**Table 6.4**   Vulnerability of all Red Data Book butterflies and a small selection of RDB macromoths to climate change using five scenarios (Table 6.3). The moths were chosen to illustrate different patterns of geographic ranges; almost all the butterflies have a strong south or south-eastern bias to their distribution. For each of the five scenarios the percentage of the current area of occupancy (number of 10-km squares), which would remain within the species' climate range, is calculated for each of the three variables (continentality, mean temperature and mean annual rainfall).

Species	Percentage of current range occupancy given climate scenario				
	1	2	3	4	5
Butterflies					
Carterocephalus palaemon	100	100	100	100	100
Nymphalis polychloros	100	100	100	100	100
Argynnis adippe	100	100	100	100	100
Hesperia comma	100	100	93	97	100
Papilio machaon	100	100	86	98	100
Cyaniris semiargus *	100	100	91	91	100
Mellicta athalia	100	100	88	95	100
Aporia crataegi *	100	100	86	94	100
Maculinea arion *	100	100	75	85	100
Lycaena dispar *	100	100	0	82	100
Melitaea cinxia	100	100	14	36	100
Strymonidia pruni	100	100	0	41	100
Macromoths					
Eustroma reticulatum	100	100	71	100	100
Orgyia recens *	100	100	88	94	100
Catocala promissa	100	100	25	62	100
Cucullia gnaphalii	100	100	15	31	100
Coscinia cribraria	100	43	0	14	0
Phyllodesma ilicifolia	71	71	57	75	14
Lycia lapponaria	56	88	88	69	44
Anarta cordigera	31	38	92	69	15
Photedes captiuncula	13	93	87	60	20
Hadena caesia	11	78	78	56	11
Lycia zonaria	22	39	39	33	33
Sabra harpagula	33	0	0	67	0

* based on former range

62

Figure 6.1 Mean altitude of 10km squares of Great Britain. Large black circles are below 5m a.O.D., small black circles are 5-10m a.O.D., open circles are 300-500m a.O.D., triangles are >500m a.O.D.
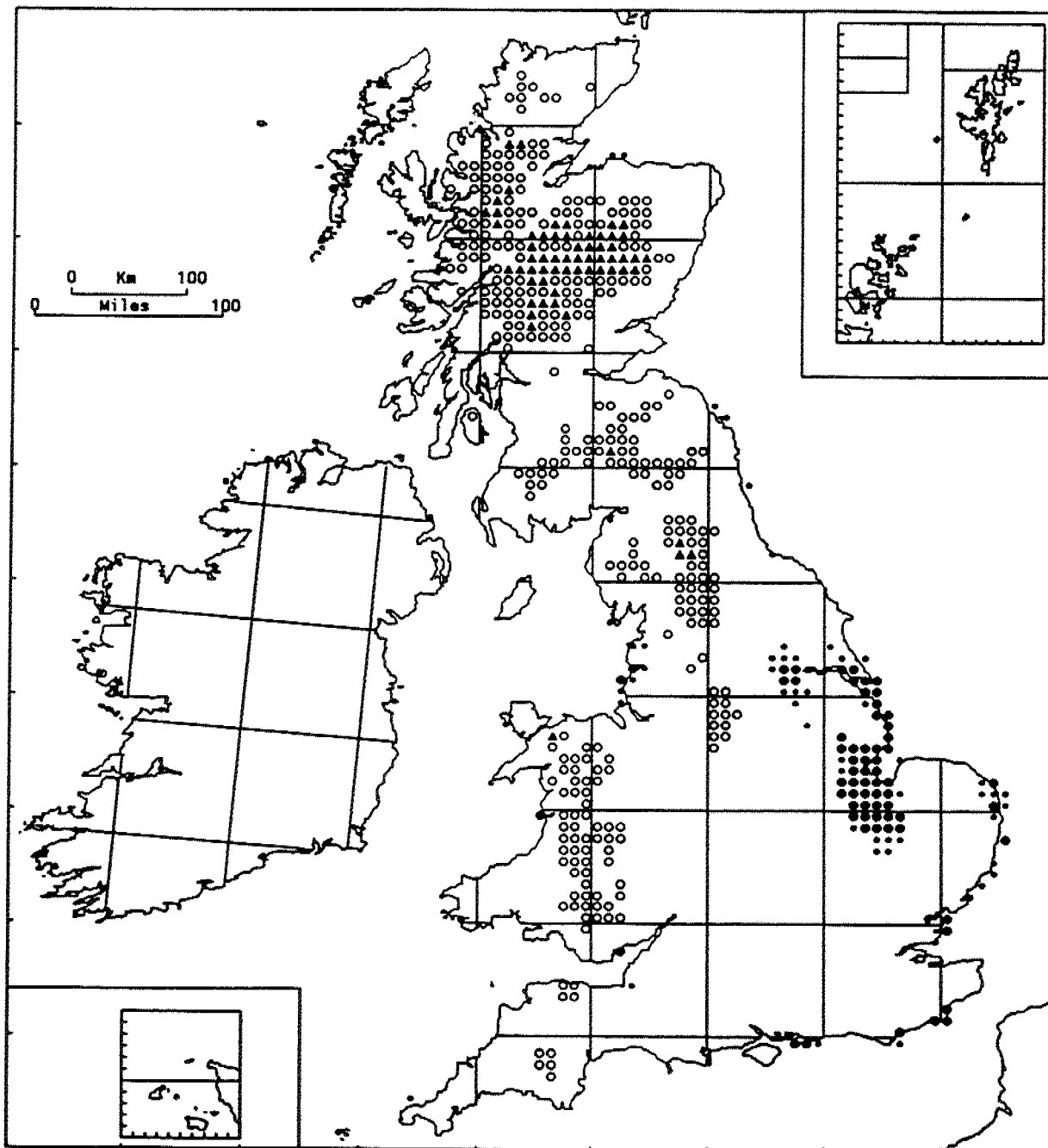
For this work, the database has been considerably updated, and translated to conform to the EC's CORINE Biotopes Classification (European Communities 1991). The CORINE system of biotope coding is hierarchical, so it is possible to specify the biotopes of a species to varying degrees of precision, eg.

