the site. Various comparisons (see next section) were made therefore to consider possible systematic biases in the data.

2.5 Dealing with potential bias (systematic errors)

Bryophytes and lichens were collected separately and specimens identified later in 1971; in the 2001 survey only a limited list of common bryophytes was recorded. To achieve consistency, all bryophytes and lichens were excluded from analyses of change.

Some species were recorded in 1971 as amalgamated taxa reflecting difficulties in their consistent separation, for example *Quercus robur* and *Q. petraea*. In the analyses amalgamated taxon codes were applied across the whole data-set where necessary, in order to remove the effect of recorders separating out such species to differing degrees. Other standard validation checks included plot and site counts to ensure no duplicate numbering and hence double counting of plots.

As another part of the quality assurance process six sites were visited by a different set of surveyors and eight plots at each site recorded within two weeks of the main survey. A mixed model ANOVA showed no overall difference in species richness between the different surveyors.

Some plot relocation error was inevitable given the limited information available and the nature of the original maps. A method was developed to measure whether the 2001 record for a plot was more similar to the record for that plot in 1971 than another (randomly chosen) position from 1971. This approach was tested on the 14 pilot resurvey sites (Appendix 3). All the sites showed higher similarity between plots as a result of the search for the 1971 plot location, and for nine sites there was significantly higher similarity.

The same analysis was carried out for all the remaining sites. Overall at 97 sites (out of 103) mean similarity was greater between ‘relocated’ plot pairs compared to random-pair comparison; for 59 sites the difference was significantly greater. The results have therefore been improved through the identification of the original plot locations. There is still a need for caution in interpreting the explanatory power of plot level variables because of the possible confounding of plot relocation error and change over time. Small differences between years in plot location, for example, from an open patch to a more shaded patch could result in lower species richness and higher woody basal area being recorded for that plot. However, given the size of the data-set, individual plot errors due to this factor are likely to be balanced out over the whole sample.

There were some marked differences in the date of surveys between 1971 and the 2001 surveys with most sites being recorded earlier in the year in 2001 (Figure 2.3). This is likely to influence the recorded presence or abundance of vernal species in particular, with more species generally detectable in the late April-July period (Sykes and Horrill 1979, 1983; Kirby and others 1986) than much later surveys. More species records would therefore be expected from the 2001 surveys. However this might be off-set to some extent by the effect of earlier springs as a consequence of climate change (Sparks and Smithers 2002; Sparks and Collinson 2003) - eg a July survey in 2001 might be equivalent to an August survey in 1971. To allow for the effects of survey season the day-difference between each survey was always fitted first before exploring other explanatory variables in the analyses.

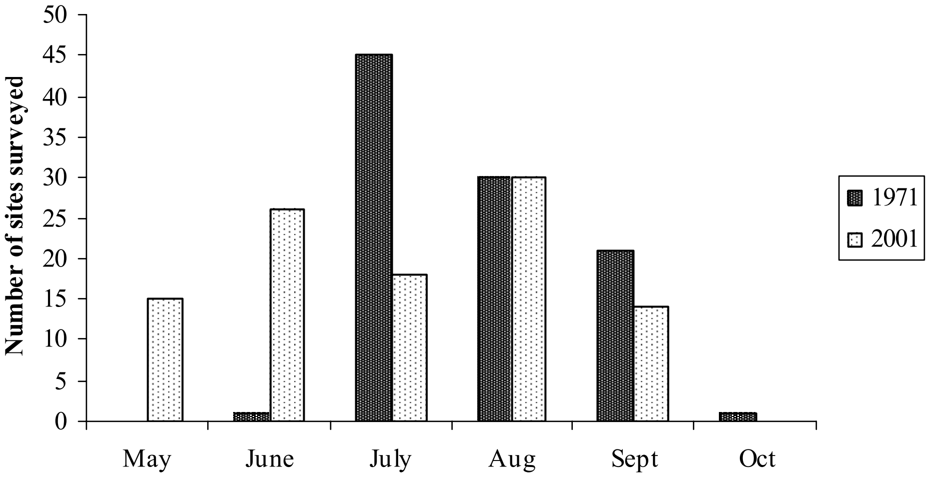


Figure 2.3 Survey dates in 1971 and 2001. In general the 1971 survey was later, as shown by the skew in the number of observations toward the right. This was therefore allowed for in the analysis.

2.6 How representative is the sample?

The woods surveyed range from 4 to 100 ha with a single outlier of 312 ha (Glen Beasdale Wood, Scotland). The mean size of the sample was 31.8ha and the median 20.4ha (Figure 2.4).The lower size cut off was determined by the minimum size (10 acres/ 4 ha) used in the original 'Steele' survey (paragraph 1.2.1). The results therefore cannot be assumed to represent what has happened in smaller woods: 48% of recorded ancient semi-natural woods in England and Wales are less than 5ha and there are many more woods less than 2 ha. However the survey is more representative of the broadleaved/semi-natural woodland resource if this is considered by total extent, rather than by the number of sites – woods in the lower size classes contribute proportionately much less to the overall area.

The 1971 sites do provide a good representation of the geographic spread of woodland cover and the range of broadleaved/semi-natural woodland types.

The number of sites recorded in the 1971 survey from each of the 32 ITE Land Classes in GB was compared with the mean percentage area of broadleaved woodland, estimated from Countryside Survey data, for each Land Class (Figure 2.5) (Bunce and others 1996). The results were also summed for each of the four major landscape types (Table 2.4). These comparisons show a good correspondence between woodland area and number of sites recorded with more woods from Land Classes or Types with a high broadleaved woodland cover.

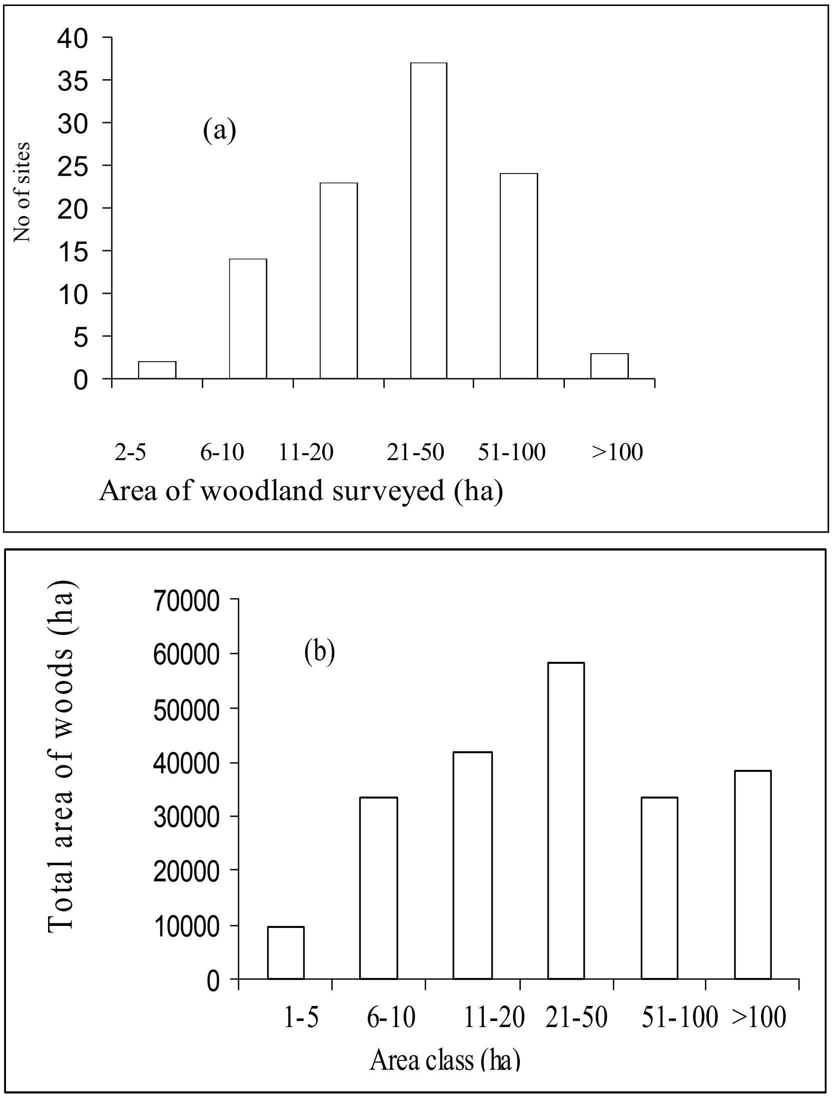


Figure 2.4 Histogram of woodland site area in 2001 (a) no of woods in this survey, (b) area of semi-natural woodland by size class in England and Wales (from Spencer and Kirby 1992).

Table 2.4 Comparisons of numbers of sites surveyed (1971) and broadleaved woodland area Countryside Survey 2000 data) for four major landscape types.

Landscape type	Lowland	Marginal upland	Pastural	Upland
Number of woods surveyed	37	14	46	3
Broadleaved woodland area (ha)	38,000	19,000	36,000	7,000

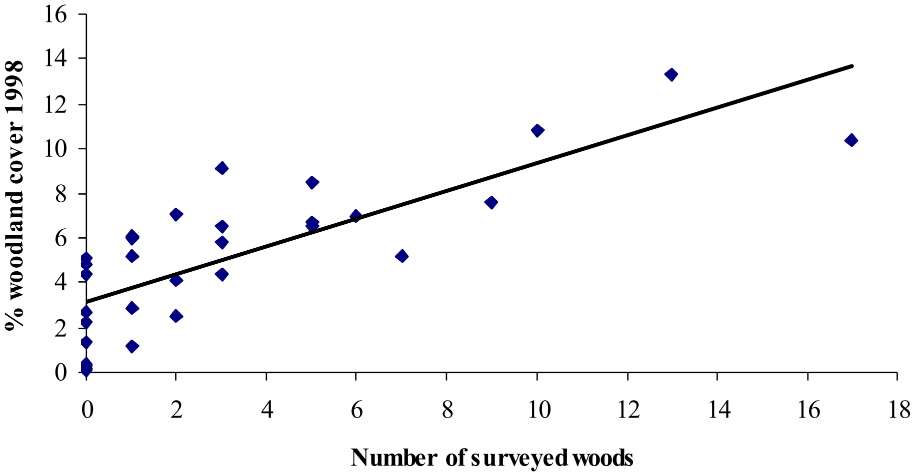


Figure 2.5 Number of woods surveyed in this project versus the percentage cover of broadleaved woodland estimated for each ITE Land Class in 1998 as part of the Countryside Survey (Haines-Young and others 2000)

The plot records from 1971 were assigned to National Vegetation Classification (NVC) types (Rodwell 1991) using standard computer programmes (MAVIS Modular Analysis of Vegetation Information www.ceh.ac.uk/products/software/land.html). The NVC types were grouped because the analysis did not include the bryophyte data and therefore the programme could not separate some types, particularly W16 and W17. The numbers of plots allocated to each group from the 1971 survey were compared with the numbers of plots included in the surveys that produced the NVC (Rodwell 1991) and rough estimates of the total area of different NVC types in ancient semi-natural woodland across Britain made by Cooke and Kirby (1994). The 1971 survey data span the broad range of types in roughly the proportions that might be expected from the Cooke and Kirby data (Table 2.5). The main difference between both of these and the Rodwell (1991) data is the much lower representation of mesotrophic oakwoods in the latter. In developing the National Vegetation Classification relatively more samples were recorded from the richer, more variable, base-rich woods because these showed a greater number of different 'homogenous' stands: the NVC was not designed to represent all types in proportion to their overall extent.

A second comparison was made with the sample of woody vegetation recorded in the Countryside Surveys of GB (Haines-Young and others 2000). The 1971 plots were grouped by Countryside Vegetation System classes (Appendix 11) (Bunce and others 1999) and their frequency was compared to the estimated national area of each class (Figure 2.6). The two data sets are generally well-correlated. The largest deviations from the line of best fit are also consistent with the differences in sampling strategy between the surveys. CVS classes that tended to have more plots than suggested by their national area (CVS 35 and 46) correspond with NVC types W8 and W7. These are community types that appear to be well-represented in large broadleaved woods, but large woods (being relatively infrequent) are not well-sampled in the Countryside Survey approach (Smart and others, in press). The CVS woodland classes that appear under-represented in the 1971 survey versus their estimated national area included mature conifer plantation and dense bracken (CVS 64) which were not the focus of the 1971 survey.

Table 2.5 Comparison of woodland NVC types identified in 1971 survey data with other estimates across the country. (Percentage values are given in brackets.)

NVC grouping	Number of NVC samples (Rodwell 1991)	1971 records	Area estimates from Cooke and Kirby 1994
Mesotrophic oakwoods (W10,11)	518 (27)	684 (53)	128500 (42)
Ash-elm woods (W8,9)	546 (28)	238 (18)	88500 (29)
Acidic oakwoods (W16,17)	186 (9)	149 (12)	41500 (14)
Alderwoods (W5-7)	267 (14)	76 (6)	17000 (6)
Birch –willow woods (W1-4)	172 (9)	23 (2)	14500 (5)
Calcareous beech-yew woods (W2,13)	131 (7)	49 (4)	7000 (2)
Acidic beechwoods (W14,15)	108 (6)	61 (5)	5000 (2)

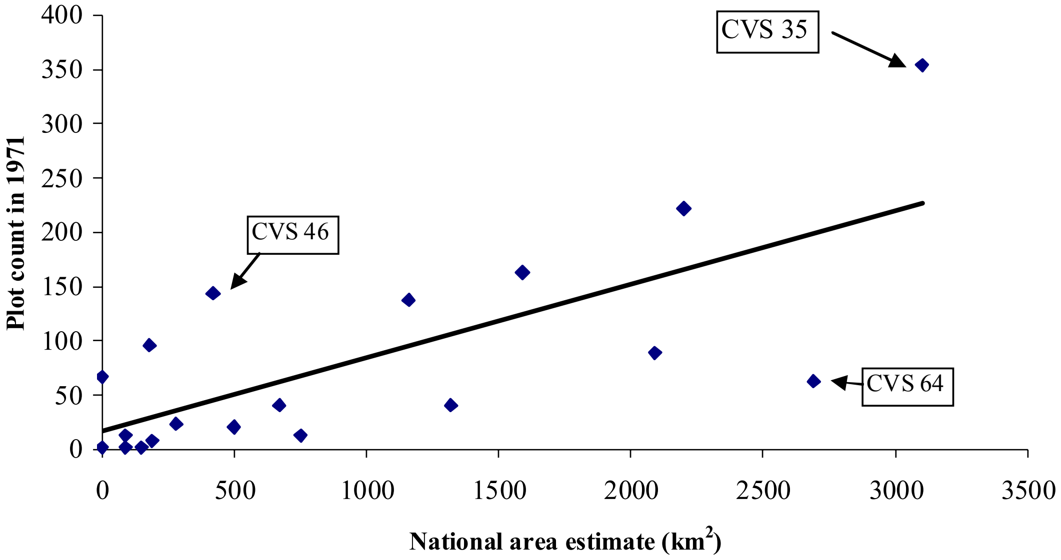


Figure 2.6 Count of woodland survey plots in 1971 by CVS class versus the estimated national area in 1990 of each CVS class (Bunce and others 1999).

The concept of ancient woodland (sites continuously wooded since 1600) (Peterken 1977; Spencer and Kirby 1992; Roberts and others 1992) had not been developed when the 1971 surveys were carried out. Both ancient and recent sites were covered, with 66 of the 103 woods being ancient at least in part. Nationally approximately 45% of the broadleaved woodland area is estimated to be ancient (based on a combination of data from the ancient woodland inventories and the Forestry Commission's National Inventory of Woodland and Trees – K J Kirby unpublished data).

2.7 Approaches to vegetation analysis

Variables analysed included soil pH, Soil Organic Matter content (SOM), change in basal area of woody species and change in the species richness of regenerating stems (>25cm in height, but less than 1.3m), species richness of higher plants in the ground layer and change in cover of potential field layer dominants.

In a separate project Corney and others (2004) used multivariate analysis to explore the variation within the 1971 data. In the study reported here ordination (Detrended Correspondence Analysis (DCA) (Hill and others 1979)) was also used on the combined (1971 and 2001) data to identify the major spatial and temporal ecological gradients in the vegetation. Analysis of a DCA bi-plot allows plant community structure to express itself without constraining the data to track predefined ecological gradients. Ordinations were carried out separately for ground flora records only. Common species of bryophytes and lichens that were thought to have been reliably recorded in both surveys were included in the ordination. In all other analyses they were omitted.

Ordination axes were interpreted by inspecting the scatter of influential species in the resulting bi-plots and by correlation with mean unweighted Ellenberg scores (Box 2.1a) for light, fertility, moisture and soil reaction added passively to the analysis. Change along ordination axes over time was assessed by plotting means and standard errors of axis sample scores in each survey. The use of weighted Ellenberg values was explored but the results were little different (Appendix 8). Weighting by percentage cover adds further assumptions and levels of complexity, and may not always improve the interpretation of the results (Kafer and Witte 2004); therefore we have used unweighted values in the main report.

2.7.1 Grouping of species by indicator values

The multivariate approach was complemented by analysis of change using pre-defined indicator variables. Univariate tests were used to quantify change over time in the floristic, soil, tree and shrub basal area and management data at both site and plot level. Plant species lists for each plot were converted into a series of summary indicator variables designed to convey implicit shifts along key environmental gradients (Table 2.6, 2.7). These included mean Ellenberg scores for each plot based on values recalibrated for the British situation (Hill and others 1999; Hill and Carey 1997) and mean values for the Unit of Comparative Plant Ecology's three established strategies of plant growth; Competitors, Stress-tolerators and Ruderals (Grime and others 1995) (Box 2.1b). Species were also classified according to their biogeographic affinities (Box 2.1c) as part of the exploration of possible climate change effects; and into 'woodland specialists' versus other species (Box 2.1d), because changes in woodland specialist species may be of greater concern from a nature conservation viewpoint.

Summarising multivariate data in this way both reduces variation, leading to more sensitive tests, and builds in an assumed link to proximal causes of change (Smart 2000). For example, increasing Ellenberg fertility score is indirect evidence for an increase in nutrient availability.

Box 2.1 Species characterisation using Ellenberg scores, UCPE Plant Strategies, Biogeographic Affinities and Woodland Specialist classifications of vascular plants

(a) Ellenberg scores. Ellenberg scores are assigned to plants to represent their affinities to different sets of key environmental factors (Ellenberg, and others 1991; Hill and others 1999). The scores are on an ordinal scale and the steps between values are not equal. Therefore, in some cases, as well as comparing mean values for the species groups, the distribution of values within the groups was tested using chi-squared analysis for each factor. Those for light and nitrogen are set out below.

L Light	**N Soil nitrogen**
1 Plants in deep shade	1 Sites poor in available N
2 Between 1 and 3	2 Between 1 and 3
3 Shade plants	3 More often N-deficient soils
4 Between 3 and 5	4 Between 3 and 5
5 Plants of half shade	5 Average N availability
6 Between 5 and 7	6 Between 5 and 7
7 Generally in well-lit places	7 More often N-rich sites
8 Light-loving plants	8 Between 7 and 9
9 Plants of full light	9 Extremely rich N soils

(b) UCPE Plant Strategies. The Unit of Comparative Plant Ecology, based at Sheffield, developed the concepts of analyzing the characteristics of different groups of species using a variety of functional attribute data (Grime and others 1995). The main strategies are C Competitive (C), Stress-tolerant (S), Ruderal (R). Intermediate forms such as CR Competitive-Ruderal, SC Stress-tolerant competitive, SR stress-tolerant ruderal, and CSR also exist. The MAVIS programme (www.ceh.ac.uk/products/software/land.html) calculates for a given species list an index of the contribution of each of the three main strategy elements to that list. Scores were not available for all species.

(c) Biogeographic affinities. Species were grouped according to their biogeographic affinities (Hill and others in press). Species in Group 1 might be expected to respond differently to (say) increased summer temperatures to those in Group 3.

Group code	Element name
1	Arctic-montane
1	Boreo-arctic montane
1	Boreal-montane
2	Temperate
2	Southern-temperate
3	Mediterranean-atlantic
3	Mediterranean

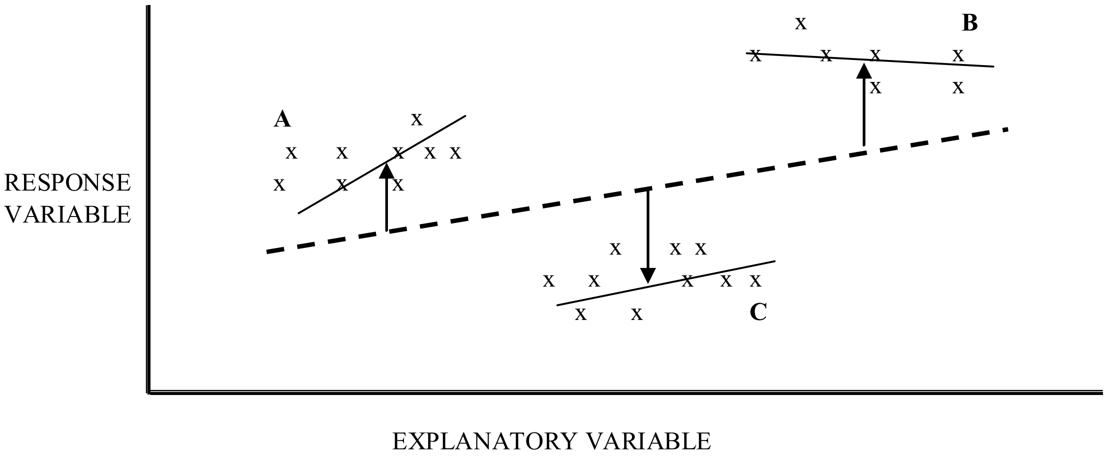
(d) Woodland specialist classification. In Lincolnshire woods Peterken and Game (1984) showed that there was a strong association between the distribution of some vascular plants and the history of the site, more particularly whether it was ancient woodland as defined by Peterken (1977). Ancient woodland indicator plants have also been identified elsewhere in Europe and North America. Kirby and others (in press) assembled various lists of ancient woodland vascular plants suggested for different parts of Britain which included 158 species. These species as a group were more strongly attached to woodland habitats and in particular appeared to be more shade tolerant than other groupings of species (woodland generalists, non-woodland habitat species). They tended to have higher Ellenberg fertility (N) scores than non-woodland habitat species, but there was no difference in the mean nitrogen scores for woodland specialists and woodland generalists.

2.7.2 Mixed modelling approach to variable analysis

Paired t-tests were carried out on differences between years for variables within each site. For cross-site tests of change a mixed-modelling approach was used (Little and others 2000) (Box 2.2). This allowed the effect of site to be taken into account when generating estimates of change for the entire population represented by the sample. The important technical issues here are threefold. First, plots within each site will be spatially auto-correlated. The degrees of freedom for estimating the size of the different components of the total variation may be much less than the number of paired plots recorded because of the lack of independence between plots within a site. Secondly, declaring an effect as random allows the correct term for unexplained variation to be used in tests of the different variance components. Thirdly, a site term enables variation across the dataset to be partitioned according to within site and between-site components (Singer 1998).

Tests of change in frequency of individual species were carried out in two ways. A simple guide as to whether a species increased or decreased more than expected by chance in the number of sites or plots occupied was conveyed by the binomial formula (Zar 1996). This approach was used to evaluate change in number of records of trees, shrubs and woody species seedlings between surveys. Inference of change in frequency and cover in the wider population of British broadleaved woodlands were estimated using the SAS %Glimmix macro (Wolfinger 2002). This implements a mixed model ANOVA but allows the binomial nature of species frequency data (ie ones or zeros) to be correctly modelled (Box 2.2).

Box 2.2 Mixed-modelling approach to analysing woodland change



The above illustrates the relationship between within-site and between site variation in an ecological response variable, such as change in ground flora species richness, along a gradient of an explanatory variable e.g. change in the basal area of trees and shrubs over time. Each 'x' represents a sample plot within a site.

The problem: Based on our sample of 103 woodlands across GB, within each of which are 16 random plots, we want to quantify the relationship between the response variable and the explanatory variable in such a way that the results can be inferred to the entire sampled population of plots in sites.

Analytical approach:

The variability in both the response and the explanatory variable can be broken down into a within-site component and a between-site component.

The within-site variation in the response is expressed as the sum of the squared deviations of each plot (x) from the overall site mean (indicated as the point where the vertical arrows meet each within-site regression lines A to C).

Within each site, regression lines can be fitted to try and explain these deviations in terms of an explanatory variable.

Since we have 103 sites, we could also use all plot level responses across all sites and fit an overall regression line (heavy dashed line). The problem with this approach is that plots within the same sites are likely to show more similar behaviour than when compared with plots in other sites. This is reflected above by the difference in overall position of each site along the y-axis. This spatial auto-correlation means that we cannot justifiably treat all plots in all sites as if they were all statistically independent from each other. An obvious example of this would be the common response of a set of plots to a storm event.

One answer to the spatial autocorrelation problem is to down-weight the influence of each plot by an amount that reflects how similar its response was to that of other plots within the site. An extreme way to do this would be to ignore within-site variation completely and just use the mean response for each site. This would amount to fitting the heavy dashed line in the graph above to just the three site means. However this would be to ignore the important variability that occurs between plots within each site. Our statistical model should acknowledge the fact that sites are internally heterogenous and that plots will therefore differ in their ecological patterns of change. So how do we incorporate both within and between-site variance components while not over-playing the role of similar within-site responses.

The **mixed-model** analysis of variance (ANOVA) takes account of all the variation in the dataset but with an adjustment to the degrees of freedom based on the similarity of within-site responses. This affects the overall statistical power of the test and also adjusts the influence of each plot response according to its contribution to within- versus between- site variation. (Note that degrees of freedom can become fractional with this method.)

It is called a mixed model because the components into which the variation in the data are broken down are a mixture of fixed and random effects. In our situation, site is a random effect because it reflects our strategy for randomly sampling broadleaved woodland across Britain. Although there is no sense in which our sites represent a designed treatment or gradient of an explanatory variable, we would expect variation to occur that reflected site-level factors such as management history, altitude, topography and exposure to local perturbations such as storm events. The fixed effects in the analysis refer to our explanatory variables. Though the extent of their replication and crossing may be accidental they represent gradients of different potential ‘treatment’ effects and we carry out and interpret analysis of variance in light of a hypothesised relationship between woodland change and each fixed effect.

2.7.3 Overview of the variables used in the analysis

Table 2.6 summarises the variables derived from site data, species lists, tree and shrub measurements and soil samples. All variables were tested for change over time, both within and between sites. Subsets of data were also entered into constrained ordinations to search for the minimum number of variables that could best explain temporal and spatial patterns in the data. The choice of which data to combine in which analyses was determined by the steering group in relation to the particular ‘drivers of change’ with which we were concerned.

Table 2.6 List of variables derived from site data, species lists, diameter breast height measurements and soil samples.

Variable name	**Role (Explanatory / Response / Stratification)**	**Weighting**	**Transformation**	**Description and interpretation**	**SITE or PLOT level**
Species richness	R	n/a		count of ground layer species excluding bryophytes, lichens and woody species recorded in the shrub layer and canopy	p
Mean Ellenberg pH score	R	cover/none		decrease = implied lowering of pH	p
Mean Ellenberg Fertility score	R	cover/none		decrease = lower fertility	p
Mean Ellenberg Light score	R	cover/none		decrease = greater shade	p
Mean Ellenberg Moisture score	R	none		decrease = drying out of woods	p
Mean Competitor proportion	R	cover/none		decrease=lower contribution from competitive traits	p
Mean Stress-tolerator proportion	R	cover/none		decrease=lower contribution from stress-tolerant traits	p
Mean Ruderal proportion	R	cover/none		decrease=lower contribution from ruderal traits	p
Soil pH	R/E	none			p
Soil organic matter	R/E	none		increase = greater carbon content in soils	p
Mean and total DBH class count	R/E	none	square root	count of woody stems more than 1.3m high by 10cm diameter classes in each plot (smallest size class 6-10cm); the mean count was taken over all the plots in which a tree or shrub species was recorded in each year.	p
Basal area (m^2)	R/E	none		woody stem diameter converted into basal area at 1.3m height; increase means greater coverage of ground by woody stems	p
Plot regeneration	E	none		count of regenerating woody species (woody stems between 0.25 and 1.3m tall) in each plot	p
Plot openness	E	code dependent		weighted average of open habitat sum[1]; higher score = greater extent of unwooded habitat	p
Site openness	E	code dependent		weighted average of open habitat sum[1]	s
Plot management	E	code dependent		count of management indicators[3]	p

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[1] Sum of scores; 5-12m glades, path<5m, track non-prepared and rocky knoll <12m - **score 1** and larger glades and knolls and ride>5m - **score 5**. Sum divided by maximum possible score to give index between 1 and zero.

[3] Count of signs of recent management; new conifer stumps, felled trees, extraction routes, chips/sawdust, vehicle tracks, new hardwood stumps, fire sites, stack/timber, recently cut coppice.

Variable name	Role (**E**xplanatory / **R**esponse / **S**tratification)	Weighting	Transformation	Description and interpretation	SITE or PLOT level
Site grazing	E	none		count of signs (cattle, sheep, red deer, other deer, horses, pigs) across all 16 plots in site in each year.	s
Site management	E	code dependent		count of management indicators (as for plots)	s
Adjacent habitats	E	none		count of adjacent habitats in 1970 and in 2001.	s
Area of land-cover around site	E	none		proportion of Broad Habitats in three circular buffer zones around each site taken from Land Cover Map 2000 (radius 1500, 2500 and 3500m). Habitats grouped into intensive (arable+improved grassland+built), broadleaved/mixed woodland and other semi-natural habitats (acid and calcareous grassland, heath, bog, fen, montane). Coniferous woodland (>80% conifer) could logically have been included in either the intensive or woodland/semi-natural sets: therefore it was excluded from this analysis.	s
Nitrogen deposition (kg N ha^{-1} yr^{-1} for woodland in 1998)	E	none		modelled estimate for 5x5km square containing a site of wet and dry deposition.	s
SOx deposition (kg N ha^{-1} yr^{-1} in 1970)	E	none		as above.	s
Growing season length (GSL)[4]	E	none		rate of change (positive or negative) in GSL for the 5x5km square containing each site over the interval 1961 to 1999.	s
Rainfall intensity (RI)[5]	E	none		rate of change in RI for the 5x5km square containing each site over the interval 1961 to 1999.	s
Minimum January temperature	E	none		average temp for 1990-1999 minus average for 1961-1970	s
Maximum July temperature	E	none		as above	s
Mean Jan-March temperature	E	none		as above	
Countryside Vegetation System class	S	none		automated allocation of each plot in each year using the MAVIS software.	p

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[4] Number of days with average daytime temperature >5°C. All weather data based on interpolated values for 5x5 km squares containing each site.

[5] Annual rainfall on rain days (days with >1mm rainfall)/number of rain days. Higher if more rain falls on fewer days.

Variable name	Role (Explanatory / Response / Stratification)	Weighting	Transformation	Description and interpretation	SITE or PLOT level
National Vegetation Classification class	S	none		as above but translating cover of species in each plot into frequency values between 1 and 5. Also *Quercus* sp. replaced with records for both *Q. robur* and *Q. petraea*. Woody species in canopy were always coded as frequency =5.	p
Environmental zone	S	none		six zones across Great Britain based on an aggregation of ITE land-classes (see Haines-Young and others 2003); for most analyses these were amalgamated into Upland and Lowland zones	s
Ancient Woodland	S	none			s
SSSI	S	none			s
Country	S	none		England, Wales or Scotland	
October 1987 storm track	E	none		sites south of a line running from Southampton via north London to Great Yarmouth (Appendix 4)	s
Grass cover	R	none	square root	sum of grass cover in each plot	p
Bryophyte cover	R	none	square root		p
Cover of potentially dominant field layer species	R/E	none	square root	*Rubus fruticosus* agg. + *Pteridium aquilinum* + *Rhododendron* spp.	p
Bare ground	R	none	square root	litter + rock + wood + bare soil + water	p
Easting	E	none		higher = more eastern	s
Northing	E	none		higher = more northern	s

2.8 Relationship between signal detection and attribution

In a designed experiment crossing and replication of treatments allows for unequivocal separation of the effects of different factors. However, this report is based on a large-scale surveillance dataset that, by definition, lacked designed controls and treatment applications. The degree of crossing and replication among potential drivers may be uneven because it is beyond the control of the observer (Gadbury and Schreuder 2003). Lack of a correlation between change and explanatory data can reflect low variation in an explanatory variable and low replication along the given gradient: it is not necessarily evidence that a driver has not, or could not have, caused change. Similarly a significant correlation between change in a potential driver and a response does not necessarily prove a causal link between the two: both may be responding independently of one another to a different variable that may or may not have been included in the survey.

Inferring causal relationships from correlative data is more reliable where we can specify a direct mechanism by which driver 'x' might influence response 'y'; and where there is internal consistency between the responses observed with respect to that driver. For example if 'x' influences 'y' and the same mechanism should affect 'z' then we would expect 'z' to show a similar response to 'y'. A series of possible drivers of change were therefore proposed as hypotheses against which expected responses were compared in order to focus on the more likely causal relationships.

Several hundred statistical tests were carried out, so that some significant results are likely to occur simply by chance. A strictly conservative approach to correcting for multiple tests would greatly decrease the study-wide p value required to declare significance. As a consequence we might miss weak signals that could nevertheless generate new hypotheses. On the other hand, leaving p values uncorrected could, with so many tests, generate spurious, apparently, significant results (Moran 2003; Garcia 2003; Yoccoz 1991). We have chosen not to correct for the many tests carried out because of the exploratory nature of this analysis (Roback and Askins 2005). Some spurious significant results may therefore be present, especially where p values are above 0.01. Where there appears to be no clear ecological basis for the statistical relationship such results should be treated with caution.

2.9 Exploring drivers of change via direct hypothesis testing

The exploration of possible causes of within and between-site change in soil and vegetation was based on hypothesised relationships between changes in specific sets of response and explanatory variables. The main drivers explored were (in no particular order):

- climate change;
- eutrophication and acidification;
- adjacent agricultural intensification;
- increased shade resulting from lack of disturbance and growth of the trees;
- changes in woodland management;
- changes in grazing regimes.

For each of these the Steering Group suggested a series of possible explanatory and response variables from Table 2.6. All the explanatory variables associated with each driver were then tested against each response variable for that driver. The overall explanatory power of the final best fitting set of explanatory variables was expressed as the r-squared value between model predictions and the observed data. Interaction terms were tested where these were consistent with hypothesised effects. The different potential drivers of change, their explanatory and response variables are now considered in turn.

Climate change - changes in temperature, rainfall and growing season
Explanatory variables
Rainfall Intensity (RI)
Growing Season Length (GSL)
Change in mean max July
Change in mean min Jan
Mean Jan-March temperature
Response variables
Proportion of biogeographic elements in each plot (three groups; see Box 2.1c).
Individual species changes

Under climate change response variables were generated summarising the proportion of higher plant species in each plot in each year that had been allocated to three biogeographic groups following Hill and others (in press) (Box 2.1c). The changes in abundance of individual species were also explored.

Five climate datasets were downloaded from the Meteorological Office UKCIP web-site. These were Growing Season Length (GSL) and Rainfall Intensity (RI) (every year from 1961 to 1999), minimum January temperature and maximum July temperature (every year from 1961 to 1970 and every year from 1990 to 1999) and mean January to March temperatures. Values were extracted for the 5x5 km square containing each woodland site. Change in GSL, RI and mean January-March temperatures over the 29 year interval were derived as the slope coefficients of each weather variable regressed on to year for each site; all coefficients, whether significant or not were used as explanatory variables. Change in mean Jan and max July temperatures were derived as the difference between the mean values for each site, again at the 5x5km square scale, across all years in each decade.

Seventeen sites had significant rates of change in GSL; all showed a lengthening of the season over time. None of the 103 sites had a shortened growing season. Twenty five sites showed a significant rate of change in RI. Of these, 21 increased in rainfall intensity. All sites that showed a reduction were in upland England and Wales. Across all 103 woodland sites, both mean minimum January temperature and maximum July temperature were higher in the nineties than the sixties. There were also increases in the mean January to March temperatures.