31 HEATH AND SCRUB
    31.2   Dry heath
        31.22 Sub-Atlantic Calluna-Genista heaths
            31.225   British *Calluna-Genista* heaths
                31.2251   East Anglian *Calluna-Festuca* heath
                31.2252   Spring squill (*Scilla verna*) heath

An extract from the dry heathland entries in the BOD is given in Table 6.6.

Table 6.5   Red Data Book species vulnerable to sea level rise

Species	Order	% 10-km squares in occupancy at alt.[*]	
		< 5m	5-10m
Sphaerium solidum	Mollusca	100	0
Thalera fimbrialis	Lepidoptera	100	0
Eupithecia extensaria	Lepidoptera	63	0
Coenagrion armatum [**]	Odonata	50	50
Iris spuria	Magnoliopsida	50	0
Equisetum ramosissimum	Equisetopsida	50	0
Alisma gramineum	Magnoliopsida	50	0
Teucrium scordium	Magnoliopsida	38	21
Valvata macrostoma	Mollusca	38	19
Eilema pygmaeola	Lepidoptera	38	0
Viola persicifolia	Magnoliopsida	37	26
Photedes brevilinea	Lepidoptera	33	33
Lactuca saligna	Magnoliopsida	30	20
Hydraecia osseola	Lepidoptera	30	0
Limonium bellidifolium	Magnoliopsida	27	9
Lycaena dispar	Lepidoptera	0	100
Senecio paludosus	Magnoliopsida	0	100

[*]   altitude above Ordnance Datum

[**]   species believed extinct; values refer to former sites

6.3.2 Biotopes of rare species

The species-complement of BOD is not yet fully representative of the British flora and fauna, but represents a broad cross-section. The species-richness data (Table 6.7) confirm widely-held beliefs, such as the ranking first of broad-leaf woodland as the richest habitat, and the higher species-richness in calcareous than in sandy grassland. It is more surprising that fens (232 species) are hardly richer than wet heath and upland moorland (228 species). However, the latter category combines lowland, southern biotopes with Pennine and Welsh moorland and the middle-altitude Scottish upland biotopes. Thus, the full quota of wet heath and moorland species will never co-occur, whereas the finest fenland sites could be expected to contain a high proportion of the full biota. Dry heath is also comparatively species-rich, but in this case it represents a single well-defined biotope rather than a combination of geographically disparate ones. However, the high species-richness is probably because dry heathland in Britain is concentrated in a few southern counties, especially Dorset, Hampshire and Surrey (Webb 1986). These counties are centres for species-richness in Britain, primarily for reasons of climate (Watt *et al.* 1990; Lawton *et al.* in press).

There are several possible explanations for the low values for aquatic biotopes. They may reflect a greater resilience of these biotopes. There may also be a taxonomic bias at work: important freshwater groups such as Ephemeroptera and Plecoptera were excluded from the Red Data Book (Shirt 1987), and others, such as aquatic Coleoptera, are not yet in BOD.

6.4 CURRENT EXTENT OF MAJOR BIOTOPES

In the same way that rare species tend to be more threatened by change than common and widespread species, a rare biotope may be deemed to be more vulnerable than a common biotope by virtue simply of its rarity. For such biotopes, the concept of corridors or stepping-stones for dispersal may be irrelevant (if, as with mountain-tops, there is nowhere for a corridor to lead), or be too late, in that the change is likely to be to severe and too rapid, that dispersal is unlikely to keep pace.

If a biotope is rare but geographically scattered (for example, raised bogs), a different problem arises. Its climate space may be wide enough to ensure that a biotope of similar character survives somewhere within Britain; but fragmentation of the biotope is already such that it will be difficult or impossible for species to move long distances through the landscape within the biotope. Corridors do not currently exist for most rare, fragmented biotopes, and it is difficult to imagine them being created in future.

Sites can be characterized according to the species-richness of species characteristic of a given biotope. A few sites will contain the majority of the assemblage, and parts of the wider countryside will support one or two of the species in the absence of what would be recognized as the biotope itself. A threshold species-richness is needed to indicate when a biotope is present in a 10km square. For this purpose, the relative extent of the biotope is defined as the number of 10km squares containing *at least 25% of the maximum number of characteristic species which co-occur in one or more 10km*

Table 6.6 Composition of some major biotopes as held in the Biotope Occupancy Database

Biotope name	CORINE code	Species numbers Characteristic	RDB
Heath and scrub	31	22	4
Dry heaths	31.2	181	11
Sub-montane *Vaccinium* heaths	31.21	11	2
North Atlantic *Vaccinium* heaths	31.211	1	
Upland British bilberry heaths	31.212	2	1
Sub-Atlantic *Calluna-Genista* heaths	31.22	1	
British *Calluna-Genista* heaths	31.225	11	4
East Anglian *Calluna-Festuca* heaths	31.2251	1	1
Spring squill heaths	31.2252	2	2
Northern *Erica vagans* heaths	31.234	1	
Lizard peninsula *Erica vagans* heaths	31.2342	2	2
Anglo-Armorican western gorse heaths	31.235	4	2
Ulex gallii-bristle bent heaths	31.2354	1	1

Table 6.7 Numbers of characteristic and RDB species in the targeted major biotopes

Biotope	Number of characteristic species	Red Data Book species Number	Percent
Calcareous grassland	337	61	18
Inland cliffs & scree	139	58	42
Fen & reedbeds	232	57	25
Broad-leaf woodland	441	53	12
Sand dune/shingle	185	49	27
Fallow fields	126	44	35
Sandy grassland	182	32	18
Damp grassland	183	31	17
Coastal cliffs	113	30	27
Dry heath	233	28	12
Woodland edge, scrub	173	27	16
Arable	87	23	20
Wet heath & upland moors	228	22	10
Flowing freshwater	203	18	9
Raised & blanket bogs	65	18	28
Urban/industrial	88	18	9
Coastal/estuarine	93	14	15
Bocage	123	11	9
Non-eutrophic freshwater	164	10	6
Montane grassland	38	6	16
Conifer woods/plantation	53	3	6
Eutrophic freshwater	66	2	3
TOTALS	1535	379	24.7

squares. This is used, rather than the total from Table 6.7, because of the different levels of species-turnover (beta-diversity) in the different biotopes. If there are two broad geographic groupings in a biotope, 'northern' and 'southern', the total species listed for the biotope may be twice what is actually found at any individual site. Conversely, if the biotope is predominantly southern, with a gradual decline in species-richness northward, the whole assemblage may be found in one place.

The relative extents of the biotopes are presented in Table 6.8, ranked in order of decreasing extent. These ranks generally concur with common belief, though a few biotopes seem out of sequence. This is a function of the biotic indicators which the method uses. 'Arable land' is less prevalent as a wildlife habitat than its actual area would suggest, because the characteristic species of arable land in BOD are cornfield and arable weeds, not the crop plants themselves. Broadleaf woodland is rather less ubiquitous than expected because the pattern of species-richness of the woodland assemblage has a strong south-eastern bias. Hence, although the biotope is present throughout the country, many areas lack the suite of associated species. This is a possible problem when applying any indicator-species criterion nationally.

6.5   DECLINES IN MAJOR BIOTOPES - AN OVERVIEW

The change in status of each biotope could in theory be quantified by combining species decline and biotope occupancy data. Unfortunately, differences in the pattern of recorder effort over time for each taxonomic group, and in the number of species in each group, render simple summing of the slopes of regression lines ineffective (it is unduly influenced by species with large total numbers of records). One alternative would be to tabulate the numbers of species showing a significant increase or decrease. For the set of species occurring in each of the 22 biotope categories, a number X will have increased significantly and Y will have decreased significantly: the difference $(X - Y)$ could provide a relative measure of the decline/increase of each biotope category (Table 6.9). The calculations have been performed for each taxonomic group separately, then the net number of species showing a significant change is expressed as a % of the assemblage in the biotope (from Table 6.7).

These statistics are difficult to interpret, and may not reflect the true pattern of change. Adjustments are needed for the species composition of the assemblages (eg. some biotopes may be characterized mainly by flora, and vascular plants have generally shown fewer statistically significant declines as assessed by the techniques used here). Further work on refining the methods of assessing biotope change using suites of characteristic species if clearly needed.

6.6   MONTANE HABITATS - A CASE STUDY

6.6.1   Distribution of mountains

The uplands cover about 30 % of the land area of Britain. The montane component of the uplands has been defined as beginning at 610m (2000 ft) and covers about 2.5 % of Britain (Ratcliffe &

Thompson 1988). Montane areas form islands rather than corridors: the higher the altitude, the more separated the islands and the smaller the area available. The uplands as a whole, however, join into extensive contiguous blocks which may be assumed to be substantially interconnected by corridors of broadly similar habitat types. Figure 6.1 shows the distribution of high-altitude areas, and Figures 6.2 to 6.4 show the patterns of upland biotopes, defined by species assemblages.

Since the development of post-glacial forest cover, montane vegetation has at times been restricted to even smaller areas than at present. There would have been no extensive montane vegetation south of North Wales at the time of maximum woodland cover during the 'climatic optimum', c. 7000-5000 years B.P. (Buckland & Coope 1991). The tree-line is estimated to have been at 419 m (100% of highest point) on Bodmin Moor, 551 m (81%) on Dartmoor, 635 m (59%) in North Wales, 595 m (94%) in the Southern Pennines, 893 m (100%) in the Northern Pennines, 454 m (100%) on the North York Moors, and 760 m (78%) in the Lake District (Birks 1988).

Table 6.8   Relative extents of major biotopes. Relative extent is the number of 10km squares containing at least 25% of the maximum species richness (maximum number of characteristic species recorded in a single 10km square).

Ranked biotope	Relative extent	Max. spp. richness
1 Eutrophic still freshwater	1188	16
2 Sandy grassland	1160	88
3 Damp grassland	1116	79
4 Urban, suburban & industrial	1100	86
5 Fallow fields & waste ground	1080	48
6 Still non-eutrophic freshwater	1075	57
7 Flowing freshwater	1015	60
8 Woodland edge, clearings & scrub	1010	101
9 Calcareous grassland	942	137
10 Arable land with weeds	924	27
11 Coniferous woodland & plantation	894	16
12 Wet heath & upland moorland	889	83
13 Bocage	879	66
14 Dry heath	873	92
15 'coastal' sand dune & shingle	808	42
16 Broadleaf woodland	744	178
17 Water-fringe, reedbed & fen	728	79
18 Coastal cliffs and rocks	364	32
19 Inland cliffs, rocks & scree	277	16
20 Intertidal coastal/estuarine	264	30
21 Raised bogs & blanket bogs	235	10
22 Mountain-top grassland & snowpatch	52	9

Table 6.9 Relative change (%) of major biotopes using characteristic species. Positive values signify an increase, negative values a decrease. Columns are: 1, Bumblebees; 2, Butterflies; 3, Macromoths; 4, Mammals; 5, Molluscs; 6, Odonata; 7, Orthopteroids; 8, Vascular plants; 9, Net change (number of species); 10, net % of characteristic species changing.

Biotope	1	2	3	4	5	6	7	8	9	10
Intertidal			-5	1	-3	1	2		-4	-4
Coastal sand	-3	1	-25		-2		-2	4	-27	-15
Coastal cliff		1	-7		1		-2	4	-3	-3
Eutrophic water				-14	5			-9		-14
Non-eutrophic water				2	-15	4	-1	3	-7	-4
Flowing fresh-water			-4	1	-28	1	-2	2	-30	-15
Wet heath	-1		-29		-2		-2	2	-32	-14
Dry heath	-2	-2	-28	-1	-2		-3	3	-35	-15
Woodland edge	-2		-1		-4		2	-3	-8	-5
Calcareous grassland	-8	-3	-17		-4			4	-28	-8
Sandy grassland	-55	2	4		4		1	-1	5	+3
Mountain-top					6			1	7	+18
Damp grassland		2	6		4		1	1	14	+8
Broadleaf woodland		-3	-38	-2	-12		-1		-56	-13
Conifer woodland		-1	-3	-1	-2			-1	-8	-15
Raised bogs			-2		-1	-2	-1	2	-4	-6
Fen etc	-1	-3	-6		-15	-3	-1	2	-27	-12
Inland cliffs		1			-3		-1		-3	-2
Arable land		1	2	1				-5	-1	-1
Bocage	-2	1	-1	1			2	-3	-2	-2
Urban	1	2	6	1	-4			-2	4	+5
Fallow & waste grnd		1	4		3		1	-2	7	+6

6.6.2 The montane environment

At British latitudes, the theoretical lapse rate for air temperature is around 6°C per 1000 m increase in altitude (Giddings 1980). The commonly accepted scenario of temperature increases of 2-3°C would be equivalent to about 300-500 m downward shift in altitude.