Eutrophication and acidification – increases in atmospheric N deposition (NHx and NOy) and decreased SOx from mid-seventies onwards.
Explanatory variables
GSL*N deposition
N deposition
Area of intensive land cover adjacent to site *N deposition
Response variables
Ellenberg N scores (see Box 2.1a)
Ellenberg R scores (see Box 2.1a)
Soil pH
Species richness
Individual species changes

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Relationships between changing pollutant deposition and woodland change focussed on the possible effects of the known reductions in SOx deposition since the early 1970s and increases in nitrogen deposition, particularly NHx, over the period of the survey (NEGTAP 2001). Various studies on the continent have reported changes in soil and vegetation characteristics that have been attributed to changes in acidity or nitrogen deposition (Brunet and others 1996, 1997; Diekmann and Falkengren-Grerup 2002, 2003; Falkengren-Grerup and others 1998; Thimonier and others 1992), although evidence from Britain is more equivocal (Smart and others 2005; Ling 2003; Kirby and Thomas 2000).

Current SOx deposition values are based on interpolation from a network of monitoring stations across Britain. The earliest data available from the current network date back to 1986 and therefore miss the period of marked reduction in deposition. Estimates from 1970, which predate the emissions reductions of the mid-seventies onward, were generated using the HARM model. However these data cannot be directly used to calculate grid cell estimates of change in deposition because they cannot be validly subtracted from contemporary deposition estimates (RI Smith and D Fowler pers comm).

The 1970 model estimates were therefore used as a surrogate explanatory variable to represent the spatial pattern of estimated SOx deposition in 1970. The change in soil pH across each site was expected to be significantly positively correlated with 1970 model estimates because the largest increases in pH ought to coincide with the largest reductions in SOx deposition, and hence with the highest depositions in 1970. Model estimates of total N (oxidised plus reduced) were acquired from CEH Edinburgh for each 5x5 kilometre-squares for 1998. It was assumed that the spatial pattern in 1998 represented the culmination of N loading to which woodland, along with other ecosystems, had been exposed up to and beyond the 1971 survey. Significant relationships between woodland change and N deposition were explored based on this assumed relationship (Smart and others 2005).

Under conditions of raised nitrogen load competitive species and those with high Ellenberg N scores (representing higher fertility conditions) might be expected to increase while stress-tolerant species decline.

Impact of intensive agriculture – increased P and N surpluses from farmland and interactions with fragmentation and isolation as surrounding land cover intensifies
Explanatory variables
Site area
Site area*Perimeter/area ratio
Area of intensive land cover adjacent to site
Site area* Area of intensive land cover adjacent to site
Perimeter/Area ratio* Area of intensive land cover adjacent to site
Response variables
Ellenberg N
C score
S score
Plot species richness

While general nitrogen inputs might be picked up in the above analysis there can also be substantial impacts from localised sources (Sutton and others 2003, 2004). Intensification of land-use surrounding each site could result in localised increased exposure to enriched run-off, ammonia deposition from intensive pig and poultry units and fertilizer over-spray (Pitcairn and others 1998; Bateman and others 2004; Gove and others 2004), as well as changes in the levels of background wet and dry deposition.

The area of adjacent intensive land cover around each site may therefore be a more spatially sensitive correlate of eutrophication than the 5x5 km square grid cell estimate because of the known sub-grid variability in N deposition (Sutton and others 2004). The impact of adjacent land may be more pronounced in small sites, or sites with relatively long edge-to-area ratio because a greater proportion of the wood is exposed to outside influences (Petit and others 2004).

Increased shade – reductions in active management of broadleaved woodland.
Explanatory variables
Weighted average of open habitats at plot and site level
Change in summed cover of Bracken + *Rubus* + *Rhododendron*
Basal area of trees and shrubs
Response variables
Plant Strategy S score (see Box 2.1.b)
Plant Strategy R score (see Box 2.1.b)
Ellenberg L score
Species richness

Change in management/use of wood – extent to which within-site management effects can explain cross-site differences in changing basal area and habitat openness
Explanatory variables
Change in number of signs of recent management (site and plot level)
Response variables
Basal area
Weighted average of open habitats in each plot

Many broadleaved stands were heavily felled during the First and Second World Wars (Richards 2003) as part of the war effort. 23% of all woodland areas were classified in the 1947 census as felled or devastated (HMSO 1952). Even if the woods most affected were excluded from the Steele surveys (section 1.2), the age range of British broadleaved woodland is strongly skewed towards relatively young stands (Forestry Commission 2003). For much of the second half of the twentieth century there has also been a decline in the management of broadleaved woods following the large-scale abandonment of coppicing. This combination of relatively young stands and lack of deliberate forestry management was expected to be reflected in changes in basal area and DBH distributions, and with fewer open habitats being recorded at plot and site level. The main response in the flora was expected to be declines in light-demanding (high Ellenberg L score) or ruderal (plant strategy R score) species (Kirby and Thomas 2000).

Changes in grazing regimes – increases in sheep in upland woods and deer in lowland ones.
Explanatory variables (by upland and lowland)
Count of grazing signs in plots (GS))
Site grid reference*Area of adjacent acid grassland
GS*Area of site
GS*Perimeter:Area ratio
GS*N deposition
Response variables
Ellenberg L score
Ruderal score
Plot level woody species regeneration
Species richness
Grass cover
Changes in individual species (eg reduced bramble)

Since the 1970s there has been an increase in sheep numbers in upland landscapes, especially in England and Wales (Fuller and Gough 1999) and increases in deer numbers in lowland England (Fuller and Gill 2001) and parts of Scotland (Deer Commission 2001). In some circumstances grazing can enhance ground flora diversity, but heavy pressure tends to have a negative effect overall, particularly in woods not previously subjected to regular grazing (Mitchell and Kirby 1990; Kirby 2001). In general, grasses tend to be favoured whilst forbs such as *Mercurialis perennis* and undershrubs such as bramble are reduced in abundance (Cooke and others 1995; Crampton and others 1998).

The above do not cover all the possible factors that could be leading to change in British woods, but the Steering Group for the project considered that these were likely to be the most important in terms of their impacts and in relation to the nature of the records that were available. For example the records collected were not appropriate for assessing changes in grey squirrel density; therefore these results cannot assess the significance of squirrels as a factor in beech growth and mortality.

3. Results

3.1 Introduction to the results

The results are presented in the following order:

- Losses of plots and changes in their classification (this section)
- the main factors correlated with variation within the 1971 survey (section 4);
- the outputs from the 2001 survey:
- changes in the soils (section 5);
- changes in the tree and shrub layers (section 6);
- the ground flora trends (section 7);
- Summary of the findings (section 8).

We looked particularly at whether the results provide evidence for the impacts of the drivers discussed in section 2.9. Analyses of change were carried out in which the proposed response variables were each tested against the proposed explanatory variables for each driver in turn.

Where appropriate the results are presented for plots across the whole data set and, separately, for 'sites' through combining the data from the 16 plots recorded in each wood. Species occurring in the wood between plots do not contribute to the site lists as used in this report.

The primary analysis is across the whole of Great Britain because the original selection of survey sites was made at this spatial scale. The original analyses carried out on the 1971 data (Bunce 1982, 1989) indicated that there was a strong division between upland and lowland samples. Tests were therefore also carried out by grouping sites into either upland or lowland zones across GB. The ITE Land Classes were aggregated into the six environmental zones used to report stock and change in Countryside Survey 2000 (Haines-Young and others 2003): the upland woodland sites were then those located in either of the two Scottish upland zones or in the one upland zone covering England and Wales (Figure 3.1). The sample sites were not evenly split between these zones; there were 84 in the lowlands and only 19 in the uplands. Results from the lowlands are therefore likely to be more robust than for the uplands.

Other subdivisions of the data (for instance into England, Scotland and Wales) are more arbitrary with respect to the numbers and types of woods represented and hence this weakens the strength of any conclusions that can be drawn from them. Country sub-sets, and sub-sets defined on the basis of ancient versus non-ancient woodland, SSSIs versus non-SSSIs were however analysed. A summary of where such results differ from GB pattern is given in Appendix 9 and only passing reference is made in this report where such differences help to clarify GB findings.

Ten woods lay in the track of the 1987 great storm in south-east England (Appendix 4). This is a relatively rare example (for Britain) of a sudden catastrophic event known to have had a major influence across a large number of woods (Kirby and Buckley 1994). Although it is a small sample, and there is not an equivalent control set of sites, the changes in this sub-set may indicate ways in which such events can have longer-term consequences for the structure and composition of woods affected.

Extracts from the records for individual sites are used to illustrate some of the issues, for example losses due to land-use change. *These are only illustrative examples, and do not define the outcome of a particular set of impacts.* The complete set of site survey reports will be made available separately.

Figure 3.1 Lowland and upland landscapes of GB defined by aggregating the six environmental zones in GB formed by an aggregation of ITE Land Classes.

3.2 Lost plots and changing land-use between 1971 and 2002 survey

Sixteen of the 103 sites were affected by major change in land use between surveys although the proportion of the woodland area affected was generally small. Thirty-eight plots (2.3%) out of a total 1,648 locations were lost to other forms of land-use (Table 3.1). An additional six were not recorded because of access restrictions.

Table 3.1 Land-use change in woodland plots not recorded in the resurvey.

	Built	Roads	Quarrying	Agriculture	No access
Number of plots	17	2	1	18	6
Number of sites	11	1	1	4	6

In Wales, Hensol Wood (site 48) was bisected by the M4 and so lost two plot locations. In four sites plots were lost to gardens while in Mulben Wood (site 98) in Scotland two plots were lost to the extension of a distillery. In Dulwich Wood, London, three plots were lost to a housing estate built in 1972. Two plots were also lost to housing development in Warren Wood in south-east England.

Other changes were a result of agricultural activity. The largest single change in land-use as a proportion of site area was at Hill Wood in Wiltshire (site 24) where half the wood had been clear-felled and converted to *Lolium perenne* ley. Two lowland sites, Love's Copse and Austy Wood, close to each other in the West Midlands, lost six and three plots respectively to wheat and set-aside. One plot location each was lost to semi-improved grassland at New Laund and High Wood in north-west England.

The impact of grazing by sheep and cattle led to a change from woodland to grassland at three sites; Fridd Wood in Wales (1 plot), Reins Wood in Northumberland (1 plot) and Compton Wood in Somerset (2 plots). Botanical evidence and surveyors' notes highlighted the fact that in all cases increased grazing pressure blurred the boundary between adjacent pasture and the woodland site rather than creating a clear discontinuity of land-use. The resulting vegetation had more of the character of secondary woodland or wood pasture with a typically grass-dominated (e.g. *Holcus lanatus* 75%), well-lit and eutrophic understorey accompanied by sparsely scattered shrubs and trees (Box 3.1)

For ancient woodland more generally between 1935 and 1985 the major source of loss of area was to agriculture (Spencer and Kirby 1992); the higher losses to development suggested by Table 3.1 may reflect that much of the loss to farmland happened prior to 1970 and was then largely halted by the 1985 Broadleaves Policy. Losses of woodland to development have however continued, including, as found in this survey, piecemeal losses of woodland edge to garden encroachment (www.woodsunderthreat.info). More general changes between woodland communities, particularly in small woods, and other land-uses are described in Haines-Young and others (2003).

BOX 3.1. The species composition of plot 9 at Compton Wood in Somerset in 1971 and in 2002. In 2002 the plot was described as being in cattle-grazed 'open wood pasture with mature widely spaced ash and planted conifer standards over a grassy sward.' The uppermost rows refer to counts of woody species stems >1.3m height. Ground flora values are percentage cover with + for species covering less than 1%.

Species	1971	2002
Prunus padus	5	
Fraxinus excelsior (c)	4	
Acer campestre	3	
Salix seedling/sp	2	
Salix caprea	1	
Quercus seedling/sp	1	
Corylus avellana	1	
Ground flora		
Agrostis capillaris	1	
Deschampsia cespitosa	1	
Mercurialis perennis	1	
Prunella vulgaris	1	
Veronica chamaedrys	1	+
Circaea lutetiana	+	+
Fragaria vesca	+	+
Fraxinus excelsior seedling	+	+
Urtica dioica	.+	1
Hyacinthoides non-scripta	+	1
Dactylis glomerata	5	5
Rubus fruticosus agg.	1	15
Holcus lanatus		75
Lolium perenne		5
Cirsium arvense		1
Ajuga reptans	+	
Brachypodium sylvaticum	+	
Carex caryophyllea	+	
Cirsium palustre	+	
Corylus avellana	+	
Crataegus monogyna seedling	+	
Dryopteris dilatata/carthusiana	+	
Dryopteris filix-mas	+	
Euphorbia amygdaloides	+	
Festuca ovina agg.	+	
Galium pumilum	+	
Geum urbanum	+	
Lysimachia nemorum	+	
Poa nemoralis/trivialis	+	
Primula vulgaris	+	
Veronica montana	+	
Viola riviniana/reichenbiana	+	
Castanea sativa		+
Ranunculus acris		+
Rumex obtusifolius		+
Senecio jacobaea		+
Taraxacum agg.		+
Vicia orobus		+

3.3 Vegetation types

Plots were allocated to National Vegetation Classification groupings (see 2.5 and Appendix 10). Individual plots might be allocated differently between the surveys because of changes in species composition over time. However 41% showed no change in allocation and the overall balance between groupings was also similar in 2001 to that in 1971 (Table 3.2).

Plots were allocated to Countryside Vegetation Survey (CVS) classes (section 2.5 and Appendix 11). 67% showed no change in allocation and the balance of CVS types was again similar between the survey years (Table 3.3). There were slight trends towards more fertile groups, particularly in the CVS allocation, and towards more open communities in both classifications. However over 90% of the variation between groups in the 2001 data could be explained by the 1971 allocations.

Table 3.2 Allocation of plots to NVC groupings, 1971,2001.

NVC Grouping/Formation	No of 1971 plots	No of 2001 plots
Mesotrophic oak-birch woodland (W10, W11)	665	528
Ash-elm woods (W8, W9)	237	157
Acid oak-birch woods (W16, W17	164	192
Alder-willow woods (W1-7)	97	97
Beech-yew woods (W12-W15)	108	147
Scrub (W19-25)	107	178
All 'woodland'	*1378*	*1299*
Open vegetation (OV formation)	187	256
Mesotophic grassland (MG formation)	17	39
Mire, heath, upland (M,H,U formations)	31	18
Swamp, sand-dune (S,SD formation	10	11

Table 3.3 Allocation of plots to Countryside Vegetation System classification 1971, 2001

CVS class	No of plots in 1971	No of plots in 2001
5. Lowland woods	895	939
6. Upland woods	588	494
All woodland	*1483*	*1433*
4. Infertile grassland	71	66
7. Moorland grass/mosaics	38	37
2. Tall herb/grassland	17	41
8. Heath/bog	11	18
3. Fertile grassland	2	25
1. Crops/weeds	1	3

3.4 Summary of changes in plot type

- Between 1971 and 2001 sixteen of the 103 sites were affected by major change in land use and thirty-eight plots (2.3%) out of a total 1,648 locations were lost, with about equal contributions from urban development and agricultural activities.

- The overall balance of plot types, classified by NVC and CVS remained the same, but with some increases in plots of more fertile and more open conditions.

4. Variation within the survey sites in 1971

As part of a separate series of analyses Corney and others (2004) assessed the factors influencing woodland vegetation composition across Britain using the 1971 woodland survey data. Indirect gradient analysis (unconstrained ordination using Detrended Correspondence Analysis (DCA)) suggested a gradient strongly associated with nutrient availability and pH. Direct gradient analysis (constrained ordination using canonical correspondence analysis) and variation partitioning were used with over 250 ecophysiologically relevant variables, including climatic, geographical, soil and herbivore data, to model the spatial variation in the woodland vegetation in 1971.

The DCA results indicated that shade-bearing species and a greater proportion of shrub layer species were associated with woodland on deep fertile base-rich soils, while less fertile woods on acid soils were associated with species of relatively open, nutrient poor environments. The upland woodlands surveyed had more rocky and shallow soils and were more likely to be grazed by sheep; they supported fewer deep-rooted herbs and shrub species.

The variation in the vegetation composition reflects the qualitative interpretation of the main divisions within the National Vegetation Classification (Rodwell 1991); for example between north-west and south-east oak and ash woods and within the south-east types, the split between base-rich, mesotrophic and acidic communities.

Sheep grazing intensity was also found to be a significant variable using forward selection. High grazing pressure was found to be associated with species typical of the western uplands; grassland, moorland mosaics and open woodland, whilst low pressures in the south and east allowed palatable species such as *Lonicera periclymenum* and *Rubus fruticosus* to thrive. Similar results have been found in the Atlantic deciduous woodlands of the western Pyrenees where sheep actively select woodlands containing an abundant herbaceous understorey and preferentially browse *Rubus*, *Ilex*, *Vaccinium*, forbs and some palatable grasses, while preferentially rejecting *Brachypodium* and *Luzula* sp. (Garin and others 2000). There was no significant correlation between species composition and the presence of deer or domestic stock recorded during the 1971 survey.

The analyses also suggested a micro-climatic effect of woodland shape, via light, temperature and humidity gradients. A high edge-to-area ratio could make woodland more vulnerable to invasion by light-demanding or nitrophilous species more typical of adjacent, earlier successional habitats (e.g. Smart and others 2001; Honnay and others 2002).

A strong environmental gradient related to soil pH was also found, from upland species assemblages on acid soils to lowland groups of species on more base-rich soils. Summer rainfall and growing degree-days (accumulated temperature) were found to be the most important factors correlated with vegetation composition in the woodlands sampled. In other studies the gradient of temperature along the length of the country has been identified as one of the primary factors controlling the distribution of *Tilia cordata* (Pigott and Huntley 1978), whilst the west-east oceanic-continental rainfall gradient influences fern and bryophyte distributions, which are most abundant in western oak woods.

How then did the variation within the woods change between 1971 and 2001?

5. Comparison of the soil records for 1971 and 2001

5.1 Soil pH

There was a significant increase in soil pH (Table 5.1), with a large reduction in the number of sites with soil pH <5 and a corresponding increase in the number of sites with higher soil pH (Figure 5.1). Of the individual sites, 38 showed significant increases in soil pH from 1971 to the 2001 survey. Of the remaining 65 sites, most showed small increases in soil pH (42 sites) while 23 sites showed small, but insignificant, declines in soil pH (Figure 5.2).

From 1971 to 2001 there was a significant increase in the mean soil pH across all plots (Table 5.2 and Figure 5.3) which is also reflected in a small increase in the maximum pH value. There was a reduction (9%) in the number of plots with a soil pH <5 and increases in the number of plots with soil pH>5.

The increases in soil pH were influenced by location and soil characteristics derived from soil sub-groups (Avery, 1980) and most marked in organic/organic-mineral, non-calcareous and free-draining soils (Table 5.3), albeit that there were unequal numbers of plots in the different categories.

The soil pH increase is consistent with other national trends from the Countryside Survey 2000 and the re-survey of the National Soil Inventory (Haines-Young and others 2000; www.silsoe.cranfield.ac.uk/nsri/services). These national-scale increases in soil pH may be signs of recovery from acid rain pollution across the UK (NEGTAP 2001) although no relationship was found between pH change and the historical estimates of 1970s SOx deposition: see later discussion on effects of atmospheric deposition.

The best predictors of soil pH change were N deposition in 1998 (positive correlation), SOM change (negative correlation) and basal area change (negative correlation), i.e. although there was a general increase in pH, the increase was less, the greater the basal area growth) (Table 5.4).

Table 5.1 Summary statistics for soil pH in 1971 and 2001 across all sites

Survey	**N**	**Mean**	**s.e.**	**P**
1971	103	4.98	0.09	
2001	103	5.31	0.10	
change	103	0.33	0.04	**<0.005**

Table 5.2 Summary statistics for soil pH in 1971and 2001 across all plots.

survey	**N**	**Mean**	**s.e.**	**P**
1971	1648	4.98	0.028	
2001	1618	5.31	0.032	
change	1618	0.33	0.023	**<0.005**

Table 5.3 Change in plot soil pH.

Stratum	**Mean 1971**	**Mean 2001**	**Mean change**	**s.e.**	**df**	**T**	**p**
GB	4.98	5.30	0.33	0.039	102	8.28	**<0.0001**
Lowland zones	5.07	5.40	0.33	0.044	83.1	7.54	**<0.0001**
Upland zones	4.57	4.88	0.31	0.094	18	3.27	**0.0043**
Free-draining soils	5.00	5.35	0.34	0.040	102	8.42	**<0.0001**
Gley soils	4.71	4.90	0.21	0.076	28.2	2.81	**0.009**
Non calcareous soils	4.63	5.02	0.43	0.044	83.9	9.65	**<0.0001**
Calcareous soils	7.19	7.11	-0.08	0.076			n.s.
Organic soil	4.89	5.18	0.29	0.079	51.4	4.53	**<0.0001**
Mineral soil	5.13	5.48	0.36	0.055	75.7	5.30	**<0.0001**
Organo-mineral soil	4.60	4.96	0.36	0.046	99.7	7.11	**<0.0001**

Table 5.4 Change in plot soil pH in relation to explanatory variables, based on mixed-model ANOVA.

All soils

Effect	**df**	**F**	**p**	**Relationship between effect and pH change**
N deposition 1998	100	7.67	**0.0067**	+
Basal area change (trees and shrubs)	1425	10.78	**0.0011**	-
Soil organic matter change	1419	24.78	**<0.0001**	-

Non-calcareous soils

Effect	**df**	**F**	**p**	**Relationship between effect and pH change**
N deposition 1998	82.3	10.99	**0.0014**	+
Basal area change (trees and shrubs)	1215	10.99	**0.0014**	-
Soil organic matter change	1200	30.18	**<0.0001**	-

Free-draining soils

Effect	**df**	**F**	**p**	**Relationship between effect and pH change**
N deposition 1998	101	7.50	**0.0083**	+
Basal area change (trees and shrubs)	1298	7.50	**0.0073**	-
Soil organic matter change	1292	20.85	**<0.0001**	-

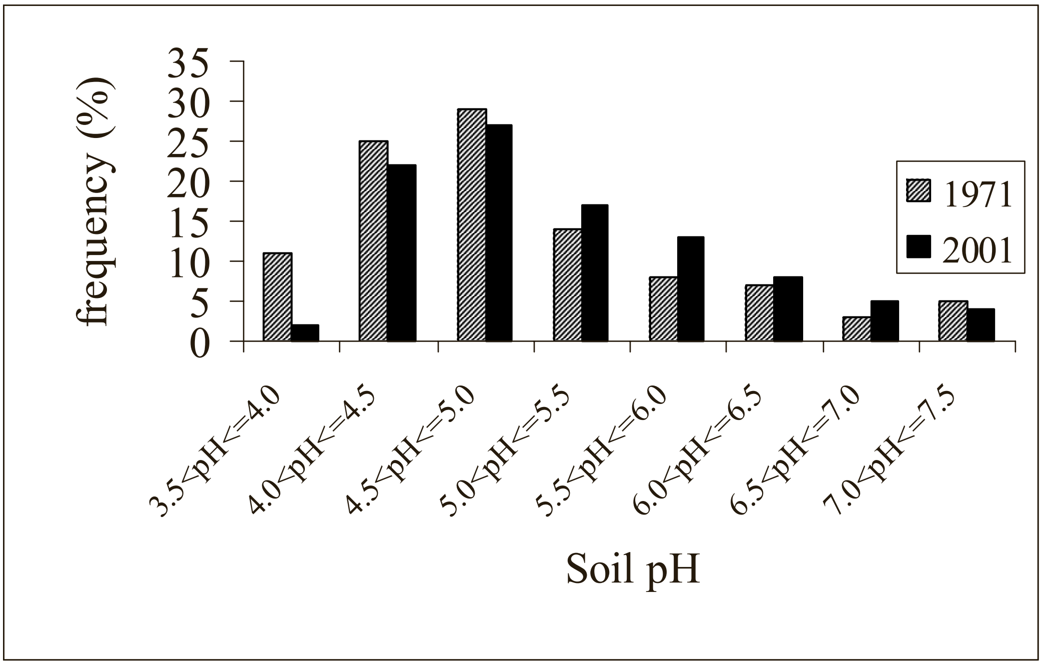


Figure 5.1 Percentage of sites showing different mean soil pH in 1971 and 2001 (n= 103).

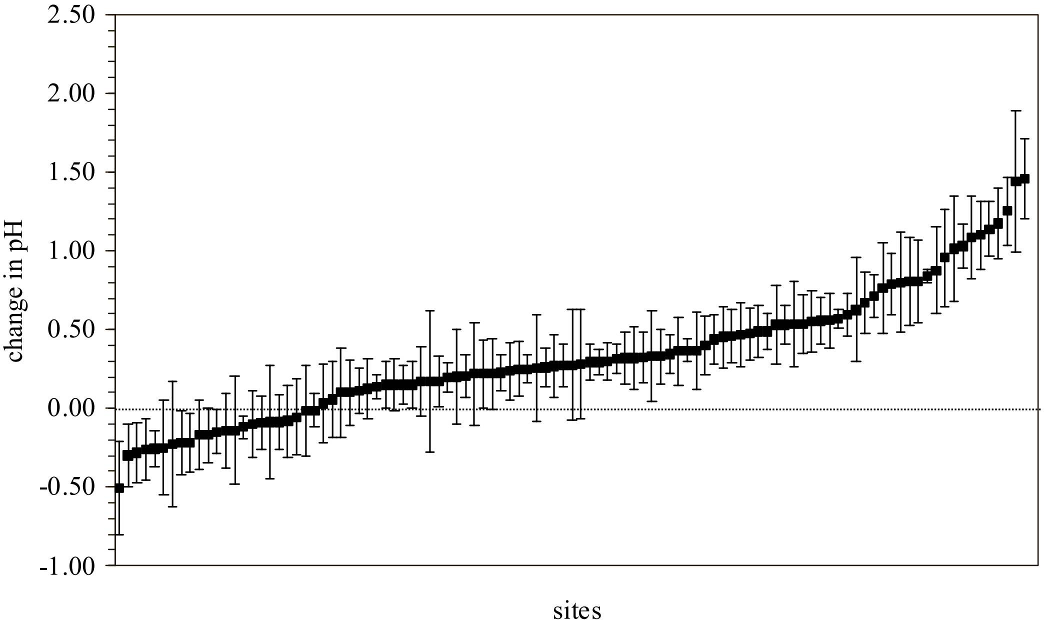


Figure 5.2 Mean change in soil pH for each site from 1971 to the 2001 survey arranged in increasing change in pH from left to right.

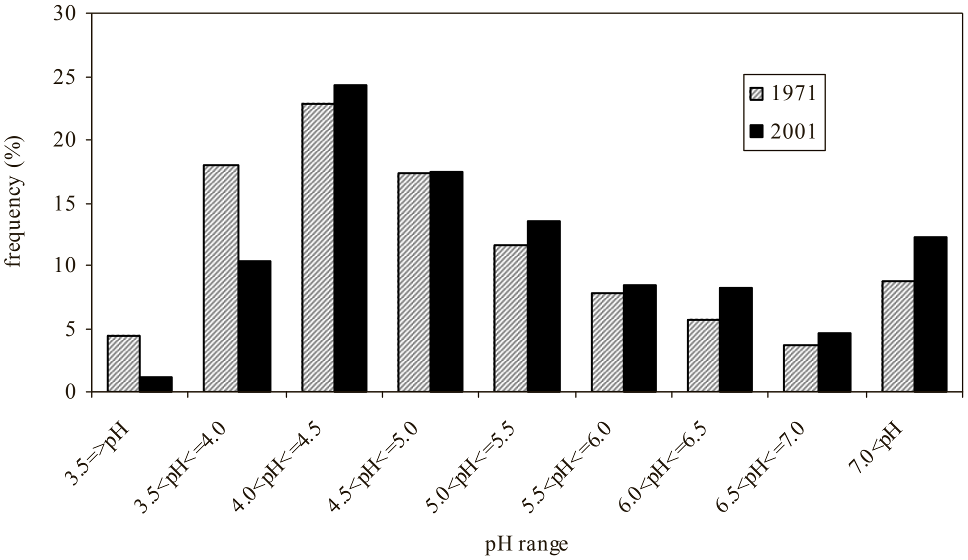


Figure 5.3 Percentage of plots in different soil pH classes in 1971 and 2001 (n= 1603 and 1598, respectively).

5.2 Change in soil organic matter (SOM)

There was no significant change in SOM across the complete site series (Table 5.5), but there were significant changes in 23 sites; of which 15 showed an increase in SOM. There was an increase in the number of sites with SOM > 15% and a decrease in the number of sites with SOM <15% (Figure 5.4).

Table 5.5 Soil organic matter content (%) in 1971 and the 2001 survey for sites.

Survey	N	Mean	s.e.	P value
1971	103	17.58	0.948	ns
2001	103	17.94	0.868	
Change	103	0.38	0.490	

Table 5.6 Soil organic matter content (%) in 1971 and 2001 for plots.

survey	N	Mean	s.e.	p
1971	1644	17.58	0.37	
2001	1620	17.96	0.37	
change	1620	0.37	0.40	ns

At the plot level there was a greater spread of variation in SOM with, in particular, a greater proportion of plots with SOM <10%. There was no significant increase in the mean SOM of all plots (Table 5.6), but there was a decrease in the number of plots with SOM <10% and a corresponding increase in the number of plots with SOM >15% (Table 5.7). Significant increases in soil organic matter content were recorded for plots from lowland sites and mineral and organo-mineral soils while there was a decline in SOM on organic soils (Table 5.8).

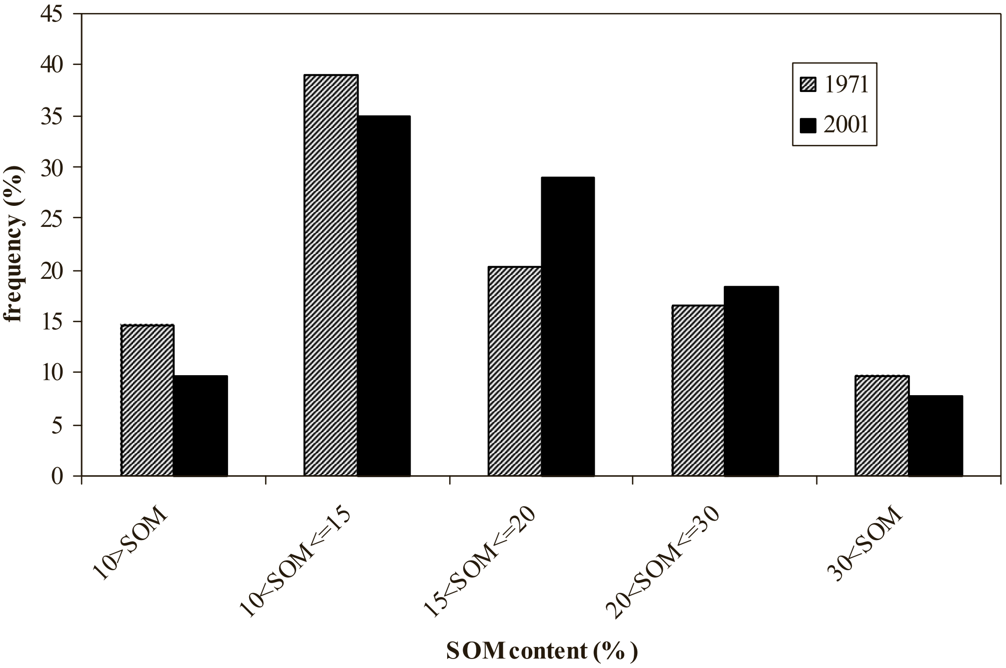


Figure 5.4 Frequency of mean soil organic matter content for sites in 1971 and 2001.

Table 5.7 Soil organic matter content (%) from all plots in 1971 and the 2001 survey, by SOM class in each survey.

SOM(%)	**N**		**Mean**		**Median**	
classes	**1971**	**2001**	**1971**	**1971**	**1971**	**2001**
<10	524	420	7.81	7.71	8.05	7.90
10 to <15	453	514	12.28	12.26	12.11	12.20
15 to <20	282	293	17.33	17.24	17.12	16.91
20 to <30	212	211	23.71	24.13	23.56	23.80
30+	173	182	53.93	51.72	48.88	44.77

Table 5.8 Changes in mean soil organic matter content at the plot level (based on mixed-model ANOVA)

Stratum	**Mean 1971**	**Mean 2001**	**Adj. mean change[1]**	**s.e.**	**p**
GB	17.58	17.96	0.45	0.500	0.3652
Lowland	15.72	16.75	1.04	0.469	**<0.029**
Upland	25.63	23.53	-2.09	1.647	0.219
Organic soil	48.89	30.4	-16.23	1.911	**<0.0001**
Mineral soil	7.81	12.86	5.74	0.774	**<0.0001**
Organo-mineral soil	15.61	17.63	2.19	0.461	**0.001**

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

Table 5.9 Correlations between change in SOM and possible explanatory variables.

Explanatory variable	**df**	**F**	**P**	**Sign of relationship**
Modelled N deposition 1998	106	7.06	**0.0091**	+

An increase in SOM in woodland on more mineral soils, corresponds with analogous results from the Countryside Survey 2000 (Haines-Young and others 2003). The latter recorded a trend for increased soil organic matter content in woodland sites since 1978, although the sample size in CS2000 was relatively small. Various drivers have been proposed for changes in soil organic matter content from local disturbance to climate change, but in this analysis the increase in SOM was only correlated (positively) with nitrogen deposition (Table 5.9).

5.3 Summary of GB level soil changes

The main GB level soil changes were as follows.

- Soil pH increased across sites, with a large reduction in the number with soil pH<5 and a corresponding increase in the number of sites with more alkali soil pH.
- The soil pH increase was more marked in organic than in mineral soil, and in non-calcareous than calcareous soils.
- The increase is consistent with other national trends from the Countryside Survey 2000 and the re-survey of the National Soil Inventory.
- There was no overall change in soil organic matter (SOM) although there were significant changes within 23 sites, of which 15 sites showed an increase.
- There was no increase in the mean level of plot SOM, but the number of plot with low levels of SOM (<10%) decreased.
- SOM increased for lowland soils and mineral and organo-mineral soils but declined in organic soils.
- There was a positive correlation between changes in modelled nitrogen deposition and increases in SOM.

6. Tree and shrub comparisons 1971 -2001 and management changes at the GB level

6.1 Overall tree and shrub composition

The tree and shrub data collected are considered under four broad headings: changes in frequency with which species were encountered, diameter distributions, basal area changes; and regeneration. Related habitat and management records are also considered in this chapter.

The main woody species recorded from the overstorey in the plots were *Corylus avellana*, *Betula pendula* & *pubescens*, *Acer pseudoplatanus*, *Quercus robur* & *petraea* and *Fraxinus excelsior* (Table 6.1*)*. Other common species were *Fagus sylvatica*, *Ulmus* spp, *Alnus glutinosa* and *Sorbus aucuparia*. The most common understorey species were *Crataegus monogyna, Ilex aquifolium* and *Prunus spinosa*.

Most species showed little change at either site and plot levels. At the plot level *Ilex aquifolium* and four conifer species increased. Plot level decreasers included *Betula* spp., *Quercus spp., Ulmus* spp. (excluding *Ulmus glabra*)*, Corylus avellana, Sorbus torminalis, Salix* spp and a range of shrubs (eight of the 18 shrub species in Table 6.1 declined). At the site level three shrub species also declined while *Pseudotsuga* spp and *Pinus nigra* significantly increased. Otherwise no major cross-site changes in canopy dominants were apparent during the thirty year interval. This is perhaps not surprising, since 30 years, while a relatively long time in ecological studies, is a only a small part of the life-span of most canopy dominants in British woodland.

Table 6.1 Site and plot counts for trees and shrubs in 1971 and the 2001. Significant changes in site or plot count are emboldened (based on a binomial probability distribution).