The climate experienced by plants and animals at or close to ground level is far more complicated than air temperature. Rainfall, wind speeds and humidity all have an effect, as will the microclimate created by small boulders, stones, or the structure of the vegetation. The soil microclimate is also critical for seed germination, root growth and for many invertebrates: large numbers of surface-dwelling or non-soil invertebrates are subterranean as larvae (Wallwork 1976; Eversham et al. 1992). Aspect and exposure in particular can have a major effect: high-altitude south-facing slopes can have consistently higher temperatures than north-facing or level-ground sites several hundred metres lower. A further complication for many invertebrate species is that basking behaviour in sunshine can raise body temperature by 10°C above ambient (Begon

1983); sunshine is one of the most unpredictable variables in future climate scenarios.

6.6.3 **Assemblage mapping of upland biotopes**
The mountain-top grassland and snowpatch biotope has been chosen as being among the most vulnerable to climate change, and will be discussed in the context of upland biotopes in general.

Figures 6.2, 6.3 and 6.4 show the national distribution of the assemblages of species characteristic of three predominantly-upland biotopes: (i) raised bog and blanket bog, (ii) inland cliffs and scree, and (iii) mountain-top grassland and snow-patches. The first is a geographically wide-ranging grouping which combines upland and lowland biota associated with permanently-wet acid peaty soils. (Many species in bogs are common to both upland and lowland, so separation of the two is impracticable.) The three maps together provide a series of assemblages which are increasingly specialized with respect to climate, and whose biotopes are less and less likely to have an effective pattern of corridors or stepping-stones along which species could disperse in response to climate warming.

6.6.4 **Loss of montane biotopes with climate change**
On the basis of temperature alone, increases in ambient values will result in the montane islands become smaller and more separated. Using Birks' (1988) tree-line estimates as the lower limit of the montane zone in England and Wales, and the altitude equivalent of a 3°C increase in temperature, it might be predicted that no montane habitats would remain. If the increase was 2°C, only parts of Snowdonia might be expected to retain any montane environment. However, if other factors prevented tree/scrub cover from developing, the complexity of the montane microclimate might allow suitable habitat for some montane species to survive.

6.6.5 **Montane animal species**
The British montane fauna is impoverished, containing no montane endemics such as occur in profusion in the Alps, Pyrenees and Sierra Nevada. Only a few representatives of the boreal fauna of Scandinavia occur in Britain, and most of these are mainly or entirely confined to Scotland. This may well be the result of the warmer period in the post-glacial: many tundra-dwelling boreomontane insects occurred in Britain for up to 3000 years after the end of the glaciation, but disappeared with the advance of forests (Buckland & Coope 1991). However, there is limited evidence for more recent boreomontane extinctions. For instance, the snail *Trochoidea geyeri* was abundant throughout lowland and upland England immediately after the glaciation, then declined rapidly and is absent from most deposits younger than 7000 B.P. However, it has recently been found in Bronze Age deposits (*c.* 3500 B.P.) from south-west England. It appears to have survived the successional changes of the 'climatic optimum', only to be exterminated by human impacts on the landscape (Kerney & Cameron 1978; M.P. Kerney pers. comm.).

Animal species living in mountains have several types of distribution. Some are found exclusively in the montane zone, whereas other, more wide-ranging species occur throughout the upland