Plant species	**Plot no 1971 n=1648**	**Plot no 2001 (n=1621)**	**Binomial - p**	**Direction**	**Site no 1971 (n=103)**	**Site no 2001 (n=103)**	**Binomial - p**	**Direction**	**Mean plots per site 1971**	**Mean plots per site 2001**
Shrubs										
Cornus sanguinea	59	12	**0.0000**	down	21	10	**0.0354**	down	2.81	1.20
Corylus avellana	733	662	**0.0304**	down	90	86	0.4106	no change	8.14	7.70
Crataegus monogyna	401	342	**0.0166**	down	81	78	0.4370	no change	4.95	4.38
Ligustrum vulgare	24	7	**0.0017**	down	12	2	**0.0065**	down	2.00	3.50
Rhamnus catharticus	16	7	**0.0466**	down	5	3	0.3633	no change	3.20	2.33
Ulex spp.	13	4	**0.0245**	down	8	3	0.1133	no change	1.71	0.00
Viburnum lantana	13	0	**0.0001**	down	7	0	**0.0078**	down	1.86	0.00
Viburnum opulus	31	14	**0.0080**	down	18	11	0.1325	no change	1.72	1.27
Clematis vitalba	2	0	0.2500	no change	1	0	0.5000	no change	2.00	0.00
Cytisus scoparius	6	4	0.3770	no change	3	2	0.5000	no change	2.00	2.00
Euonymus europaeus	30	27	0.3957	no change	11	12	0.5000	no change	2.73	2.25
Juniperus communis	14	7	0.0946	no change	3	2	0.5000	no change	4.67	3.50
Sambucus nigra	176	150	0.0830	no change	51	46	0.3424	no change	3.45	3.26
Frangula alnus	1	2	0.5000	no change	1	2	0.5000	no change	1.00	1.00
Prunus laurocerasus	9	14	0.2024	no change	5	4	0.5000	no change	1.80	3.50
Prunus spinosa	59	60	0.5000	no change	27	26	0.5000	no change	2.19	2.31
Rhododendron ponticum	35	40	0.3222	no change	15	16	0.5000	no change	2.14	2.29
Prunus padus	16	21	0.2557	no change	12	10	0.4159	no change	1.33	2.10
Trees										
Betula pendula & pubescens	647	521	**0.0001**	down	84	83	0.5000	no change	7.70	6.43
Populus tremula	33	12	**0.0012**	down	14	9	0.2024	no change	2.36	1.33
Quercus robur & petraea	871	779	**0.0125**	down	95	95	0.5289	no change	9.17	8.28
Salix spp.	126	39	**0.0000**	down	51	17	**0.0000**	down	2.47	2.29
Salix viminalis	5	0	**0.0313**	down	4	0	0.0625	no change	1.25	0.00
Sorbus torminalis	11	1	**0.0032**	down	4	1	0.1875	no change	2.75	1.00
Ulmus sp.	202	149	**0.0027**	down	47	42	0.3359	no change	4.30	3.24
Malus sylvestris	12	17	0.2291	no change	11	13	0.4194	no change	1.09	1.31
Prunus avium	34	41	0.2443	no change	18	21	0.3746	no change	1.89	1.95

Plant species	**Plot no 1971 n=1648**	**Plot no 2001 (n=1621)**	**Binomial - p**	**Direction**	**Site no 1971 (n=103)**	**Site no 2001 (n=103)**	**Binomial - p**	**Direction**	**Mean plots per site 1971**	**Mean plots per site 2001**
Acer campestre	145	136	0.3166	no change	29	32	0.3991	no change	5.00	4.25
Castanea sativa	68	60	0.2681	no change	24	19	0.2712	no change	2.83	3.16
Larix sp.	80	80	0.5315	no change	28	31	0.3974	no change	2.86	2.58
Picea abies	30	22	0.1659	no change	20	11	0.0748	no change	1.50	2.00
Sorbus aucuparia	269	263	0.4142	no change	60	58	0.4634	no change	4.48	4.53
Tilia sp.	5	5	0.6230	no change	2	2	0.6875	no change	2.50	2.50
Tilia x vulgaris	4	1	0.1875	no change	4	1	0.1875	no change	1.00	1.00
Ulmus glabra	22	13	0.0877	no change	6	3	0.2539	no change	3.67	4.33
Acer pseudoplatanus	412	441	0.1689	no change	68	75	0.3080	no change	6.06	5.88
Aesculus hippocastanum	12	16	0.2858	no change	7	6	0.5000	no change	1.71	2.67
Alnus glutinosa	160	164	0.4338	no change	47	48	0.5000	no change	3.40	3.42
Carpinus betulus	22	35	0.0556	no change	4	11	0.0592	no change	5.50	3.18
Fagus sylvatica	258	272	0.2862	no change	58	72	0.1271	no change	4.45	3.78
Fraxinus excelsior	728	722	0.4478	no change	88	89	0.5000	no change	8.27	8.11
Picea sitchensis	13	17	0.2923	no change	7	7	0.6047	no change	1.86	2.43
Salix caprea	59	65	0.3268	no change	24	32	0.1748	no change	2.46	2.03
Salix cinerea	13	14	0.5000	no change	6	11	0.1662	no change	2.17	1.27
Taxus baccata	26	27	0.5000	no change	9	12	0.3318	no change	2.89	2.25
Thuja plicata	0	2	0.2500	no change	0	2	0.2500	no change	0.00	1.00
Tilia cordata	8	12	0.2517	no change	7	5	0.3872	no change	1.14	2.40
Tsuga heterophylla	2	5	0.2266	no change	1	5	0.1094	no change	2.00	1.00
Abies sp.	6	19	**0.0073**	up	5	6	0.5000	no change	1.20	3.17
Pinus nigra	1	12	**0.0017**	up	1	8	**0.0195**	up	1.00	1.50
Pinus sylvestris	49	69	**0.0399**	up	17	20	0.3714	no change	2.88	3.45
Pseudotsuga spp.	6	28	**0.0001**	up	6	15	**0.0392**	up	1.00	1.87
Ilex aquifolium	117	264	**0.0000**	up	46	61	0.0878	no change	2.54	4.33

6.2 Diameter distribution of trees and shrubs

The size distribution for woody species based on counts of stems greater than 5cm, showed a tendency for net loss of stems from the smaller size classes and net recruitment to the 30-60cm interval (Figure 6.1). Thus most shrubs and small trees showed marked declines in stem numbers (Figure 6.2).

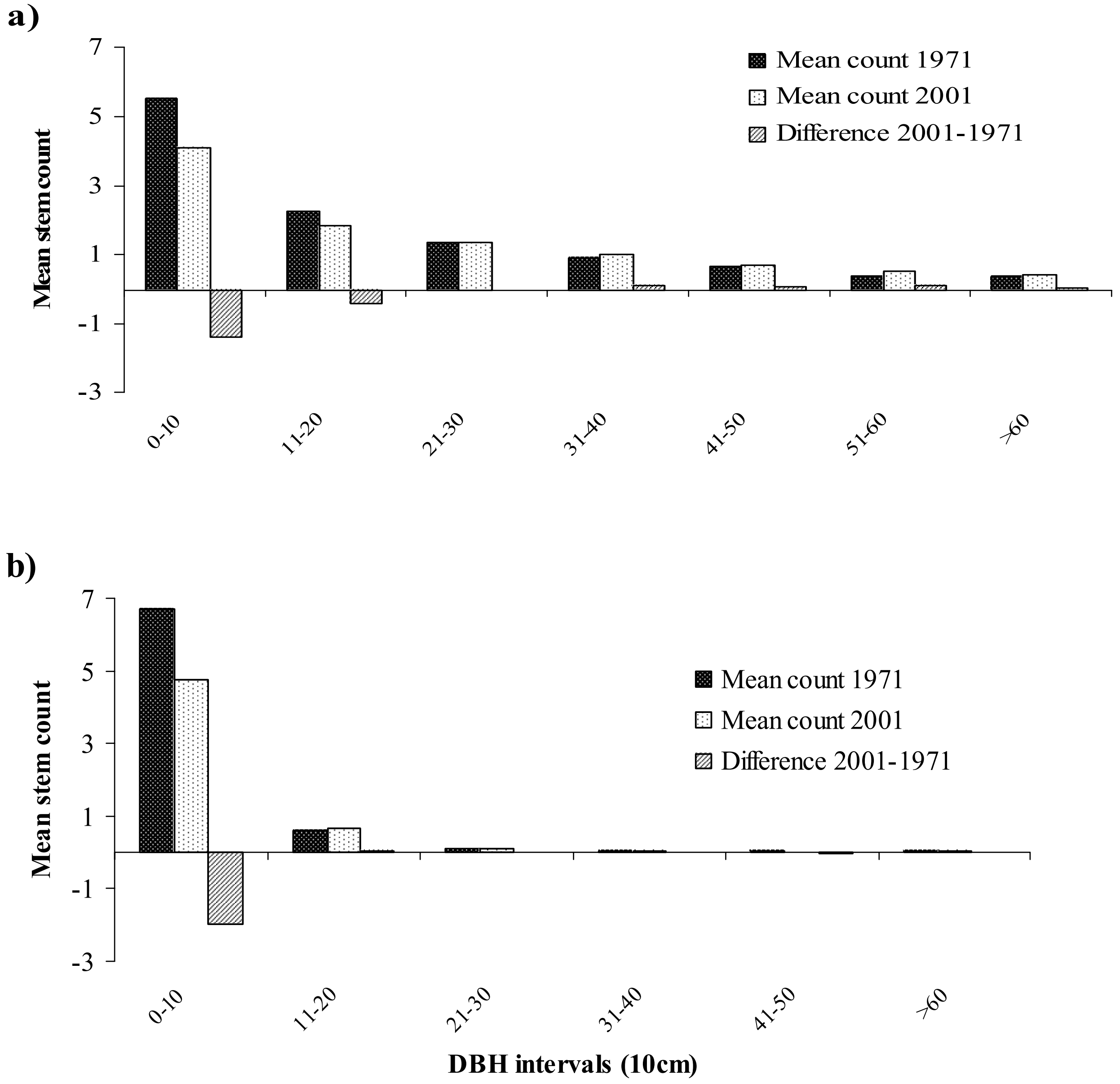


Figure 6.1 Mean count of woody stem per DBH class (10cm intervals) between 1971 and 2001. Note the geometric scale on the y-axis. (n=1647 plots in 1971 and 1610 in 2001). a) trees, b) shrubs. Stems below 6cm were not included.

There was the expected difference in the contribution of species to broad size categories. In the smallest size classes (less than/equal to 30cm) *Corylus avellana* was the biggest contributor in both upland and lowland zones (Figure 6.2). Next highest contributors were *Betula* spp., *Fraxinus excelsior* and *Acer pseudoplatanus*. *Ilex aquifolium* increased its proportional contribution in 2001 particularly in lowland sites. *Quercus* spp made a small contribution to the smallest DBH interval in both zones and both surveys, while *Betula* stems contributed more in the upland woods (albeit a smaller sample of 19 woods), but declined markedly between 1971 and 2001.

The main difference between lowland and upland profiles in the small stems category was the high contribution from *Juniperus communis* in 1971. However this was entirely due to high counts of young stems at Tynron Wood (site 83) where sampling coincided with the juniper woodland on the site. The surveyor's report and Bunce and Hurst (2000) stated that the area the area of juniper woodland within the 1971 boundary had halved by 2001.

In the DBH interval between 30 and 60cm (Figure 6.2b), *Quercus* spp contributed the majority of counted stems in both zones and both surveys. *Betula* spp., *Fraxinus excelsior* and *Acer pseudoplatanus* were the next highest contributors with a larger contribution from *Fagus sylvatica* in lowland sites. Conifers made small contributions overall with *Pinus sylvestris* being the biggest contributor in both zones, but outstripped by *Larix* spp in upland sites in the 2001 survey.

Individuals with DBH greater than 60cm made up a very small fraction of the total count over both surveys; 0.25% of stems in both upland and lowland zones, a reflection of the young age of most broadleaved woods. Older *Quercus* spp, and *Fraxinus* dominated this DBH interval in both zones with *Fagus* again more common in lowland sites (Figure 6.2c).

These general patterns however mask considerable differences between species (Figure 6.3) and between woods as is illustrated for one upland and one lowland site (Boxes 6.1 and 6.2).

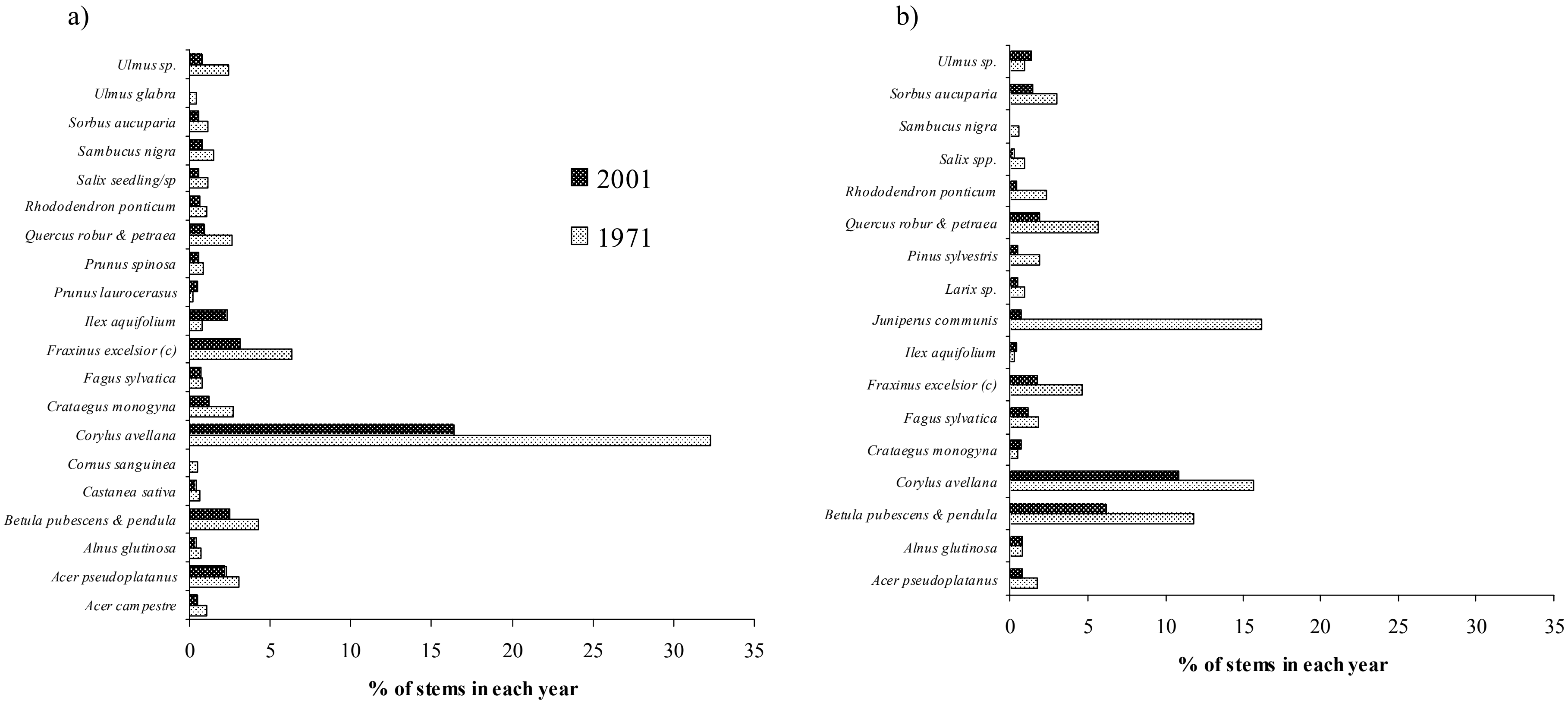


Figure 6.2a Proportional contribution of trees and shrubs to total stem counts **<=30cm DBH**. The percentage is taken over the total stem count in both years. a) Lowland environmental zones (number sites= 84, total stems=180,667, both years combined). b) Upland zones (number sites=19, total stems=18,305, both years combined). Species with <1% contributions have been omitted. Because these are proportions a decline in the contribution of one species can lead to an increase in the relative contribution of another.

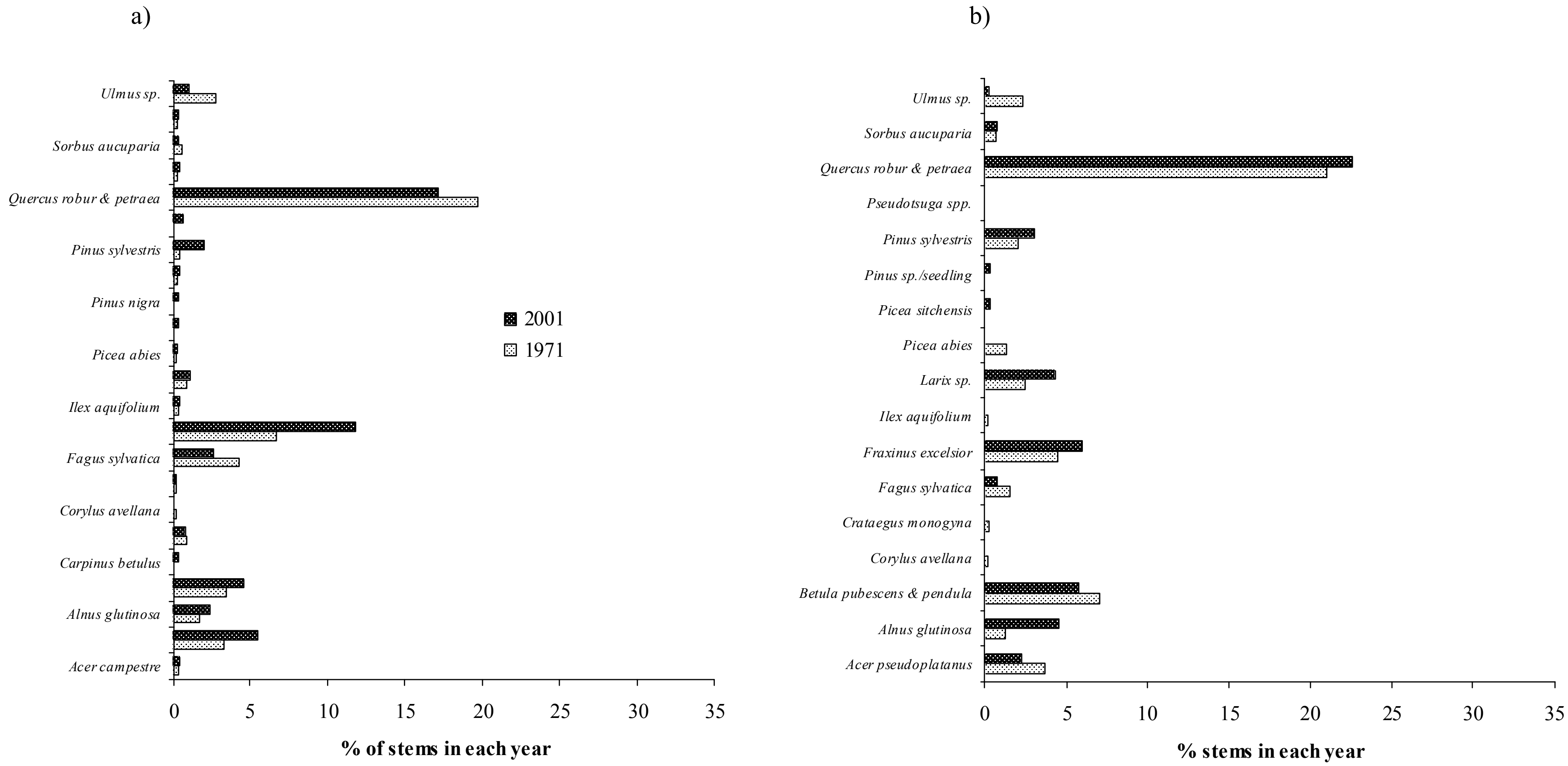


Figure 6.2b Proportional contribution of trees and shrubs to total stem counts between **31-60 cm DBH**. The percentage is taken over the total stems in both years. a) Lowland environmental zones (number sites= 84, total stems=6,055, both years combined). b) Upland zones (number sites=19, total stems=1,161). Because these are proportions a decline in the contribution of one species can lead to an increase in the relative contribution of another.

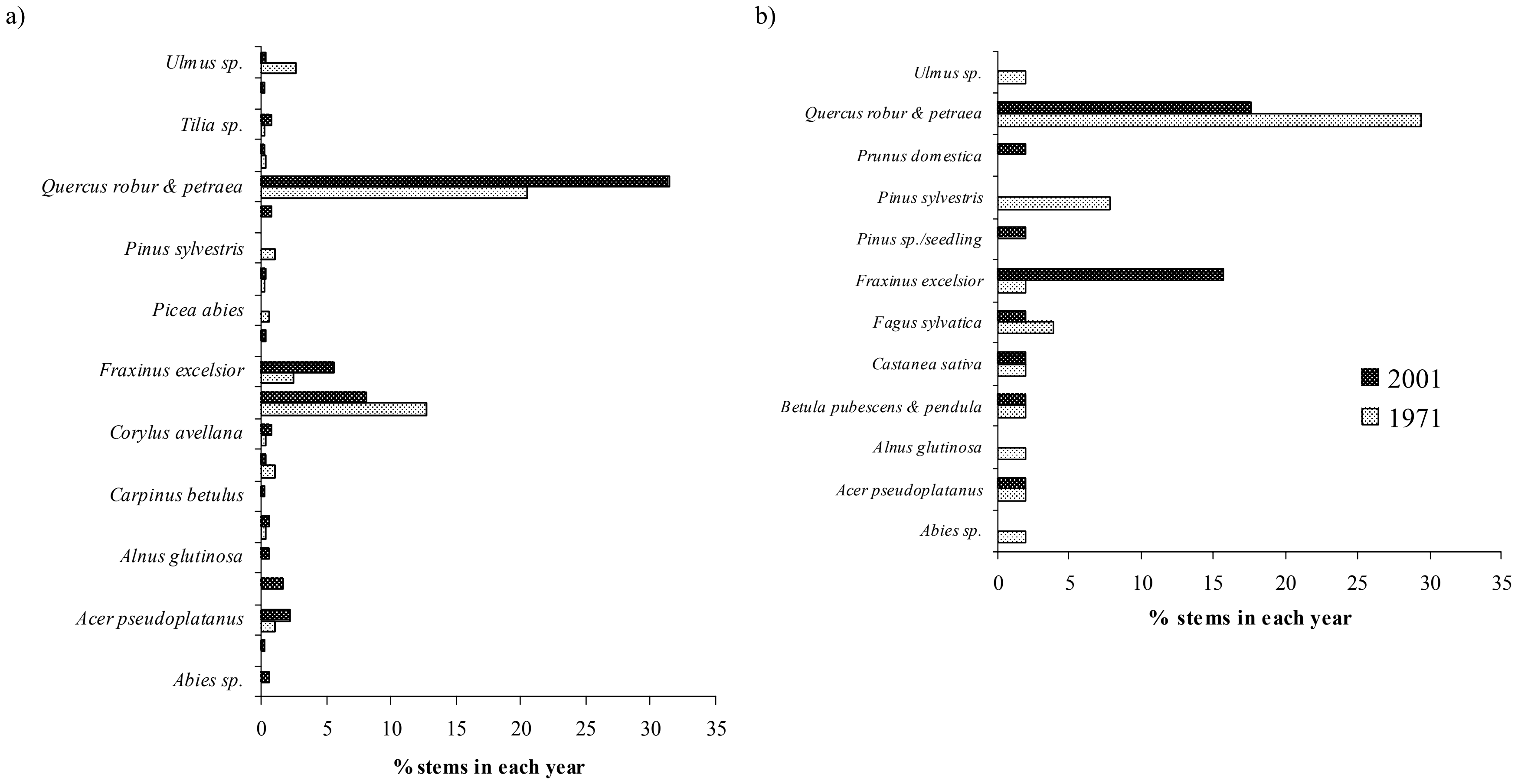
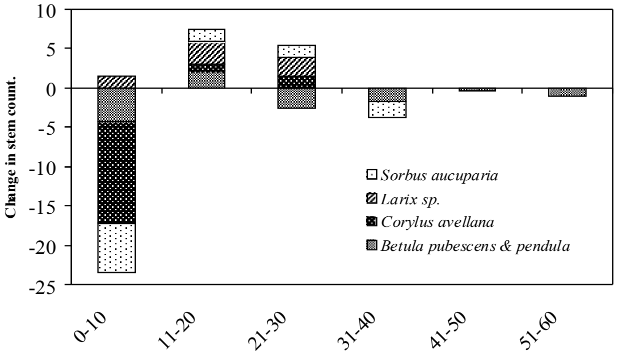


Figure 6.2c Proportional contribution of trees and shrubs to total stem counts **>60 cm DBH**. The percentage is taken over the total stems in both years. a) Lowland environmental zones (number sites= 84, total stems=478, both years combined). b) Upland zones (number sites=19, total stems=51, both years combined). Because these are proportions a decline in the contribution of one species can lead to an increase in the relative contribution of another.

Box 6.1 Glen Orchill Wood (98)

W11 *Quercus petraea-Betula pubescens-Oxalis acetosella*

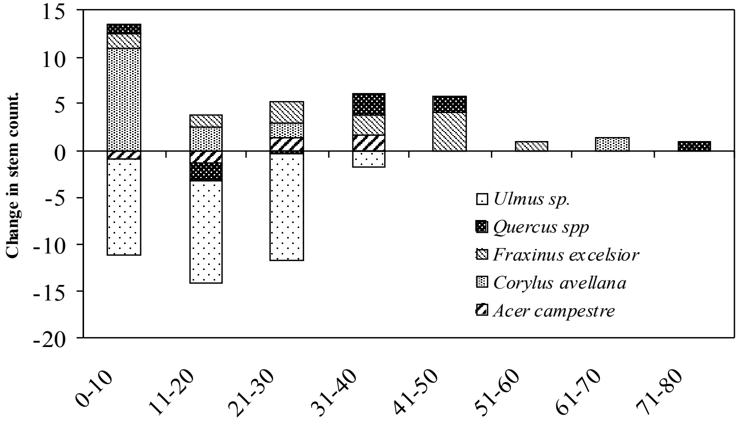
This upland woodland was dominated by *Betula pubescens, B. pendula, Sorbus aucuparia* and *Corylus avellana* covering the steep slopes of several small, narrow valleys. Low ground flora diversity was associated with drier surfaces dominated by *Luzula sylvatica* or *Pteridium aquilinum,* while much richer ground flora communities were associated with localised flushes and the banks of the stream. No active management was apparent at the time of the 2001 survey. Changes between DBH intervals reflect the GB-wide tendency for loss of the youngest stems and modest recruitment to older age classes. *Corylus avellana* was the highest proportional contributor to the <=10cm and also saw the highest proportional reductions in small stems, a pattern also seen at the GB cross-site level. The gain to *Larix* spp across DBH classes also mirrors the GB wide trend. Loss of larger broadleaved trees was attributed to deliberate extraction according to the surveyor's report.



Box 6.2 Papworth Wood (25)

W8 *Fraxinus excelsior-Acer campestre-Mercurialis perennis*

This lowland Cambridgeshire woodland was situated on level ground surrounded by an earthen bank and ditch. When surveyed in 2001 it was found to be part of a much larger woodland following new, adjacent broadleaf planting. Within the original site boundary, the canopy was dominated by *Fraxinus* and *Acer campestre* with *Crataegus monogyna* and patchy *Corylus avellana*. However, the most notable feature, according to the surveyors, was the abundance of largely-dead suckering *Ulmus* sp as well as older collapsed and fallen trees. It was used for recreation and disturbed by fly tipping along parts of the site boundary, but most of the woodland was judged to have had little recent management. Change in the DBH distribution of stems among the canopy dominants clearly indicate the major loss of *Ulmus* between 1971 and 2001. However, in contrast to the GB-wide trend, *Corylus* saw net recruitment to the smallest size-classes as did *Fraxinus*. It was not clear whether the dense growth of young *Corylus,* noted by the surveyors, coincided with canopy gaps from the dead *Ulmus*.



Changes in DBH class for species showing some contrasting patterns for different species are set out below. Oak *Quercus* spp., as one of the commonest species overall (871 plots in 1971 on 95 sites), represents the general trend for loss of individuals from the smallest classes (Figure 6.3a). While there has been little recent recruitment growth into the larger size classes is continuing.

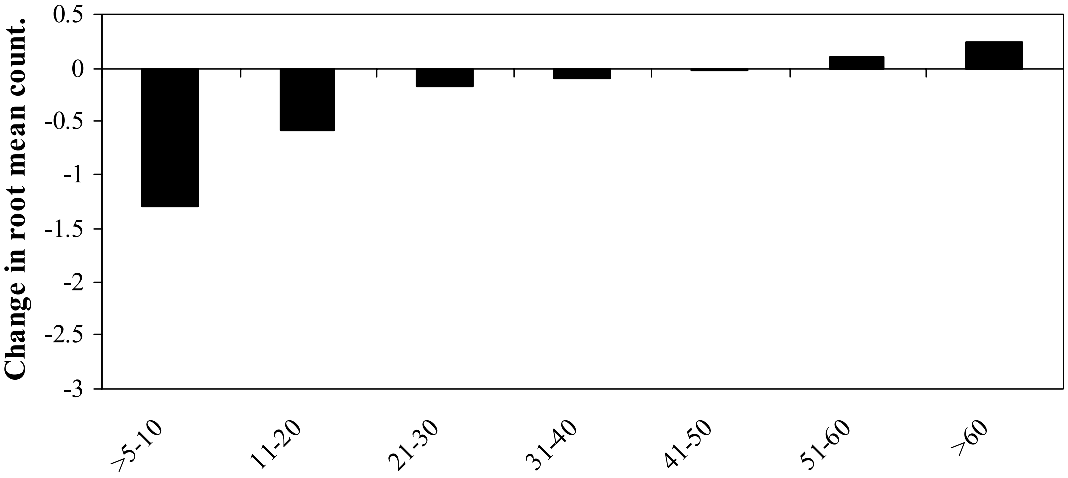


Figure 6.3a Change in size-class (cm, dbh) for *Quercus robur* and *petraea* between 1971 and 2001.

Beech *Fagus sylvatica* showed a pronounced loss across the size range (Figure 6.3) despite a (non-significant) increase in both the number of sites and plots from which it was recorded (Table 6.1). This was also reflected in the increase in the number of plots assigned to beech woodland NVC types (Table 3.2). The tendency for loss across all classes rather than loss and recruitment to older classes as cohorts age, may reflect the various factors that have contributed to increased mortality of beech in recent years including severe drought, wind-storms and attacks by grey squirrels (Mountford 2004; Peterken and Mountford 1998a).

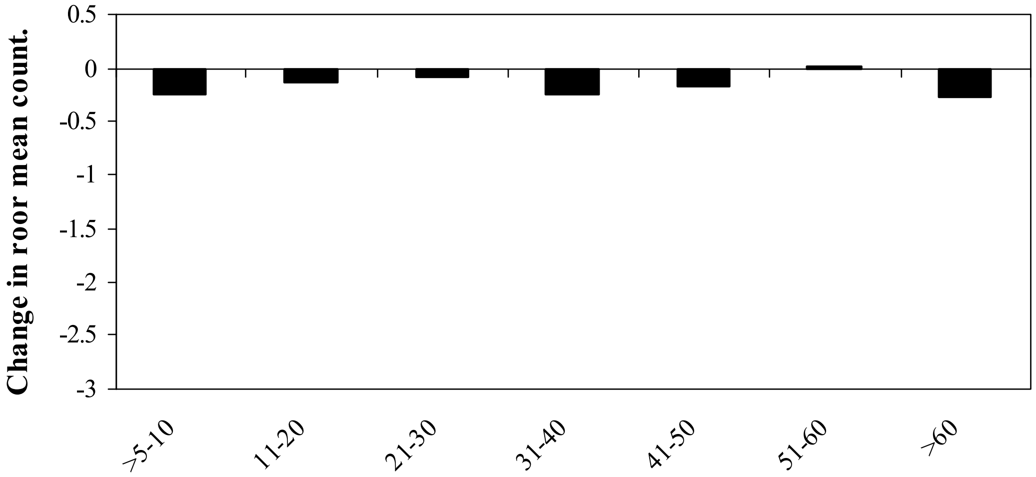


Figure 6.3b Change in size-class (cm, dbh) for *Fagus sylvatica* between 1971 and 2001.

Suckering elm *Ulmus* spp. showed an even more marked decline across all age-classes, consistent with the impact of Dutch Elm Disease (Figure 6.3c), which also affected wych elm

Ulmus glabra. By contrast fir *Abies* spp. (Figure 6.3d) showed an increase in young and older stems although found in few sites and plots overall.

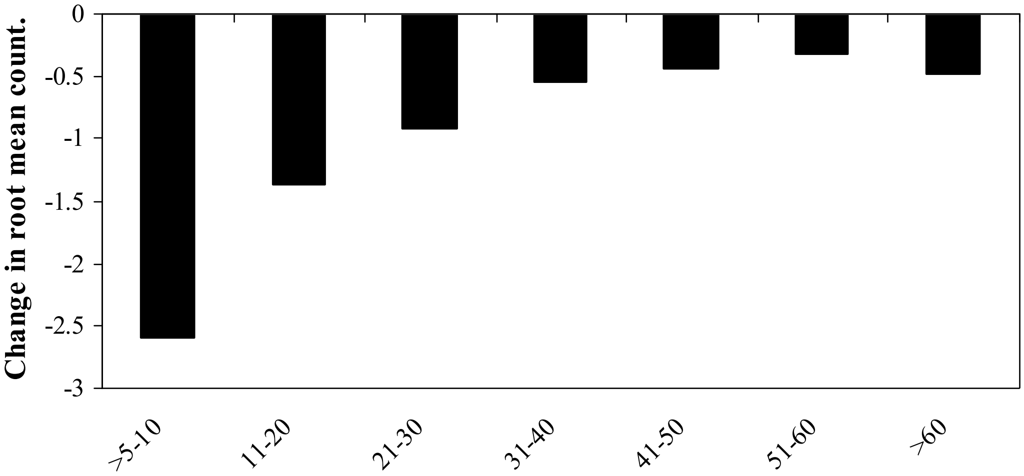


Figure 6.3c Change in age-class (cm, dbh) for suckering elms *Ulmus* spp between 1971 and 2001.

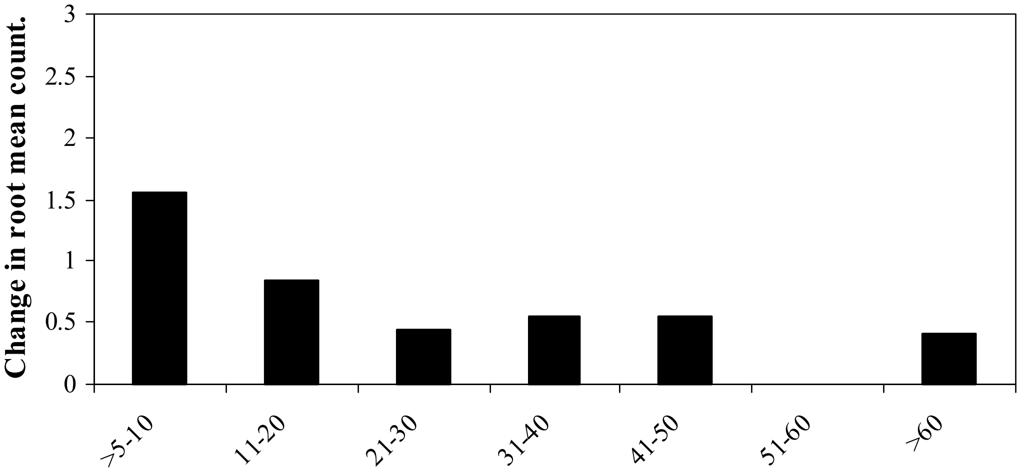


Figure 6.3d Change in size-class distribution for *Abies* spp between 1971 and 2001.