70

Figure 6.2 Species-richness of the assemblage characteristic of raised bogs and blanket bogs.
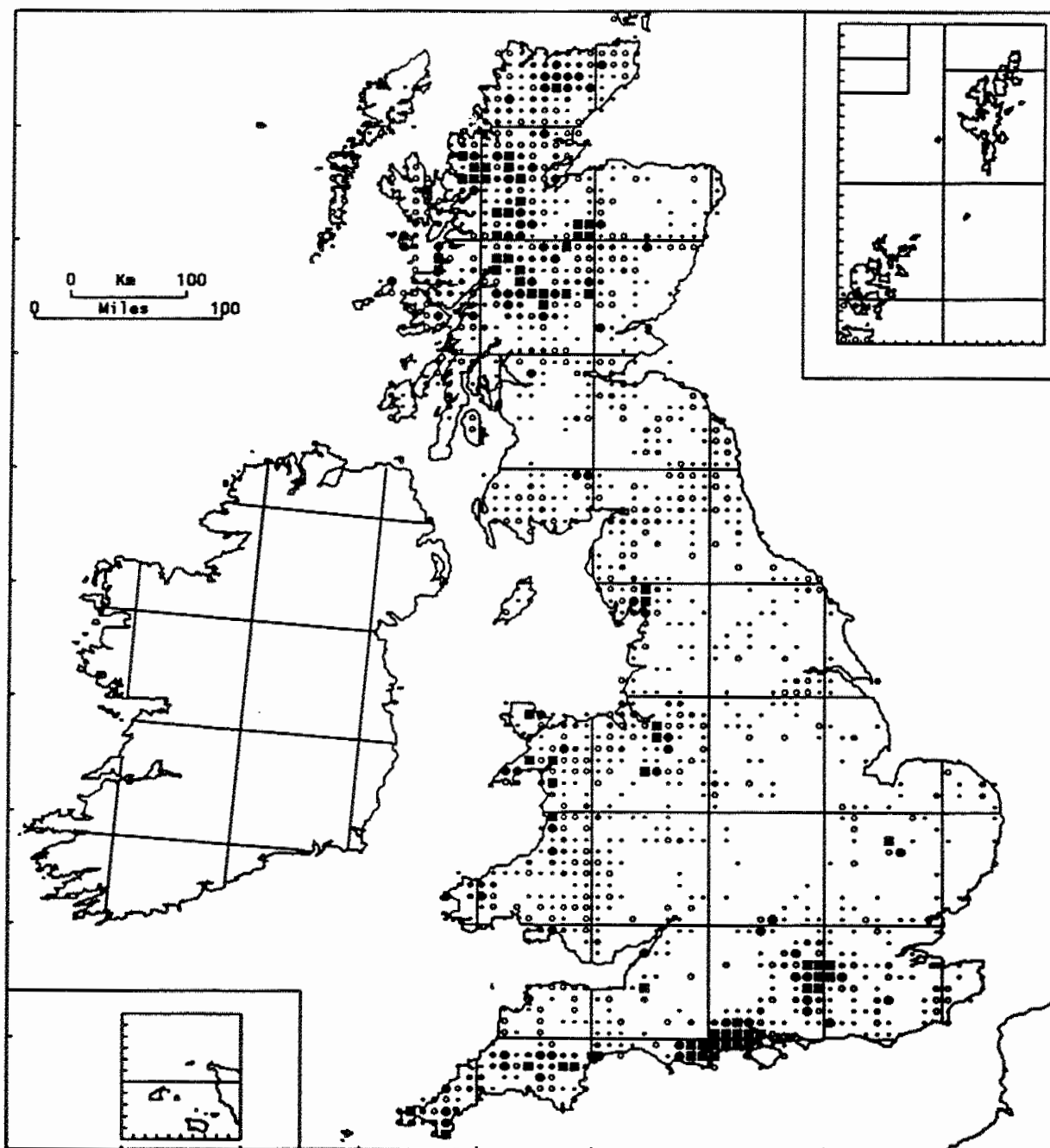
Figure 6.3 Species-richness of the assemblage characteristic of inland cliffs, rocks and scree.
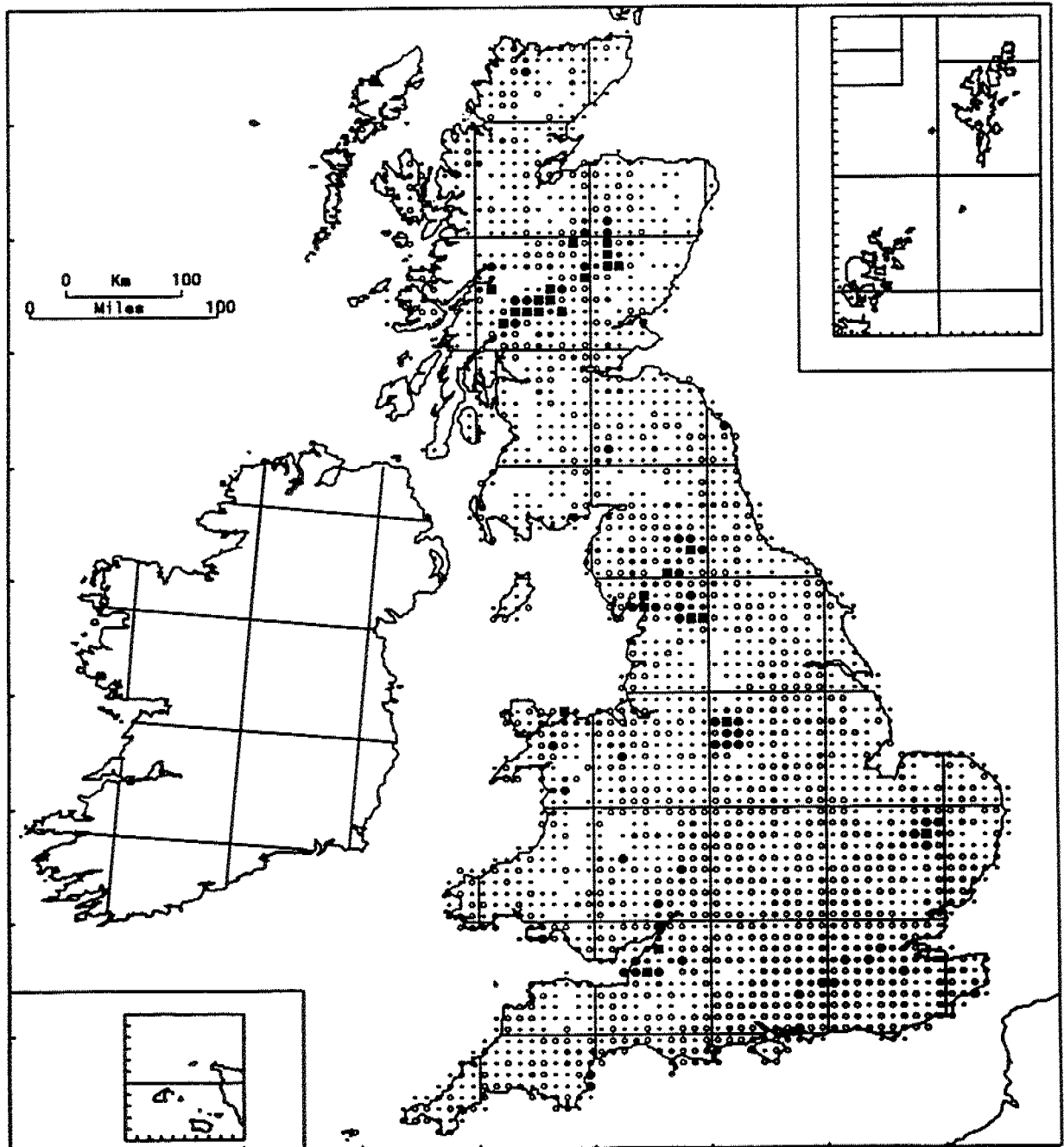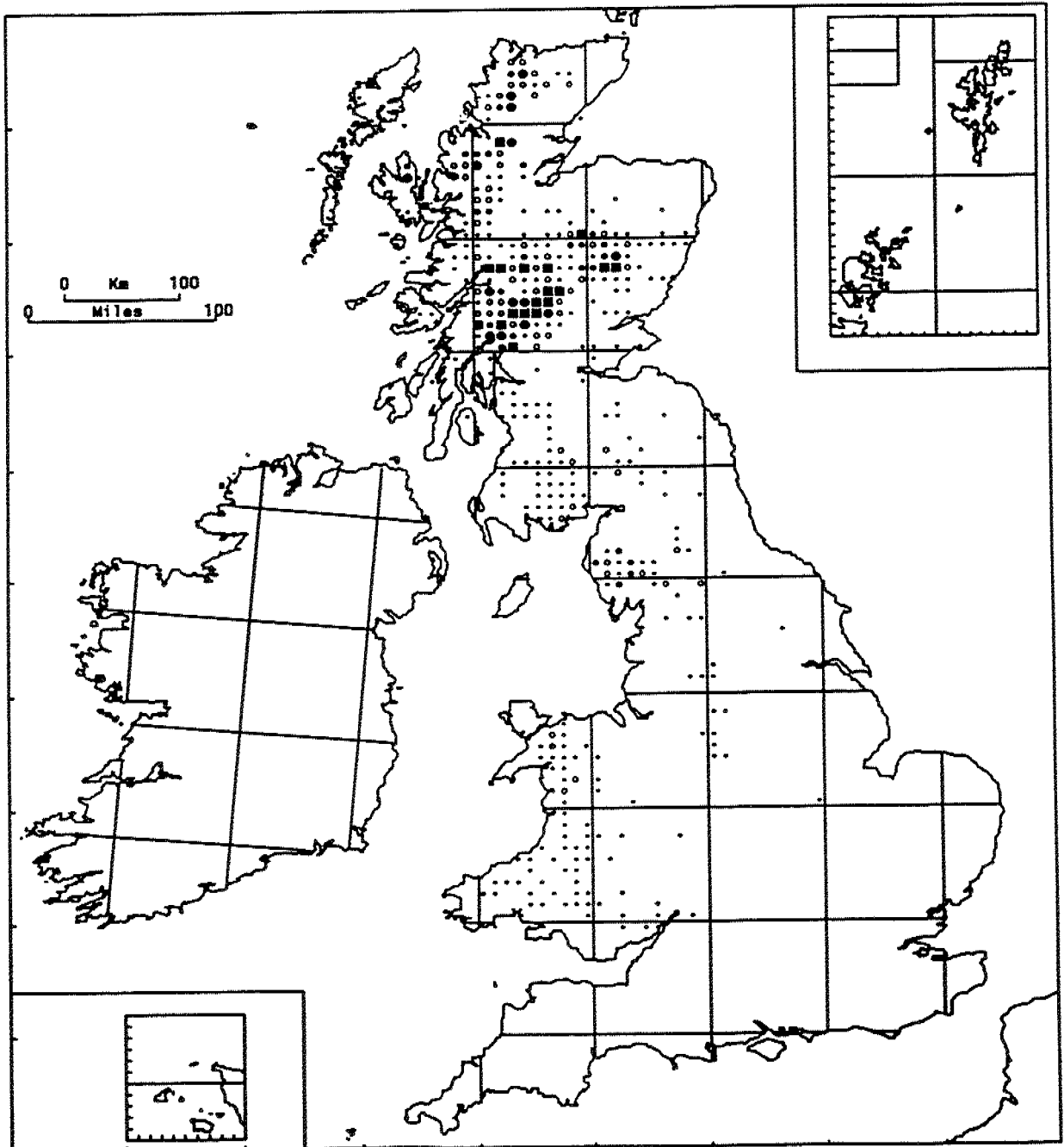
Figure 6.4 Species-richness of the assemblage characteristic of mountain-top grassland and snow-patches.