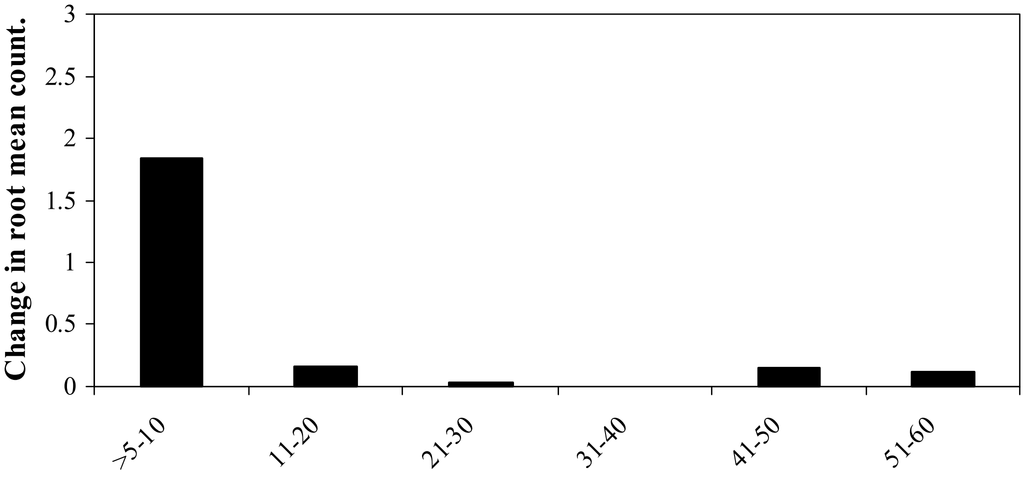
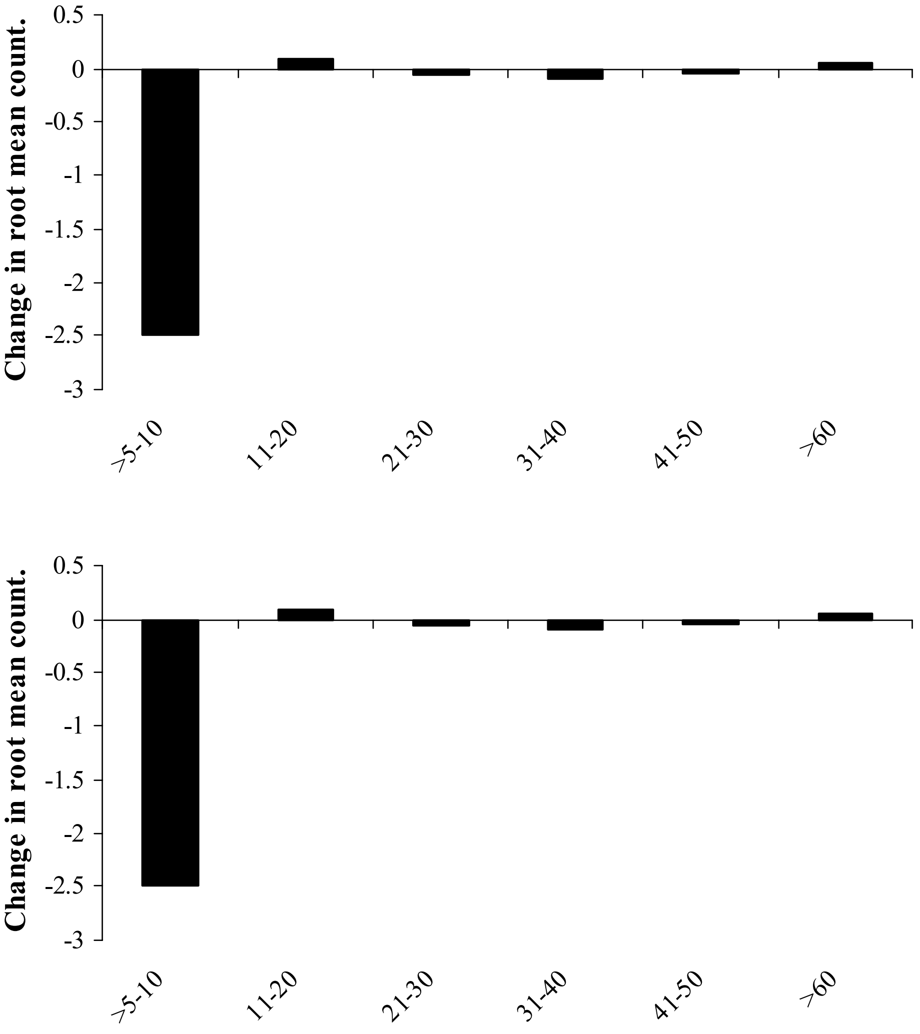


Figure 6.3e Change in age-class distribution for *Ilex aquifolium* between 1971 and 2001.

The recruitment of holly *Ilex aquifolium* in the youngest DBH class (Figure 6.3e) is in line with anecdotal reports of extensive increases in holly abundance across a range of sites in GB (e.g. Mountford 2004). It may have increased because of long-term reductions in grazing levels in some former wood-pasture sites; reduced winter mortality (it is frost-sensitive) because of milder winters; or its strong ability to develop under the shade of mature broadleaved canopies. However, no significant relationships were found between any of the management and climate variables considered and the level of holly increase. Possibly different factors are involved at different sites, hence blurring the signal.



6.3f Change in age-class distribution for *Corylus avellana* between 1971 and 2001

In contrast to holly, small stems of *Corylus avellana* hazel declined.

6.2.1 Diameter distribution and abundance of dead stems

Dead stem numbers in absolute terms were only a small proportion of the total count - 15% of stems counted in 1971 and 6% of those counted in 2001. The distribution of these stems among tree and shrub species reflected the abundance of live stems, being dominated by very high numbers of small diameter hazel *Corylus avellana* (Figure 6.4, 6.5).

Dead stem counts offer an incomplete perspective on the mortality of trees and shrubs because fallen stems and those <1.3m high are not counted. These items were however counted as present when surveyors recorded plot descriptors (Appendix 2). Change in counts for the three categories of larger woody debris showed a significant increase in occurrence in plots. This change was concentrated in lowland sites with particularly large increases in the October 1987 storm sites (Appendix 4).

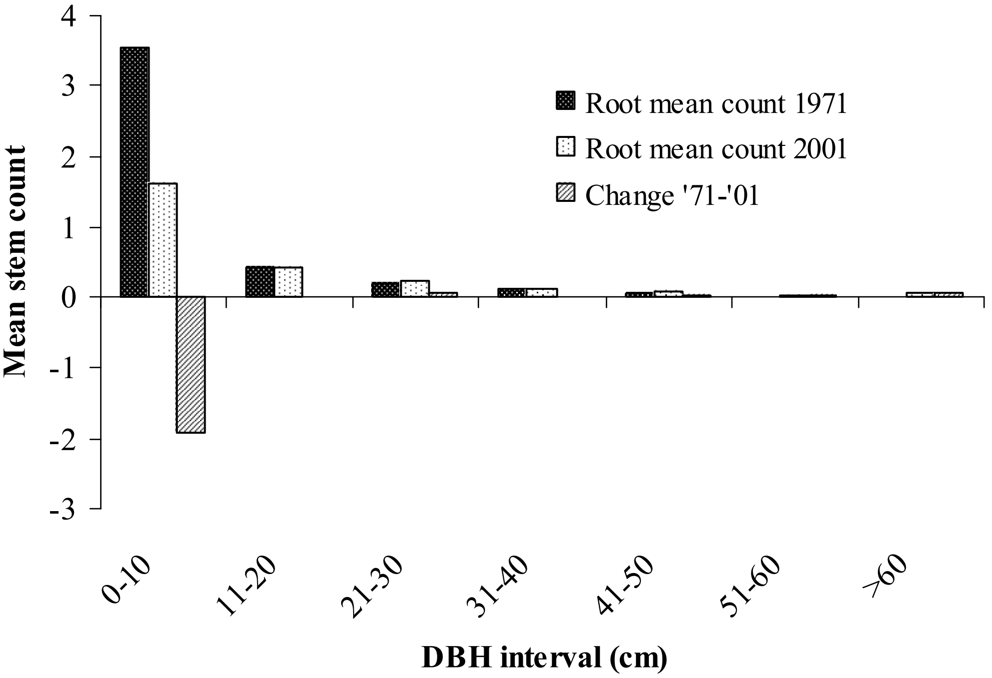


Figure 6.4 Mean count of dead woody stems per DBH class (10cm intervals) between 1971 and 2001. Counts were square root transformed to allow the very high counts in the smallest classes to be graphed alongside older classes. (n=1647 plots in 1971 and 1610 in 2001).

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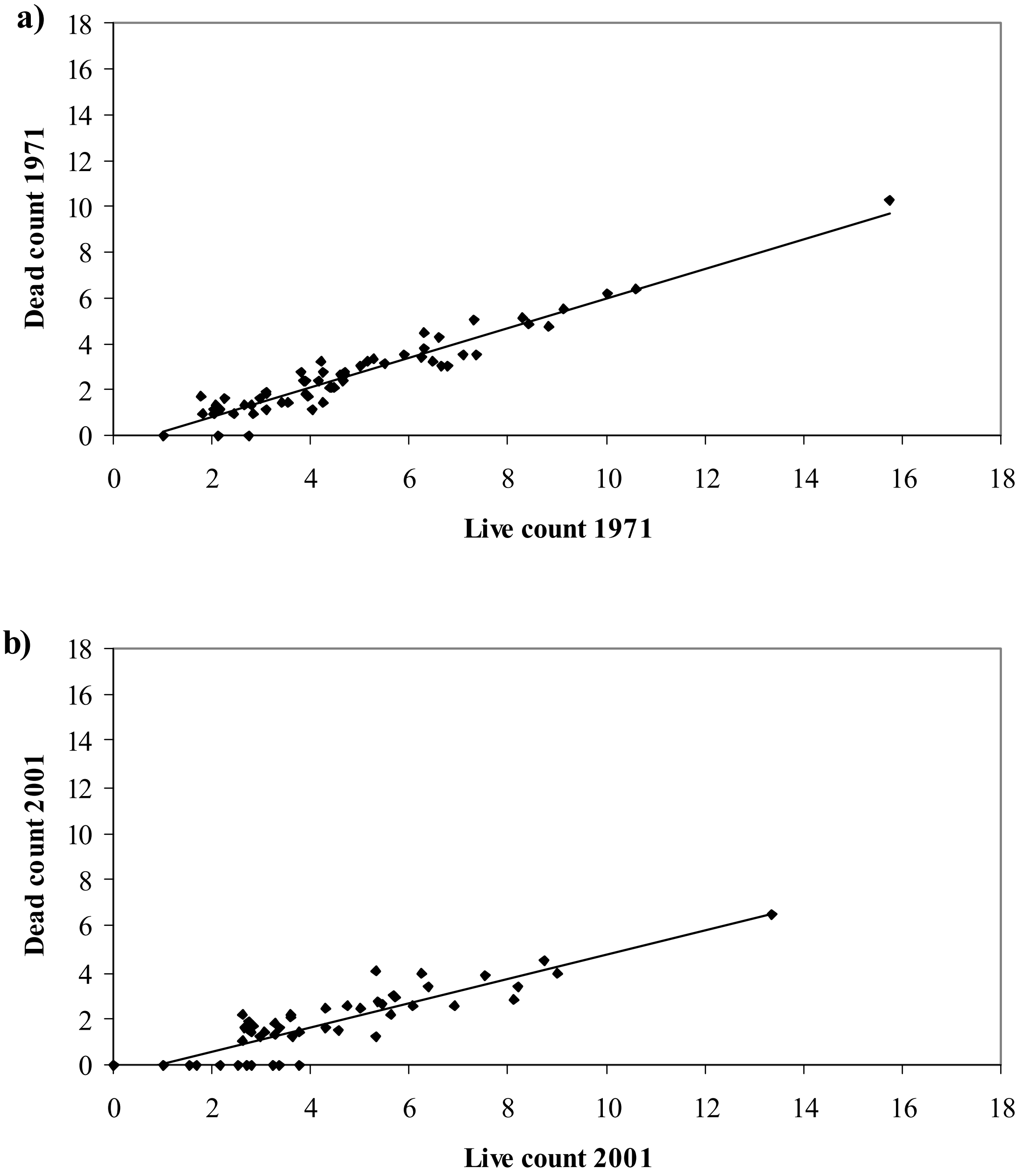


Figure 6.5 Numbers of dead stems versus live stems in a) 1971 and b) 2001, for each tree and shrub species. Each point is a species. Counts have been 4th-root transformed so that the top right outlying point for *Coryllus avelana* could be plotted on the same graph.

6.3 Change in basal area of trees and shrubs between 1971 and 2001

Basal area of trees and shrubs was calculated from the stem diameter measurements in each plot at each time point. Across all strata, mean basal area increased between surveys (Table 6.2). The values should be seen as an index of basal area change rather than an absolute measure. They are based on estimates from small plots and also include the small stems and shrubs that are not normally considered in forestry basal area calculations. This has inflated the values compared to those normally found in forestry yield tables.

From an ecological perspective the contribution of shrub stems to the biomass of the stand or to changing shade at ground level is no different in principle to the contribution of tree stems, hence their inclusion in this analysis. However since numbers of stems of the main shrub hazel, and of small stems more generally declined (Figure 6.1, 6.2) the net increase in basal area represents an increased contribution from the larger stems.

Table 6.2 Change in basal area of shrubs and trees (m^2 per $200m^2$ plot).

Stratum	**Mean 1971**	**Mean 2001**	**Adj. mean change[1]**	**se**	**Df**	**P**
GB	1.31	1.55	0.24	0.052	104	**<0.0001**
Lowland zones	1.37	1.63	0.28	0.067	84.1	**<0.0001**
Upland zones	1.04	1.16	0.17	0.115	17.7	0.1679

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

Changes in basal area were reasonably normally distributed at both plot and site level. Hence, the results did not reflect the large influence of a few high magnitude changes but rather change was more evenly spread across the whole sample (Figures 6.6, 6.7). There was no change in the basal area of dead woody stems.

Basal area change was negatively correlated with plot level management changes, based on records of recent activity such as cut stumps, as summarised on the tick sheets for each plot (Table 6.3). This only explained 0.4% of the variation, but there were few signs of recent management recorded in either year. The other variable tested against basal area was site level management change, but no significant relationship was found.

Table 6.3 Change in basal area in relation to possible explanatory variables

Explanatory variable	**Df**	**F**	**P**	**Relationship between effect and basal area change**
Change in number of plot level signs of recent management	1343	5.20	**0.0227**	-

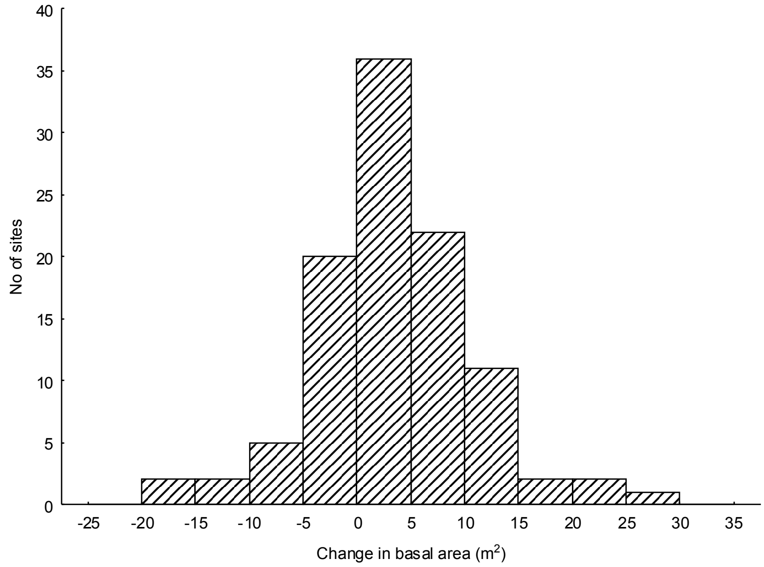


Figure 6.6 Distribution of site-level change in basal area between surveys (m^2 per 16 x $200m^2$ plot).

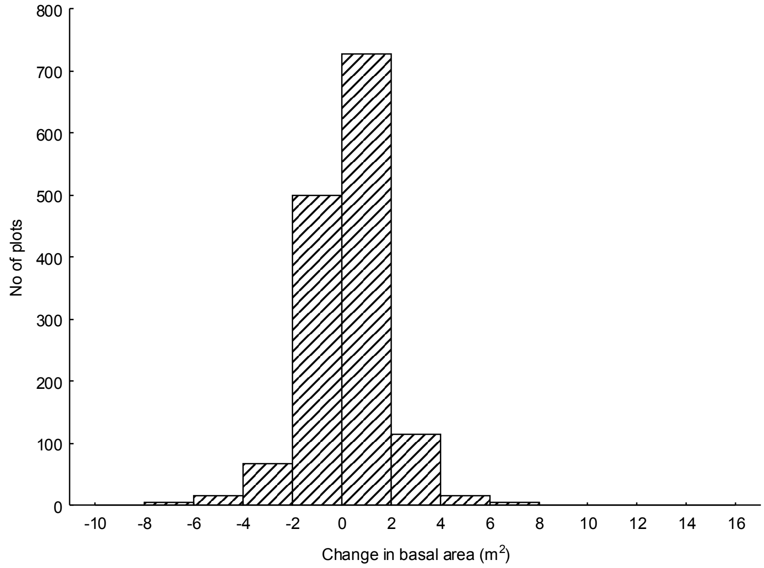


Figure 6.7 Distribution of plot level change in basal area between surveys (m^2 per $200m^2$ plot).

6.4 Regeneration

6.4.1 Site level sapling regeneration

Species richness of regenerating trees and shrubs (stems between 25 cm and 130 cm high) declined significantly at plot and site levels across GB and in the Lowland zone generally (Table 6.4), but not in the October '87 storm sites (see Appendix 4). The decline in the abundance and diversity of the youngest woody stems in British woodlands parallels the losses in the 6-10 cm dbh category for stems greater than 1.3m high (Figure 6.1).

6.4.2 Plot level sapling regeneration

None of the explanatory variables considered were significantly correlated with change in species richness of saplings at the plot level. Some of the more shade tolerant species did however show small increases including *Taxus baccata, Fagus sylvatica* and *Ilex aquifolium* (Figure 6.8) against the general tendency for a decline shown by other species.

Table 6.4 Changes in species richness of saplings at the site level.

Stratum	**Mean 1971**	**Mean 2001**	**Adj. mean change[1]**	**se**	**df**	**p**
GB	8.07	6.38	-0.13	0.022	102	<0.0001
Upland zones	7.56	5.22	-0.12	0.060	18	0.0533
Lowland zones	8.19	6.65	-0.13	0.023	83	**<0.0001**

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

Table 6.5 Changes in species richness saplings at the plot level.

Stratum	**Mean 1971**	**Mean 2001**	**Adj. mean change[1]**	**se**	**Df**	**p**
GB	0.12	0.08	-0.04	0.006	101	<0.0001
Upland zones	0.08	0.05	-0.02	0.012	17	0.0994
Lowland zones	0.13	0.09	-0.04	0.007	81.6	**<0.0001**

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

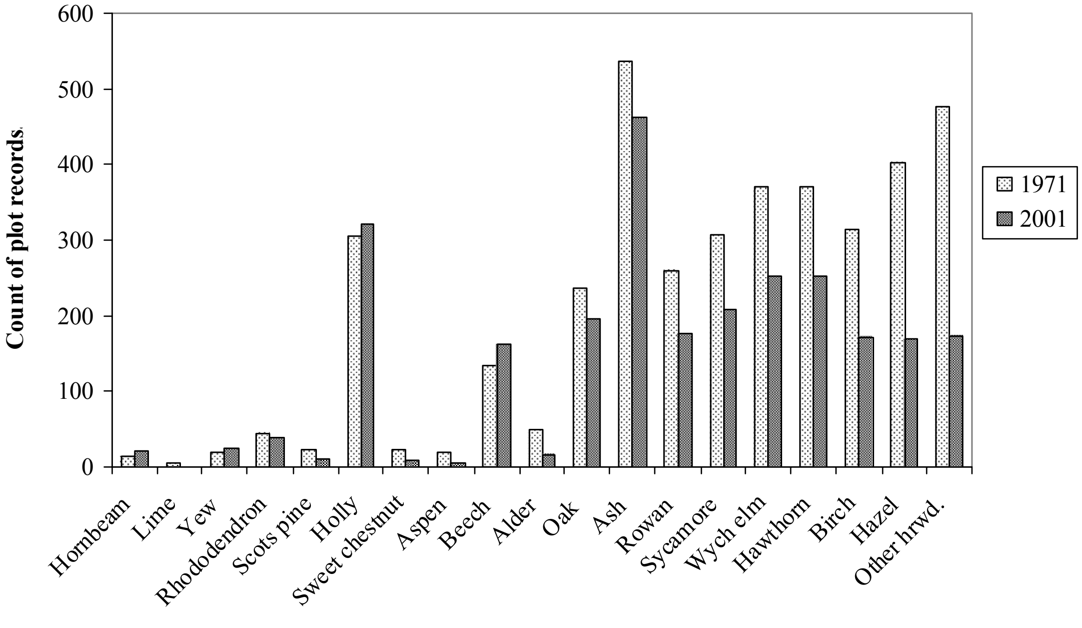


Figure 6.8 Change in recorded occurrences of saplings (stems between 25 and 130 cm high) between 1971 and 2001.

Table 6.6 Counts of tree and shrub seedlings (or vegetative regrowth) less than 25 cm high in 1971 and 2001. The probability of a significant difference between surveys is given by the binomial formula.

Plant species	Plot count '71	Plot count '01	binomial p	Direction
Acer campestre	78	56	**0.0346**	down
Acer pseudoplatanus	396	213	**0.0000**	down
Betula pendula & pubescens	371	248	**0.0000**	down
Castanea sativa	290	134	**0.0000**	down
Cornus sanguinea	36	12	**0.0004**	down
Corylus avellana	194	153	**0.0158**	down
Crataegus monogyna	380	224	**0.0000**	down
Cytisus scoparius	23	12	**0.0448**	down
Fraxinus excelsior	812	651	**0.0000**	down
Ligustrum vulgare	85	38	**0.0000**	down
Lonicera periclymenum	699	508	**0.0000**	down
Prunus padus	16	2	**0.0007**	down
Prunus spinosa	67	48	**0.0464**	down
Quercus robur & petraea	585	339	**0.0000**	down
Rosa spp.	288	145	**0.0000**	down
Sorbus aucuparia	440	230	**0.0000**	down
Taxus baccata	47	21	**0.0011**	down
Ulmus glabra	69	14	**0.0000**	down
Viburnum opulus	85	23	**0.0000**	down
Alnus glutinosa	12	9	0.3318	no change
Euonymus europaeus	21	13	0.1147	no change
Fagus sylvatica	157	171	0.2365	no change
Pinus sylvestris	28	19	0.1215	no change
Prunus avium	28	34	0.2629	no change
Rhododendron ponticum	47	58	0.1646	no change
Ribes uva-crispa	23	21	0.4402	no change
Salix spp.	44	35	0.1841	no change
Sambucus nigra	68	58	0.2114	no change
Ulex sp.	23	16	0.1684	no change
Carpinus betulus	4	15	**0.0096**	up
Ilex aquifolium	328	392	**0.0094**	up
Prunus laurocerasus	2	10	**0.0193**	up
Ulmus sp.	7	40	**0.0000**	up

6.4.3 The abundance of tree and shrub seedlings

The majority of woody species saw seedling frequency decline between surveys (Table 6.6). *Ilex aquifolium* increased in frequency consistent with its general increase in other layers. The other significant increasers were shade-tolerant species, but being present in small numbers may simply reflect within-site patterns in few locations rather than a more geographically widespread pattern. By contrast the shade-tolerant *Taxus baccata* showed a decline.

6.5 Changes in indicators of woodland disturbance

6.5.1 Open habitats

The number of Open Habitats per plot as recorded on the tick cards in sample plots, declined significantly at the GB level and in both upland and lowland zones (Table 6.7). Large and small glades, and paths less than 5m wide declined most (Figure 6.9). Most aquatic habitats also declined in occurrence (Figure 6.10). Open Habitats also declined at site level (Table 6.8).

Table 6.7 Changes in the occurrence of Open Habitats within plots.

Stratum	Mean 1971	Mean 2001	Adj. mean change[1]	Se	df	P
GB	0.13	0.08	-0.03	0.007	101	**<0.0001**
Upland zones	0.19	0.10	-0.07	0.017	17	**0.0013**
Lowland zones	0.11	0.07	-0.02	0.008	81.9	**0.0060**

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

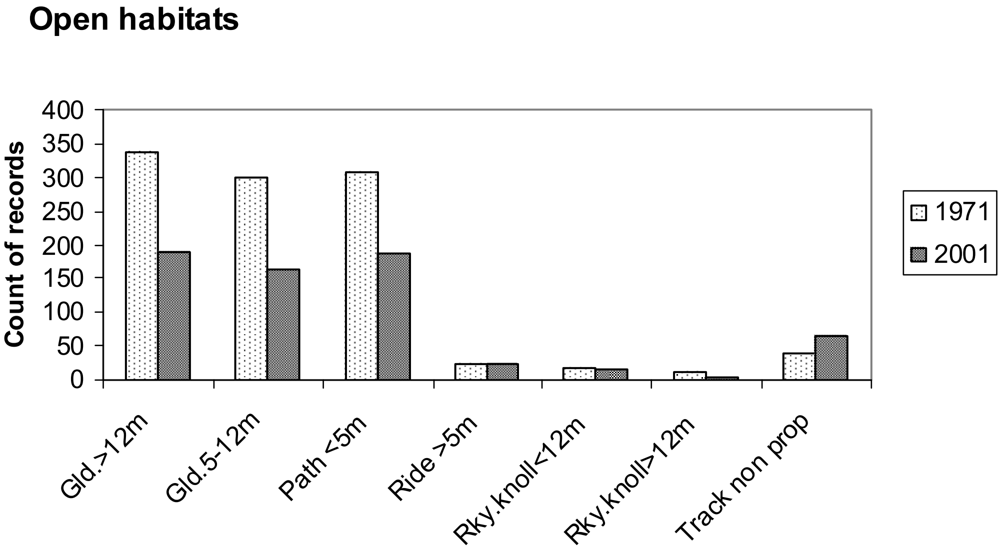


Figure 6.9 Changes in occurrence of open habitats in woodland survey plots (gld = glade, rky = rocky, track non prop = unmade up track).

Table 6.8 Change in occurrence of Open Habitats across sites.

Stratum	Mean 1971	Mean 2001	Adj. mean change[1]	se	df	p
GB	0.38	0.27	-0.13	0.029	102	**<0.0001**
Upland zones	0.49	0.40	-0.15	0.085	18	0.096
Lowland zones	0.36	0.24	-0.13	0.030	83	**<0.0001**

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

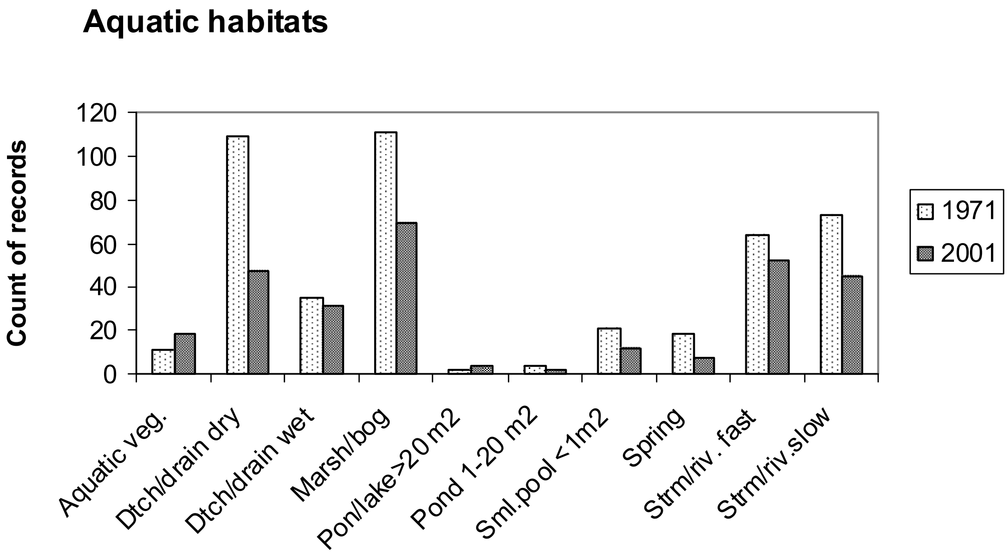


Figure 6.10 Change in occurrence of aquatic habitats in woodland survey plots (Dtch = ditch, sml = small, strm = stream, riv = river).

Table 6.10 Change in signs of recent management at site level.

Stratum	Mean 1971	Mean 2001	Adj. mean change[1]	se	Df	P
GB	0.28	0.28	-0.006	0.058	102	0.9116
Upland zones	0.30	0.11	-0.20	0.069	18	**0.0089**
Lowland zones	0.28	0.31	0.04	0.069	83	0.5919

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

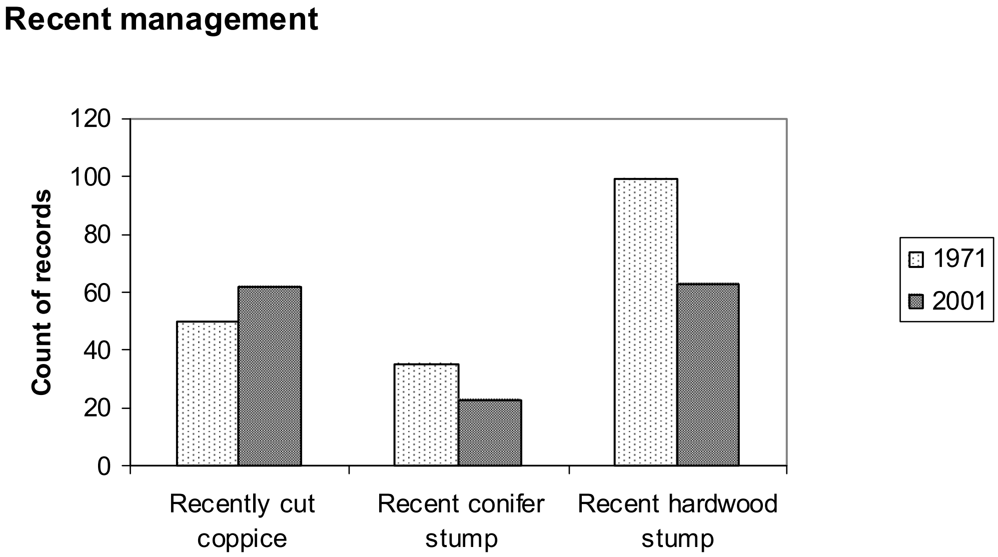


Figure 6.11 Changes in plot level management signs between 1971 and 2001

6.5.2 Site level signs of recent management

At the GB level, site level counts of management signs did not change significantly although there was a significant reduction in upland zone sites. When broken down by type of plot-level management, slightly more counts of recently cut coppice were recorded in 2001 but substantially fewer signs of large stumps.

6.5.3 Woodland activities noted in the surveyors' reports

In 2001 surveyors produced reports for each wood to complement the data from the tick cards. The reports included sections relating to recent management, but for 68 sites nothing was noted (Table 6.10, Appendix 12)

Public recreation featured more prominently in the reports than in the tick card analysis, where for example the count of paths had declined. In many cases the prevalence of public recreation centred on public or permissive paths traversing an otherwise impenetrable and unmanaged wood. However, there were also woods where recreational activity was an integral part of the management of the site particularly those that were local authority owned or associated with nearby urban centres (eg Lower Nut Hurst, Den of Alyth and Dulwich Wood).

Table 6.10 The most common management and related activity from the surveyors' reports for 2001. All other activities were reported from seven or fewer sites.

		Number of sites
a.	*Forestry management*	
	No recent management noted	68
	Selective felling	23
	Coppicing	19
	Broadleaf planting	18
b.	*Removal of introduced species*	
	Exotic removal (unspecified)	9
	Sycamore/rhododendron control	5
c.	*Grazing/browsing issues*	
	Overgrazing – sheep	10
	Overgrazing – deer	7
	Deer fencing	9
d.	*Recreation/sporting/access issues*	
	Public recreation	41
	Pheasants	19
	Rubbish dumping	9
e.	*Storm damage*	9

6.5.4 Grazing signs

There was no overall change in the number of grazing signs recorded on the tick cards at a GB level (Table 6.11) but there was an increase for the lowland zone. The biggest increase was for 'Other deer' (i.e not red deer), with little change for other animals (Figure 6.12). As deer were the most common grazing animal recorded in the Lowland zone (Figure 13) in 2001 this is consistent with other evidence for increases of deer in lowland woods (Fuller and

Gill 2001). There is however considerable variation with some lowland sites showing deer signs in most plots and others with none.

Table 6.11 Number of signs of grazing (site level tick cards)

Stratum	Mean 1971	Mean 2001	Adj. mean change[1]	se	df	p
GB	3.78	4.41	0.81	0.599	102	0.1817
Upland zones	5.11	3.37	-2.21	1.244	18	0.0925
Lowland zones	3.87	4.26	1.48	0.659	83	**0.0266**

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

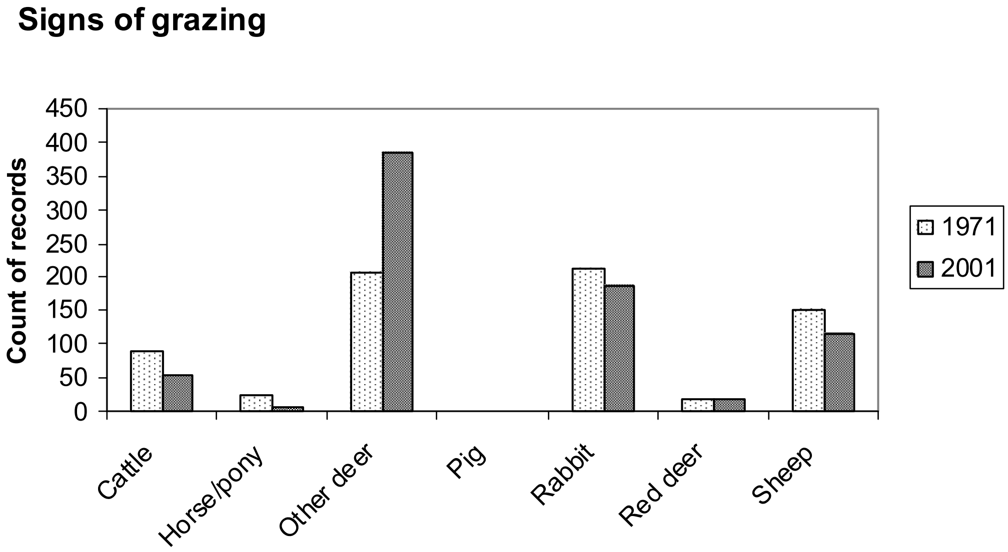
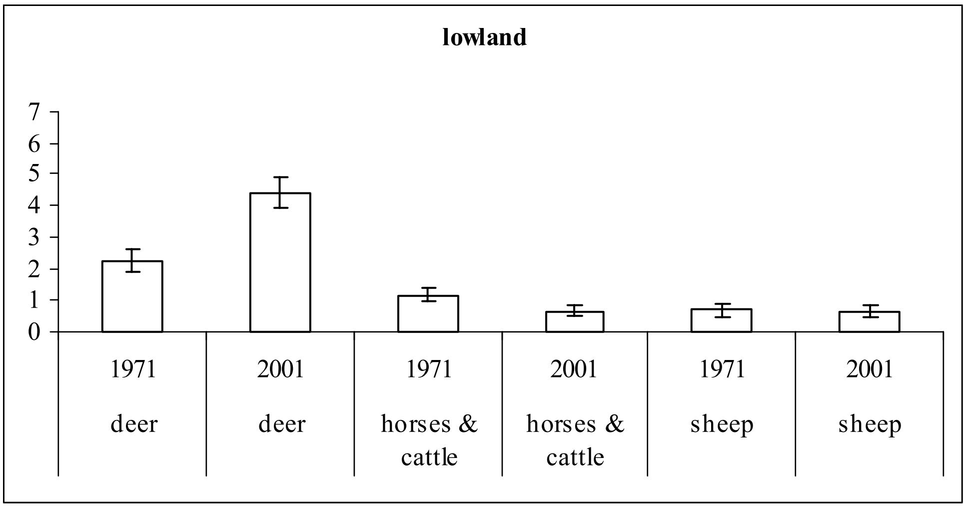


Figure 6.12 Change in count of grazing signs between 1971 and 2001.