zone. In the hill pasture or moorland zone, the latter would co-exist with species restricted to this zone. There are also eurytopic species, such as the carabid beetle *Nebria salina*, which occur in many biotopes from seashore to mountain summits. Some eurytopes, such as the beetle *Carabus problematicus*, are most abundant in upland areas, and may become less frequent under climate warming, especially if more lowland competitors begin to colonize the uplands.

Common upland species ranging from, say, 300 m to the summits, eg. *Nebria gyllenhali*, should easily survive an increase in temperature. Those "trapped" on lower hills may be able to use corridors which are widespread at 300m. Similarly, those species almost exclusive to the hill pasture and moorland zone, such as the elaterid *Ctenicera cuprea* or the carabids *Trichocellus cognatus* and *Calathus micropterus* would still have suitable habitat.

Ratcliffe and Thompson (1988) state that there are, for example, 13 strictly montane Lepidoptera, 13 montane Diptera, 17 montane spiders, 22 montane beetles and 14 montane sawflies in Britain. There are, in Wales, about 21 montane beetles (Goodier 1968). The fate of these truly montane species, such as the carabids *Nebria nivalis*, *Leistus montanus* or the staphylinid *Geodromicus longipes*, might primarily depend on the extent of the present montane zone and the extent of the elevation in temperature. Many montane invertebrates are unable to fly, and therefore would be unable to move to remaining suitable habitats. Indeed, their present distributions may already be fragmented and lacking in high-altitude corridors. Suitable microhabitats might remain within the areas of their current distribution: there are, however, features of present distributions which are difficult to understand.

Many upland and montane animals have very disjunct distributions, and are absent from some mountain ranges which would appear to provide suitable habitat. Most striking of these is the Rainbow Leaf-beetle (*Chrysolina cerealis*) which occurs only in a very limited area of North Wales, and is apparently absent from English and Scottish mountains (Shirt 1987). Conversely, two snail species, *Vitrea subrimata* and *Clausilia dubia*, are abundant in the Pennines but absent from Scotland and Wales (Kerney 1976).

Fragmentation of ranges and isolation of mountain-top populations may appear to threaten the survival of invertebrate species. However, there are a few instances which suggest that isolated populations in less than ideal climate may have some hope of survival even if corridors and not available and dispersal is impossible. A small population of the snail *Clausilia dubia* has persisted at Dover Castle in Kent, almost 400km from the nearest native population, for over 150 years It is believed originally to have been introduced there. Populations of the glacial-moraine carabid *Miscodera arctica* have been found at an isolated sea-level moraine 50km away from (and 400m lower than) their nearest neighbours. The same lowland moraine and adjoining bog, in lowland south Yorkshire, supports the predominantly Scottish-montane shore-bug *Salda muelleri*, as well as a large and genetically very diverse population of the upland moorland Large Heath Butterfly (*Coenonympha*

tullia) (Skidmore *et al.* 1987; Eversham 1992). Fortuitously, the bog fauna at this site is known to have persisted for over 3000 years: it is substantially similar to that associated with a Bronze Age trackway across the mire (Buckland & Kenward 1973; Buckland 1979).