There were only a small number of indications of cattle and horse grazing. Sheep grazing signs, surprisingly, declined slightly over time (Figure 6.12), even though there have been dramatic increases in the early to mid-eighties (Fuller and Gough 1999) in sheep numbers in parts of upland England and Wales.

An increase in grasses is often linked to changes in grazing levels (e.g. Kirby and Thomas 2000). In this study there was no detectable cross-site increase in grass cover or significant relationships between plot-level variation in grass cover change between surveys and any of the expected drivers of change (including nitrogen deposition, growing season length or grazing signs). There was however a negative correlation between change in grass cover and tree/shrub basal area (increased woody basal area being associated with decreased grass cover).



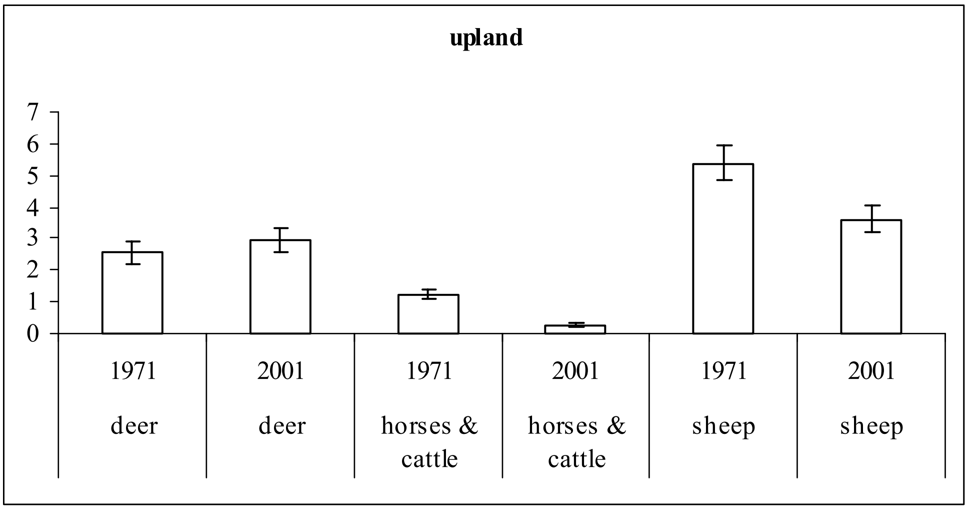


Figure 6.13 Counts of grazing signs across all plots in each site in the 2001 survey.

6.6 Summary of tree and shrub/management changes

Between 1971 and 2001 the main changes were:

- Most tree and shrub species remained stable in terms of their frequency of occurrence at plot and site levels, although 15 species (eight of these shrubs) declined, whilst five other species (four conifers) increased.
- There was a net loss of stems from the smallest size classes (particularly less than 10 cm dbh) with some smaller gains in the 30-60 cm classes. Stems greater than 60 cm remained scarce.
- Individual species showed distinct patterns: for example
  - oak lost stems in the lowest size classes but gained in the larger ones;
  - elm and beech lost stems across the size class range;
  - holly increased substantially in the smallest size class;
  - hazel lost small stems.

- Mean basal area of trees and shrubs increased both for individual plots and across most sites.
- Species richness amongst saplings (25-130cm high) decreased, but small increases in frequency were shown by some shade tolerant species including yew, beech and holly.
- Seedling (< 25 cm high) frequency declined for most species, but holly showed a notable increase.
- Open habitats (rides, glades etc) and some wet habitats (ditches, boggy patches) became less common.
- Grazing signs increased in the lowlands; across GB the biggest increase in recorded grazing signs was for ‘Other deer’ (i.e. species other than red deer).

7. Ground flora changes

This section looks at the changes in the ground flora, including occurrence and abundance of different species and variations in their Ellenberg values and other plant attributes. The most frequently recorded species across both survey are shown in Table 7.1.

Table 7.1 The most frequently recorded species (no of sites) across both years.

Lowland sites (n=84)	1971	2001
Rubus fruticosus agg.	84	83
Dryopteris filix-mas	81	75
Quercus spp	79	61
Dryopteris dilatata/carthusiana	78	77
Viola riviniana/reichenbiana	77	71
Fraxinus excelsior (c)	76	76
Circaea lutetiana	76	71
Lonicera periclymenum	75	73
Urtica dioica	74	71
Geum urbanum	74	67
Crataegus monogyna	74	60
Pteridium aquilinum	72	64
Epilobium montanum	72	38
Poa nemoralis/trivialis	72	53
Hyacinthoides non-scripta	72	69
Hedera helix	71	67
Acer pseudoplatanus	71	50
Deschampsia cespitosa	69	60
Geranium robertianum	68	68
Oxalis acetosella	68	57
Holcus lanatus	66	54
Dactylis glomerata	65	50
Mercurialis perennis	65	56
Carex sylvatica	65	55
Ajuga reptans	64	48
Fragaria vesca	63	30
Galium aparine	62	62
Holcus mollis	62	54
Agrostis capillaris	62	55
Ranunculus repens	60	62
Ilex aquifolium	58	59

Upland sites (n=19)	1971	2001
Agrostis capillaris	19	18
Oxalis acetosella	19	19
Anthoxanthum odoratum	19	16
Sorbus aucuparia	19	15
Viola riviniana/reichenbiana	18	15
Holcus mollis	18	16
Galium saxatile	18	15
Juncus effusus	18	14
Dryopteris filix-mas	18	15
Dryopteris dilatata/carthusiana	18	17
Pteridium aquilinum	18	18
Holcus lanatus	18	17
Deschampsia flexuosa	18	17
Ranunculus repens	18	12
Rubus fruticosus agg.	17	17
Vaccinium myrtillus	17	17
Epilobium montanum	17	13
Athyrium filix-femina	17	13
Cirsium palustre	17	11
Betula spp	17	12
Calluna vulgaris	17	13
Potentilla erecta	16	15
Digitalis purpurea	16	15
Cardamine hirsuta/flexuosa	16	8
Fraxinus excelsior (c)	16	14
Blechnum spicant	16	13
Agrostis canina sens.lat.	16	13
Lysimachia nemorum	15	14
Lonicera periclymenum	15	11
Luzula campestris/multiflora	15	14
Ilex aquifolium	14	14

7.1 Changes in ground layer species richness (excluding bryophytes and lichens)

A highly significant reduction in mean plot-level species richness occurred across GB (Table 7.2). The results, after taking into account day differences in survey date indicate that, on average, eight higher plant species were lost per plot from broadleaved woodlands in the thirty year interval, that is about 36% of the initial number per plot. This was true for both ancient and recent woods within the samples (Appendix 9).

Table 7.2. Change in average ground flora species-richness per plot.

Stratum	Mean 1971	Mean 2001	Adj. mean change[1]	se	Df	p
GB	22.6	16.7	-8.14	0.828	99.8	**<0.0001**
Upland zones	28.6	18.9	-6.93	0.816	81.4	**<0.0001**
Lowland zones	21.2	16.2	-9.94	1.752	16.9	**<0.0001**

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

Table 7.3 Change in ground flora species-richness per site (sixteen plots)

Stratum	Mean 1971	Mean 2001	Adj. mean change[1]	se	Df	p
GB	90.1	77.3	-12.79	2.771	102	**<.0001**
Upland zones	104.7	85.32	-19.40	6.448	18	**0.0076**
Lowland zones	86.8	75.5	-11.30	3.063	83	**0.001**

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

Significant reductions were also seen across GB level (Table 7.3) in the species richness change at the site level (ie counting species only once across all plots in each site in each year). The reduction in overall number does not necessarily mean that an average of 12-13 species have become extinct from each site but it does show a reduction in general species abundance. As a species becomes less abundant across a site then the chances of it being recorded in one or other of the 16 plots at a site is reduced.

Previous analysis of the 14 pilot sites (Smart and others 2001) indicated a positive relationship between species-richness in 1971 and the proportion of species lost per site. This was in turn associated with location in that northern sites tended to be more species-rich than southern ones. With the larger sample size, the species-loss versus latitude correlation is somewhat changed (Figure 7.1); in particular southern sites showed more variation in how they have changed. Where losses occurred these were often larger than northern sites, but a greater number of sites in the south also gained species, including the ten sites within the October 1987 storm track.

Relationships between plot-level species-richness in 1971 and change in richness were not explored because a statistical 'regression to the mean' effect is likely to produce a negative and highly significant correlation. This is a statistical artefact because unusually high or low values at time one may reflect sampling variation and will rebound back to average levels at time two (Smart and Scott 2005).

Despite the general cross-site pattern of species-richness decline there were eight sites where significant increases had occurred and 37 where mean change was so variable within a site that no statistically significant change could be detected. Increases in richness or stability were mainly associated with central and south east England (Figure 7.1), partly attributable to the effects of the October 1987 storm (see Appendix 4, 9).

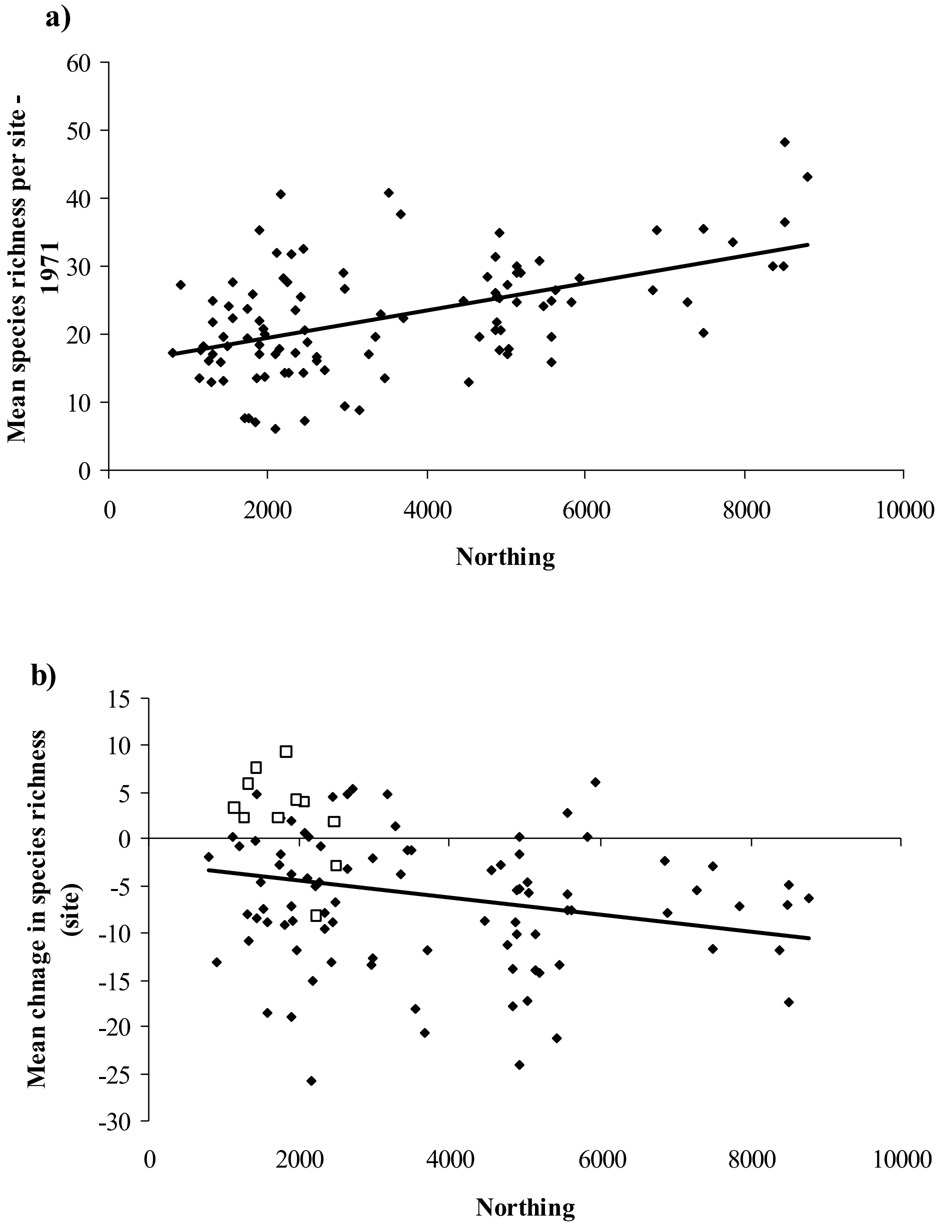


Figure 7.1 a) Mean species richness at site level (n=103) versus northing. b) Change in mean species richness averaged across sites (n=103) versus northing. Open squares are the ten storm track sites.

7.1.1 Explaining change in species richness

Early season surveys tend to pick up more of the vernal species, and so higher species richness would tend to be expected from the 2001 surveys (see section 2.5). Survey-date difference was a significant factor explaining differences in species richness.

Other factors were tested only after taking account of the survey-date effect, including variables related to eutrophication, climate change, woodland size and shape, surrounding land-cover and changes in stand growth. The best fitting model found significant positive effects on species richness for whether or not the site was in the October '87 storm track (Figure 1), increases in site-level signs of grazing had increased, or in the count of open habitats. Decreased richness was associated with an increase in woody basal area (Table 7.4). A model including these terms explained 40.7% of the observed changes, the highest explanatory power achieved for any of the ANOVA models constructed to explore the change in response variables.

Table 7.4 Change in level ground layer species richness in relation to explanatory variables

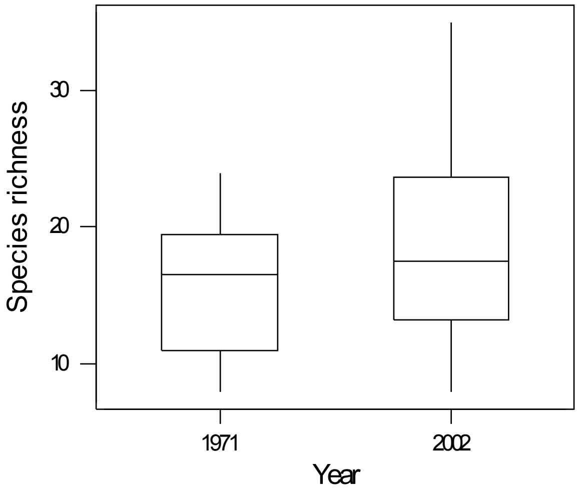
Explanatory variable	df	F	P	Relationship between effect and response
Survey day difference	95	29.48	**<0.0001**	+
1987 Storm site	93.8	15.89	**0.0001**	+
Change in site level grazing signs	96.8	6.55	**0.0121**	+
Change in plot level presence of open habitats	1436	57.14	**<0.0001**	+
Change in woody basal area	1392	8.33	**0.004**	-

Each woodland showed an individual pattern of change, so no single site can be representative of the whole. To illustrate this variation three woodlands were deliberately chosen to illustrate different types of vegetation change at the site level. Surveyors' reports have been used to provide information on apparent management history.

Priestfield (site 95) – re-survey 2002

W14 *Fagus sylvatica-Rubus fruticosus* agg. woodland. Ancient woodland site

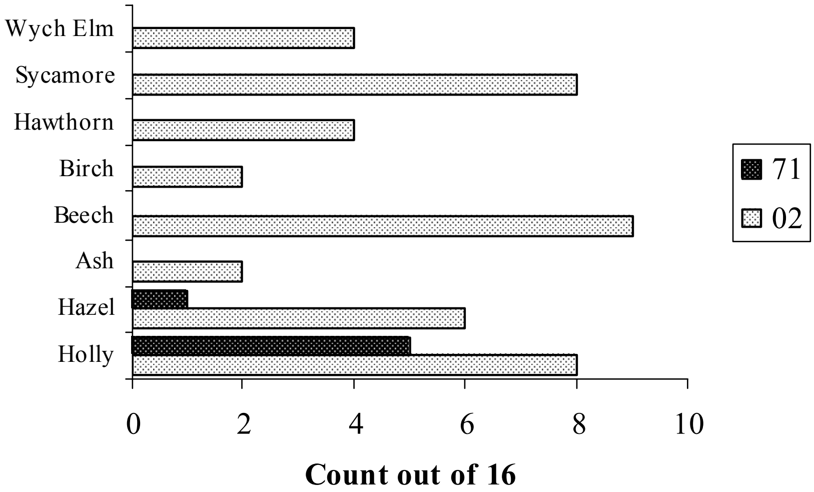
Priestfield is located in north eastern England, occupying marginal agricultural land over the slopes of a small valley. Slopes steepen considerably towards the two streams in the valley bottom. In 2002 there had been recent felling of ash and beech across half of the site in the previous five years. The remaining areas comprised undisturbed and mature larch plantation plus unworkable broadleaved woodland on the steepest slopes. Birch saplings and trees had greatly increased as had holly.



Ground flora species-richness at Priestfield in 1971 and 2002. (Box plots show the range of richness values (vertical lines), upper and lower quartiles and the median (horizontal line in middle of box).

On average ground flora species richness increased by 2.8 species per plot. The species richness changes were unevenly distributed across the site. Major disturbance (felling of mature beech and ash) was reported for nine plots out of 16 and in three of these 14, 15 and 9 species were gained. Other plots saw smaller increases or decreases. 34 species were recorded only in 2002 versus just nine recorded only in 1971. All the additional species were present at low frequency – recorded just once or twice. Some *Cirsium arvense, Epilobium montanum, Lolium multiflorum, Anchusa arvensis, Polygala oxyptera, Rumex obtusifolius* were typical of open or disturbed habitats which may be linked to the felling. Other woodland species that increased included *Dryopteris dilatata/carthusiana, Galium odoratum, Geranium robertianum, Hypericum pulchrum, Lonicera periclymenum, Milium effusum, Melica uniflora, Melampyrum pratense, Mercurialis perennis* and *Moehringia trinervia*.

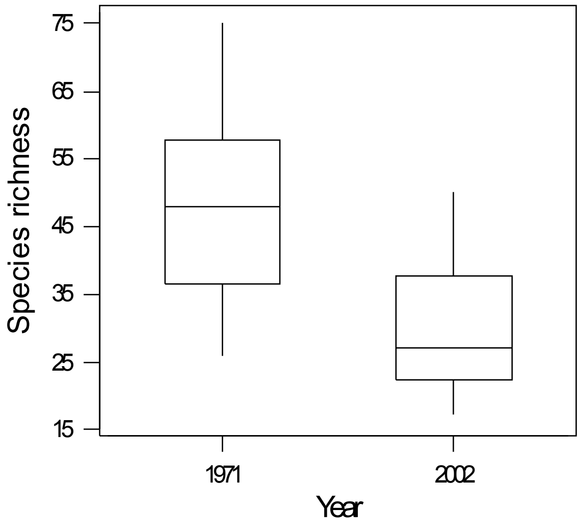
Recruitment of juvenile trees was also very apparent from the figure below showing the number of plots containing saplings (25-130 cm high).



Mill Wood (site 40) – re-survey 2002
W11 *Quercus petraea-Betula pubescens-Oxalis acetosella* woodland plus local W9 *Fraxinus excelsior-Sorbus aucuparia-Mercurialis perennis* woodland. Part ancient woodland site.

Mill Wood covers a narrow series of steeply sided small valleys in north-east Scotland. Although located in the uplands, the site was surrounded by improved pasture and barley at the time of the 2002 survey. Surveyors reported little sign of recent management.

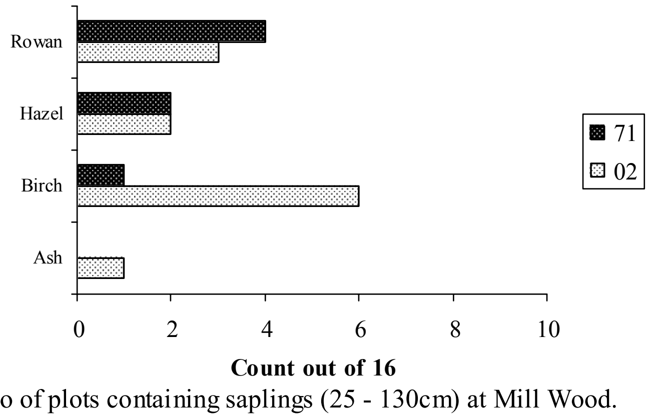
The wooded valley cuts through a sandstone cap and into underlying limestone on the valley floor and lower slopes. These exposures of rock had been well-worked in the past. Artificial extraction ramps made of rock debris occurred and in the valley bottom was what appeared to be an old field, regenerating with trees and shrubs amongst tall-herbs and wet grassland. Much of the woodland appears to be early to mid-successional colonisation of the former quarry workings. Records for open habitats in the plots declined in 11 out of 16. There were three records of recent management in plots in 1971, none in 2002.



Ground flora species-richness at Mill Wood in 1971 and 2002. Box plots show the range of richness values (vertical lines), upper and lower quartiles and the median (horizontal line in middle of box).

Only two plots gained species – one each – while the mean loss was 18 per plot. The species that saw the largest reductions in frequency included *Anthoxanthum odoratum, Lysimachia nemorum, Angelica sylvestris, Conopodium majus, Potentilla erecta, Galium saxatile, Geranium sylvaticum, Cerastium fontanum, Cirsium palustre, Veronica officinalis* and *Lathyrus pratensis.* The only species to see modest increases were *Poa trivialis, Festuca arundinacea, Galium aparine* and *Stellaria holostea.* Out of 149 species recorded across the 16 plots in 1971, 39 were not recorded again in 2002.

Young rowan and particularly birch were associated with the young woodland development on the grassland in the centre of the site.

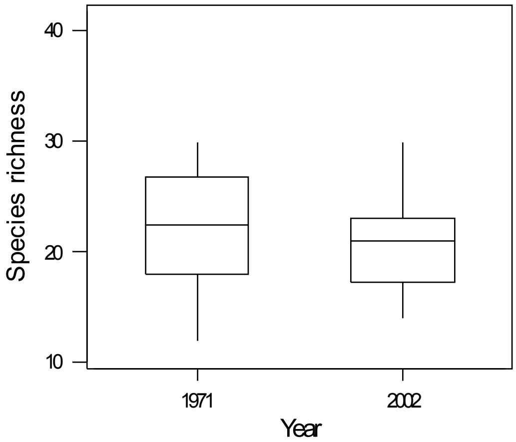


No of plots containing saplings (25 - 130cm) at Mill Wood.

Bubney Moor Wood (site 53) – re-survey 2002
W5 *Alnus glutinosa-Carex paniculata* woodland plus W6 *Alnus glutinosa-Urtica dioica* woodland.
Recent woodland

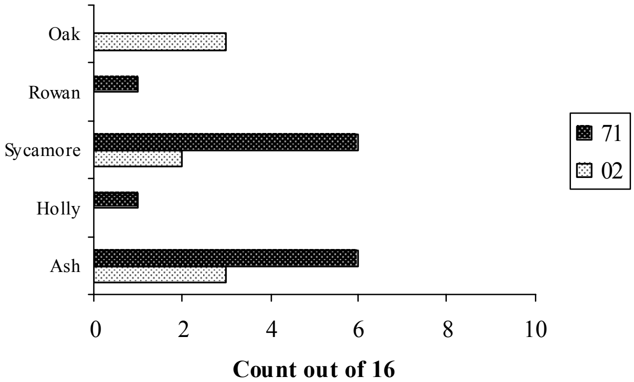
Bubney Moor is a recent wood located in the Welsh borders, one of a number of narrow, linear woodlands between Wrexham and Whitchurch occupying the steep slopes of small brooks that feed the River Dee, north of Bangor on Dee. The wood was estate-owned in 2002.

The western third of the site comprised a steep, unstable, sandy slope with extensive badger and rabbit burrows. Bracken was dense where the canopy was open, but for much of its length the ground flora was sparse, *Galium aparine* combining with bramble to form an impenetrable tangle. Adjacent to the brook, humic heavily flushed soils supported extensive alder and willow with planted hybrid black poplar. Recent signs of management were only recorded in one plot in 1971 and none in 2002 suggesting a site that even in 1971 had experienced little recent disturbance. There was a reduction in abundance of open and aquatic habitats between surveys.



Ground flora species-richness at Bubney Moor Wood in 1971 and 2002. (Box plots show the range of richness values (vertical lines), upper and lower quartiles and the median (horizontal line in middle of box).

Mean species richness in the ground flora declined by 1.2 species per plot on average but there was marked turnover in the species within plots; 47 species present in 1971 were not recorded again 2002 while 22 were newly recorded during the resurvey. Many of those gained were aquatic and swamp species including *Caltha palustris, Carex acuta, C. riparia, Mimulus guttata, Oenanthe crocata* and *Sparganium erectum*. In contrast, species not recorded again in 2001 were more typical of unimproved, wet grasslands such as *Stellaria graminea, Juncus articulatus/acutiflora, J. bulbosus, Myosotis scorpioides, Trifolium pratense, Succisa pratensis* and *Geum rivale*. The biggest increasers overall were *Ranunculus repens, Chryosplenium oppositifolium* and *Galium aparine*.



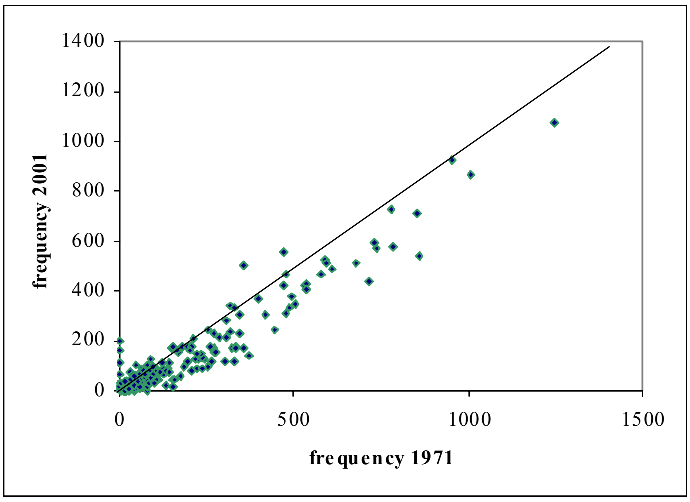
Number of plots containing saplings (25cm - 130cm high) for different species at Bubney Moor Wood. The regeneration was concentrated on the drier parts of the woodland rather than in the alder stands.

7.2 Change in frequency of individual species

Many species declined in frequency between 1971 and 2001 (Figure 7.2; Appendix 5). At the site level 63 species decreased significantly, 247 had not changed and 16 increased; at the plot level, 162 species decreased, 127 showed no significant difference and 37 increased. Both species common and rare in the samples showed declines.

Increasers and decreasers were grouped by Ellenberg and UCPE Plant Strategy scores (see Box 2.1) (Figure 7.3) to explore whether there were any overall difference between the characteristics of species significantly increasing or decreasing. Using linear logistic regression species with a higher 'Stress-tolerator' score were more likely to have declined. Further discussion on the changes in individual species is given in later sections.

a)



b)

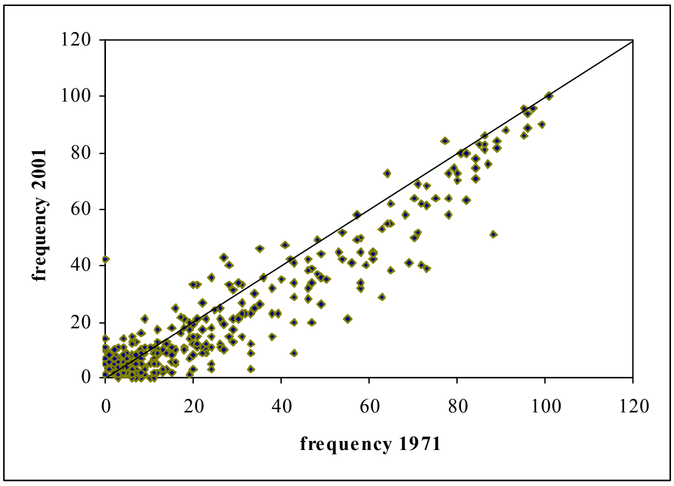


Figure 7.2 Frequency of higher plant species in the ground layer in 1971 versus 2001. a) Frequency between plots. b) Frequency between sites. The line represents equal frequency in both years, so species that increased are above the line, decreasers below it.

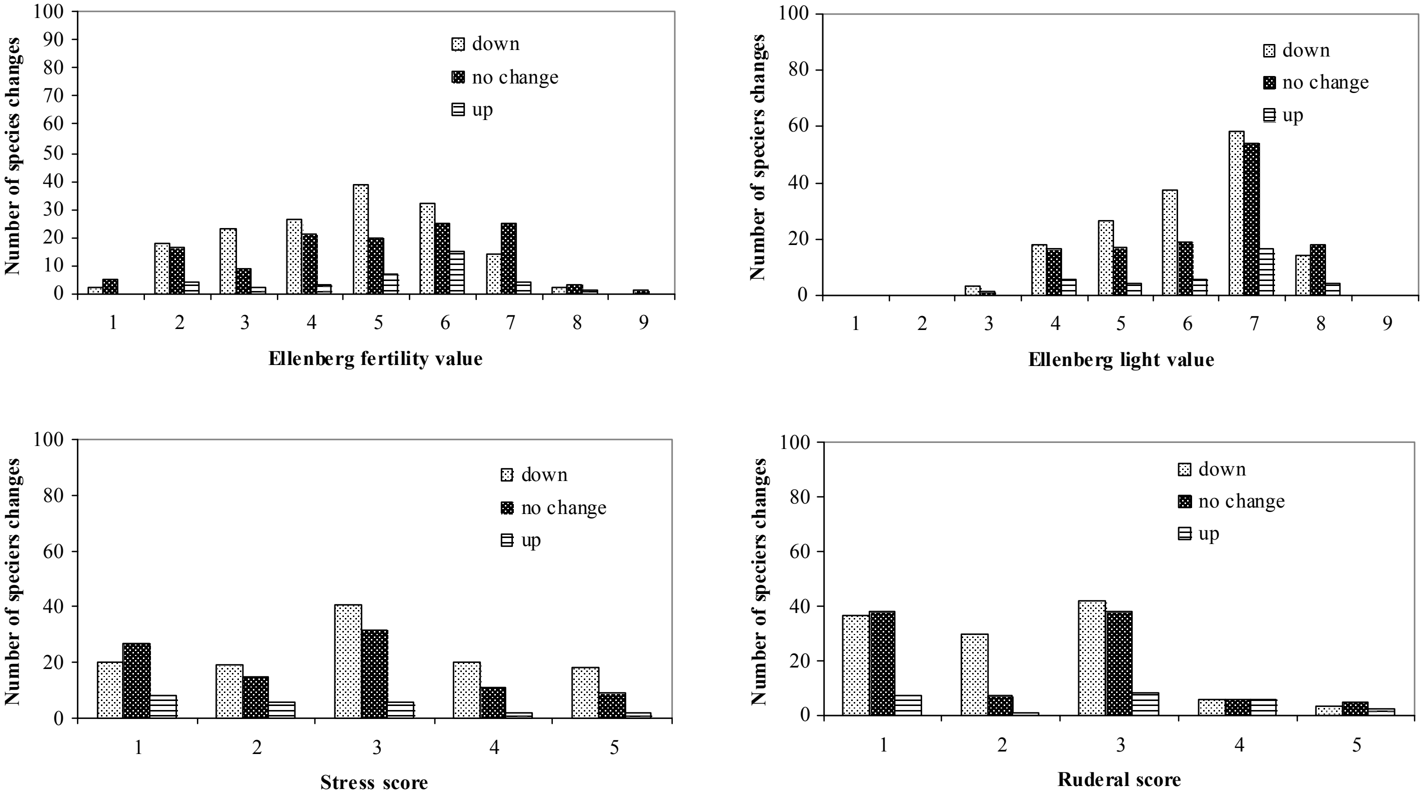


Figure 7.3 Number of species analysed for significance of change in frequency, grouped by (a) Ellenberg fertility(N) score, (b) Ellenberg Light score, (c) Plant Strategy Stress-tolerator score, (d) Plant Strategy Ruderal score. (For explanation of the scores see section 2.1.)

7.2.1 Change in frequency of Woodland Specialist species

Seventy-two species classed as woodland specialists (see Box 2.1, Kirby and others (in press)) were recorded in either 1971 or 2001. Between surveys, 56 species decreased significantly at plot level in frequency, seven showed no change and nine increased (full list in Table 7.6). Four of the nine increasers are vernal species *Narcissus pseudonarcissus, Allium ursinum, Hyacinthoides non-scripta* and *Anenome nemorosa,* but difference in date of survey was fitted as a model term before testing change in time.

An analysis of change was also carried out at the site level, based on presence or absence in at least one plot. Of the 72 species tested, 13 were too rare to be analysed, 33 showed no significant change, 25 disappeared from site samples and only 1 increased.

The number of decreasing species was larger in the woodland specialist group than expected compared with the change in the total species pool while the number of No Change tests was smaller (Table 7.5). This suggests that woodland specialists were more likely to have decreased in abundance than the complete species pool.

Table 7.5 The proportion of species showing No Change, Increases or Decreases for Woodland specialists compared to all other species.

	All species	**n**	**Woodland specialist**	**n**	**Binomial p**
Decrease	0.50	162	0.78	56	**0.0000**
No change	0.39	127	0.10	7	**0.0000**
up	0.11	37	0.13	9	0.7018

7.3 Change in cover of individual plant species

The frequency changes (section 7.2) provide a general measure of the changing composition of the woods. However, there might also be changes in the relationship between species because of differences in the relative growth of dominant species. This was explored by analysing cover values for abundant species only.

Cover estimates are subject to recorder error and are also inherently correlated with plot occupancy, and with frequency change. Analysis of cover change was therefore based only on those paired plots in which cover was estimated at a minimum 1% in both years. Small changes, such as where a species is present in one year and absent in the other, are excluded. The disadvantage of the approach is that sample sizes are greatly reduced and uncommon species could not be tested. However where tests were possible they should represent real differences in the abundance of species, not just in their occurrence. All cover tests were carried out after fitting day differences in survey date as a covariate.

Table 7.5 Changes in frequency of woodland specialist species.
(Too few = too few to test)

	sites occupied			plots occupied		
Woodland specialists	**1971**	**2001**	**SIG**	**1971**	**2001**	**SIG**
Athyrium filix-femina	73	61	down	477	314	down
Blechnum spicant	58	50	no change	317	237	down
Brachypodium sylvaticum	65	62	no change	343	305	down
Bromopsis ramosa	29	17	down	88	31	down
Campanula latifolia	8	2	too few	17	9	down
Carex sylvatica	70	64	no change	308	217	down
Conopodium majus	30	22	no change	84	46	down
Convallaria majalis	5	1	too few	12	2	down
Epipactis helleborine	20	3	down	50	4	down
Equisetum sylvaticum	11	11	no change	39	33	down
Festuca gigantea	46	28	down	176	59	down
Fragaria vesca	73	39	down	264	120	down
Galium odoratum	31	23	no change	102	57	down
Geranium robertianum	81	80	no change	395	371	down
Geranium sylvaticum	6	1	too few	17	3	down
Geum rivale	16	5	down	41	9	down
Gymnocarpium dryopteris	10	11	no change	21	16	down
Hordelymus europaeus	5	2	too few	8	4	down
Hypericum androsaemum	15	6	down	28	10	down
Hypericum pulchrum	46	42	no change	131	82	down
Iris foetidissima	4	1	too few	25	1	down
Lamiastrum galeobdolon	30	34	no change	198	176	down
Lathyrus linifolius	13	10	no change	42	19	down
Listera ovata	14	9	no change	50	16	down
Luzula pilosa	65	38	down	327	121	down
Luzula sylvatica	26	21	no change	106	84	down
Lysimachia nemorum	71	52	no change	333	167	down
Melampyrum pratense	23	9	no change	59	25	down
Melica uniflora	26	25	no change	78	68	down
Mercurialis perennis	75	64	down	534	426	down
Moehringia trinervia	31	15	down	80	34	down
Oxalis acetosella	87	76	down	782	575	down
Paris quadrifolia	2	2	too few	11	3	down
Polypodium vulgare sens.lat.	33	12	down	86	34	down
Polystichum aculeatum	21	12	down	46	29	down
Potentilla sterilis	63	29	down	236	87	down
Primula elatior	4	1	too few	14	5	down
Primula vulgaris	63	53	down	272	154	down
Ribes nigrum	24	3	down	56	4	down
Rubus saxatilis	6	1	too few	11	1	down
Sanicula europaea	43	20	down	161	41	down
Scrophularia nodosa	42	42	no change	92	85	down
Silene dioica	59	40	down	276	157	down
Solidago virgaurea	33	9	down	132	21	down
Stachys officinalis	15	8	no change	44	11	down
Stachys sylvatica	64	55	no change	204	163	down
Stellaria holostea	61	42	down	232	149	down
Tamus communis	39	23	down	141	76	down
Vaccinium myrtillus	47	34	down	221	148	down
Valeriana officinalis	34	25	no change	94	66	down
Veronica montana	68	58	down	287	214	down
Vicia sepium	29	13	down	58	17	down
Viola odorata	13	3	down	26	7	down
Viola palustris	22	17	no change	55	34	down
Lonicera periclymenum	89	84	no change	679	509	down
Campanula trachelium	6	0	too few	23	0	down
Adoxa moschatellina	3	5	too few	9	8	no change
Hypericum tetrapterum	4	3	too few	11	9	no change
Polygonatum multiflorum	1	3	too few	11	8	no change
Chrysosplenium oppositifolium	41	47	no change	178	181	no change
Euphorbia amygdaloides	14	12	no change	43	43	no change
Carex pallescens	3	8	no change	11	11	no change
Hypericum hirsutum	8	9	no change	15	22	No change
Allium ursinum	21	21	no change	80	92	up
Anemone nemorosa	43	41	no change	148	174	up
Carex pendula	23	21	no change	66	92	up
Carex remota	36	36	no change	92	123	up
Hyacinthoides non-scripta	82	80	no change	470	552	up
Milium effusum	18	21	no change	78	102	up
Narcissus pseudonarcissus	0	1	too few	0	13	up
Phyllitis scolopendrium	17	22	no change	40	61	up
Polystichum setiferum	3	11	up	4	23	up

All significant cover changes were increases (Table 7.7). Changes in mean species cover were summed within each score class for Ellenberg Light (L), Fertility (N) and soil pH (R) (for species with cover of a minimum of 1% in both years) (Figure 7.4), as well as being considered in terms of other characteristics.

- There were increases amongst plants of semi-shade to shaded conditions (Ellenberg scores of 6 or less), although some of the vernal increases, such as *Allium ursinum* and *Hyacinthoides non-scripta,* may be partly a survey-date difference effect. The large increase at Ellenberg Light Score 7 reflected mainly a non-significant increase in cover of *Poa trivialis*.
- There were more increases in cover of plants of more fertile conditions (Ellenberg N values 6 or more). The species concerned included *Allium ursinum* (Ellenberg N=7), *Carex pendula* (6), *Urtica dioica* (8) and *Hyacinthoides non-scripta* (6); the increase in the first three of these species was partly explained by modelled N deposition in 1998 (see Table 7.10).
- Species associated with higher pH (Ellenberg R score 7) increased although Ellenberg N and Ellenberg R values are strongly correlated.
- Grazing-tolerant species such as *Brachypodium sylvaticum* and *Carex pendula* increased but so also did the more grazing-sensitive bramble *Rubus fruticosus* and dog’s mercury *Mercurialis perennis*.
- The largest changes involved widespread woodland dominants and half are classed as ‘woodland specialists’.

Table 7.7 Significant changes in mean species cover between 1971 and 2001. Analysis based on plots in which cover was at least 1% in both years. *= Woodland specialist (as defined in Box 2.1).

Species	**T**	**p**	**Mean cover 1971**	**Mean cover 2001**	**No plots**	**Ellenberg N score**	**Ellenberg L score**	**Strategy type**
Ajuga reptans	2.13	0.0337	1	2	26	5	5	CSR
*Allium ursinum**	3.98	0.0001	2	24	17	4	7	CR/CSR
*Brachypodium sylvaticum**	4.31	0.0000	3	8	59	6	6	S/SC
*Carex pendula**	3.60	0.0003	4	18	14	5	6	S/SC
Dactylis glomerata	2.32	0.0205	4	6	24	7	6	C/CSR
Dryopteris dil./ carthusiana	3.90	0.0001	7	11	236	5	5	SC/CSR
Fraxinus excelsior	2.64	0.0083	1	4	41			S
*Geranium robertianum**	3.27	0.0011	1	9	66	5	6	R/CSR
Geum urbanum	4.73	0.0000	2	5	66	4	7	S/CSR
*Hyacinthoides non-scripta**	6.41	0.0000	4	22	162	5	6	SR/CSR
*Mercurialis perennis**	2.55	0.0107	30	33	269	3	7	SC
*Oxalis acetosella**	6.95	0.0000	3	6	294	4	4	S/CSR
Ranunculus repens	2.91	0.0037	2	6	56	6	7	CR
Rubus fruticosus agg.	4.13	0.0000	18	21	628	6	6	SC
Rubus idaeus	2.46	0.0140	1	6	15	6	5	SC
Urtica dioica	2.80	0.0051	9	19	139	6	8	C
*Veronica montana**	2.46	0.0140	2	3	25	4	6	S/CSR
*Viola riv./reichenbiana**	2.92	0.0036	1	3	86	6	4	S

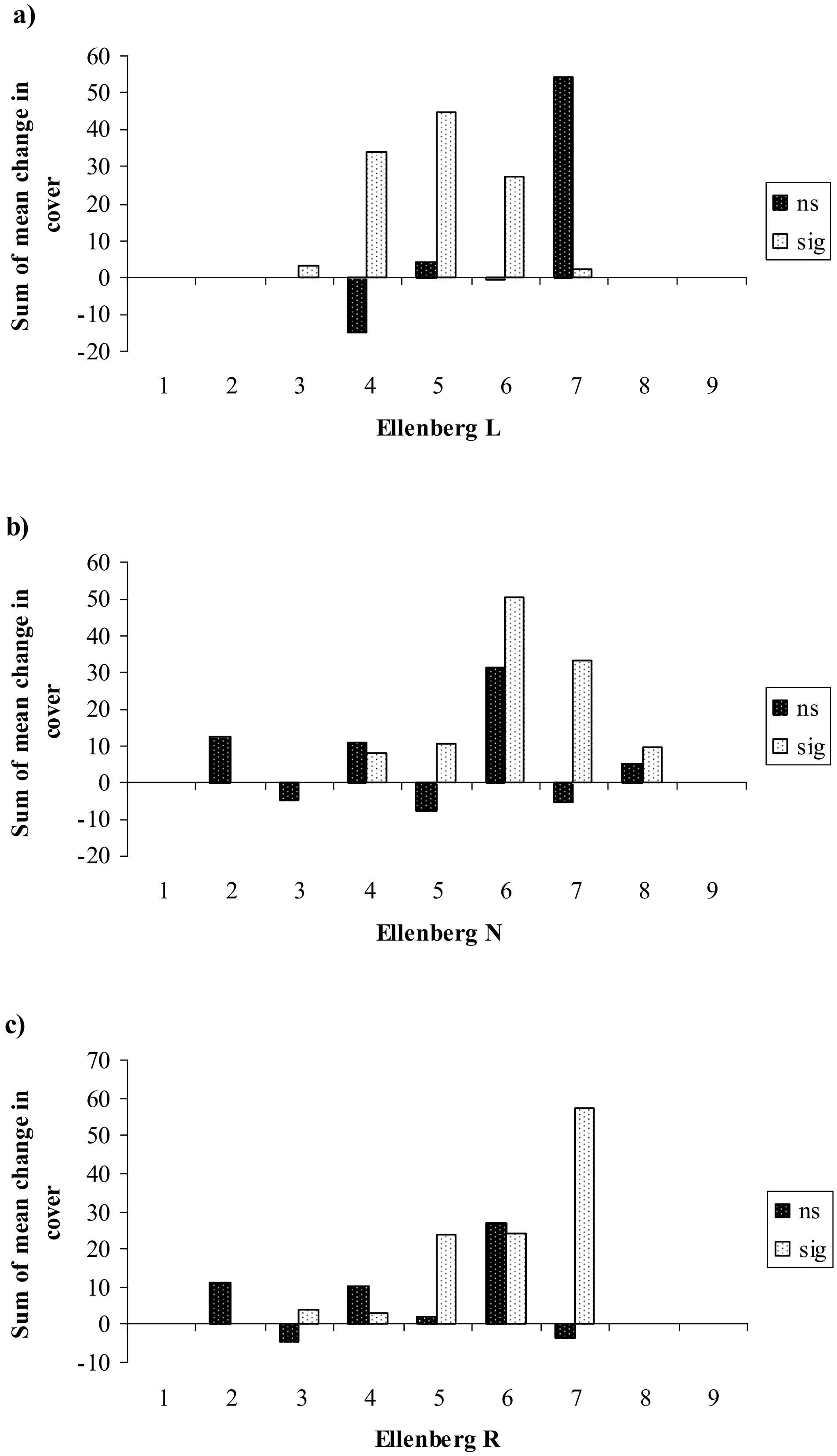


Figure 7.4 Sum of the mean change in cover for groups of species with the same Ellenberg score class. a) Ellenberg light scores L, b) fertility score N and c) soil pH score R.

7.4 Changes in species composition between 1971 and 2002 - Multivariate analysis

An unconstrained ordination on ground flora data from both 1971 and the resurvey (DCA) showed two primary gradients in the data. Axis 1 was highly correlated with mean Ellenberg pH (R) and fertility scores (N) for sample plots and explained 6.1% of the species variation data, while axis 2 (3.7% of variation) was correlated with Ellenberg light scores (L) (Figure 7.5). These were also the primary gradients identified in the Countryside Survey 2000 data-set covering a wide range of vegetation types (Bunce and others 1999; Haines-Young and others 2000, 2003).

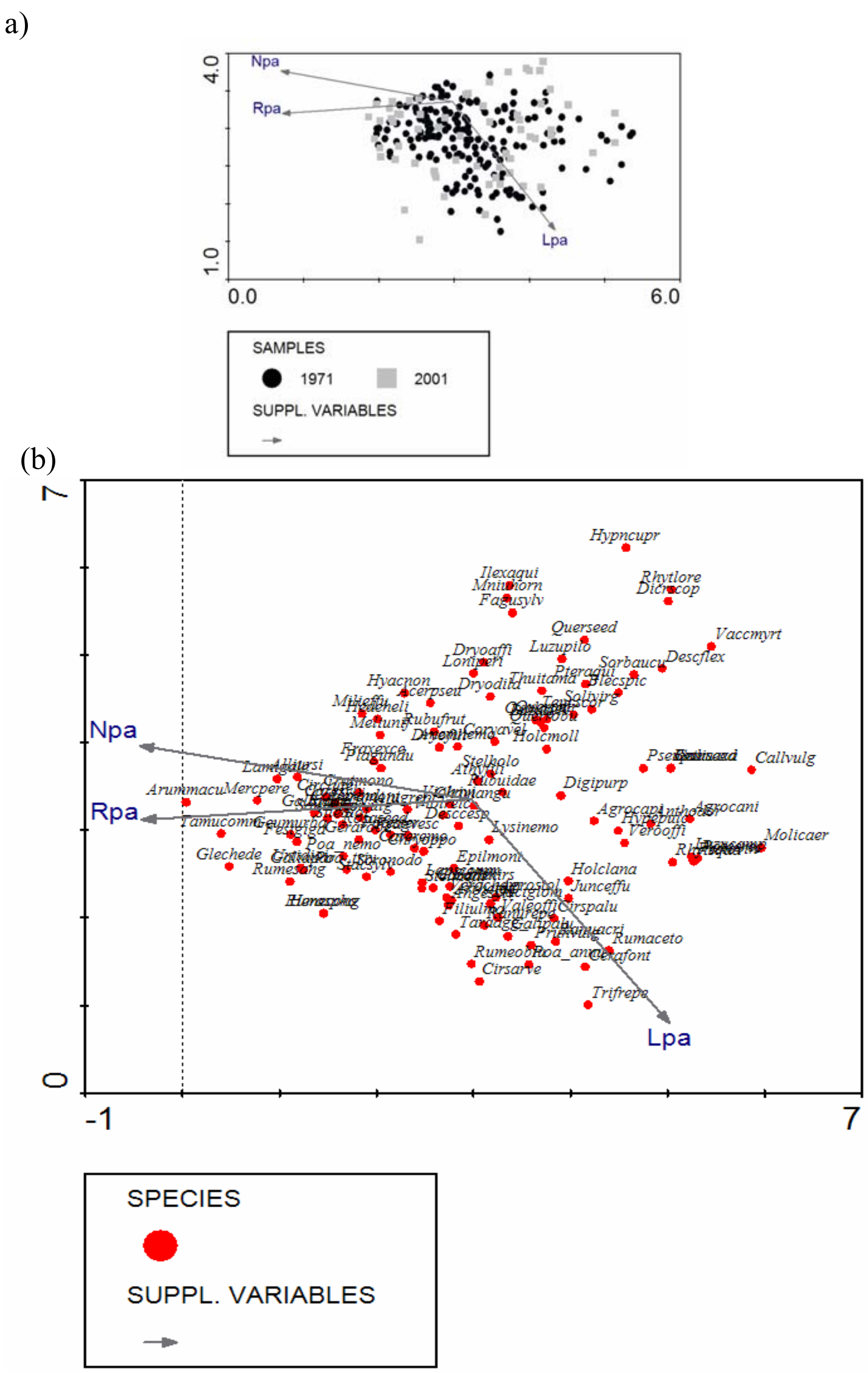


Figure 7.5 DCA biplot showing samples a) and species scores b) for ground flora data recorded in both 1971 and 2001. Mean unweighted Ellenberg scores for light (Lpa), fertility (Npa) and soil pH (Rpa) were added passively.

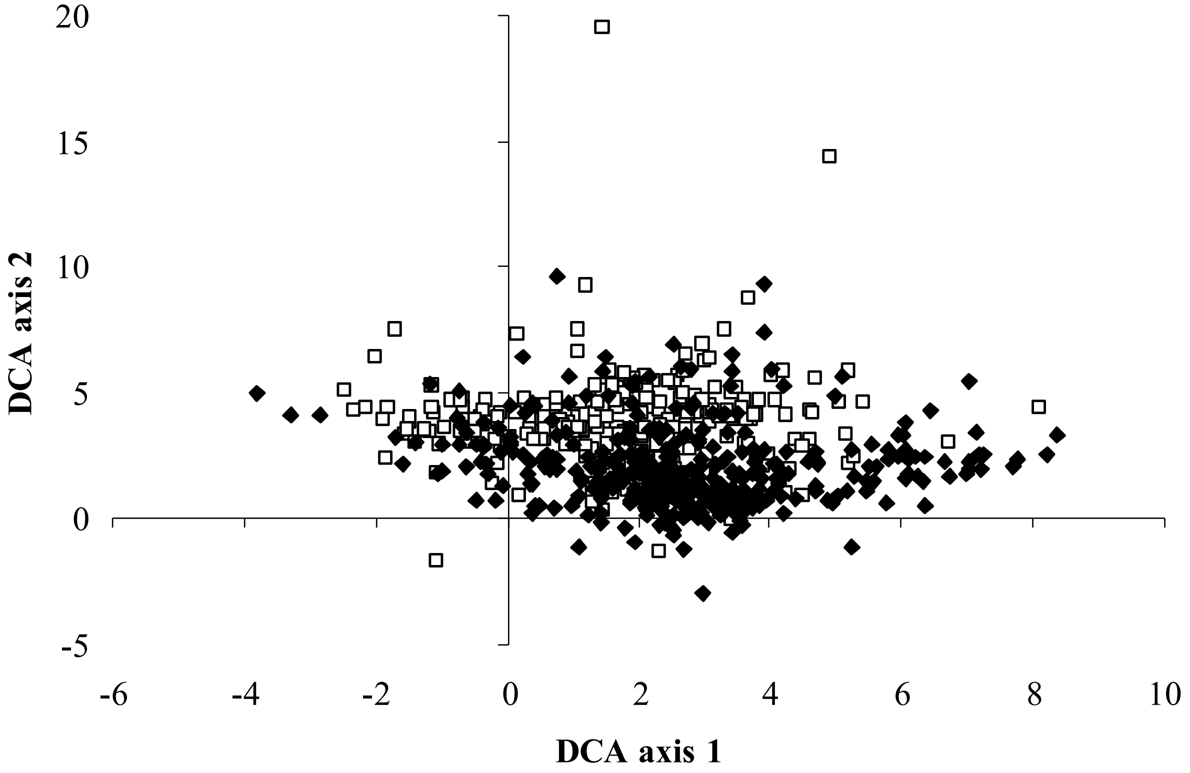
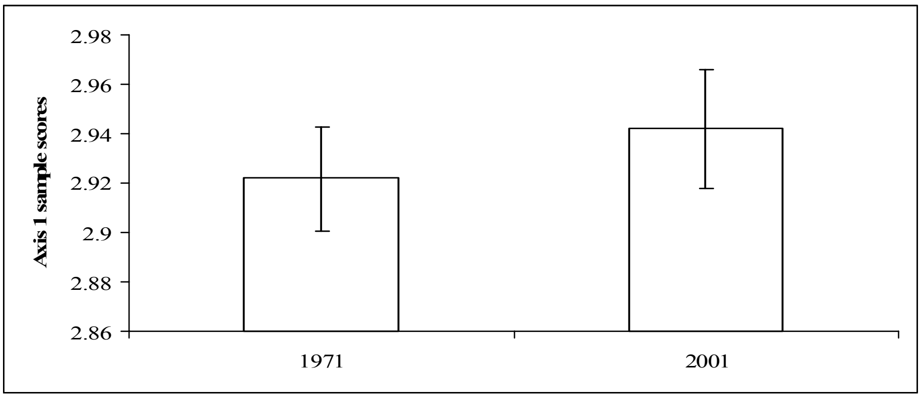


Figure 7.5c DCA species scores for the ground flora in sample plots in both surveys. Axis 2 is correlated with Ellenberg L scores and axis 1 with Ellenberg R/N scores. Each species has been categorised by its Ellenberg L score, into those with a score of 6 or less (**open squares** more shade tolerant) and those with a score of 7 or more (**filled diamonds**, more light demanding).

Between surveys there was a significant movement of plots along axis 2 suggesting a shift toward species assemblages associated with more shaded conditions (Figure 7.6). Despite the increases in soil pH, there was no significant movement of plots along the inferred pH or Nitrogen axis (Axis 1) (Figure 7.6).

(a)



(b)

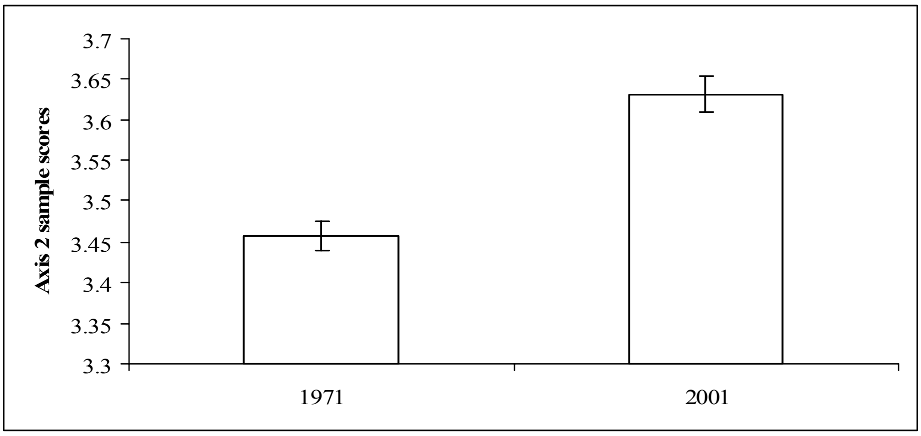


Figure 7.6 Change in DCA axis scores between 1971 and 2001 in paired woodland sample plots. In (a) low axis 1 scores are associated with high Ellenberg R/N scores, i.e more base-rich/fertile conditions; in (b) high axis 2 scores are associated with low Ellenberg Light Scores i.e more shaded conditions. Only the difference between the years on axis 2 is significant.

7.5 Changes in the distribution of different Ellenberg scores

This section is based on unweighted Ellenberg scores (see section 2.7). Analyses using weighted values were carried out but showed little difference (see Appendix 8).

7.5.1 Ellenberg fertility (N) and pH (R) scores

Changes in mean Ellenberg N and R scores are generally highly correlated (Figure 7.7). Mean unweighted Ellenberg fertility and pH scores decreased in the uplands (Table 7.8), consistent with the shift on Axis 1 of the DCA towards a more nutrient-poor/acidic flora.

There was no evidence for overall cross-site shifts towards species assemblages associated with higher nutrient availability or higher pH even though soil pH had generally increased (see section 5.1). However change in soil pH and the area of intensive land cover surrounding the site in 2000 (based on the satellite Land Cover Map 2000) explained 12.6% of the temporal change in mean Ellenberg fertility score (Table 7.9); i.e. where there was an increase in the fertility score this was most likely to be in sites with a large soil pH increase and adjacent to more intensive land-use.

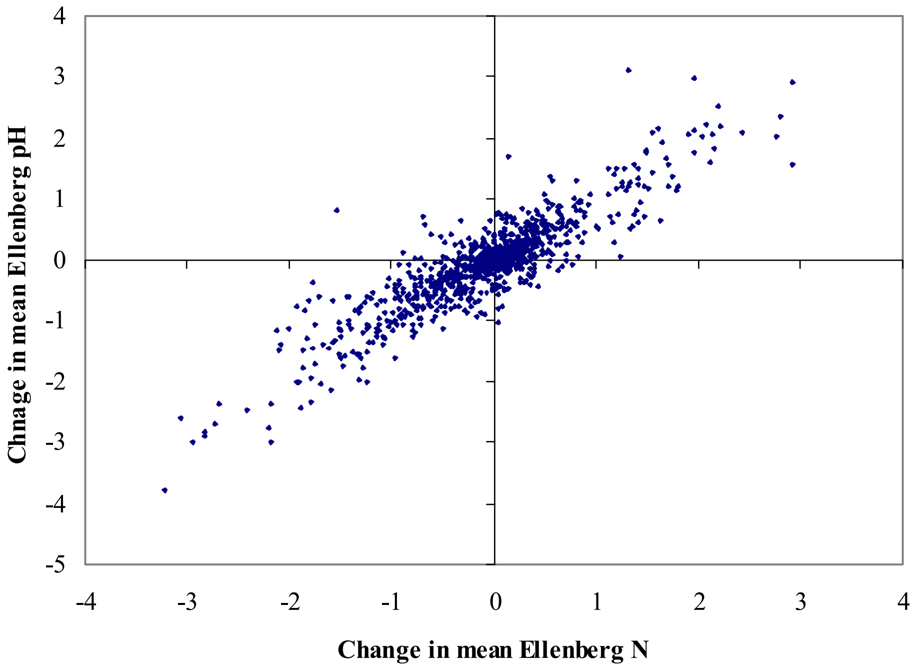


Figure 7.7 Change in mean unweighted Ellenberg fertility (N) score per plot versus change in mean unweighted Ellenberg pH score between 1971 and 2001.

Table 7.8. Changes in mean Ellenberg pH and fertility score.

Stratum	**Mean 1971 (+/-se)**	**Mean 2001 (+/-se)**	**Adj. mean change**[1]	**se**	**df**	**P**
Fertility						
GB	5.55 (0.018)	5.54 (0.100)	-0.04	0.023	100	0.0976
Lowlands	5.74 (0.012)	5.74 (0.014)	-0.004	0.023	82.4	0.3797
Uplands	4.78 (0.063)	4.66 (0.067)	-0.19	0.066	16.6	**0.0097**
pH						
GB	5.70 (0.017)	5.68 (0.019)	-0.04	0.020	101	0.0754
Uplands	5.03 (0.056)	4.95 (0.062)	-0.11	0.051	16.3	**0.0438**
Lowlands	5.86 (0.013)	5.85 (0.014)	-0.02	0.022	82.4	0.3797

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

Table 7.9 Change in mean Ellenberg fertility score in relation to possible explanatory variables.

Explanatory variable	df	F	P	Significance and sign of relationship
Survey day difference	97	3.75	0.0557	Ns
Soil pH change	156	6.34	**0.0119**	+
Area of intensive land cover in 3500m radius buffer (from LCM 2000)	97.5	5.4	**0.0222**	+

Earlier analyses suggested that some high Ellenberg N species were increasing in cover (section 7.3). For twelve species there was a significant relationship between the changes in their cover and modelled N deposition (Table 7.10), consistent with their Ellenberg N scores and with indicator values developed for Swedish woods that reflect species preferences for more or less fertile soil conditions (Diekmann and Falkengren-Grerup 1998). These FNIS values were based on the association between species occurrence and measured soil N mineralization (nitrate and ammonium). Values are higher for more nitrophilous species (Table 7.10).

Species that tended to decline with higher N deposition had Ellenberg N scores of 5 or less, while the increasers had Ellenberg scores of 6 or more. These latter included established nitrophiles such as *Galium aparine* and *Urtica dioica,* and *Athyrium filix-femina* that has emerged as an indicator of eutrophying N deposition in Scandinavian woodlands (Diekmann and Falkengren-Grerup 2002). However some species that show overall declines with increasing nitrogen, for example *Agrostis capillaris, Holcus lanatus* and *Deschampsia flexuosa* have shown positive responses to nitrogen deposition in at least one other study (NEGTAP 2001).

Table 7.10 Species showing a significant correlation between change in cover between 1971 and 2001 and modelled N deposition in 1998 for the local 5x5 km square. (*= woodland specialist as defined in Box 2.1)

Plant species	Ellenberg N	FNIS value[1]	P value
Increasing cover as N deposition increases			
Poa nemoralis/trivialis	6	6	**0.0487**
Galium aparine	8	8	**0.0000**
*Allium ursinum**	7	-	**0.0041**
*Athyrium filix-femina**	6	7	**0.0494**
*Carex pendula**	6	-	**0.0008**
Urtica dioica	8	9	**0.0007**
Decreasing cover as N deposition increases			
Deschampsia flexuosa	3	2	**0.0090**
Agrostis capillaries	4	4	**0.0023**
Ajuga reptans	5	-	**0.0373**
Holcus lanatus	5	3	**0.0034**
Pteridium aquilinum	3	2	**0.0256**
*Vaccinium myrtillus**	2	2	**0.0001**

[1] See Diekmann and Falkengren-Grerup (1998) for details.

7.5.2 Ellenberg light scores

Despite some species with low Ellenberg light scores showing increases in cover (Figure 7.4) and the movement along Axis 2 of the ordination (Figure 7.6) there were no overall significant changes in mean Ellenberg light scores (Table 7.11). However plot level relationships between mean Ellenberg L score change and explanatory variables were consistent with effects of shade at ground level (Table 7.12). Increased count of open habitats had a positive relationship and change in basal area of trees and shrubs a negative effect, explaining 16.8% of the variation in observed data.

Table 7.11 Changes in mean Ellenberg light scores.

Stratum	**Mean 1971 (+/-se)**	**Mean 2001 (+/-se)**	**Adj. mean change[1]**	**se**	**Df**	**p**
GB	5.82 (0.009)	5.80 (0.012)	-0.01	0.023	102	0.6107
Lowland zones	5.78 (0.009)	5.74 (0.012)	-0.04	0.023	82.6	0.1019
Upland zones	5.98 (0.023)	6.07 (0.029)	0.10	0.064	17	0.1517

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

Table 7.12 Change in mean Ellenberg light score in relation to possible explanatory variables

Explanatory variables	**df**	**F**	**P**	**Significance and sign of the relationship**
Survey day difference	99.7	0.000	0.9521	Ns
Plot level change in count of open habitats	1413	42.25	**<0.0001**	+
Change in woody basal area	1437	8.86	**0.0030**	-

7.5.3 Ellenberg wetness scores

Aquatic habitats were recorded in 28% of paired plots and their abundance declined between surveys. However mean Ellenberg wetness scores did not change significantly even within the subset of plots in which aquatic habitats occurred (Table 7.13).

Table 7.13. Cross-stratum changes in mean Ellenberg wetness score.

Stratum	**Mean 1971 (+/-se)**	**Mean 2001 (+/-se)**	**Adj. mean change[1]**	**Se**	**df**	**p**
GB	5.91 (0.007)	5.89 (0.008)	0.02	0.017	99.5	0.3198
Lowland zones	5.91 (0.007)	5.89 (0.008)	0.03	0.019	80.9	0.1794
Upland zones	5.95 (0.018)	5.91 (0.023)	-0.02	0.041	16.8	0.7025

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

7.5.4 Interactions between Ellenberg Light, Fertility and Reaction Scores

Some of the changes in mean Ellenberg scores above did not match expectations and so the possible suppressive effect of shading on ground flora response to changing fertility or pH conditions was explored. The joint behaviour of change in soil pH, and change in Ellenberg light, on change in Ellenberg R was tested using a mixed-model ANOVA. Both factors were significant and had the expected positive relationships, i.e. an increase in soil pH might not produce a correlated increase in Ellenberg R score if Ellenberg light decreased at the same time (Figures 7.8, Table 7.14a). The variance explained is very low, but these results indicate that vegetation response can be uncoupled from the effects of increasing soil pH by increased shading.

Table 7.14a Change in Ellenberg pH (R) score in relation to changes in soil pH and Ellenberg light score.

Explanatory variable	df	F	p	Significance and sign of the relationship
Survey day difference	99.6	2.69	0.104	Ns
Change in soil pH	1540	25.46	**<.0001**	+
Change in Ellenberg L	1539	29.67	**<.0001**	+

Table 14b. Changes in Ellenberg N score in in relation to changes in soil pH and Ellenberg light score.

Explanatory variable	df	F	p	Significance and sign of the relationship
Survey day difference	97.9	3.58	0.0616	Ns
Change in soil pH	1572	6.48	**0.0110**	+
Change in mean Ellenberg L	1570	12.98	**0.0003**	+

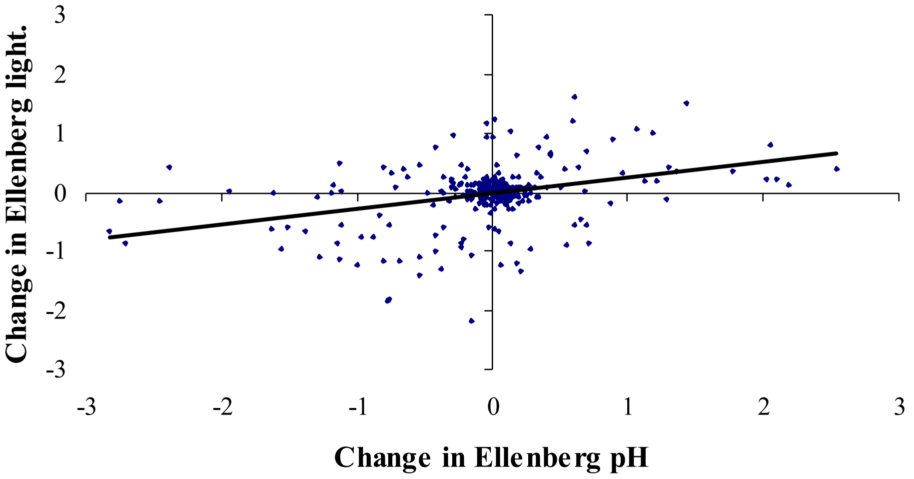


Figure 7.8 Change in mean Ellenberg pH (R) score versus mean Ellenberg light (L) score. Correlation significant p<.0001, r-sqrd = 0.017.

The same negative effect is seen between Ellenberg L and N scores because species of more open conditions (ie. higher Ellenberg L scores) tend to be more nutrient-demanding (higher N scores) and hence likely to be favoured by less-shaded woodland gaps.

7.6 Changes in UCPE Plant Strategy types

The species lists from the plots were classified according to the UCPE Plant Strategy types into Competitor, Stress-tolerator and Ruderal elements (Box 2.1; Grime and others 1995) and mean values calculated for each plot.

No significant changes were detected in the Competitor scores.

Significant reductions in Stress-tolerator scores were seen at the GB level (Table 7.15, 7.16). There was a negative correlation with change in count of open habitats, ie stress-tolerators appeared to do less well where there were more open habitats. The combination of variables explained 19.4% of the observed data.

There was no overall change in the Ruderal score (Table 7.17), but the spatial variation in the scores was positively correlated with increases in open habitats and negatively correlated with increased basal area (Table 7.18). Overall, change in count of plot-level open habitats and change in woody basal area accounted for 26.8% of the variation in observed data.

Table 7.15 Changes in Stress-tolerator score.

Stratum	**Mean 1971 (+/-se)**	**Mean 2001 (+/-se)**	**Adj. mean change**[1]	**se**	**df**	**P**
GB	2.76 (0.011)	2.66 (0.012)	-0.08	0.021	100	**0.0005**
Lowland zones	2.69 (0.011)	2.62 (0.012)	-0.06	0.023	81.7	**0.0094**
Upland zones	3.07 (0.024)	2.88 (0.033)	-0.12	0.044	16.8	**0.0145**

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

Table 7.16 Change in mean Stress-tolerator score in relation to possible explanatory variables

Explanatory variables	**df**	**F**	**P**	**Sign and significance of relationship**
Survey day difference	15.5	2.57	0.1124	Ns
Plot level change in count of open habitats	1414	25.22	**<0.0001**	-
Change in woody basal area	1432	2.39	0.1222	Ns

Table 7.17 Changes in Ruderal score.

Stratum	Mean 1971 (+/-se)	Mean 2001 (+/-se)	Adj. mean change^1	se	df	P
GB	1.91 (0.010)	1.96 (0.012)	0.02	0.028	98.9	0.5734
Upland zones	1.91 (0.022)	2.05 (0.034)	0.13	0.078	16.9	0.1218
Lowland zones	1.91 (0.011)	1.94 (0.013)	-0.01	0.029	79.7	0.6934

[1] 'Adj. mean change' may differ from the recorded difference between the means for each year because it is the adjusted mean after taking account of the effect of the day-difference between survey dates (see 2.5).

Table 7.19 Change in mean Ruderal score in relation to possible explanatory variables

Explanatory variables	df	F	P	Sign and significance of relationship
Survey day difference	96.2	9.83	**0.0023**	+
Plot level change in count of open habitats	1440	40.89	**<0.0001**	+
Change in woody basal area	1405	4.04	**0.0445**	-

7.7 Climate variables and changes in ground flora species composition

No significant changes in the proportion of biogeographic elements (see Box 2.1d) in plots were seen in the initial analyses using Rainfall Intensity, Growing Season Length, July maximum and January minimum climatic variables.

Changes in the timing of growth, in particular the start of the growing season, are occurring (Sparks and Smithers 2002; Collinson and Sparks 2003). Such shifts could alter the competitive relationships between species, which might be reflect in changes in species abundance. Change in mean cover was tested for 65 sufficiently-abundant plant species. Of these, 13 showed a negative relationship with lengthening growing season and four an increase in cover with lengthening growing season (Table 7.19). When arranged by biogeographic element, neither increasers or decreasers were clearly differentiated by affinities for northern, southern, continental or oceanic distributions. Both early flowering and late flowering species, and early growing, e.g. *Allium ursinum,* and late growing species, e.g. *Pteridium aquilinum* were among those that decreased. Some species were at odds with other climate model predictions. For example, *Pteridium aquilinum* is expected to increase in vigour and abundance with greater oceanicity and more frost-free days (Pakeman 1995).

Table 7.19 Plant species showing a significant correlation between change in cover and change in growing season length between 1971 and 2001. * = woodland specialist as defined in Box 2.1.