Rare species, such as the Rainbow Leaf Beetle on Snowdon, present a different problem for conservation. It is not clear to what extent their present isolation depends on the uniqueness of the combination of abiotic and biotic factors at their specific site. Even if, in temperature terms, sufficient habitat remained, it seems unlikely that a similar combination of factors would occur given a climate change. A list of exclusively montane Lepidoptera and spiders of England and Wales is given by Ratcliffe (1977).

The mobility of birds allows them to forage over wide areas and no species is confined to the montane zone. Only three species, Ptarmigan (*Lagopus mutus*), Dotterel (*Charadrius morinellus*) and Snow Bunting (*Plectrophenax nivalis*), breed exclusively in the montane zone in Scotland (Fuller 1982). Golden Plover (*Pluvialis apricaria*), Twite (*Carduelis flavirostris*), Ring Ouzel (*Turdus torquatus*) and Wheatear (*Oenanthe oenanthe*) breed in both the montane and submontane zone. These species, together with those, such as Raven (*Corvus corax*), nesting on rocky outcrops or crags, depend more on the availability of suitable habitat than on altitude. There are, for instance, breeding populations of Twite at sea level in the Hebrides (Sharrock 1976) and in southern Yorkshire (Marshall *et al.* 1989). It therefore appears that an increase in temperature might have little effect on the bird species living in the mountains of England and Wales.

The only 'montane' mammal in England and Wales, the Mountain Hare (*Lepus timidus*), is thriving in the southern Pennines where it was introduced *c.* 1880. It seems unlikely to be affected directly by climate change as it occurs abundantly in the lowlands of Ireland, where the Brown Hare (*Lepus capensis*), which replaces it in lowland Britain, is a scarce introduction (Barrett-Hamilton 1898; Ni Lamhna 1979; Arnold 1993). It is conceivable that there could be climate-modified competition effects between the two in future (Fargher 1977).

6.6.6   **Montane plant species**
The British montane flora, like the fauna, is species-poor compared with mountains on the European mainland. Only 15 montane plant species occur more plentifully above than below the tree-line in England and Wales (Ratcliffe 1977, 1991). These species would be more affected by climate change than the more widespread species which occur both above and below 600m. Individual populations of these high-montane species will, of course, be isolated from each other and it would not be possible for them to use the upland corridors to move to a new location. Evidence for the duration of current isolation is hard to find; but in at least one case, *Lychnis alpina*, there is evidence for genetically distinct edaphic races at each existing site, which may point to prolonged isolation (Proctor & Johnson 1977).

75

It is, however, difficult to predict what effect increased temperature might have on montane species, which occur at the extremes of their distribution. Some species are fairly plentiful where they occur, but need special soil conditions. Others do not appear to be limited by lack of suitable habitat, but by unknown limiting factors, or perhaps by chance (Ratcliffe 1977). Sheep grazing might be an important factor in some areas, survival being due to protection from grazing on cliffs or in crevices rather than to temperature.

Many montane plants can be transferred to higher temperatures, in, for example, rock gardens, where they seem to thrive. A rich assemblage of montane species occurs at sea level in the warm, wet Atlantic climate of the Burren in Co. Clare, where geology and land-use history have permitted the survival of a diverse flora containing elements from the early post-glacial (Webb & Scannell 1983).

The ecophysiology of the montane flora is not thoroughly known, but there is often no inherent complication with germination. *Saxifraga hypnoides*, for example, was shown to disperse viable seed in its natural environment, although none germinated. The seed germinated easily in the laboratory at 18°C (Ripley 1989). In addition to the montane species being adapted to the harsh environment, reduced competition probably plays an important part in their survival. As with animals, it is difficult to predict the effect of a change in one factor, i.e. an increase in temperature, on species which are surviving in suitable micro-habitats.

6.7   CURRENT EXPANSIONS OF RANGE OF SPECIES IN BRITAIN
At any point in time, few species have an entirely static geographic distribution. BRC recording schemes continually reveal apparent changes, some species being found in 'new' areas, while others become extinct at sites where they have long been known. Although it is difficult to be certain that a species has arrived recently rather than been overlooked by previous recorders, or to show that it is no longer present, there is some evidence that the proportion of species changing significantly at any time may be rather high. Eversham and Arnold (1992) tabulated some 31 species of British breeding bird which have expanded their range markedly since 1700. At least 20 of these had shown a similar increase in Europe over the same period.

A few species stand out from the ordinary mode of gradual change, and appear to be expanding their range so rapidly annual differences may be detected. In a handful of cases alerting recorders to such change has generated data of sufficient quality to chart the expansions of range unequivocally. For example, the gradual northward extension of the hedge brown butterfly (*Pyronia tithonus*) has been analysed by Pollard (1991, 1992), using both BRC distribution data and censuses from individual sites within the Butterfly Monitoring Scheme. It appears that the expansion began in the 1920s or 1930s, following half a century of southward retreat. Also the post 1940 colonization and spread of the migrant hawker dragonfly (*Aeshna mixta*) has been described in the literature

(Merritt *et al.* in press; map of the post-1980 expansion by Branson 1990).

The long-winged conehead (*Conocephalus discolor*), a bush-cricket, has been known from several southern counties in England since the 1920s, but had never been found more than 10-15km from the coast. During the hot summers of 1989 and 1990, records of actively-flying, extra-macropterous forms were received, and the species was found for the first time at sites up to 50km inland (Figure 6.5). Fortunately, the counties of Sussex and Surrey had been thoroughly surveyed during the 1970s and early 1980s, and *C. discolor* was not present in the areas in question. In the same two seasons, the species was recorded in Cornwall and on the Scilly Isles for the first time. Laboratory studies have shown that both high temperatures and high population densities give rise to longer-winged forms, with larger wing musculature and greater dispersive powers (Ando & Hartley 1982), hence long-distance dispersal during hot summers might be expected. Coincidentally, large areas of agricultural land in Surrey had, in the previous three years, been turned over to set-aside, and by 1989 provided ideal habitat for *C. discolor*. Substantial populations became established, and two unexceptional summers later, the species appears to be maintaining itself. Thorough surveys have shown that the distribution is patchy; there are large areas between the newly-founded populations and the source populations in which *C. discolor* cannot be found despite the many suitable areas in between the established colonies. Although the evidence is circumstantial, it would appear that corridors played no part in this range expansion. Appropriate 'stepping stones' of habitat were up to 30 km apart.

A final example is provided by a tree-wasp (*Dolichovespula media*; Figure 6.6), first recorded in Britain in 1980 (Else 1989; Edwards 1989). It is unlikely to have been present for very long prior to this, as a thorough survey in the 1970s failed to reveal it (Archer 1979). No specimens had been found by environmental health officers or pest control companies, who are called on to destroy large numbers of wasp nests each year. By 1987 *D. media* was well established in Kent and Sussex. In the succeeding five years, the species has expanded its range north and west by 150km. Most individuals and nests recorded so far have been in gardens, and the species is now a common suburban insect in many parts of the south-east (Else 1992, and in press). However, there are large gaps in its range, and a thorough survey of wasp nests disposed of by Rentokil shows it to be very patchy. As with *C. discolor*, circumstantial evidence suggests that individuals of the species are travelling long distances, and establishing colonies well beyond the current range (giving an expansion rate of about 30km per year). Whether such dispersing queens travel along linear landscape features, and whether they feed on the way is unknown.

The evidence from the last two examples of dispersal is not supportive of the need for corridors, at least for the more mobile and eurytopic insect species. It is debatable how widely applicable this conclusion is. The two species are both generalists, with a broad tolerance of common habitats. They are clearly highly mobile, at least in suitable weather conditions, and highly fertile, being

77

able to build up high population densities in only one or two generations. They are thus likely to be very different from stenotopic species which seldom stray from specific habitats, and have little in common with less mobile or fecund organisms.

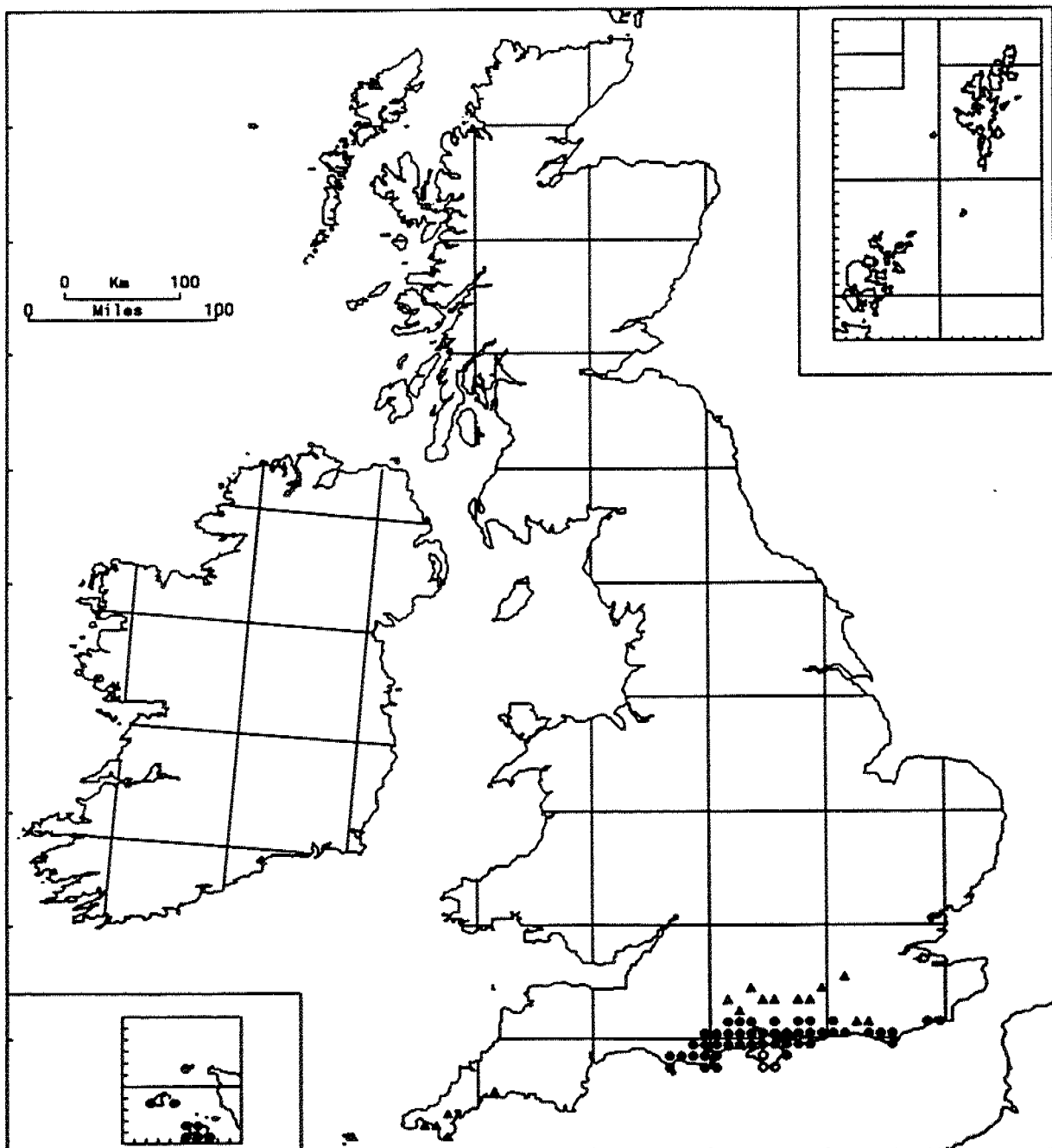**Figure 6.5** The distribution of *Conocephalus discolor*.

Figure 6.6 the distribution of *Dolichovespula media*.

Range edge in 1987