Decreased in cover with longer growing season	**Biogeographic element**	**Start of flowering**
Erica cinerea	Oceanic Temperate	July
Dryopteris filix-mas	Circumpolar Temperate	n/a
*Brachypodium sylvaticum**	European Temperate	July
*Athyrium filix-femina**	Circumpolar Boreo-temperate	n/a
*Allium ursinum**	European Temperate	April
*Viola riviniana/reichenbiana**	European Temperate	April
*Oxalis acetosella**	Eurasian Boreo-temperate	April
Deschampsia cespitosa	Circumpolar Wide-boreal	June
*Mercurialis perennis**	European Temperate	February
Lonicera periclymenum	Suboceanic Southern-temperate	June
Geum urbanum	Eurosiberian Temperate	June
Pteridium aquilinum	Circumpolar Temperate	n/a
*Luzula pilosa**	Eurosiberian Boreo-temperate	April
Increased in cover with longer growing season		
*Lysimachia nemorum**	Suboceanic Temperate	May
*Milium effusum**	Circumpolar Boreo-temperate	June
Agrostis capillaris	Eurosiberian Boreo-temperate	June
*Geranium robertianum**	European Temperate	May

7.7.1 Effect of mean Jan-March temperature change

Following the initial analyses a mixed-model ANOVA was used to test whether change in mean ‘spring’ temperature was better correlated with change in frequency of individual plant species in woodland plots between the initial survey in 1971 and the recent re-survey. Spring temperature change was defined as the slope of a linear regression fitted to mean Jan, Feb and March temperatures for 5km sqrs containing each of the 103 woodland sites for all years from 1961 to 1999. These data were obtained from the UKCIP web-site.

Day difference between the dates of each survey was always fitted first. Change in woody basal area was fitted before Jan-March temperature change in the model statement.

In total 51 species out of 332 with sufficient records for analysis showed a significant relationship between Jan-March temperature change and change in frequency within woodland plots after fitting the two covariates.

Only four species, *Aira praecox, Equisetum sylvaticum, Oxalis acetosella and Oreopteris limbosperma* showed a negative correlation with temperature change. The other 47 showed a positive positive relationship. Sample sizes and the length of the temperature change gradient were small for many species, but for twenty-four species there was a significant Pearson correlation coefficients between frequency change across each site and Jan-March temperature change.

Table 7.20. Species showing the most significant relationships between Jan-March temperature change and frequency changes. * = woodland specialist as defined in Box 2.1.

	Correlation coefficient	P value
*Anemone nemorosa**	0.46	0.0003
Anthoxanthum odoratum	0.344	0.0043
Campanula rotundifolia	0.502	0.015
Carex nigra	0.725	0.003
Carex pilulifera	0.576	0.012
Cerastium fontanum	0.266	0.046
Cynosurus cristatus	0.433	0.02
Danthonia decumbens	0.832	0.002
*Equisetum sylvaticum**	-0.575	0.04
Galeopsis tetrahit agg.	0.377	0.034
Galium aparine	0.24	0.025
*Geranium robertianum**	0.266	0.01
*Hypericum androsaemum**	0.564	0.015
*Hypericum pulchrum**	0.405	0.0008
Luzula campestris/multiflora	0.55	0.0001
*Melampyrum pratense**	0.49	0.0095
Plantago major	0.411	0.0023
Prunella vulgaris	0.553	0.003
Prunus avium	0.524	0.01
Rubus idaeus	0.407	0.0008
Senecio jacobaea	0.361	0.0173
*Tamus communis**	0.334	0.027
Taraxacum agg.	0.281	0.018
Veronica chamaedrys	0.341	0.0047

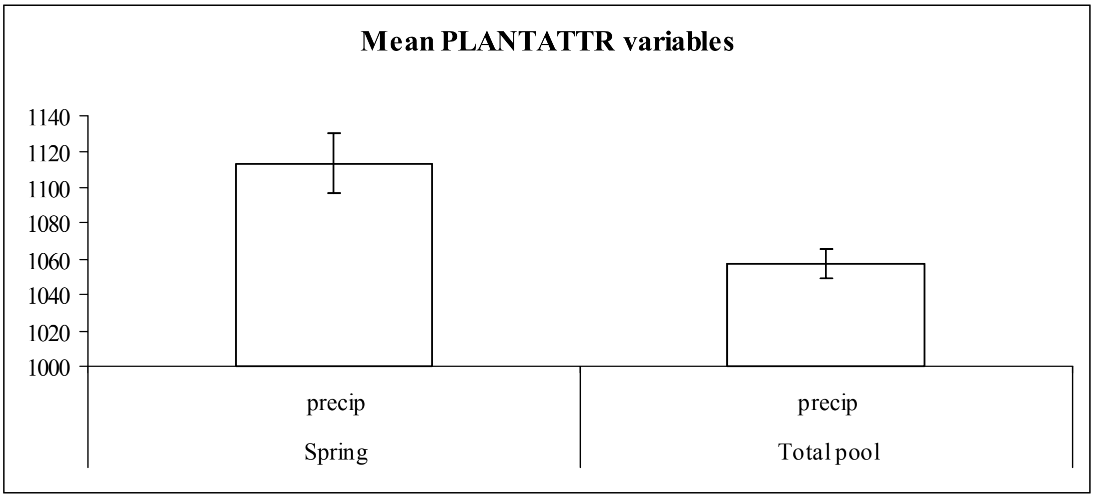


Figure 7.9 Comparison of characteristics of ‘spring change’ species with the total pool: association with higher mean precipitation conditions.

The characteristics of those species that showed a positive correlation between frequency and spring temperature change were exampled in comparison to a wider pool of species. The only apparently significant differences were that the Spring Change group had a higher mean precipitation value (data taken from PLANTATTR) and a lower maximum July value. There were no significant differences in either Ellenberg Light or Grime CSR values although the data suggested that the Spring group were somewhat less competitive and more stress-tolerant than the total species pool.

7.8 Analysis of change in total ground flora cover

A second piece of additional analysis tested for change in total ground flora cover as a potential discriminant between eutrophying effects and shading effects.

The hypothese were
H1 – As soil pH increases (assumed to drive an increase in nutrient availability given its positive correlation with N deposition) ground flora cover increases.
H2 – As basal area increases, more shade leads to reduced ground flora cover.
H3 – There should be a negative interaction between basal area change and soil pH change in their relationship with ground flora cover change. Hence, ground flora cover should show a lower increase or even no change in response to increased soil pH where woody basal area increased the most.
The mean ground flora cover was 72% in 1971 and 80% in 2001, (very similar to the figures reported for Wytham Woods from 1974, 1991 and 1999 (Kirby 2004)). The analysis suggests that both H1 and H2 are supported, (Table 7.21), but there is no evidence for any interaction between the effects of pH and basal area.

Table 7.21 Effects of change in pH and basal area on ground flora cover.

Effect	**Df**	**F**	**p**	**Sign and significance of the relationship**
Change in soil pH	1427	5.07	**0.0245**	+
Change in basal area	1398	10.36	**0.0013**	-
Interaction term	1383	0.48	0.4907	Ns

7.9 Summary of ground flora results

The main ground flora changes were as follows.

- Overall species richness per plot and per 'site' (set of 16 plots) declined markedly by 36% and 12% respectively. Considerable variation in species richness occurred between sites: increases were more common in the south and east, particularly in woods affected by the 1987 storms.
- The surveys in 2001, being earlier in the season, were expected to detect more species, particularly in the vernal flora. The changes in species richness, cover and frequency reported here were significant after allowing for this survey date difference.
- Although species frequency in 2001 was strongly correlated with that in 1971, more species declined in frequency than increased at both site and plot levels. 'Woodland specialists' were more likely than other species to show decreases in frequency.

- Some widespread species increased in cover, with woodland specialists again disproportionately represented among those increasing.
- Increases in tree basal area were associated with species richness declines; other variables relating to disturbance (1987 storm damage, grazing, open habitats) were associated with increased richness.
- There was an overall shift across the data set towards more shaded assemblages of plants, but no change in mean Ellenberg light score. Reductions in the numbers of open habitats recorded per plot and increases in basal area were associated with declines in Ellenberg light scores. Species increasing in cover were more likely to be those associated with semi-shaded (rather than open) conditions.
- There was no overall shift in species towards more fertile/eutrophic assemblages and no change in mean Ellenberg fertility score. Increasing soil pH and high levels of intensive land surrounding the wood were however associated with increases in Ellenberg fertility scores. Species increasing in cover were more likely to be associated with high (rather than low) nutrient status conditions. Changes in abundance for some species were correlated with increases in modelled nitrogen deposition.
- There was evidence that the vegetation response to increasing soil pH and increased fertility could be partly uncoupled by increased shading.
- Stress-tolerator species scores declined and were negatively associated with changes in open habitats. Competitor and Ruderal species scores did not show any overall change, but the Ruderal changes were positively correlated with changes in open habitats and negatively with basal area change.
- 51 species out of 332 showed a significant relationship (in all but four cases positive) between spring temperature change and change in frequency within woodland plots. Other species changed in cover in relation to increased growing season length, with both increases and decreases being found.
- Overall cover of the ground flora was positively correlated with increasing pH and negatively correlated with increasing basal area.

8. Summary of key results

8.1 Summary of changes

Changes in the plot vegetation overall:

- Between 1971 and 2001 sixteen of the 103 sites were affected by major change in land use and thirty-eight plots (2.3%) out of a total 1,648 locations were lost, with about equal contributions from urban development and agricultural activities.

- The overall balance of plot types, classified by NVC and CVS remained the same, but with small increases in plots of more fertile and more open conditions.

The main soil changes were as follows.

- Soil pH increased across sites, with a large reduction in the number with soil pH<5 and a corresponding increase in the number of sites with more alkali soil pH.
- The soil pH increase was more marked in organic than in mineral soil, and in non-calcareous than calcareous soils.
- The increase is consistent with other national trends from the Countryside Survey 2000 and the re-survey of the National Soil Inventory.
- There was no overall change in soil organic matter (SOM) although there were significant changes within 23 sites, of which 15 sites showed an increase;
- There was no increase in the mean level of plot SOM, but the number of plot with low levels of SOM (<10%)decreased.
- SOM increased for lowland soils and mineral and organo-mineral soils but declined in organic soils;
- There was a positive correlation between changes in modelled nitrogen deposition and increases in SOM.

The main changes to the tree and shrub species were as follows.

- Most tree and shrub species remained stable in terms of their frequency of occurrence at plot and site levels, although 15 species (eight of these shrubs) declined, whilst five other species (four conifers) increased.
- There was a net loss of stems from the smallest size classes (particularly less than 10 cm dbh) with some smaller gains in the 30-60 cm classes. Stems greater than 60 cm remained scarce.
- Individual species showed distinct patterns: for example
    - oak lost stems in the lowest size classes but gained in the larger ones;
    - elm and beech lost stems across the size class range;
    - holly increased substantially in the smallest size class;
    - hazel lost small stems.
- Mean basal area of trees and shrubs increased both for individual plots and across most sites.

- Species richness amongst saplings (25-130cm high) decreased, but small increases in frequency were shown by some shade tolerant species including yew, beech and holly.
- Seedling (< 25 cm high) frequency declined for most species, but holly showed a notable increase.
- Open habitats (rides, glades etc) and some wet habitats (ditches, boggy patches) became less common.
- Grazing signs increased in the lowlands; across GB the biggest increase in recorded grazing signs was for 'Other deer' (i.e. species other than red deer).

The main ground flora changes were as follows.

- Overall species richness per plot and per 'site' (set of 16 plots) declined markedly by 36% and 12% respectively. Considerable variation in species richness occurred between sites: increases were more common in the south and east, particularly in woods affected by the 1987 storms.
- The surveys in 2001, being earlier in the season, were expected to detect more species, particularly in the vernal flora. The changes in species richness, cover and frequency reported here were significant after allowing for this survey date difference.
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- Some widespread species increased in cover, with woodland specialists again disproportionately represented among those increasing.
- Increases in tree basal area were associated with species richness declines; other variables relating to disturbance (1987 storm damage, grazing, open habitats) were associated with increased richness.
- There was an overall shift across the data set towards more shaded assemblages of plants, but no change in mean Ellenberg light score. Reductions in the numbers of open habitats recorded per plot and increases in basal area were associated with declines in Ellenberg light scores. Species increasing in cover were more likely to be those associated with semi-shaded (rather than open) conditions.
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- Overall cover of the ground flora was positively correlated with increasing pH and negatively correlated with increasing basal area.

8.2 Summary of changes by numbers of sites

In the main analyses trends across all plots and sites have been considered. An alternative way of looking at the data is presented in Table 8.1. This shows the number of 'sites' ie sets of 16 plots where the change, just within that site, was significant. This highlights the following: the decline in ground flora richness, and in Stress-tolerator species scores, the decline in woody regeneration; the decrease in the Open habitat counts and increase in basal area of woody species; and the increase in soil pH. In the other tests the increases and decreases were more-or-less equally matched across sites.

Table 8.1 Summary table of outcomes of within-site tests of change in floristic, environment and management variables.

Variable	Increase	Decrease	No significant change
Ground layer species richness	8	58	37
Mean stress-tolerator score	4	25	74
Plot level woody regeneration	9	48	46
Basal area (trees and shrubs)	18	0	85
Plot level count of open habitats	2	25	76
Soil pH	41	1	61
Mean Ellenberg N (cover-weighted)	16	8	79
Mean Ellenberg R (cover-weighted)	14	13	76
Mean Ellenberg L (cover-weighted)	11	10	82
Mean Ellenberg N (unweighted)	11	10	82
Mean Ellenberg R (unweighted)	11	8	84
Mean Ellenberg L (unweighted)	12	16	75
Soil Organic Matter	10	15	78
Mean ruderal score	22	14	67
Mean competitor score	20	11	72
Plot level signs of recent management	7	8	88
Cover of grasses	18	17	68
Cover of Bracken + *Rubus fruticosus* + *Rhododendron*	16	15	72
Bryophyte cover	17	15	71

8.3 Summary of results in relation to potential drivers of change

In section 2.9 we set out potential drivers of change in the woodland samples and which variables might be expected to show some response to those drivers. Each is these is now considered in the light of the results described above. In the 'summary of relevant results' box the section number is given for where the result comes from.

Climate change

Explanatory variables	Response variables considered
Rainfall Intensity (RI)	Proportion of biogeographic elements in each plot
Growing Season Length (GSL)	Individual species changes
Change in mean max July	
Change in mean min January	
Change in the mean Jan-March temperatures	

Summary of relevant results

- Individual species responses, mainly decreases in cover and increases in frequency in response to Growing Season Length and mean Jan-March temperature (7.7).
- Holly increase (possible response to milder winters) (6.2, 6.4)

The correlations between climate factors and individual species changes indicate individualistic growth responses. The lack of an obvious ecological link between the species that showed a response makes it difficult to draw general conclusions about how the woodland flora as a whole will change in future.

Eutrophication and acidification.

Explanatory variables	Response variables consdered
GSL*N deposition	Ellenberg N score
N deposition	Ellenberg R score
Area of intensive land cover adjacent to site *N deposition	Soil pH
SOx deposition in 1970	Species richness
Soil pH category in 1971	Individual species changes

Summary of relevant results

- Soil pH increase positively correlated with modelled nitrogen deposition (5.1).
- Change in soil organic matter positively correlated with modelled nitrogen deposition (5.2).
- No overall change in plot Ellenberg fertility (N) score (7.4).
- Increased Ellenberg fertility (N) score where soil pH increased or woods were surrounded by intensive land use (7.5.1).
- Increases in mean Competitor score at 20 sites (declines at 11) (8.2).
- Increased ground cover with increasing pH (7.8).
- Increase in cover by some high Ellenberg fertility (N) score species (7.2).
- Individual species increases and decreases correlated with modelled nitrogen deposition change (7.5.1).
- Evidence for 'masking' of species response to fertility changes by shade increase (7.5.4).
- Some shift to more 'fertile' types in the CVS classification of the plots (3.3).

The soil changes are generally consistent with the modelled changes in the atmospheric deposition regime over the period, including with the reduction in SOx, although no significant correlations were found with this factor. The vegetation response is more complicated because of the interactions with other factors. Clear site and species specific responses to the changes in modelled nitrogen deposition were however detected.

Impact of intensive agriculture

Explanatory variables	Response variables consdered
Site area	Ellenberg N score
Site area*Perimeter/area ratio	Competitor score
Area of intensive land cover adjacent to site	Stress-tolerator score
Site area* Area of intensive land cover adjacent to site	Plot species richness
Perimeter/Area ratio* Area of intensive land cover adjacent to site	

Summary of relevant results

- Increased Ellenberg fertility (N) score where soil pH increased or woods were surrounded by intensive land use (7.5.1).
- No overall change in plot Ellenberg fertility (N) score (7.4).
- Increases in mean Competitor score at 20 sites (declines at 11) (8.2).
- Decline in Stress-tolerator scores (7.6)

There was evidence for impacts of intensive adjacent land-use on the Ellenberg fertility (N) score of the flora in the woods. Other increases in high ‘N’ score species were correlated with spatial variation in more general nitrogen deposition.

Increased shade

Explanatory variables	Response variables consdered
Weighted average of open habitats at plot and site level	Stress-tolerator score
Change in summed cover of Bracken + *Rubus* + *Rhododendron*	Ruderal score
Basal area of trees and shrubs	Competitor score
	Ellenberg Light (L) score
	Species richness

Summary of relevant results

- Increase in basal area of trees and shrubs (6.3)
- Loss of open habitats (6.5.1)
- Overall shift towards more shaded assemblages in the DCA (7.4).
- No overall change in Ellenberg Light (L) score (7.5.2)
- Increases in basal area at plot level correlated with declines in L score (7.5.2).
- Increases in cover of some semi-shade species (7.3)
- Decline in species richness negatively correlated with increased basal area (7.1.1)
- Reductions in Ruderal score correlated with fewer open habitats and increased basal area (7.6)
- Positive correlation between species richness and disturbance such as 1987 storm (7.1.1)
- Increase in holly and some other shade-tolerant species in regeneration (6.4)
- Grass cover declined with increased basal area (6.5.4).
- Decreasing overall ground flora cover with increasing basal area (7.8)
- Increase in more ‘open’ vegetation types in the CVS/NVC plot classification (3.3)
- Decline in Stress-tolerator scores, negatively associated with presence of open habitats (7.6)
- Greater proportionate decline in frequency of woodland specialists (7.2.1)

There is strong evidence for reductions in species associated with increased basal area of trees, reductions in open habitats in many woods, and increases associated with more openness. These are consistent with changes in the degree of shading in the woods. At the same time some plots have become more open. The decline in Stress-tolerant species and in the frequency of many woodland specialists also goes somewhat against the indications of a shade effect because these species might be expected to have benefited from increased shade.

Change in management/use of wood

Explanatory variables	Response variables
Change in number of signs of recent management	Basal area Weighted average of open habitats in each plot

Summary of relevant results

- Basal area has increased (6.3)
- Open habitats have declined (6.5.1)
- No overall trend in management sign counts, but fewer 'cut stumps' (6.5.1)
- Less evidence of management in upland woods (6.5.2)
- A minority of woods showed evidence of recent management (6.5.3)

The overall impression is that relatively little active forestry management has taken place in most of the woods surveyed over the last 30 years.

Changes in grazing regimes

Explanatory variables	Response variables
Count of grazing signs in plots (GS)	Ellenberg light (L) score
Site grid reference*Area of adjacent acid grassland	Ruderal score
GS*Area of site	Plot level woody species regeneration
GS*Perimeter:Area ratio	Species richness
GS*N deposition	Grass cover
	Changes in individual species (eg reduced bramble)

Summary of relevant results

- Increased grazing signs in the lowlands (6.5.4)
- Increase in 'Other deer' signs 6.5.4)
- No correlations between grazing signs and Ellenberg Light (L) or Ruderal scores (7.5, 7.6)
- Woody species regeneration declined, fewer small trees/shrub stems, but not correlated with grazing signs (6.2, 6.5.4).
- Changes in species richness correlated to internal changes including grazing (7.1.1)
- No significant links found between individual species changes and grazing signs (7.3)
- Shifts in some plots to more grassland-type vegetation (3.2, 3.3)

Increases in grazing levels are occurring in the lowland woods (where it was generally formerly low), but little sign of change in grazing in the uplands (where grazing has traditionally been high). Across all the woods no significant impact on the flora was detected. There are clear impacts on the vegetation of a small number of plots and woods where shifts to 'pasture' have occurred, and grazing may also be implicated in the woody regeneration declines.

8.4 How successfully has the linking of responses and variables been?

In many cases the amount of variation explained in the models linking variables to drivers of change was low. The variation explained by the different parameters tested was highest for change in ground layer species richness, followed by changes in Ruderal species scores and soil pH (Figure 8.1). With the proviso that 50% of the observed species richness change remained unexplained, the data can be used to suggest the basis for a conceptual model of how species-richness changes (Figure 8.2) within a site.

The low level of change that could be ‘explained’ by the variables is not unexpected, despite the high number of woods and plots. Other studies have similarly found only low explanatory power between long-term woodland change and potential correlates (eg Diekmann and Falkengren-Grerup 2002). The various explanatory variables used may not accurately reflect the impacts of the drivers they were selected to represent. An additional feature of this study is that the woods were deliberately chosen to have a high degree of variation in their composition; each wood potentially had a unique history up to 1971 and pattern of change thereafter. Even within woods Rackham (1975) notes how individual coppice plots in Hayley Wood respond differently to cutting. A further complication may be that the woods are in part responding to wider-scale changes in the landscape than have been considered, or to changes over longer time periods.

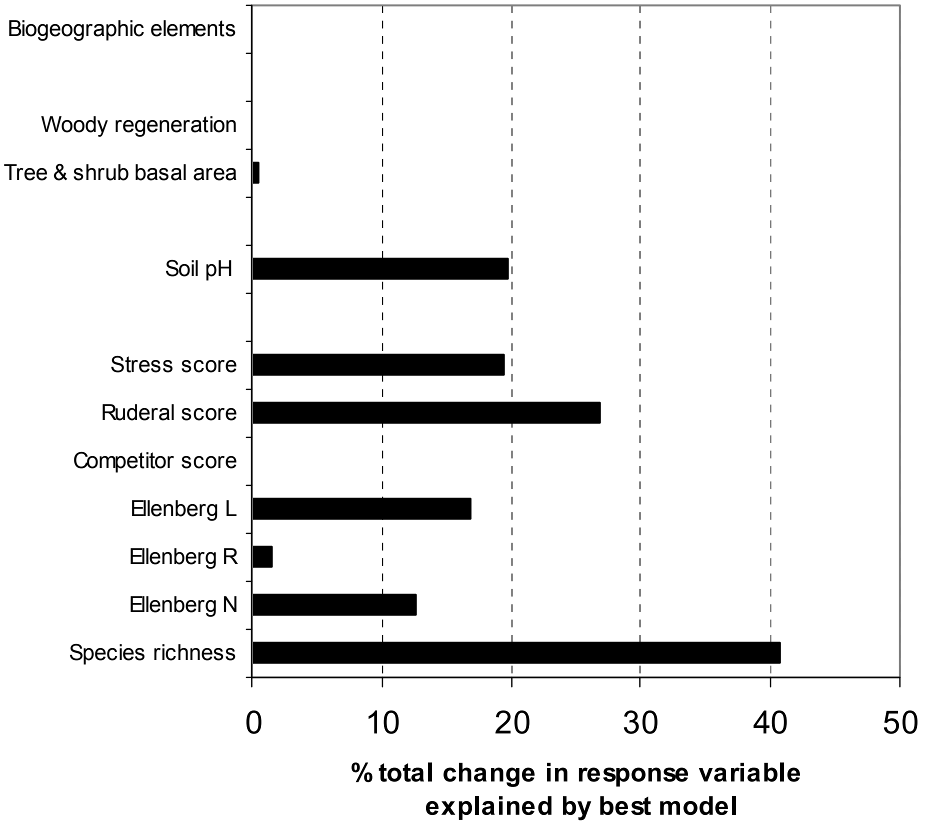

Figure 8.1. Explanatory power of the best fitting models for variables relating to broadleaved woodland change between 1971 and 2001. The y-axis shows the percentage of the observed variation in time accounted for by explanatory variables.

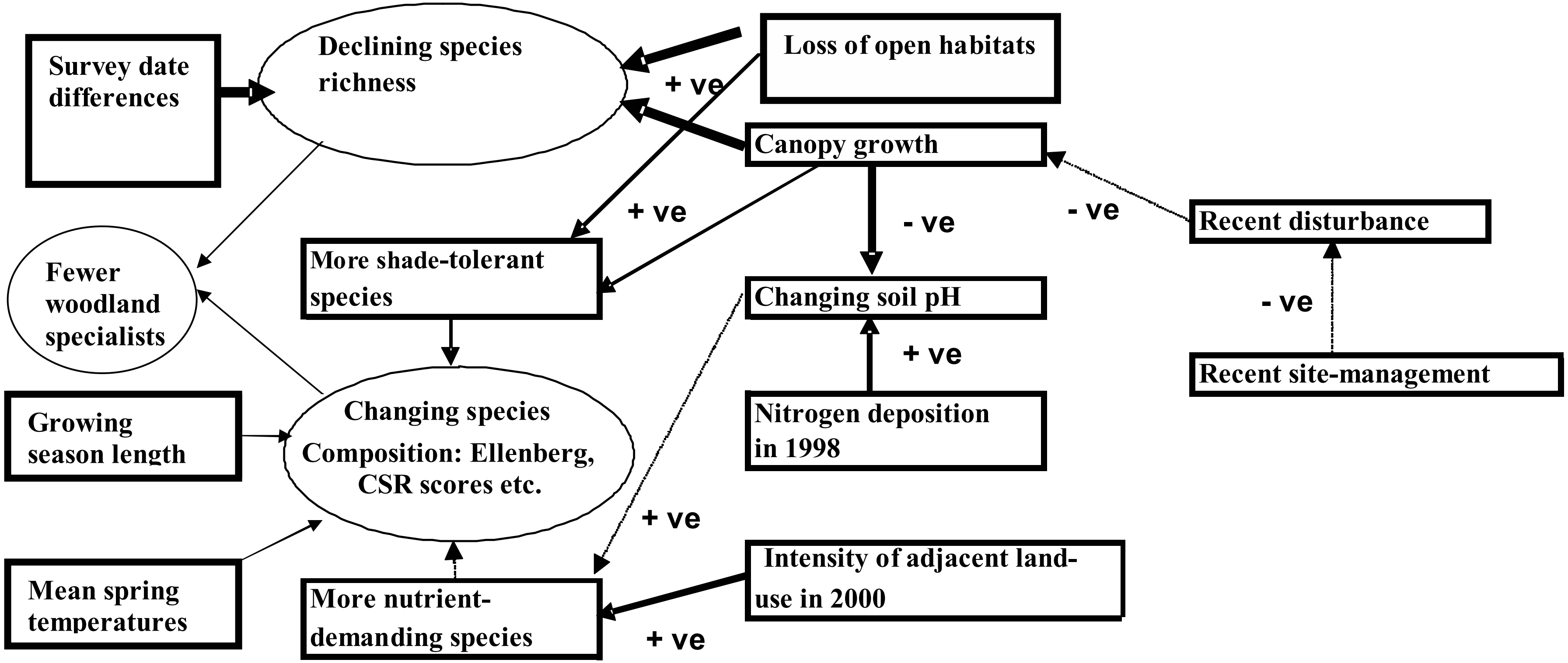

Figure 8.2. Conceptual diagram summarising significant correlative relationships found in this study between changing species richness and species composition and driving variables across British woodlands between 1971 and 2002. Thicker arrows indicate stronger correlations. + ve indicates a positive relationship e.g. both increasing soil pH and a greater area of intensive land-cover around each site in 2000 were associated with increasing Ellenberg fertility, and hence more nutrient-demanding plants.

9. Discussion

In the remaining sections of the report we bring together the results from the analyses of change and other relevant studies to try to provide an overall picture of what has been changing in British woodland between 1971 and 2001. Some areas for future research are highlighted. We suggest what the results mean in terms of the overall condition of the woods (chapter 10) and some possible implications for the ways in which future management and conservation might change (chapter 11). First however some of the inherent strengths and weaknesses of the study are revisited.

9.1 Strengths and weaknesses of the study

The study deals with one of the largest woodland data-sets yet analysed in Britain, covering 103 woods and 1648 plots. The woodland section of the National Vegetation Classification was based on about 2500 plots; the Peterken Stand Type system on about 800 plots (Peterken 1981). The Countryside Survey has about 536 plots within woodland. The 103 sites, if not fully representative, nevertheless cover a wide range of conditions in terms of their distribution, origins, size and composition (section 2.6). Other data often quoted in support of forestry policies frequently come from smaller, less representative studies of, for example the impact of intensive agriculture or diffuse pollution on woodland in Britain, the effects of deer grazing, or removal of conifers on woodland ground floras.

Nevertheless the sample size is limited, particularly in some of the comparisons where only sub-samples of the woods were considered (e.g. only 19 upland woods, only 10 woods in the 1987 storm track). In addition the initial selection procedures were based on incomplete surveys – notably the omission of small woods. While there are risks in extrapolating from these data to national trends for broadleaved woodland, the limitations should not be over-emphasised in comparison to other sources of woodland trends or data.

A second strength of this study is that the same sites have been studied over time, albeit with only two time points. We have not had to rely on ‘space-for-time’ chronosequence substitutions, as for example where stands of different ages are compared to infer the progress of change within a stand, or points along a transect are compared to infer the changes produced close to a source of pollution.

The study has concentrated on the changes that have taken place within the woods themselves over a 30-year period. Only limited consideration has been given to changes in the landscape around the woods over this time or to changes that may have taken place in the 50-100 years prior to 1971. Changes at both wider spatial and temporal scales may be significant factors in woodland dynamics. For example Petit and others (2004) and Lindborg and Eriksson (2004) suggest long-term and landscape effects on woodland and grassland floras, while Woodland Trust (2000) argue for the importance of cumulative core areas of semi-natural habitat. However Rackham (2003) questions the application of Island Biogeography theory to lowland woods. Peterken and Game (1984) similarly found little evidence for species ‘relaxation’ in Lincolnshire woods: parts of woods did not contain more species than isolated woods of the same total extent. These are areas worth further exploration.

Most of the analyses deal with correlations between different possible ‘driver’ variables and responses. Correlations, no matter how statistically significant, do not prove causation. The stronger the evidence for a direct mechanism linking the driving variable and a particular response the more reasonable it is to infer causation. Therefore in the following discussion we have tried to identify other studies that support (or conflict with) our inferred conclusions, or which point to the mechanisms that may be involved.

Some changes were correlated with several potential drivers; in some cases the inferences from different tests were contradictory. The effect of one change may mask the response to another, as with the correlation between Ellenberg pH (R), Fertility (N) and light (L) scores (section 7.5.4). The observed response in the vegetation to increased soil pH or fertility (as measured by the Ellenberg R and N scores) may be reduced if there is simultaneously a tendency to increased shade (favouring low L score species) and *vice versa.* We have cross-referenced the main instances where this might be important in interpretation of the results.

Finally we reiterate that there are further analyses that might be done, that might help to off-set some of the weaknesses identified above, or clarify areas where the data are ambiguous.

9.2 Relating possible drivers of change via landscape, woodland and stand level processes

In Figure 8.2 we summarized the statistically significant correlations found in this study between some of the variables as they related to species-richness. Figure 9.1 represents these and other possible interactions in a different way. This brings together drivers that tend to act through similar woodland processes – wherever the extra nitrogen comes from for example it tends to act on the flora in a similar way. We start with the factors that operate at the broadest scale, and work down to within-stand interactions. Climate change, external nutrient input and output fluxes, grazing regimes and stand dynamics are thus considered in turn, bringing together both results from this study and those from other work.

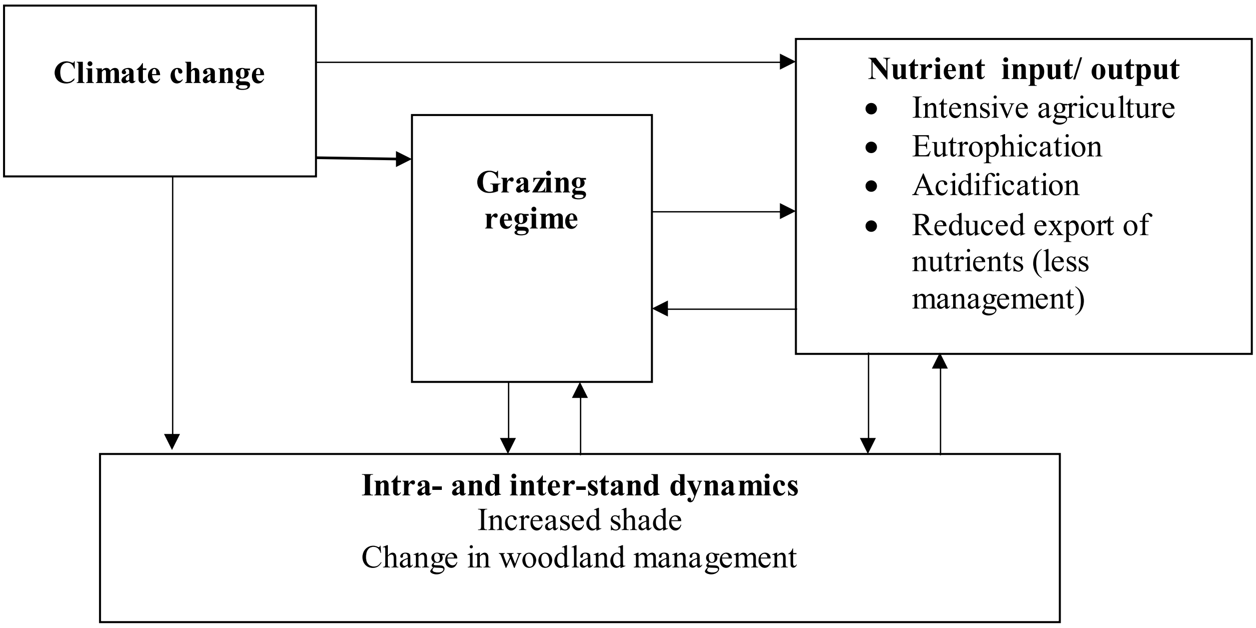

Figure 9.1. A simple model of the interaction of major processes and the possible drivers of change within woodland.

9.3 Climate change

Climate plays a key role in shaping the variation in woodland plant communities across Britain, as is apparent from the primary gradient revealed in the National Vegetation Classification (Rodwell 1991) and in the analysis of the variation in the original 1971 survey results (section 4). This effect arises from the direct impacts of climate on the distribution of individual species, and through effects on the outcome of the competition between species. For example the apparent northerly limit to the distribution of small-leaved lime is related to summer temperatures (Pigott and Huntley 1978); Rackham (2003) makes the case that *Primula* spp. could be particularly sensitive to an increase in summer droughts; while Peterken and Mountford (1998a) note the impact of the 1976 drought on beech *Fagus sylvatica* and birch *Betula* spp. in Lady Park Wood.

The growing season across Britain has advanced by, on average, three weeks in the past 60 years and this has driven well-documented phenological changes in plants and animals (Collinson and Smithers 2002; Collinson and Sparks 2003; Crick and Sparks 1999; Roy and Sparks 2000; Sparks and Smithers 2002). Over the next 100 years further change in the timing of growth and life cycle events, or in growth and survival rates could lead to major shifts in the distribution and abundance of species (Broadmeadow and others 2005; Harrison and others 2001).

The potential for climate change to influence the future composition and structure of our woods is thus considerable. As an example of the potential for multiple effects, a reduced dominance of beech, brought about by more frequent summer droughts, and its replacement by ash *Fraxinus excelsior* could lead to a greater ground flora cover, largely because of increased light penetration; in turn this could lead to the stands being more attractive to grazing animals. Less severe winters may mean less winter death of deer, higher deer populations, and hence more impact on ground vegetation and reduced regeneration of ash.

In this study direct correlations were found between climate change variables and the ground flora species responses, but no obvious shared ecological characteristics have yet been identified for the species showing such correlations. On the whole the evidence was for more increases in frequency than decreases, by both woodland specialists and more generalist species, but changes in cover were more mixed, with more decreases than increases. These individualistic species-responses make it difficult to suggest which species might be most affected in future. They do however support the idea that woodland species assemblages as they occur at present are likely to break-down and re-assemble in new ways.

9.4 Changing nutrient inputs and outputs

Soil nutrient and pH status is, with climate, a major factor explaining the distribution of woodland communities and species (Rodwell 1991; section 4). Within a stand soil nutrient and pH changes occur throughout its development cycle (Mitchell and Kirby 1989; Oliver and Larson 1996). This section is however more concerned with the additional variations introduced through changes to the nutrient regimes over the last 50 years; these include increased inputs from the atmosphere or surrounding farmland and reduced output of nutrients as harvested wood and litter.

9.4.1 Soil pH changes

A major finding this study was an increase in woodland soil pH during the last 30 years. Recent increases in soil pH have also been identified in the Countryside Survey 2000 and the re-survey of the National Soil Inventory (Haines-Young and others 2000; www.silsoe.cranfield.ac.uk/nsri/services), in contrast to earlier studies showing declines in woodland soil pH (e.g.Farmer 1995; Falkengren-Grerup 1990). The soil pH increase could be a sign of recovery from acid rain pollution (NEGTAP, 2001). We did not find any significant correlation with the estimates of SOx deposition, but this may be because the variable used was not an adequate measure.

Soil pH increase in our analyses was correlated with higher nitrogen deposition estimates (based on Countryside Survey 1998 data). Increased nitrogen deposition, as well as reduced acid rain pollution, can lead to a reduction in soil Carbon/Nitrogen ratio and increased nutrient availability (Diekmann and Falkengren-Grerup 2002). Most macro-nutrients reach maximum availability at soil pH values about 5.5-6.0 (Schaffers 2002). The mean change in this study was from 4.98 to 5.30 which suggests that any future increase in pH could take soils into and in some cases beyond the optimum for nutrients: in the 12% of plots where the change was from <5.75 in 1971 to >=5.75 in 2001 this may already have happened.

The pH increase in British woods suggests that acid rain may no longer be a significant issue, but the potential interaction of increasing pH with changing nitrogen levels is of concern. If, in the past, acidification buffered sites against increased nitrogen, then pH recovery may increasingly compound future nitrogen deposition effects.

9.4.2 Nitrogen enrichment

Sutton and others (2004) estimated that 90% of woodland received nitrogen deposition in excess of the estimates for critical loads (NEGTAP 2001). This, combined with the effect of changing pH regime, is likely to encourage species able to exploit more fertile surroundings. The Countryside Survey 2000 results showed a tendency towards more eutrophic vegetation although this was less clear in the ‘woodland classes’ (Smart and others 2005).

The potential for woodland vegetation change has also been demonstrated close to point sources of high ammonia such as intensive pig and poultry units (Pitcairn and others 1998). The impact of fertilizer over-spread into woodland has been examined; Gove and others (2004) concluded that there was little impact beyond the first 10m; Bateman and others (2004) report potential effects 20-30 m into woodland; while Kennedy and Pitman (2004) suggest even greater impacts.

In this study the eutrophic character of the ground flora was often mentioned in surveyors’ reports. Change in mean Ellenberg fertility (N) score was associated with the area of intensive land-cover surrounding sample sites and soil pH change was correlated with modelled nitrogen deposition for the surrounding 5x5 km sq in 1998. At Spital Wood, a narrow site, only *c*100m wide, the vigorous dominance of the ground layer by *Urtica dioica* was thought by the surveyors to be a consequence of the surrounding intensive arable land. More generally across the data-set there were some species (Table 7.10) that did show consistent increases or decreases in abundance correlated with modelled nitrogen changes. However there was not the expected shift in the overall flora to more competitive species or to a more eutrophic flora (Figure 7.4). In England, where the impact of increased nitrogen inputs might be expected to be greatest the overall change in Ellenberg N score was significant, but the reverse of that expected (Appendix 9). Kirby and Thomas (2000) were unable to detect a response in the vegetation to observed changes in soil pH and nitrogen content at Wytham Woods and Smart and others (2005) did not detect a eutrophication signal in woodland vegetation, unlike for some other habitats. These findings, however, contrast with Ling's (2003) work in Cotswold beechwoods and some continental European studies (e.g. Thimonier and others 1992; Diekmann and Falkengren-Grerup 2002; Brunet and others 1996, 1997,1998; Lamiere and others 2000).

Three effects may help to resolve the differences in these various studies.

- Firstly, compared to the flora of some other habitats such as heathland (where eutrophication effects are common), woodland plants as a group, and including the woodland specialists (defined as in Box 2.1) are more towards the middle and upper part of the Ellenberg fertility spectrum (Figure 9.2). This may make changes in woodland communities less easy to detect because more moderate to high fertility species are already part of the woodland assemblage.
- Secondly impacts, particularly on small woods (covered more by the Countryside Survey 2000 than by this study) may often be from localised sources of ammonia, so are less likely to register in a national survey focussed more on large sites.
- A third cause is the possible damping effect of increased shade on response to changes in fertility (Aarssen and Schamp, 2002) (section 7.5.4). Increasingly favourable conditions for ' base-rich' species along one habitat dimension represented by Ellenberg fertility (N) score might be outweighed by unfavourable conditions along another, in this survey represented by the Ellenberg Light score.

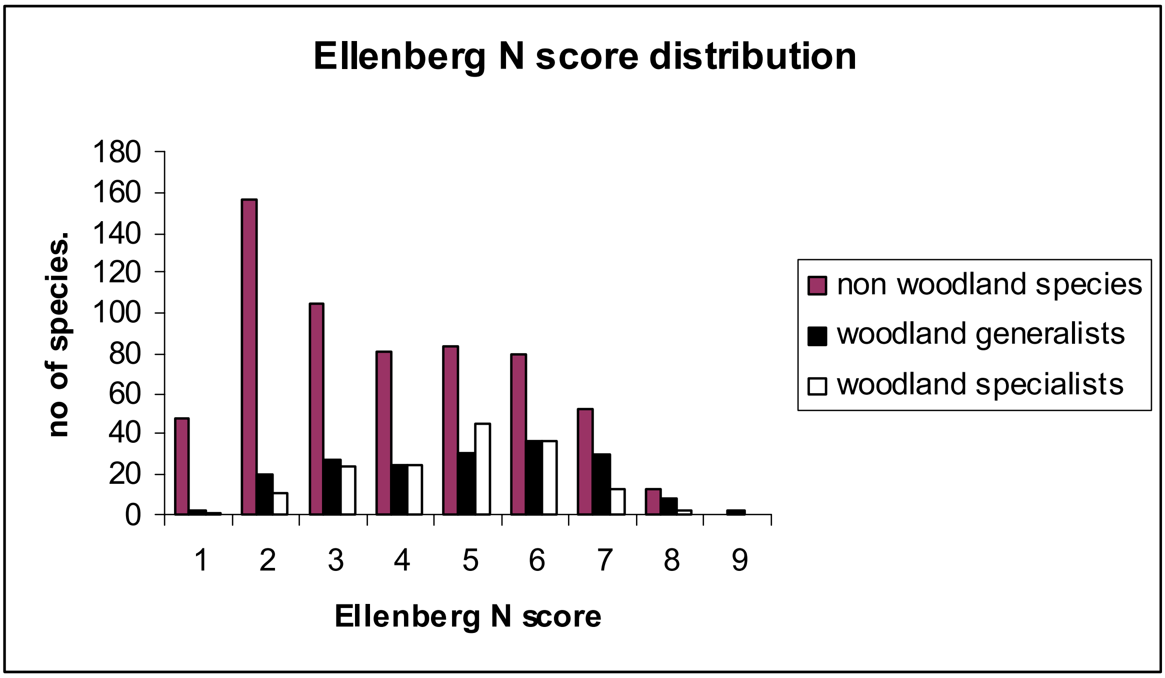

Figure 9.2 Ellenberg N scores for woodland specialists, woodland generalists and non-woodland species (Kirby and other in press).

Change in Ellenberg pH and Ellenberg light scores are negatively correlated so that shifts toward more shade-tolerant assemblages (e.g. movement along DCA axis 2, Figure 7.5) tended to favour a lower contribution from calcicoles and circum-neutral species. This may have preferentially filtered out some of the species most likely to take advantage of increased pH and nutrient availability. Diekmann (2003) recently raised the same confounding of light and acidity scores as a problem in signal attribution within Scandinavian woodland ecosystems although the interactions were different to those observed here.

The dampening effect of one effect on another could work either way – responses to increased nitrogen could be dampening a light-effect response. However as the light response tends to be more obvious we have emphasised it this way round. In the same way Kennedy and Pitman (2004) sought to factor out the effects of canopy gaps on the vegetation first in their study of the relationship between the ground flora and soil nitrogen.

9.4.3 Relations between nitrogen, pH and soil organic matter

Increased nitrogen deposition could drive down the soil C:N ratio, particularly if soil pH increased, reducing the soil carbon pool and accelerating decomposition processes (Chapin 2003; Paul and Clark 1996); hence soil organic matter would be expected to decrease.

More rapid litter decomposition and nutrient cycling however also depends upon the quality of litterfall. Persistent inputs of relatively high lignin content and low Specific Leaf Area litter from trees and shrubs slow any response to increasing nutrient availability because such tissues are harder to break down than, for example, nutrient-demanding grasses and forbs (Cornelissen 1996; Melillo and others 1982). If nitrogen deposition enhances the growth of trees, such that their dominance suppresses herbs, or shrubs such as hazel (Mohr and Topp 2005) with more readily decomposed leaves, the greater recalcitrance of the litter will result in an increase in soil organic matter.

Changes in the amounts of soil organic matter in relation to changing nitrogen or pH levels are therefore difficult to predict. In this study there was an inverse relationship between soil pH and SOM change, consistent with the first expectation, but SOM change was also positively correlated with nitrogen deposition (the second explanation).

Soils appeared to be responding differently in relation to location and soil characteristics.

- Organic soils were more prone to loss of SOM, which would be consistent with recovery from acidification, nitrogen deposition and also climate change (Evans and others 2001).
- Lowland and mineral soils saw increasing SOM which may reflect litter accumulation associated with greater canopy closure and lower nutrient turnover in surface soil layers, and perhaps less removal of material through timber harvesting. However, future canopy disturbances, such as gap creation, could trigger increases in litter breakdown and reverse this trend, leading to more nutrients being made available to the ground flora (Augusto and others 2002; Guo and others 2004).

9.4.4 Increased inputs or reduced outputs?

Concern about the changing nutrient status of woodland soils and vegetation has concentrated on the effect of increased inputs from atmospheric pollution. However a less-studied phenomenon is the degree to which lack of removal of material from the woods may be a contributory factor to nutrient enrichment. Up until the end of the nineteenth century, apart from the wood harvested, litter and bracken were often regularly collected for animal bedding, while bramble stems might be bundled into faggots for burning. Stock that grazed in the woods and dunged at night elsewhere, or were removed for slaughter, would also have been a long-term drain on the nutrient stocks, particularly of acidic soils.

The associated nutrient removals appear to have been balanced in most cases over a production cycle by aerial inputs or soil weathering (Mitchell and Kirby 1989). However the amounts of nutrients that might be removed through coppicing or other harvests – and now no longer are – may be much greater than the additional inputs (Hofmeister and others 2004). In one Polish study, in the plots where litter removal was stopped the ground flora became more 'eutrophic', but where litter removal continued the traditional woodland species were retained (Dzwonko and Gawronski 2002).

While not discounting the role of increased nutrient inputs, the possible role of reduced outputs should not be ignored.

9.5 Grazing in British woodland

Large herbivores are a natural part of woodland systems and have been part of the cultural management of British woodland for millennia (Rackham 2003). There is currently debate about the degree of openness of the post-glacial landscape and interest in sustainable and cost-effective mechanisms for maintenance of woodland and mid-successional mosaics using grazing (Bokdam and others 2002; Hodder and others 2005; Kirby 2004; Vera 2000).

There is also increased appreciation of how grazing and browsing may influence the biodiversity of woods, not just through effects on tree regeneration, but through direct impacts on other species and indirectly through changing the woodland structure (Kirby and others 1994; Kirby 2001). Grazing can promote species richness in the ground flora by checking the vigour of suppressive dominants; in other studies grazing has led to losses in species richness. Other changes reported include the spread of grasses such as *Poa trivialis, Deschampsia cespitosa* and *Brachypodium sylvaticum* a decline in bramble *Rubus fruticosus* and forbs such as *Mercurialis perennis* (Crampton and others 1998; Cooke and others 1995; Kirby 2001; Kirby and Thomas 2000).

In 2001 only a few woods with deer grazing signs were judged to be over-grazed by surveyors. There was however an increase in grazing signs in the lowland zone. Across GB there was a large increase in signs of 'Other deer' (i.e not red deer), with little change for other animals. As deer were the most common grazing animal recorded in the lowland zone in 2001 this is consistent with other evidence for increases of deer in lowland woods (Fuller and Gill 2001). There was a positive relationship between change in signs of grazing and ground flora species richness, as has been found in some, but not all, other studies (Kirby 2001). No effect of grazing on grass cover was detected, nor were any individual species cover changes significantly positively or negatively associated with changes in grazing signs.

A much larger proportion of sheep-grazed sites were judged over-grazed, even though there was some indication that sheep grazing signs had decreased in parts of the uplands. Surveyors' reports indicated that localised switches to grassland at the expense of woodland were associated with sheep or in a few cases, cattle rather than deer, along with in some cases localised felling. On sites such as Fridd Wood in Wales, and Reins Wood in Northumberland, grazing appeared to be blurring the boundary between open grassland and broadleaved woodland resulting in the development of eutrophic wood pasture.

9.6 Stand growth and succession in the woods

Woodland stands typically go through a series of stages, from open conditions immediately after felling or natural gap creation, through a dense thicket stage, then opening-up as the trees mature (Bormann and Likens 1979; Mitchell and Kirby 1989; Oliver and Larson 1996). Young stands typically consist of many small stems, whose numbers are later reduced during the stem exclusion phase. Later as gaps start to appear in the canopy some regeneration of trees and shrubs may again be possible – understorey re-initiation.

In the nineteenth century many woods were treated as coppice or coppice with standards. Even in the 1947 forestry census 140,000 ha was still classed as coppice or coppice with standards (HMSO 1952) compared to 24,000 ha in 2001 (Forestry Commission 2003). During both the first and second world wars there was extensive felling of privately owned woods (where most broadleaved woods occurred); the 1947 census records 60,000 ha as 'devastated' and a further *c* 268,000 ha as felled.

Consequently most broadleaved stands are less than 100 years old: 48% of the broadleaves have been planted/regenerated since 1940 (Forestry Commission 2003). Woods entirely composed of young, very open stands in the late 1960s would probably not have been selected by the surveyors in the 'Steele' surveys (section 1.2) and hence would not have been considered for the 1971 survey (R.G.H. Bunce personal communication). However the scale of the second war fellings was such that few woods escaped without some cutting; some reports (Richards 2003) refer to half the trees or half the woods being cut-over: 46 % of broadleaved parcels in 1947 were classed as uneven-aged (HMSO 1952). Many broadleaved stands in the mid-twentieth century would probably therefore be closest to the early stages of stand initiation and stem exclusion in the Oliver and Larson (1996) model.

In this study the overall tendency was as expected from the above history of woodland management: in 1971 small stems predominated and large stems (over 60 cm diameter) were very rare. Between 1971 and 2001 the smallest stems tended to be lost, but there was recruitment into the middle-sized classes, with a general trend towards upward shifts in diameter distributions and increases in basal area. In terms of forest succession models the cross-site change reflects movement from a reverse J-shaped age-class distribution where recruitment outweighed mortality in 1971 to, in 2001, the stem-exclusion stage.

This successional trajectory is consistent with some of the principal changes seen among woody species such as the decline of birch and hazel and the increase in juvenile holly. Other observed changes that were consistent with the increased basal area include:

- increases in litter production which might contribute to the soil organic matter increases;
- changes in soil pH. Overall in this study soil pH tended to increase but there was a negative association between pH change and basal area increase. Soil pH decline is common in conifer stands through the growth cycle and there are some indications for this effect under oak and beech stands (Mitchell and Kirby 1989).

Different stages in the stand cycle support different suites and abundances of species: for example the difference between the birds, butterflies or plants at different stages in the coppice cycle are well-described (Buckley 1992). Chronosquence studies in plantations indicate marked changes in the character, richness and cover of the ground flora between the open ground/gap stage, the thicket stage and the maturing stand (Kirby 1988, 1990, 1993).

The richness of a plot at any one time will therefore depend on the stage that it is at in this cycle: the richness of the wood, as assessed from a set of plots, will depend on how these are distributed across the growth stages – if most are in the open phase the overall richness is likely to be higher than if most are at thicket stage. Moving up the scale to a set of woods, the mean richness recorded will be lower if most woods are relatively dense and dark because of their age structure, than if the woods between them span the range of age classes from relatively young to open, veteran stands.

Forestry census data and the results from this study suggest a degree of synchronicity in British woodland in the 1971-2001 period, with more of them being in the younger, denser stages and showing basal area growth, than in the post-mature/veteran stage. Basal area and canopy cover are closely linked (e.g. Savill1991), so increases in basal area would be likely to mean that the woods had become more shady – the surveyors reports do describe many of the woods as ‘dark, overgrown and hard to access’. This natural trend in the growth of the stands coincided with a period in which broadleaved silviculture has generally been seen as unprofitable; the general impression gained by the surveyors was that most woods had had little management since 1971. There may therefore have been less opening up of gaps the canopy by thinning, felling or coppicing than occurred in the past. The effect of changes in the canopy cover on species richness is illustrated directly in the results from Wytham Woods over a similar period (Figure 9.3).

If part of the reduction in plot richness is due to the pattern of stand development in the woods then we might expect future increases (other factors being equal) as the stands subsequently open out. This will happen, even without active management, for a variety of reasons. Loss of individual trees through disease can create small gaps, a notable example in this and other surveys being the loss of elm through Dutch Elm Disease (Kirby and Thomas 2000; Peterken and Mountford 1998b). Other natural disturbances can also create the open space and young growth conditions (Buckley and others 1994; Evans and Barkham 1992; Mountford 2004; Pickett and White 1985). The 1987 storm for example had a major effect (see Appendix 4) to the extent that the basal area and species richness trends in the woods affected were the reverse of those for the rest of the country.

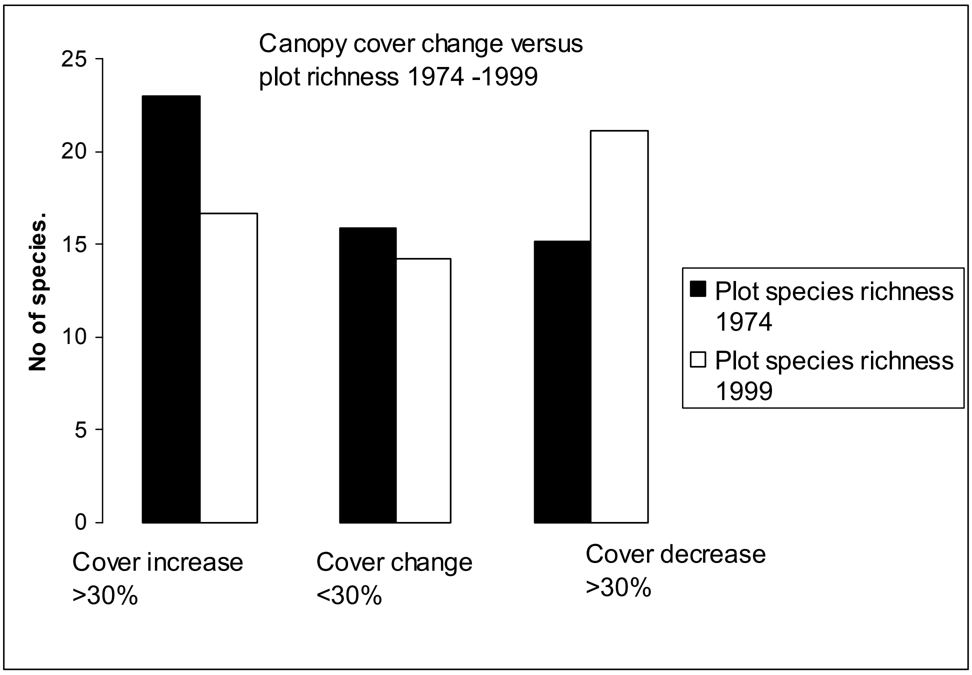

Figure 9.3 Mean plot richness increased or decreased significantly in plots where there was >30 change in estimated canopy cover over the period 1974-2001 (Wytham Woods, unpublished data, K J Kirby)

However most broadleaved woods are small: in England and Wales for example 66% of ancient woods, by number of sites, are in the 2-10 ha class (sites less than 2 ha not included) (Spencer and Kirby 1992). The likelihood that gaps will occur regularly in any particular wood through natural processes are rare which increases the risk that there will be a loss of continuity of open space conditions in many sites. Such continuity is particularly important for many invertebrates, such as woodland butterflies (Warren and Key 1991), because of their annual life cycles, but may also be valuable for some plants.

9.6.1 Why have woodland specialists declined so much?

The disproportionate reduction in the frequency of woodland specialists, and also the reduction in stress-tolerators, was not expected. If one of the major drivers of change was an increase in shade, these species would be expected to be amongst the best able to cope with the change. More work is needed on this question, but the following is put forward as a preliminary set of ideas.

- The woodland specialist definition used is broad and includes the species that make up much of the ground flora in many woods. As they are the most common species it is more likely that declines in them can be detected than for less common non-woodland species.
- Many of the species are primarily species of dappled shade and woodland edge. These species do not necessarily thrive in continuous deep shade, but equally may not be suited by the plots that have opened up to the extent that an increased number have been classified as grassland communities. Barkham and Hance (1982) for example showed that alternating periods of open and closed conditions were most suitable for maintaining populations of daffodils *Narcissus pseudonarcissus* in woodland. Half-

shade species may be particularly vulnerable to periods when there is little disturbance to the canopy in the wood.

- Not all woodland specialists have declined; they are well represented in the species that have increased in cover generally (Table 7.7) and for example in relation to increased nitrogen (Table 7.10) deposition. It may also be significant that overall the ground flora cover has not reduced (Table 7.21). Under conditions of relatively uniform shade, with reduced disturbance both to the canopy and to the ground vegetation itself, species such as *Allium ursinum, Mercurialis perennis, Carex pendula*, are capable of forming dense cover that excludes other species. Competition within the ground flora may be one of the factors that we have not considered sufficiently in our analyses (Ernst 1979; Kirby and Woodell 1998; Pigott 1977).

9.7 Further research questions

This has been a preliminary set of analyses. Our intention is that the data will be made available for further study. The following are areas that we think are worth more detailed investigation.

A difficulty in identifying general trends within the data-set is that it does cover a wide range of conditions. One solution to this would be to sub-divide the data, for example according to woodland type or soil type, and look in more detail at these sub-groups. Ultimately each individual wood might be looked at as a case-study in how different factors may influence changes at a particular site.

We have carried out only very limited work on looking at wider landscape or temporal effects: there is scope for more analysis, for example looking at the influence of regional species pools (eg from data in the latest BSBI atlas); changes in land-use during the 1971-2001 period (from Countryside Survey data); and possible influences of longer term land-use change – for example making use of the 1947 forestry census reports for the woods and the 1930s Land Use Surveys of Sir Dudley Stamp.

Some of the analyses suggest an influence of adjacent land-use on the flora: this might be explored further by using different measures of woodland core area or cumulative core area of semi-natural habitats (Woodland Trust 2000) in analyses. Other types of landscape metrics might be applied to the data.

The measures of diffuse pollution used were necessarily somewhat crude to cover the whole of GB. More sophisticated models of deposition, concentrating on woods in those areas where high levels of pollution are known to have occurred might produce clearer results.

Climate models are becoming more sophisticated. The links between phenological changes and growth patterns of different ground flora species might help us to identify which species are likely to respond most under the different scenarios. The data collected here may then be useful in testing some of the hypotheses that are generated through ‘hind-casting’: do the observed changes match the predictions of which species might change?

Some of the changes identified are comparable to results from European continental studies; in other cases different trends have been identified. A more systematic comparison with continental work would be desirable, concentrating on similar environmental zones.

Finally at least some of the changes in the woods reflect stand growth and succession processes. The data here present opportunities for exploring the development of predictive models of how the ground flora responds to such processes. The current pattern of size classes could also be projected backward to explore when different stands were initiated.

10. Using the results to explore changes in the 'condition' of broadleaved woodland 1971-2001

The conservation agencies have developed an approach to the assessment of woodland condition in Sites of Special Scientific Interest based on five broad attributes: their extent, structure and natural processes, tree and shrub composition and quality indicators such as their ground flora (JNCC 1998; Kirby and others 2002). Where woods are judged to be meeting the targets for each of these attributes they are classed as in Favourable Condition.

In England this has been translated into a government target that 95% of SSSIs should be in favourable condition (or recovering towards that state) by 2010. The Habitat Action Plans for woodland priority habitats also include targets for improving woodland condition, based on a similar approach to that used for the SSSIs.

For the woods in this study that are also Sites of Special Scientific Interest further work is planned to compare the changes at a site level with the results from the more general condition assessment surveys. In the following section we explore the extent to which the trends identified across the woods sampled in this study indicate overall changes in the condition of broadleaved woodland more generally.

10.1 Woodland extent

Nationally the trend has been for the area of broadleaved woodland to increase in the last 30 years (Forestry Commission 2003). However new woodland will takes decades or centuries to reach a comparable level of community development to that of existing sites (Peterken 1977).

Complete destruction of woodland SSSIs is now relatively rare; most of the recent cases involve loss through planned development, for example at Birkham Wood (N Yorkshire) or along the line of the Channel Tunnel High Speed Rail Link. Other ancient woods under threat are detailed on a web-site (www.woodsunderthreat.info). However at least in England the draft Planning Policy Statement 9 proposes further protection for ancient woodland through national planning guidance.

The nature of this survey was such that extent could only decline: 38 plots were lost (Table 3.1).

10.2 Structure and natural processes

Different woodland structures favour different groups of species: some use temporary bare ground patches, or dense shrubby growth, others depend on large dead logs or veteran trees. At the individual wood level therefore the appropriate balance of structures for it to be in favourable condition will depend on the particular interest of that site (Kirby and others 2002).

However across the population of woods in this study it is possible to suggest likely consequences of the changes in structure identified, notably the impact of a generally aging tree population with loss of the smallest size classes of stems, often shrubs; and reduction of open space.

Although there was an increase in holly, overall this study showed a loss of small stems and shrubs. The decline in shrub layers, which has been noted in studies of individual woods (eg Kirby and others 1996; Crampton and others 1998), is implicated in the declines in woodland birds (Fuller and others 2005). The loss of open space, both temporary and permanent, is a factor in the decline of many woodland butterflies (Warren and Key 1991). Therefore, there are likely to have been reductions in the suitable conditions for these two groups (woodland birds and butterflies).

Dead wood levels showed some reductions in small diameter deadstems in this study, but increases in other measures, particularly in the south-east (storm-track woods). Even so the overall amounts of dead wood were relatively low. The significance of retaining dead wood for biodiversity has received more attention recently (Butler and others 2003), not least because of evidence for the degree to which it is lacking in most managed forests (Kirby and others 1998).

Within the woodland SSSI condition assessments poor woodland structure is often put down to lack of recent management and efforts have been made to address this through, for example encouraging coppicing (Kirby and Solly 2000). However there is increasing interest in exploring more the degree to which natural processes may be able to deliver nature conservation objectives and reducing the degree of human intervention (Peterken 2000). The results from the woods in this study in the 1987 storm track suggest how natural processes might operate to maintain ground flora richness.

10.3 Regeneration potential

Regeneration in woodland is often sporadic both in time and space in any one wood. Therefore in any one wood there may be an imbalance between signs of regeneration and the tree and shrub composition, although it might be expected that there would be more coherence across a woodland population as a whole.

In this study there has been a reduction in saplings and seedling abundances between 1971 and 2001, except for some shade bearers, notably holly. The lack of natural regeneration of appropriate species may be the reason for new broadleaf planting reported on 18 sites. Scarce regeneration might be in part just a consequence of a bias in the age structure across the woods and is therefore not necessarily a cause for concern. However the increase in grazing in lowland woods and signs of deer poses a potential threat to future regeneration. In England this has lead to the development of a joint strategy for sustainable deer management.

10.4 Tree and shrub composition

In the SSSI condition assessment surveys favourable condition with respect to the tree and shrub composition generally means a predominance of native species (Kirby and others 2002). Sites may be classed as unfavourable under this attribute because for example they have been replanted by conifers or an understorey of rhododendron. In ancient woods outside SSSIs recent policy and practice changes are leading to restoration of native trees and shrubs in place of planted conifers (Curtis and others 2002; Goldberg 2003; Pryor and Smith 2002; Thompson and others 2003).

In this study there was a small increase in conifers and introduced shrubs, such as rhododendron and cherry laurel on some sites, but also there were signs of active management to remove introduced trees and shrubs (Table 6.11).

There were changes in the abundance of native trees and shrubs in this study which could be important at the individual site level, for example the loss of elm and hence the food source for associated invertebrates, or, more generally, the increase in holly. An increase in the shrub layer (mainly holly) at Nagshead Wood has been implicated in a reduction in pied flycatchers because of increased competition from tits (Stowe 1987); whereas at Ebernoe Common the increased holly understorey is seen as favouring rare bats over more common species.

10.5 Quality indicators

Even if there are no changes in the tree and shrub layers there may be changes in the ground flora and in the SSSI condition assessment surveys these are recorded under this attribute. The concern is not just with shifts in individual (perhaps rare) species. Changes in the overall abundance and composition of the ground flora may be important in themselves and fluctuations in the ground flora can be quite large over periods of decades (eg Kirby and Thomas 2000). Such changes may represent cyclical patterns related to the tree growth stages, or they may provide pointers to other processes taking place in the woodland system.

In this study the range of species recorded and the pattern of abundances were similar in the two surveys – species common in 1971 tended to still be common in 2001. The range of vegetation types, whether assessed by the National Vegetation Classification or the Countryside Vegetation Survey classes showed little change. However the overall species-richness per plot declined. The reduction was strongly marked in woodland specialist species, not just those associated with open spaces. Individual sites and plots showed a variety of patterns of species change and the decline might have been even more pronounced except that the block of sites in south-east England (largely within the 1987 storm track) maintained their richness.

The species lost from the samples may not have been lost completely from the site (the plots are only a sample of the vegetation); and some of the reduction in abundance may only be temporary. However reduced abundance increases the vulnerability of species populations to external pressure and there were indications that changes in individual species and groups of species were occurring in response to climate change and increased nitrogen inputs.

10.6 Overall assessment

The broad composition and structure of the whole suite of woods was not dramatically different in 2001 from that in 1971. Some of the results indicate some recovery from past damage – for example the increase in soil pH. Other changes, most particularly the decline in woodland specialist richness, represent deterioration in the quality of the woods. The final section explores whether these trends are likely to continue and what might be done about them if they are judged to be damaging.

11. Future trends?

Some potential drivers may not yet show an impact because there is a time lag between their observed operation and the woodland response. In woodland, more perhaps than in other habitats, long-term studies have a critical part to play in ensuring that such lags can be identified. If we do not understand the past we cannot prepare for the future.

All long-term woodland studies gain in value with time: equally we are conscious that some of our analyses would be more useful if there were not such a long gap between the survey times. Therefore consideration will be given to

- making the data available for further work;
- linking at least some of the sites surveyed into other woodland surveillance programmes.

11.1 Climate change

We found effects on species distributions and abundance correlated with climate change over the last 30 years. Given the changes that are already being observed in the phenology of species, it seems likely that effects on woodland species abundance will become even more common in the next 50 years (Harrison and others 2001). If the rate of global warming accelerates in the coming decades, longer-growing seasons and more temperature extremes may exert greater impact upon the competitive balance between canopy dominants, for example via outbreaks of disease or drought-stress. In turn this will impact on the assembly and dissolution of dominance hierarchies among herbs and shrubs following canopy gap formation and eventual closure, as well as through direct impacts.

Stress-tolerators that are poor dispersers, and also tend to be less able to capitalize on increased nutrient availability, may lose out to more competitive species that can accumulate more biomass as the growing season lengthens, or can take advantage of more fertile, lower carbon-nitrogen ratio soils. On the other hand there may be opportunities for some species to spread; warmer summers for example might enable small-leaved lime to regenerate more freely in the north than it does at present (Pigott and Huntley 1978).

No simple woodland management response can be made to offset the future impacts of climate change, because we cannot predict with any certainty what the impacts will be in terms of species responses.

- In general larger populations (or meta-populations) are more likely to survive and spread than small ones. This supports the case for an increase in woodland area and for improving connectivity within the landscape, but also for addressing other causes of species decline that do have a clear management solution.
- Encouraging appropriate heterogeneity in woodland structures at stand, site and landscape scales may increase the likelihood that at least some populations of species will remain in suitable conditions as climate changes.
- As part of the above, the reasons why certain management regimes are favoured needs to be revisited: where coppice has been encouraged to provide local hot-spots, e.g. for

fritillary caterpillars this may not be so necessary if summers become warmer. Maintaining humidity within woods may become more critical.

11.2 Nutrient status

Signs of eutrophication in the ground flora were detected that were correlated with models of diffuse pollution and the management of adjacent land. Changes are taking place in agricultural practices as a result of changes to the Common Agricultural Policy and incentives. However these are unlikely to result in major reductions in intensive agricultural practice, at least in the lowlands in the short term. Many woods therefore remain vulnerable to accidental spread of agricultural chemicals, particularly fertilizers.

- Increased nitrogen inputs (and also spray drift effects) at the edges of woods from adjacent agriculture can be addressed in part through:
- developing buffer strips next to woodland and
- developing dense vegetation at the wood edge: in effect establishing a scrub/grassland 'ecotone'.

The significance of diffuse pollution impacts is becoming more widely appreciated. While some effects, such as acid rain, may be decreasing others are not. Landscape-scale changes, such as those being considered under the Water Framework Directive, may be needed to reduce the impact of widespread nitrogen inputs on woodland flora.

- Increased nutrient input might be compensated in a few situations by increasing nutrient output (as coppice, litter, animals in wood pasture systems).
- Impacts may also be reduced by maintaining high shade levels – in effect ensuring that light levels rather than nutrients act as the limiting factor.

11.3 Management changes

Given the generally young nature of most of the stands (Forestry Commission 2003), then without deliberate management intervention broadleaved woods are, on average, likely to become older and darker in the next twenty years. This could benefit some species and communities – those of fallen dead wood and shade-loving conditions - but may lead to continuing decline in much of the ground flora and also other groups associated with open space and young growth. By contrast there was evidence in this study that disturbance/openness of woodland did increase ground flora richness, particularly the ruderal elements of the woodland flora. Some woodland specialists may also respond best under varying open and closed phases (Barkham and Hance 1982).

In the longer term, even in woods left undisturbed there could the development of new equilibria between open areas and closed stands as gaps form, for example from windthrow. However we have few stands that have been left alone for more than a century to act as references as to what such new equilibria might be. Will natural disturbance provide enough openness for the species that we have come to value, that depend on such conditions? Is it feasible to return to the high levels of species richness shown in the 1971 survey for higher plants (and the same argument might also apply to butterflies and birds) if this was a consequence of one-off set of circumstances with their origins in the mid-twentieth century?

Much conservation effort has in the past gone into opening out woods, through for example coppicing and ride management (Fuller and Warren 1990; Peterken 1991; Peterken and Francis 1999), but the environment under which coppicing developed has changed (Kirby 1998). Now the increased light associated with coppice coupes may interact with grazing to favour, not the woodland specialists but more competitive species; under warmer summers (as in some climate models) more open woods may expose the ground flora to greater drought effects; and increases in available nitrogen may also favour competitive species rather than the more stress-tolerant ground flora.

- Opening out the wood temporarily may increase the abundance of some species, which is desirable if existing woods are to act as sources for the colonisation of new woodland. It may increase the likelihood that species can spread under the influence of changing environmental conditions.
- In undertaking such management we must be aware of the increasing potential of interactions with and between other drivers (climate change, nitrogen deposition, deer grazing) to influence the outcome.

11.4 Endnote

Some of the woods studied have been present since the Medieval period. Thirty years is just a tiny part of their history. All woods are unique and each time period is different. We have identified major drivers and patterns of change over the last thirty years, but there are limits to how far we can go in extrapolating such patterns to other woods and to future years from this study alone.

Integration and comparison with other long-term monitoring work, such as the Environmental Change Network, the UK Phenology Network, the long-term plots and transects on National Nature Reserves/Sites of Special Scientific Interest, and with broader-based surveys such as the National Inventory of Woodland and Trees and Countryside Survey 2006 will extend our ability to predict future changes.

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